

**Ecosystem Services – Theories and
Applications: Opportunities for Humanity
to Regain Paradise**

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ABSTRACT

Ecosystem services, the benefits humans derive from nature, represents a radical departure in our perception of linked environmental and social problems and the actions we need to undertake to address those urgent challenges. Due to its increasingly widespread policy prominence, understanding and appraising its conceptual and practical benefits whilst at the same time acknowledging its potential pitfalls represents an important endeavour. Comprising seven parts and sixteen chapters, the first five parts of the thesis outline the main environmental and social challenges we face, presenting the core foundations, contemporary debates and developments in ecosystem services scholarship, whilst also underlining its increasing coalescence with sustainability discourse. In Part 6 we focus on a key application of ecosystem services with respect to its translation into incentive-based environmental management schemes, namely: payment for ecosystem service programmes and agri-environment schemes. We present a systematic global analysis of payment for ecosystem services programmes, highlighting the successes and challenges they face, whilst also providing an approach to improve their design and evaluation as a route to maximise their effectiveness. Turning our attention to a globally significant ecosystem, the thesis assesses the prospects for jointly developing seagrass Blue Carbon initiatives and payment for ecosystem service schemes, arguing that complementing these activities would produce significant climate, conservation and livelihood benefits. Switching contexts, from focusing on incentive schemes primarily in operation in developing countries to those designed to balance productivity and conservation matters in the agricultural sector of developed countries – the thesis explores the stakeholder and institutional factors affecting agri-environment scheme operation and implementation through the eyes of key operatives. Finally, in Part 7, I argue that a landscape framing and approach to ecosystem services provides an effective route to improve environmental management decision-making and policy as well as comprehensively addressing the linkages between ecosystem services and human-wellbeing.

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All accompanying material (e.g. additional raw and processed data), which have been clearly referred to in the text as Supplementary Material, are included on the accompanying CD.

PREFACE

Today we live in a world facing dramatic change as a consequence of myriad complex environmental and social challenges. The ecosystem services paradigm is presented as a progressive and pragmatic scientific framework and policy approach for developing solutions to many of these linked global social-ecological issues, and as a consequence has been increasingly embraced by the wider environmental and conservation science community. However, because the ecosystem services paradigm co-aligns economic and environmental science, in many quarters, it remains a hotly debated and contested field of inquiry even though it has seen rapid policy uptake over recent years. This thesis explores the social-ecological dimensions of ecosystem services as well as its practical implementation in the form of incentive-based instruments for natural resource management. Its purpose, therefore, is to advance a broader understanding of ecosystem services and its applications, with a view to providing knowledge and information relevant to improving the use of ecosystem services in policy and decision-making arenas.

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Secondly, I wish to extend a heartfelt thank you to my family, and in particular my mum, for always being there to support and encourage me in whatever path I chose to follow, for never failing to be anything other than positive, and always being on hand at the end of the phone. Likewise, I also wish to thank my sister for her love and support, usually shown in the form of baked goods – the long days staring at the computer were very often only made manageable by having some delicious cakes to nibble on! I also want to say thank you to my friends and family who are no longer here, but who, from childhood into adulthood, have along the way imparted their love, knowledge and friendship – in a sense this thesis is as much a product of you as it is of me, and thus it belongs to you as much as it does to me.

Thirdly, I would like pay a special tribute to my partner Hannah who has shared the last 18 months of my PhD journey with me and who has always been unwavering in her understanding and support – it is finally done!

Fourthly, I want to thank Prof Mark Huxham for bringing me on board with the CESEA project, it was a fortuitous meeting at the BES Annual Conference in Birmingham back in December 2012. It has been a productive engagement and I wish to extend my thanks to the other colleagues involved in the project who I have gotten to know over the last few years, in particular, Dr Martin Skov, Prof Hilary Kennedy, Dr Caroline Upton and Dr Fiona Nunan. I think they have all appreciated York and Kings Manor as a base for our regular project meetings.

Fifthly, I would like to thank all my fellow colleagues in the Environment Department, past and present, and particularly those that I started this PhD journey with and who have also reached journey's end: Dr Lucy Anderson, Dr Ricky Lawton, Dr Steve Roccliffe and Dr Sarika Cullis-Suzuki – we made it and we survived!

Sixthly, a PhD is to a significant extent an individual pursuit and as befits such a solitary endeavour can be quite isolating, so having friends and family on tap is a necessity - so for those that have not garnered a specific mention I wish to thank you too – you have all helped to make this PhD journey a success.

Finally, I wish to acknowledge both the ESRC and NERC for supplying the funding necessary to undertake this PhD.

AUTHOR'S DECLARATION

I declare that this thesis is a presentation of original work and I am the sole author. This work has not previously been presented for an award at this, or any other, University. All sources are acknowledged as References.

The following Chapters 11, 12 and 13 are based on first authored peer-reviewed published works. Specifically:

Chapter 11:

Hejnowicz, A.P., Raffaelli, D.G., Rudd, M.A., White, P.C.L., 2014. Evaluating the outcomes of payments for ecosystem services programmes using a capital asset framework. *Ecosystem Services*. 9, 83-97.

Chapter 12:

Hejnowicz, A.P., Kennedy, H., Rudd, M.A., Huxham, M., 2015. Harnessing the climate mitigation, conservation and poverty alleviation potential of seagrasses: prospects for developing blue carbon initiatives and payment for ecosystem service programmes. *Frontiers in Marine Science*. 2:32.

Chapter 13:

Hejnowicz, A.P., Rudd, M.A., White, P.C.L., 2016. A survey exploring private farm advisor perspectives of agri-environment schemes: The case of England's Environmental Stewardship programme. *Land Use Policy*. 55, 240-256.

In Memoriam

I dedicate this PhD Thesis to the memory of my father Peter Francis Hejnowicz (1953-1990).

Over the last twenty five years life has not always been easy but, I hope – in this work – to
have made you proud dad.

*“You did not choose Me, but I chose you. And I appointed you to go and bear fruit--fruit that will
remain--so that whatever you ask the Father in My name, He will give you.” (John 15:16)*

Introduction: Paradise Lost Or Paradise Regained: Re-establishing The Garden Of Eden After The Fall

The Garden of Eden is the classic metaphor of an idealised state of nature: a place of plenty, of all creatures great and small and; perhaps, most importantly, a state of nature in which man is an intimate and integral part:

“How first began this Heav'n which we behold Distant so high, with moving Fires adornd Innumerable, and this which yeelds or fills All space, the ambient Aire, wide interfus'd Imbracing round this florid Earth..[...]. And Earth be chang'd to Heav'n, & Heav'n to Earth, One Kingdom, Joy and Union without end.” (John Milton, Paradise Lost (1667), Book VII)

This is the mythical phase of man’s history where “man and beast”, before The Fall, reside in a state of complete harmony – as Carolyn Merchant explains:

“The Garden is filled with spring-fed water out of which the four rivers flow. It contains the “beasts in the field”, “fowls of the air”, cattle, snakes, and fruit trees, including the fig, as well as humans “to dress and keep it.” (Merchant, 2008 [2003] pg. 313)

Reconceptualising this foundational Christian narrative through the prism of today’s environmental language results in a re-imagined harmonised utopian environment (i.e. a balanced human-natural-coupled system): a pristine world (i.e. no resource exploitation, pollution or species extinction) full of rich complex landscapes (i.e. multiple ecosystems and habitats where the natural functioning of ecological processes is maintained); where morality rules (i.e. human behaviour and decision-making is based purely on a moral-ethical framework and underlying precepts); where interactions between people are equitable and fair (i.e. no social or class divisions or centralised power structures and bureaucracies); and where abundance and prosperity are everywhere around (i.e. all needs are catered for, and there is a complete absence of famine, disease, war and poverty). In this Edenic pre-lapsarian state the landscape is plentiful and nature is regarded positively (Merchant, 2008 [2003]).

Of course, as we know, this plentiful, peaceful and serene state of affairs was not to last: soon we fell from grace. In this post-Edenic world, the post-lapsarian landscape is very different, as Merchant (2008:315) describes:

“With the Fall from Eden, humanity abandons an original, “untouched” nature and enters into history. Nature is now a fallen world and humans fallen beings.”

In the new “post-Edenic adama” we became the slaves of our own wayward natures; expelled, our savagery was outed and we lost our way. In the eyes of the 17th Century English Philosopher Thomas Hobbes, we could only be kept in check and saved from our natural beastliness by the “Leviathan”. Man had disgraced himself after trying to set himself apart from creation, and soon we divorced nature and we forgot our place in the world:

“Those pure immortal Elements that know No gross, no unharmonious mixture foule, Eject him tainted now, and purge him off As a distemper, gross to aire as gross, And mortal food, as may dispose him best For dissolution wrought by Sin, that first Distemperd all things, and of incorrupt Corrupted.” (John Milton, *Paradise Lost* (1667), Book XI)

In our downtrodden state we soldiered on and we picked ourselves up entering into a kind of human Genesis project driven by the Agricultural Transition where we toiled to make the land plentiful and abundant. Upon and from which the green stems and shoots of civilisation were born. We developed societies and Great cities, we mechanised and harnessed nature for our own ends, and we grew and prospered on the wealth we created. But, many times over, we rose and fell with tragic consequences for countless societies as their time upon this Earth waxed and waned. Yet, perhaps as a consequence of these events we found redemption and we became inspired. However, our inspiration did not produce a simple easy trajectory where everyone’s a winner; in the process we created all manner of social divisions (societies riven with inequalities) as well as persistent and harmful environmental problems that threaten to undermine life itself.

This narrative is obviously highly simplistic and is, in essence, a traditional mythical story. Since the first telling of this story we now know of our evolutionary ancestry and development (our Descent through the Hominid lineage) and, more recently, the ways we have interacted with and moulded the environments around us over millennia, particularly during the Holocene period (11,500 years ago to the Present). However, though mythical and highly stylized, this narrative construct is not devoid of merit. As a device it has some basis in reality. The central theme of a prehistoric idealised state, at the dawn of time, in which man could aspire to return, inspired the European Romantic tradition in literature, art, music and culture and its conflicts with Nineteenth Century mass industrialisation, squalor and poverty. And, on the back of a growing Enlightenment discourse, especially in science, and the founding of “Modern Man”, generated the first forays into conservation with the establishment of a number of Great American National Parks such as Yellowstone (1872) and Yosemite (1890). The “conservation project” (though it was perhaps not as organised as that) continued into the Twentieth Century and was re-invigorated and re-energized by such leading lights as John Muir (who founded the Sierra Club), Aldo Leopold (who popularised his holistic view of nature in his *Sand County Almanac*) and Rachel Carson (who in her book *Silent*

Spring voiced some of the increasing environmental problems wrought by human activities). The underlying platform of Twentieth Century conservation has been that there is an idealised state of nature, before man's intervention, to which we can return.

Today we are witnesses to a time of great social-environmental upheaval: a planet, in the words of Paul Collier, increasingly "plundered" and "under pressure". Our planetary Eden remains threatened. Global climate change, large scale land conversions, agricultural expansion, deforestation, over-fishing, mining and mineral extraction, and water contamination etc., to name but a few, are just some of the many varied examples of how human activities are undermining ecosystem processes, human livelihoods, and broader social and ecological sustainability. Part of the transformation of Twentieth Century conservation has been the recognition that attempting to return to a better functioning natural environment does not mean that man needs to continue to remain divorced from it. Protected areas, at least in the terrestrial sphere, have been transformative and remain important conservation tools but they alone are not the answer to a sustainable future. Especially when for the majority the future is an increasingly urban one.

Clearly the landscapes which we co-inhabit and co-produce with "nature" are to a large extent human-dominated, shaped and derived. The future fate of these landscapes is inextricably linked to human development (i.e. the developmental routes and pathways we pursue in the realms of politics, economics and governance factoring in our socio-cultural traditions). Globalisation has shown that the patterns of Nation State interactions, for example through trade, which drive as well as feed production and consumption activities can have far reaching consequences for natural resources and ecosystems located half a world away. Through this realisation we have entered a phase where we are increasingly thinking in terms of social-ecological systems, of a return to the idea of "stewardship" rather than "dominion", of recognising the mutualism between society and nature and how the future of the former lies in the healthiness of the latter. This is the legacy of the last 30 years in particular.

It is a legacy that since its inception has increasingly observed a coalescence of environmental/ecological science, conservation, economics and social science. Environment problems and concerns have become mainstreamed as increased emphasis has been placed on the importance of the science-policy dialogue. Environment and development concerns have moved front-and-centre in an era of more widespread and potent advocacy, civil society, and social media. In the last 25 years, the merging and broadening of disciplines and disciplinary interactions has led to what we now regard as dual environment-development problems (and I use the term *development* in the widest sense possible).

Human-wellbeing has become the pivot around which we frame these issues, challenges and debates. Applying an anthropocentric lens to the way we seek to improve the lot of the natural environment and people has led to a much greater focus on the economics of environmental problems, natural resource use, biodiversity conservation, sustainable development and sustainability. We now view, especially for the expediency of policy, the environment in terms of natural capital. An asset, which for much of our history we have taken for granted, mainly because we did not perceive it as being finite or thought we could substitute its depletion through technological replacement. The language which we use to describe the “capital flows” that derive from this natural asset stock that benefit human-wellbeing is ecosystem services (ES). And like any asset, if it is to be maintained and allowed to grow then it needs to be well managed.

The ecosystem services paradigm, like the protected area paradigm of the previous Century, represents a transformative shift in how we approach and conduct the business of conservation and environmental resource management in the Twenty First Century. We have now come to conceive the environment and biodiversity in terms of the ecosystem services they supply, and whether or not our actions contribute to a decline in ecosystem services provision and/or lead to the production of unwanted trade-offs between various ecosystem services.

Like State-provided public services, managing ecosystems effectively to ensure they are able to function in a way that sustains ecosystem services means making choices and taking decisions regarding the provision of those services. Often such choices are made under highly constrained circumstances, based on the values we attach to them and how we see society developing. Implementing these choices requires the devising of policy. Traditionally, environmental policy instruments have tended to be regulatory top-down command-and-control measures. But increasingly, voluntary incentive-driven initiatives with more inbuilt flexibility and creativity are being used in developed and developing country settings to manage the provision of environmental public goods and common pool resources. It is against this backdrop that this PhD locates itself.

Returning to our Garden of Eden device seems quite natural in the context of ecosystem services. After all, it is a highly anthropocentric narrative of human-environment relations. And, like any garden, it needs to be managed if it is to continue to flourish year on year and paradise be regained.

Viewing the Garden of Eden from a “management” perspective contextualises the underlying theme to this thesis. Namely, how best are we to conceive of this garden and its workings to manage it in a sustainable and effective way? And secondly, how best can we

encourage individuals, households, communities, farmers and landowners to manage the environment sustainably in a way that maximises the ecosystem services it provides, what options are available? The purpose of the thesis therefore is to provide an overview of the theory and applications of ecosystem services, to offer a practical and policy-relevant evaluation of the ecosystem services concept and its implementation.

I feel it important to point out of course that, unlike an individual garden, the environment is far more complex and uncertain, and that any actions we carry out (e.g. the particular management regimes we undertake) are often conducted under conditions of significant uncertainty. Secondly, unlike the Garden of Eden perhaps, there is no single idealised “static” state of nature. Nature and life are ever changing. The “ideal” will be conditioned by scale (spatial and temporal), geographic location and context (many aspects being contingent). Ultimately, there will be many Gardens of Eden across the world, inhabited by many different gardeners.

With this in mind, the thesis presented is composed of seven Parts. Parts 1 to 5 survey the underlying foundations of ecosystem services. Part 1: The State of the Planetary Garden is composed of two chapters: Chapter 1 *The Garden we Find Ourselves In* describes the present condition of our post-Edenic environment, having briefly discussed some of the principal environmental challenges; in Chapter 2 *Connections in the Living Garden: Biodiversity and Ecosystem Services* we highlight recent research exploring the relationships between biodiversity, ecosystem functioning and ecosystem service provision. From there, in Part 2: The Human Garden, we reflect on the complexity of our Human Genesis and the social-ecological character of our post-Edenic landscape overviewing current themes in social-ecological thought, in particular, we place specific emphasis on the urban gardens we have created in which most of us now find our home (Chapter 3 *The Garden as a Social-Ecological System*). Relatedly, in Chapter 4 *The Stress and Strains in Social-Ecological Systems* we discuss two principal themes in social-ecological research that have guided and underpinned social-ecological appraisals of environmental challenges and opportunities, namely resilience theory and regime shift theory. In Part 3: The Garden in the Age of Sustainability, Chapter 5 *Sustainable Development of the Garden – Ecosystem Services and Human Wellbeing* discusses the growing interconnections between environment and development challenges, and against this background identifies the increasing coalescence of sustainability and ecosystem services narratives in global environmental discourse and policy.

In Part 4: Assessing the Garden, we move on from these discussions to consider how we might investigate how the Garden is changing, and so we highlight the importance of quantifying and assessing changes in the Garden. Specifically, Chapter 6 *Quantifying the Garden* considers three main technical and methodological developments used to measure and

evaluate changes in ecosystem services, namely: trade-offs, mapping and modelling and indicators. An important facet in how the ecosystem services paradigm functions and is implemented in a policy context revolves around attributing values to services as a way of informing decision-making processes and choices – as such environmental valuation is a dominant element in the theory and practice of ecosystem services. In Part 5: Valuing the Garden we take a look at this important element, first by considering its value-based dimensions – documenting intensive debates between monetary and non-monetary valuation approaches and highlighting key technical developments (Chapter 7 *The Nature of Value*). Then, in Chapter 8 *The Value of Nature* we provide a brief overview of recent valuation assessments detailing the global and local importance of ecosystems and the services they provide. In the final chapter of Part 5, Chapter 9 *Problems Come in Threes*, we discuss in more detail the underlying issues of uncertainty, discounting and benefit transfer and how these affect and influence the process of valuation and the confidence we can have in the outputs of valuation exercises.

Having outlined these basic foundations, discussions and issues Part 6: Managing the Garden sets out three case studies that investigate how the ecosystem services paradigm is implemented in an environmental management context, and in particular, focuses on two different (but overlapping) incentive-based policy instruments – payments for ecosystem services (PES) and agri-environment schemes (AES). Prior to the exploration of these case studies, Chapter 10 *Providing Public Goods – Ecosystem Services and Externalities* provides a brief exposition outlining what environmental public goods are and the basic theory behind how incentive programmes function to enable the provision of goods and services.

Moving on, Chapter 11 *Payments for Ecosystem Services: An Assessment of Global Outcomes* then proceeds to present a systematic analysis of global payment for ecosystem services programmes. Using a capital asset framework to assess the effectiveness of these programmes in terms of their capacity to deliver natural resource management and development objectives, the analysis focuses on evaluating programme outcomes in terms of their social, human, natural, financial and institutional capital elements.

Maintaining a concern for PES, Chapter 12 *Seagrasses and Incentives: Uniting Climate Mitigation, Conservation and Poverty Alleviation* investigates a globally important ecosystem and the possibilities of developing combined carbon management and PES programmes. More specifically, we review the prospects of further inclusion of seagrass ecosystems in climate policy frameworks, focusing particularly on carbon storage and sequestration, and the potential for developing payment for ecosystem service schemes that are complementary to carbon management, thereby jointly aiding conservation, climatological and development objectives.

In the final chapter of Part 6, Chapter 13 *Intermediaries and Agri-environment Schemes: Private Farm Advisor Perspectives on England's Environmental Stewardship Schemes*, we switch our attention away from PES schemes, which are predominantly employed to address environment-development natural resource issues in the “Global South”, and instead focus on the agricultural sector in the “Global North”. In particular, we consider the employment of agri-environment schemes as policy instruments to mitigate competing production and conservation objectives in a UK and England context by considering the example of Environmental Stewardship schemes. Specifically, we are concerned to highlight the role and views of intermediary actors in the operation and implementation of these schemes, and so our case study is based on a survey of private farm advisors.

Collectively, these case studies highlight the differing situations in which ecosystem services, in the form of incentive-based schemes, is increasingly applied across low, middle and high income countries, in informal and formal land management contexts, to address the interrelated issues of ecosystem service provision, conservation, production and development. Simultaneously, these case studies also serve to illustrate both the pitfalls and importance (in terms of design, operation, implementation and effectiveness) of elements such as governance; institutional arrangements; accountability and transparency; stakeholder participation and involvement; benefit sharing; social and ecological targeting, monitoring and compliance; private and public sector involvement; and viable and secure sources of funding.

In the final part of the thesis Part 7: A Proposal for a Landscape Approach to Future Garden Management we offer a conceptual and practical reformulation of ecosystem services rooted in a landscape framing, as a way of addressing many of the issues that the ecosystem services paradigm fails to fully grasp and implement. In essence, the integrative landscape approach we present argues that the most appropriate way to manage ecosystem services provision sustainably is via an integrated social-ecological systems approach that has the notion of landscape at its heart. Landscape being the point of intersection and articulation between man and the environment, it thus has the potential to represent a far more appropriate, coherent and tangible vehicle for the ecosystem services concept in both policy and management arenas than current ecosystem service frameworks.

As such, in Chapter 14 *Landscape: Meaning, Narrative and Unification* we explore the complex conceptual constructions of landscape, especially its rich tapestry of meanings. In particular, we explore the concepts immediacy and relationship to notions of identity and place, the fundamental connections it has with individual psychology, wellbeing, cultural identities and religious and spiritual practices, as well as its implications through legal, social, economic and political processes for shaping, propagating and embedding issues of social inequalities, power asymmetries and exploitation in relation to aspects such as gender, race,

justice and natural resource access. From these rich discussions Chapter 15 *Outlining a Landscape Approach* presents a brief overview of our notion of a landscape approach to ecosystem services, identifying its advantages over the present ecosystem approach to articulating the concept of ecosystem services. Having sketched out our broad conception of a landscape approach, Chapter 16 *Landscaping Ecosystem Services* presents a detailed interrogation of how ecosystem services maps onto our landscape approach, and the implications our framework has for advancing the conceptual, practical and applied dimensions of ecosystem services in terms of its capacity to further the sustainable management of our Garden (or Gardens).

In the final chapter, the Discussion, we present an overview and synopsis of the significance of each Part of the thesis: highlighting the salient points to acknowledge, and documenting the main advances that this thesis has made. Subsequently, we then explore, based on this assessment, some key future research directions that ecosystem services researchers should focus their attention on.

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Origins: Introducing Ecosystem Services

The purpose of this brief chapter is to acquaint the reader with the concept of ecosystem services and to sketch out a short development trajectory, highlighting key conceptual origins and identifying core research foundations, aspects that will be taken forward and discussed more deeply in subsequent chapters.

i. Origins And Foundations

By most measures humankind is continuing to fall short in its role as planetary steward (Stafford-Smith et al., 2012). Our impact on the earth-system is now so extensive (e.g. Rockström et al., 2009) that the negative ecological and social consequences of continuing on a business-as-usual trajectory are widely reported (e.g. Bogardi et al., 2012; Kovats and Butler, 2012; Misselhorn et al., 2012). However, amid the gloom there remains a glimmer of hope. Averting the worst aspects of the ‘global environmental crisis’ many argue requires a comprehensive, inter-connected, policy-oriented and trans-disciplinary approach locating biodiversity, social sustainability, human-wellbeing, green economics and governance at its core (Kosoy et al., 2012; Rogers et al., 2012). Moreover, as the TEEB (2012) report acknowledged better information (e.g. more accurate data concerning ecosystem processes and their contribution to human-wellbeing) as well as improved institutional structures are needed to improve the management and governance of ecosystems. For many, such an approach is available and embodied in the ecosystem services paradigm (Larigauderie et al., 2012).

Ecosystem services is a child of our time: a synthesis of ecological science and economic rationalization (Figure I). It owes its birth to the field of environmental economics¹ and its neoclassical economic² heritage and, more latterly, to the continued influence of ecological economics³ (Gomez-Baggethun et al., 2010; Häyhä and Franzese, 2014). Following its inception in the 1970s ES has evolved along two fronts, namely: the assessment and valuation of so-called natural capital and its ‘stock-flow’ dynamics (Burkhard et al., 2012) and its conceptual mainstreaming into practical conservation efforts and broader science-policy circles (Kok et al., 2010).

The dawning of the 1980s witnessed the continued growth of the international environmental movement at the same time the term “ecosystem services” was coined by Ehrlich and Ehrlich (1981) outlining the potential effects of biodiversity depletion on natural service provisions and the capacity for technology to substitute for natural capital. The ecosystem services mantra entered the lingua franca of academia and was elucidated by a

number of authors throughout the 1980s. Following the new language of sustainability towards the end of the 1980s, the idea that human-nature relations existed in a dynamic social-ecological set of relationships took hold, and the research and policy agenda of ecosystem services started to become much more favoured and widespread. Subsequently, it continued along a pathway of increased mainstreaming throughout the 1990s supported by the work of Costanza and Daly (1992), Perrings et al., (1992), Schulze and Mooney (1993), Daily's *Natures Services* (1997) and Costanza et al. (1997) landmark global ecosystem service valuation paper, alongside significant policy developments such as the Convention on Biological Diversity's (CBD) ecosystem approach enacted in the Malawi Principles (1998).

However, the main ecosystem services paradigm shift did not occur until several years later. In 2005 the publication of the Millennium Ecosystem Assessment (MA) radically altered the policy landscape: demonstrating the centrality of ecosystem services to human-wellbeing and societal prosperity the MA told the story of the dramatic decline in global ecosystem services resulting from human action over the last fifty years (MA, 2005; Gomez-Baggethun et al., 2010; Braat and de Groot, 2012). This landmark collaborative study created a new discipline and put ecosystem services front-and-centre on the environmental agenda. Much of the post-MA landscape has been focused on “ironing out” the conceptual “creases” in ES, as well as mainstreaming the concept into environmental policy and decision-making processes (e.g. Larigauderie and Mooney, 2010; Perrings et al., 2010; TEEB, 2012; UK NEA 2011; 2014). In addition, recent work has also increasingly involved as well as advocated for greater trans-disciplinary research (Costanza and Kubiszewski, 2012).

In a short period of time ES has become a productive and established discipline based around four principal areas, namely: environmental service valuation; social-ecological systems; biodiversity ecosystem functioning and ecosystem services generation; and ecosystem services, human well-being and sustainable development (Daily et al., 2012). Ironically, given the holistic foundations of ES these areas are often pursued separately making it difficult to connect a central body of knowledge, research and enterprise in a manner that delivers effective environmental management and decision-making (Nahlik et al., 2012). The primary purpose of Parts 1 to 5 is to grapple with these foundations; however, before launching into these discussions it is necessary to spend a little more time digging deeper and expanding upon recent lexicon and framework developments in ecosystem services, as this provides important contextual perspectives framing the debates and discussions that are the focus of these chapters.



Figure I Historical developments in the fields of environmental and ecological economics pertaining to the evolution of ecosystem services (adapted from Liu et al., 2010; Parks and Gowdy, 2013; Häyhä and Franzese, 2014)

ii. Framings And Meanings

The main transitions and progressions in ecosystem service conceptual evolution have been witnessed in relation to framework typologies and definitions (de Groot et al., 2010; Gomez-Baggethun et al., 2010; Salles, 2011).

ii.i Frameworks

Proto-ecosystem service frameworks were framed in terms of renewable and non-renewable natural capital stocks and flows, where ecosystems provided goods through harvesting (e.g., timber) and services when left in place to yield a sustainable flow (e.g.,

nutrient cycling), with the combination of natural, human and manufactured capital determining the magnitude of economic goods and services (Figure II, Costanza and Daly, 1992). Over the course of several years such frameworks underwent a degree of formalization and standardization, in part to enable comparative ecological economic analysis of these goods and services, leading to the production of more integrated ecosystem function, service and goods frameworks (Figure III, de Groot et al., 2002). The “classical” representation of the ecosystem services framework, presenting it as a multidimensional and spatial-framework, describing what ecosystem services are and for the first time making explicit the linkages between these services and human-wellbeing was constructed by the MA (Figure IV, MA, 2005). More recent progress in framework evolution has focused on refining the MA characterisation of ES, these reforms to the MA depiction have resulted in the development of what has been termed the “service cascade” model - now regarded as more conceptually useful (Figure V, de Groot et al., 2010).

Developed by researchers in the UK, this model explicitly separates ecosystem processes, functions, services, benefits and values (Haines-Young and Potschin, 2010). By doing so it is argued that this model captures the key components of the ecosystem service paradigm, whilst also enabling the discussion of capital stock flows, supply, depletion, restoration and human-wellbeing linkages (Potschin and Haines-Young, 2011). The model has been further elaborated by Salles (2011) who presented it within a more tightly focused construction of ecosystem and biodiversity human-wellbeing linkages. Moreover, the cascade model underpinned TEEB’s conceptual approach (TEEB, 2012). Nevertheless, there remains much discussion regarding whether to: (i) differentiate processes from functions (de Groot et al., 2010; Braat and de Groot, 2012); (ii) distinguish intermediate services from final services (Fisher et al., 2009); (iii) adopt core ecosystem processes, beneficial ecosystem processes and ecosystem benefits (Balmford et al., 2011); (iv) discriminate “services” from “goods” on the basis goods require human capital transformation (Bateman et al., 2011; Mace et al., 2011) and, (v) if conceptual ruminations aid valuation assessments (Potschin and Haines Young, 2011).

ii.ii Definitions

ES definitions have developed into a veritable smorgasbord over the last two decades, and some notable examples are presented in Table I. Their primary distinguishing feature is the extent of their ecological and anthropological centrality. For example, Daily’s (1997) definition emanates from an ecological construction of ES highlighting the centrality of ecosystem processes and functioning as determinants of human-wellbeing.

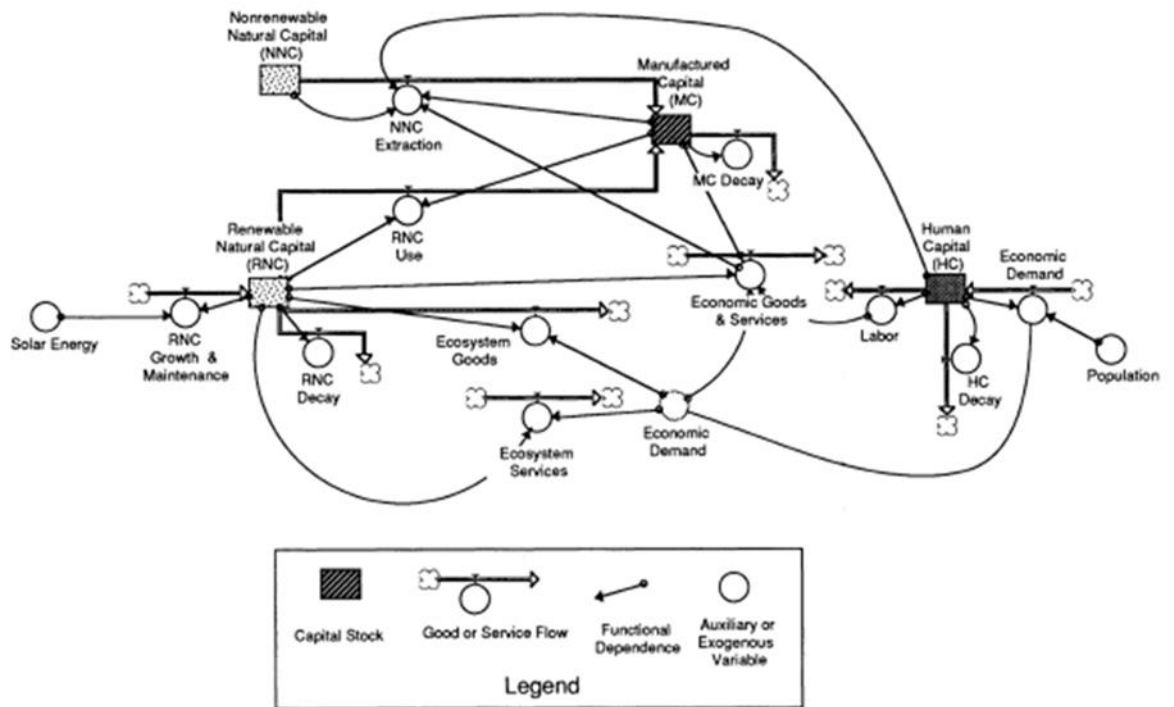


Figure II Natural capital stocks and flows, ecosystem goods and services representation (source: Costanza and Daly, 1992)

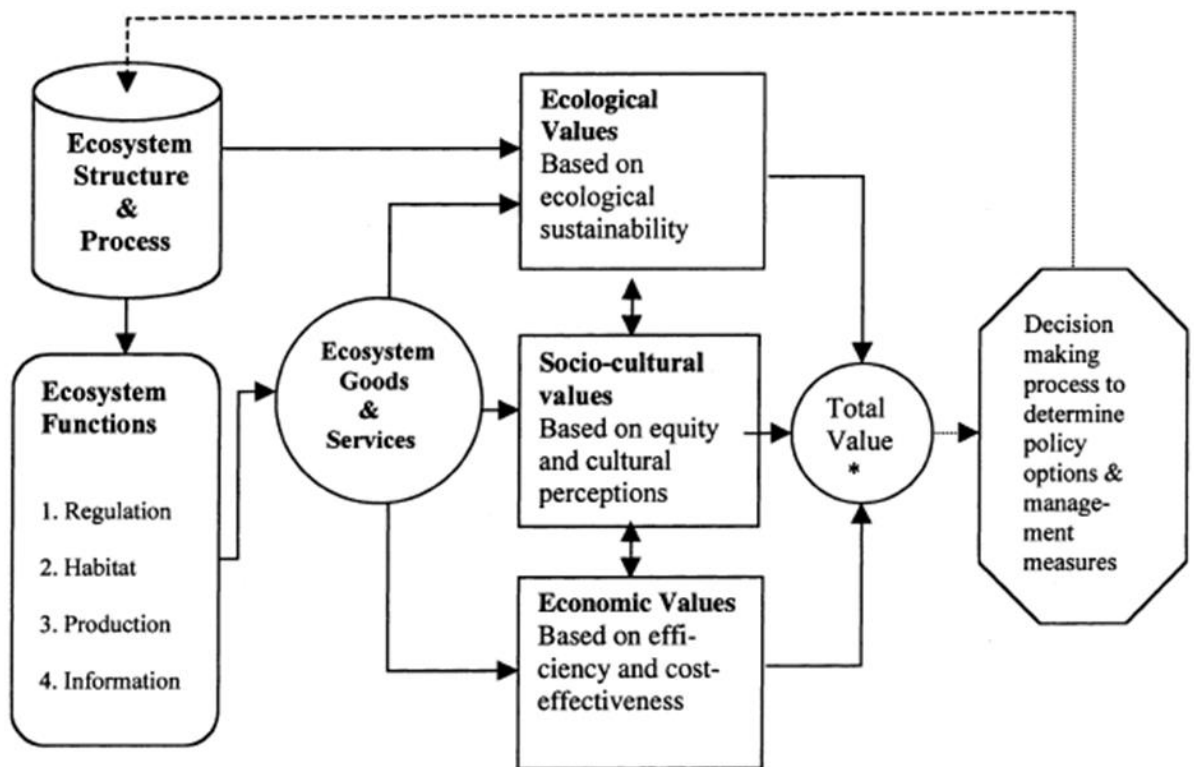


Figure III Integrated ecosystem function, goods and services framework (source: de Groot, 2002)

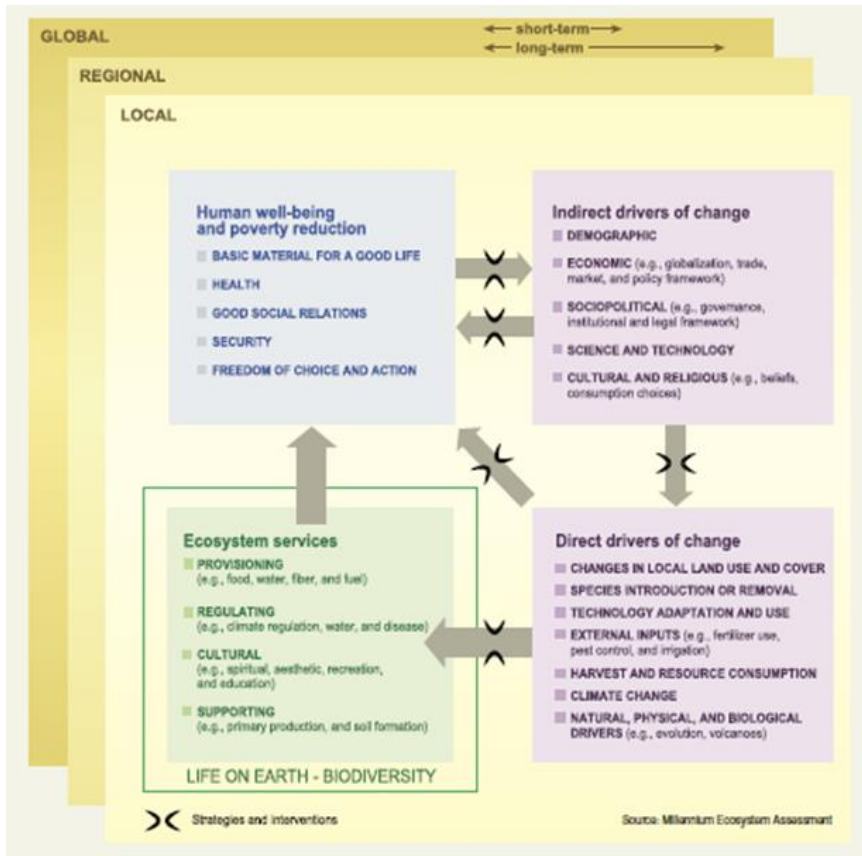


Figure IV 'Classical' representation of the ecosystem services framework (source: MA, 2005)

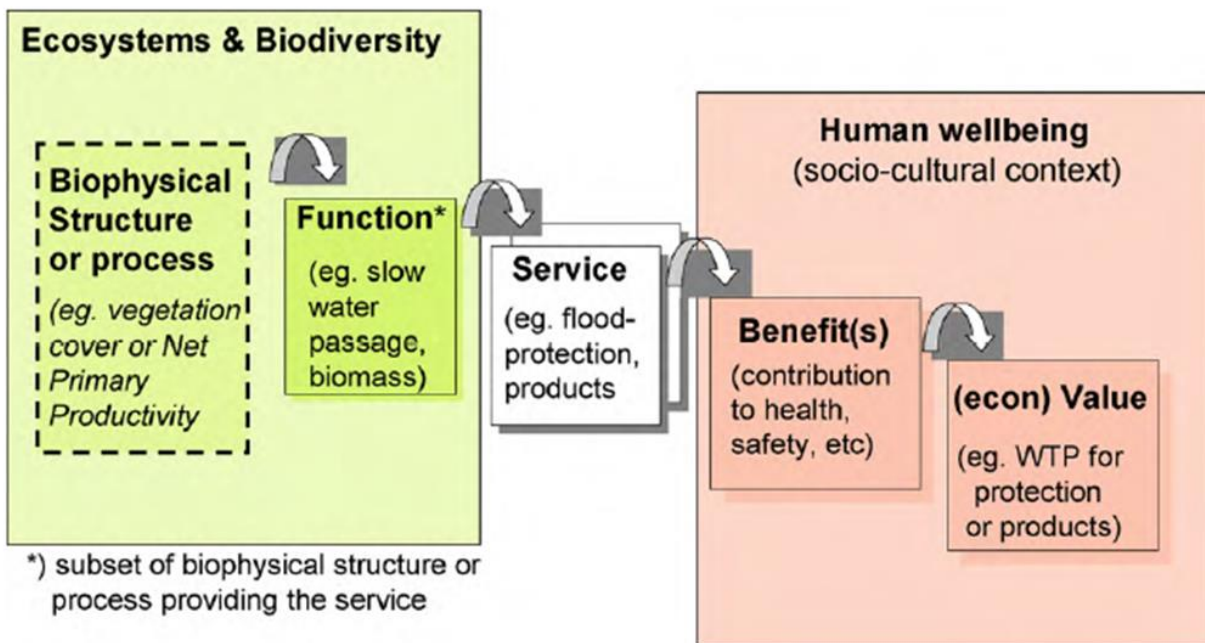


Figure V 'Cascade model' of ecosystem services (source: adaptation by de Groot et al., 2010)

Whereas for Jenkins et al., (2010) it is the consumptive attributes of ES that are of special significance with particular emphasis placed on the links between “goods” and “services” and human-wellbeing.

Table I Progressions in ES definitions

Ecosystem Service Definitions	Source
The conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life	Daily (1997)
Benefits human populations derive, directly or indirectly, from ecosystem functions	Costanza et al., (1997)
The aspects of ecosystems utilised (actively or passively) to produce human-wellbeing	Fisher et al., (2008)
A range of goods and services generated by ecosystems that are important for human-wellbeing	Nelson et al., (2009)
Benefits that humans recognise as obtained from ecosystems that support, directly or indirectly, their survival and quality of life	Harrington et al., (2010)
A collective term for the goods and services produced by ecosystems that benefits mankind	Jenkins et al., (2010)
Ecological services, landscape services, land function (terms used both synonymously with but also to differentiate from ecosystem services)	Lamarque et al., (2011)
Ecosystem services are the direct and indirect contributions of ecosystems to human-wellbeing	TEEB (2012)
Contributions of ecosystem structure and function – in combination with other inputs – to human well-being	Burkhard et al., (2012)

Adapted from Braat and de Groot (2012); Nahlik et al., (2012) and Häyhä and Franzese (2014)

Some authors have argued that the definitional lexicon of ecosystem services is too broad, in other words, rather like an over-stuffed Christmas stocking bursting at the seams far too laden with various meanings to be equally useful and helpful in all the areas it is applied, thereby meriting one definition for research and accounting and another for policy (Lamarque et al., 2011). Others, however, such as Nahlik et al., (2012) contend that ES’s conceptual foundations are too ambiguous and relies on an inappropriate classification system. In the event they propose their own classification system grounded in four underlying principles and six evaluative criteria (i.e. ecosystem service definition, transdisciplinarity, community engagement, resilience, coherence and policy-relevance).

The problem of inadequate guidelines and lack of guidance to construct effective ES typologies has also been highlighted as a stumbling block to operationalisation (Johnston and Russell, 2011). To prevent sliding into a “can’t see the wood for the trees” moment Potschin and Haines-Young (2011) make the argument for a pragmatic approach to definitional and conceptual progressions in ecosystem services focusing on the validity of analytic outputs. Ultimately, as Braat and de Groot (2012) calmly suggest, there are likely to be continued fluxes

in ecosystem service framework and definitional developments, particularly with regards to how best to differentiate the connections between goods and services, and they ought to be seen as part of the course for a growing and broadening discipline.

Notes

1. Environmental economics came of age in the 1950s and 1960s and its view of the environment is heavily influenced by its mainstream neoclassical economic foundations. Market failures (i.e. externalities) represent the core concept of the discipline, whereby the reason environmental problems occur and persist is due to the inefficient allocation of natural resources by markets (primarily because these markets do not exist). Moreover, natural resources and man-made technology are regarded as generally substitutable, which is to say, environmental economists advocate a weak sustainability position. Cost benefit analysis and monetary valuations are the principle tools of environmental economics (Hanley et al., 2013).

2. Neoclassical economics was ushered in towards the latter part of the Nineteenth Century, based on the work of economic luminaries such as William Stanley Jevons (*A General Mathematical Theory of Political Economy*, 1862), Carl Menger (*Principles of Economics*, 1871) and Léon Walras (*Elements of Pure Economics*, 1874-1877), in what has subsequently become known as the ‘Marginalist Revolution’ (Sandmo, 2011). This period was marked by an increasing professionalization of economics and mathematical approaches to economic problems, with a second wave of ‘marginalism’ occurring in the early Twentieth Century through the highly influential work of Alfred Marshall (*Principles of Economics*, 1890), Pigou (*The Economics of Welfare*, 1920) and Pareto (*A Course on Political Economy*, 1897; and *Manual of Political Economy*, 1909) (for a more in depth picture regarding the history of economic and political economy thought see Agnar Sandmo’s *Economics Evolving: A History of Economic Thought*, 2011; and Roger Backhouse’s *The Penguin History of Economics*, 2002). Neoclassical economics is mainly concerned with the determination of prices, income distribution and outputs in a supply and demand framing facilitated by utility maximising and income-constrained individuals that operate rationally (i.e. according to rational choice theory). At its core neoclassical economics makes three assumptions: (i) people display rational preferences for outcomes with attached values; (ii) firms seek to maximise profits and individuals seek to maximise their own utility and, (iii) people make decisions independently and with full information. This framework underpins the concept of ‘goods’ and ‘services’ as applied to the ecosystem services paradigm and the widespread application of environmental valuation (Hanley et al., 2013).

3. Formed as a break-away discipline from orthodox environmental economics towards the late 1980s, ecological economics is a transdisciplinary field which, in stark contrast to environmental economics, regards the economic system as a sub-component of the environment or ecosystem, and so emphasises a strong sustainability position where manufactured capital and natural capital (i.e. natural resource endowments) are only weakly substitutable. Moreover, ecological economics is far more concerned with issues of environmental justice, equity and sustainable development in the allocation of natural resources with a stronger focus on the use of social valuation methods (Daly and Farley, 2010). Some have recently argued however (e.g. Spash, 2013; Plumecocq, 2014) that the ‘radical’ character of ecological economics has diminished or at least softened over the years as it has become progressively more aligned with mainstream environmental economics.

Part 1: The State Of The Planetary Garden

This section illustrates the current environmental state of the planet we have inherited and the post-Edenic world we have shaped. Following, we explore the relationships between biodiversity in the Garden and the generation and provision of ecosystem services, not only because biodiversity represents a crucial element in the narratives of conservation, sustainability and ecosystem services; but also, because the linkages between biodiversity and ecosystem services represent fundamental sets of connections underpinning the continued supply of ecosystem services that support human-wellbeing.

Chapter 1: The (UN)Natural Garden We Find Ourselves In

“So that finding ourselves in the midst of the greatest wilderness of waters in the world, without victual, we gave ourselves for lost men, and prepared for death. Yet we did lift up our hearts and voices to God above, who ‘showeth his wonders in the deep’, beseeching him of his mercy, that as in the beginning he discovered the face of the deep, and brought forth dry land, so he would now discover land to us, that we might not perish.” (Francis Bacon, *New Atlantis*, 1627)

“Humans are the only creatures to have cumulative culture, allowing us to build on the past rather than continually reinvent the wheel. But, as we fumble about on the Earth’s surface, hostage to the whims of our phenomenally powerful brains, humanity is undertaking a brave experiment in remodelling the physical and biological world. We have the power to dramatically shift the fortunes of every species, including our own. Great changes are already being wrought. The same ingenuity that allows us to live longer and more comfortably than ever before is transforming the Earth beyond anything our species has experienced before. It’s a thrilling but uncertain time to be alive. Welcome to the Anthropocene: the Age of Man.” (Gaia Vince *Adventures in the Anthropocene*, 2014, pg.4)

Our purpose here is to explore and map out, albeit rather briefly, the current state of the “Natural Garden” we have inherited – the garden we have shaped. Stretching across oceans and forests and delving into the world of biodiversity we illustrate how we humans continue to transform and fashion the world around us and, in particular, detail the influence this has wrought on the biosphere.

1.1 Biodiversity

Accumulating evidence continues to chart the global decline in biodiversity¹ across terrestrial, freshwater and marine systems, from Northern to Southern latitudes (UNEP, 2012). In fact, to put it more starkly, according to Sachs (2015:453) biodiversity is:

“...already being reduced, degraded, and hugely threatened across the planet.”

This sentiment is further echoed in the United Nation Environment Programme’s (UNEP) most recent Global Environment Outlook Report: *Environment for the Future We Want* (UNEP, 2012:156), which argues strongly, building on earlier GEO reports, that:

“The state of global biodiversity is continuing to decline, with substantial and ongoing losses of populations, species and habitats.”

For example, the report indicates that since the 1970s there has been a reduction of 30% in vertebrate species populations and a further 66% of various taxa teetering on the brink of extinction, with the most severe declines witnessed in tropics and freshwater and marine habitats (UNEP, 2012)². Declines such as these are indicative of the broad negative trends witnessed in a raft of biodiversity trend indicators (Figure 2.1). These trends are the outward

expression of a set of interlinked human global drivers of biodiversity change such as habitat loss, agriculture, overexploitation, invasive species, climate change and pollution (UNEP, 2012). Particularly worrying is the extent of reach of these drivers of biodiversity loss, for example, the Human Footprint Index (pioneered by Columbia University's Earth Institute) has demonstrated with remarkable clarity that human activity and impacts on the Earth system are all pervasive and growing, as Sachs (2015:454) explains:

“The human impact is significant in all parts of the world except the most extreme environments, notably the desert regions, some parts of the equatorial rain forests (though these are also under threat), and the poleward (high-latitude) regions that are currently too cold for agriculture. All of the rest of the planet exhibits a heavy human footprint.”

Human appropriation of natural resources is so absolute that we are taking up to almost half of the photosynthesis on the Earth, we are “literally eating other species off the planet” (Sachs, 2015). As indicated by the International Union for Conservation of Nature (IUCN) Red List³ of the world's endangered species, there has been a consistent growth across groups, from mammals to plants, in the number considered vulnerable, endangered and critically endangered (IUCN Red List, 2016):

“Currently there are more than 79,800 species on The IUCN Red List, and more than 23,000 are threatened with extinction, including 41% of amphibians, 34% of conifers, 33% of reef building corals, 25% of mammals and 13% of birds.”

Take amphibians for example, a recent review remarked that present amphibian extinction rates, calculated for the period 1984-2004, were 173 to 2,351 times higher than the historical average of 2,162 extinctions per million years for 7,388 known species (Catenazzi, 2015). Whilst the reasons for accelerated amphibian extinction rates are complex observations indicate that a significant number of global extinctions are the result of the deadly fungal disease chytridiomycosis (Catenazzi, 2015). Underscoring the plight of amphibian species Catenazzi (2015:110) concludes that:

“The state of the world's amphibians is dire and the prospects are not improving.”

The outlook for global bird species, though perhaps not quite as dismal as that for amphibians, nevertheless presents cause for concern. As a recent BirdLife International report chronicles, European farmland birds have consistently declined over the past thirty years (40% of a 148 species), whilst water bird species, particularly in Asia (over 50%), have also demonstrated significant reductions as have long distance migrants between Europe and Africa (23% reduction between 1980 and 2010). Overall, species in the Pacific and ocean-going birds more generally have witnessed the most rapid declines (BirdLife International, 2013).

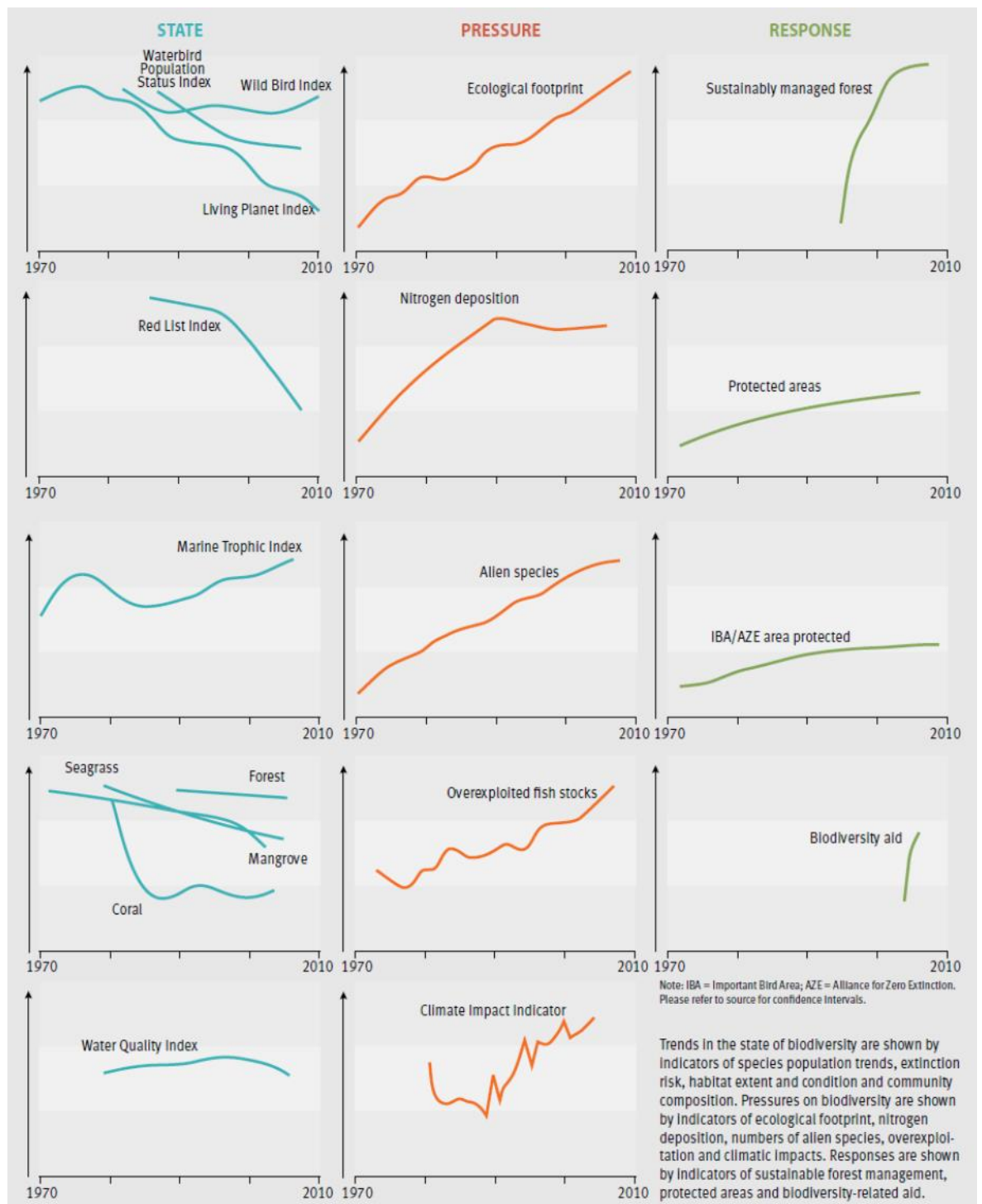


Figure 1.1 Trends in biodiversity indicators and global drivers of biodiversity change (source: UNEP, 2012)

In the case of mammalian species, between 1996 and 2008, the Red List Index noted a change of -0.8 percent: a figure equivalent to a worsening of category status for 156 species over that period, with deterioration in mammal extinction risks particularly prominent in Indo-Malayan and Australian regions (Hoffman et al., 2011).

These frightening statistics support the palpable note of alarm sounded by Sachs (2015:457):

“We are destroying the habitats of species and appropriating their water and food supplies faster than we can even identify and name the species that are threatened by our actions [...] the numbers are frightening, because even in the very short period of time covered by the Red List, the numbers of critically endangered species have soared”

Exploitation of species underpins the demand for wildlife products, which has had a detrimental impact on terrestrial vertebrates and bird species, consumption demands have also fuelled the overexploitation of capture fisheries leading to the global plummeting of marine fish stocks (UNEP, 2012)⁴. In a similar manner habitat loss and degradation has also resulted in large scale decimation of many globally significant ecosystems, biodiversity and natural resources, as the UNEP GEO 5 Report (2012:134) summaries:

“The world lost over 100 million hectares of forest from 2000 to 2005, and has lost 20 per cent of its seagrass and mangrove habitats since 1970 and 1980 respectively. In some regions, 95 per cent of wetlands have been lost. The condition of coral reefs globally has declined by 38 per cent since 1980. Two-thirds of the world’s largest rivers are now moderately to severely fragmented by dams and reservoirs.”

1.2 Forests

Forests are home to most of the terrestrial biodiversity on Earth, and in their Northward boreal range they can be found knocking on the door of the Arctic wilderness, whilst in the South forested landscapes of the Valdivian temperate rainforest breathe life into the challenging environment of Southern Chile and Argentina. The greatest concentrations of biodiverse forest and therefore life still remain in the colossal tropical expanses of the Amazon, the Congo Basin and Indonesia, whilst the largest tracts of forest occur in the vast coniferous expanses of Canada and Russia, as Vince (2014:264) remarks with regards to the relationship between biodiversity and forests:

“Around 70% of all land creatures and plants live in forests. Tropical rainforests, especially those in the Amazon, Congo, Borneo, Sumatra, Madagascar and New Guinea, support unparalleled biodiversity including many species as yet unknown to science.”

The UNEP GEO 5 Report (2012:72) also presents a similar analysis, remarking that:

“Forests cover just over 4 billion hectares, 31 per cent of the world’s total land area (FAO 2011). The majority of these are boreal forests extending across northern and central Russia and much of Canada and Alaska. Large expanses of tropical forest are found in the Amazon, Africa’s Congo Basin and parts of South East Asia. Temperate forests remain in a patchy distribution across the United States, Europe and the Asian mid-latitudes.”

Forests represent the life blood of terrestrial ecosystems: their resplendent verdant hues soak up the sun and enrich their intricate and majestic architecture casting emerald light on the humble homes they have forged over centuries. Forests are both dwellers and creators; they bind the earth, they are the great metamorphics – transforming solar energy into food and breathable oxygen whilst absorbing carbon dioxide from the air. They create a myriad of niches for other creatures to exist and they even affect local and regional climate and weather patterns – generating their own rain they also feed streams and rivers. It is quite apparent then that forests perform many globally significant functions from water and climate regulation through to soil formation⁵ (Muirra et al., 2015:35):

“The world’s forests provide fundamental protection of soil and water resources and provide multiple ecosystem services...”

The recognition of these multiple functions partly explains why approximately 25.1% of total global forest area (1.002 billion ha) is designated as protected for the purposes of soil and water resource preservation, up 0.181 billion ha on 1990 figures (Miura et al., 2015)⁶. Indeed, overall global protection of forests for reasons of ecological importance has risen over the last 25 years from just 7.7% of forests in 1990 to 16.3% of forests in 2015, with the largest increases occurring in tropical regions (Morales-Hidalgo et al., 2015).

As Vince (2014:264-265) once again elegantly expresses:

“Forests play an important role on the local and global climate. The world’s forests absorb 8.8 billion tonnes of carbon dioxide each year through photosynthesis – about one-third of humanity’s greenhouse gas emissions. And, like all living organisms, trees respire, taking oxygen and emitting water vapour from their leaves. At the same time, their canopies provide shelter from the sun and wind, making forests much wetter, cooler environments than surrounding treeless areas. This nurtures streams and rivers, provides habitat for a range of amphibians and other life, helps cools the regional and global atmosphere, and recycles water.”

This is a view further shared by Sachs (2015:471) who cites the example of the importance of forest in maintaining a globally consistent climate:

“...rain forests serve to keep the planet cool by maintaining extensive cloud cover that reflects incoming ultraviolet radiation back into space rather than allowing the ultraviolet radiation to reach and warm the Earth. If the Amazon dries out (due to human-induced climate change) or disappears as the result of the forest being cleared to make way for farmland, the Amazon’s cloud cover would shrink as well, thereby changing the Earth’s reflectance (albedo) and causing a potentially large positive feedback to warm the planet further.”

Forests are of course not just home to a rich and bountiful flora and fauna they are also home to us, to humankind: for many millions of people forests are a home, an essential component of their livelihoods, a source of food, water and income as well as shelter and a quintessential part of their identity and culture (Vince, 2014; Köhl et al., 2015)⁷. For example,

over 10 million people are directly employed in the forestry sector, with most of these individuals located in Asia (Whiteman et al., 2015):

“In 2010, employment in the forestry sub-sector reached 12.7 million employees or about 0.4% of the global workforce. The countries with the highest numbers of employees were India, with 6 million, Bangladesh with 1.5 million and China with 1.1 million. These three countries accounted for 70% of global forestry employment in 2010.” (pg. 104)

Despite the plentiful goods and services forests provide and the essential functions they perform, to say nothing of their criticality in supporting human livelihoods and economic growth, swathes of the world’s forest continue to undergo significant levels of degradation and deforestation: this is justifiably as cause for concern but at the same time is not necessarily a surprising find given that it reflects the broader historical trend of continued human utilisation and occupation of forested environments (Sachs, 2015):

“Humanity has been in the business of clearing forests for thousands of years.” (pg. 469)

A metaphor Vince (2014:265) extends, noting that:

“Over recent centuries, we have burned and chopped our way through forests, particularly in Europe, the Middle East and China, with periods of intensive deforestation [...] The most dramatic and global deforestation, however, has occurred since the 1850s.”

Vince goes on to say that:

“About half of the forests that once covered the Earth have already gone because of humans and each year, another 16 million hectares disappear.” (pg. 267)

Deforestation occurs for a number of complex and interwoven reasons represented by a multiplicity of underlying and direct drivers (d’Annunzio et al., 2015). For example, broad contextual factors such as economic development and population growth as well as more specific drivers like agricultural expansion and urban infrastructure developments (d’Annunzio et al., 2015). In many cases it is a combination of internal socio-economic factors twinned with domestic and international markets for forest goods that drives a significant proportion of deforestation, as Sachs (2015:472) illustrates:

“Some of the human impact of deforestation is internally driven, mainly by growing populations within countries. Yet a huge amount is also coming from international trade, from the demands halfway around the world for forest products. This is very difficult to control, because it means the high levels of demand, often from rich countries or rapidly growing economies like China, overwhelm local protective services, often through illegal means.”

This sentiment is further echoed by Sloan and Sayer (2015:141) who note that:

“Increasing demand for forest and agricultural products threatens to undermine efforts to arrest biodiversity decline and maintain the integrity of the forest estate.”

It remains the case that the vast majority of deforestation occurs in the tropics as a result of agricultural activities; approximately 70-95% of tropical forest loss is a consequence of agricultural conversion for both arable and pastoral reasons, particularly in poorer regions (Vince, 2014; d’Annunzio et al., 2015; Sloan and Sayer, 2015). A significant fraction of this loss is associated with the accompanying road and transportation infrastructure⁸, as Vince (2014:282) explains with particular reference to the situation in South America:

“Road creation is the primary driver of deforestation. Whether it is to provide access for mines and dams or to link towns and villages, a road enables loggers, animal poachers and traffickers, and small-time miners to enter virgin territory. Following in their wake are the farmers, who deforest for cropland, and drug growers and processors. Scientists have shown that 95% of all deforestation takes place within twenty five kilometres of a road and, on average, every road carved into the Amazon – and there have been 50,000 kilometres built in the past three years – is followed by a fifty-metre-wide halo of deforestation.”

Forest losses also occur as a result of a combination of other influences such as fire, disease, pests and climatological factors (Lierop et al., 2015):

“Between 2003 and 2012, approximately 67 million hectares (1.7%) of forest land burned annually, mostly in tropical South America and Africa. In a similar reporting period, in total 142 million hectares of forest land were affected by other disturbances than fire. Insect pests affected more than 85 million hectares of forest, of which a major part was in temperate North America. Severe weather disturbed over 38 million hectares, mostly in Asia. About 12.5 million hectares were reported to be disturbed by diseases, mostly in Asia and Europe.” (pg. 78)

For all these reasons, and probably others too, the last 25 years has continued to observe a global net decline in forests, with a 3% reduction in forested area of approx. 4,128 million ha down to 3,999 million ha occurring between 1990 and 2015, and natural forests in particular have seen the reaches of their greenery reduce from 3,961 million ha to 3,721 million ha over this same period (Keenam et al., 2015). Notably, primary forests have declined by 2.5% across the world – a figure that reaches 10% when only tropical forested regions are considered (Morales-Hidalgo et al., 2015). Due to the rapid growth in afforestation, however, the annual net rate of forest lost actually halved between 1990 and 2015, from 7.3 million ha per year to 3.3 million ha per year (Keenam et al., 2015). In terms of the overall balance of forested area, which is to say nothing about the quality of ecosystem services and biodiversity, the growth in afforested regions has offset the decline in natural forest (Payn et al., 2015).

Plantations have grown rapidly, with planted forest area expanding from 168 million ha in 1990 to 278 million ha in 2015 (Keenam et al., 2015; Payn et al., 2015). Initial expansion occurred at a rate of 2.5% per annum, however, after 2005 rates fell to approximately 1.5%

per annum and then to 1.2% between 2010 and 2015 (Payn et al., 2015; Sloan and Sayer, 2015). The majority of plantation species are natives, however, just under 20% are considered to be introduced species, and most of these are located in the Southern Hemisphere, in particular, parts of South America, Southern Africa and Oceania (Payn et al., 2015). The point has been made that in many cases these afforested regions are increasingly supplying some of the services that natural forests have traditionally supplied (Sloan and Sayer, 2015). Yet, at the same time, their capacity to do so long-term is questionable given that plantation forests are far more susceptible to a range of disturbances such as disease, pests and climatological factors compared to natural primary forests (Payn et al., 2015). In addition to purely plantation-based forests, mosaic agricultural forests are also increasing, and in recent years are beginning to constitute an important, albeit still modest, fraction of the global “forest estate” (Sloan and Sayer, 2015:140):

“In North America and Europe (including Russia), where many forests are under explicit ‘nature-oriented’ production regimes multiple-use forests account for 58% and 24% of the total forest estate, respectively, whereas in South America they account for only 13% despite having expanded 5.5-fold there since 1990.”

Changes in global forest composition and area are not homogenous; instead they display quite stark geographical differences as Keenam et al., (2015:9)⁹ remark:

“From 2010 to 2015, tropical forest area declined at a rate of 5.5 M ha y^{-1} – only 58% of the rate in the 1990s – while temperate forest area expanded at a rate of 2.2 M ha y^{-1} . Boreal and sub-tropical forest areas showed little net change. Forest area expanded in Europe, North America, the Caribbean, East Asia, and Western-Central Asia, but declined in Central America, South America, South and Southeast Asia and all three regions in Africa. Analysis indicates that, between 1990 and 2015, 13 tropical countries may have either passed through their forest transitions from net forest loss to net forest expansion, or continued along the path of forest expansion that follows these transitions.”

The changes that have occurred in natural and plantation forests over the past quarter century have also had a direct impact on forest carbon stocks (Federici et al., 2015). In general, forests are considered to be important sinks for organic carbon and therefore to positively contribute to lowering atmospheric carbon dioxide (UNEP, 2012). The picture overall is one of a reduction in organic carbon, predominantly in living biomass, of approximately 13.6 Pg C (Köhl et al., 2015). Nevertheless, this global scale picture masks important regional scale differences, as Köhl et al., (2015:24) explain:

“North and Central America, Europe and Asia report total forest C stock increases while South America and Africa report strong decreases and Oceania reports stable forest C stocks. The rate of forest C stock decline decreased from 0.84 Pg C yr^{-1} during 1990–2000 to 0.34 Pg C yr^{-1} during 2010–2015. Forest C stock densities vary sub-regionally, with the highest reported densities in Western and Central Africa, and the lowest densities in Western and Central Asia.”

On the other hand, the total amount of growing stock has increased since 1990 by 0.7% to reach 530.5 billion m³ in 2015, although between 1990 and 2000 this growing stock was slightly decreasing it has subsequently been on the increase – yet again however there are considerable geographical differences (Köhl et al., 2015). In addition, the demand for industrial wood and fuelwood increased by almost 35% between 1990 and 2015, mainly in poorer regions, for example, as Sloan and Sayer (2015:141) state:

“Combined industrial and fuelwood removals in the tropics increased by 35% (3,928,650 m³) over 1990–2015 or 1.4% per annum whilst either holding constant or declining slightly in the other climatic domains, so that tropical domain is currently the greatest source of removals globally. Similarly, industrial and fuelwood removals over 1990–2015 increased most rapidly in lower-middle and lower income countries.”

Timber has multiple uses including for fuel, pulp, paper and other wood-based material all of which have seen increasing demand, driven predominantly by urbanization and economic growth over recent years. For example, between 2002 and 2008 fuelwood consumption rose by 4%, wood-based materials by 34% and paper and paper board by 20% (UNEP, 2012).

1.3 Land And Agriculture

We have moulded and shaped the surface of our planet into a food producing engine; virtually 40% of the world’s land area (50 million square kilometres) is devoted to agricultural production activities (Sachs, 2015). The vast bulk of this comprises pasture and meadowland, approximately 34 million square kilometres, while the rest roughly 14 million square kilometres is arable land (Sachs, 2015). Our agricultural production activities have transformed the biosphere, creating a patchwork mosaic landscape that has seriously impacted many globally important ecosystems, as Vince (2014:109) explains:

“Agriculture has already cleared or cultivated 70% of grassland, 50% of savannah, 45% of temperate deciduous forest and 27% of the tropical forest biome. 80% of new tropical croplands are replacing forests. And 70% of our fresh water is used for agriculture.”

At the global scale, the period from the 1960s through to the mid-1990s saw the amount of land devoted to agriculture steadily increase, peaking around 1998, after which there has been a steady but slight decline of approximately 0.80% between 1998-2009 - this despite the fact there has been a 4.4% increase in per capita calorie production – indicating that agricultural yields have also grown (FAO, 2011). This is significant as the demand for agricultural food and livestock feed have risen substantially over recent years due to a combination of global economic development, population growth and urbanization: crucially

demand is not uniform and similarly neither is the distribution and composition of agricultural land (UNEP, 2012).

There are substantial differences in the ratio of cropland to pasture across the globe, reflecting, in part, past historical trends as well as environmental and biophysical constraints and market forces. For example, in Europe, 8.5% of the total land area is designated for pasture whilst 13.9% is earmarked for crops, on the other hand, in Asia and the Pacific region 36.0% of land is devoted to pasture with just 15.7% under cultivation for crops (GEO 5, 2014; Sachs, 2015). Taking, as an example, the geographical differences in the main global crops the UNEP's GEO 5 report (2012:69) states:

“Maize is an important crop in all regions other than West Asia, with the area harvested increasing by more than 25 per cent across Africa and Asia and the Pacific between 2001 and 2010. In total, approximately 160 million hectares of maize were harvested in 2010. Asia and the Pacific have the largest area of rice, but Europe and Africa experienced the greatest percentage increases between 2001 and 2010 – about 30 and 20 per cent respectively. The dominant soybean-producing regions are Latin America and the Caribbean and North America, with the United States, Brazil and Argentina the three largest producers. Asia and the Pacific and Europe are the primary producers of wheat.”

Farmland represents about 10% of the world's land area, but farming itself is exceptionally diverse, both in terms of what farmers grow and how they grow it as well as the different environmental challenges they face (Sachs, 2015). The determination of crop types or what livestock can be feasibly raised is often dictated by various ecological, climatological and topographical conditions, which can result in quite stark differences in farming systems – even across small areas (Sachs, 2015):

“A country like Ghana has tree crops in the humid south near the Gulf of Guinea and maize and dryland crops in the north. Mali has irrigated rice in the south and pastoralism in the north. Kenya is a mosaic of farm systems, as is Ethiopia, with deserts, pastoralism, lowland and highland crops (crops are graded by elevation as well as by latitude).” (pg. 332-333)

The biophysical constraints on agricultural production can lead to widely different yields across regions, for example, although agricultural expansion has increased in South America and Africa over recent years and decades yields are still lower than when compared to North America and Europe, and efforts to increase yields through new technologies, mechanization and use of chemical fertilizers can have detrimental effects on soil structure and fertility, increase erosion¹⁰, as well as have other additional and significant ecological and climatological impacts:

“Extending conventional agriculture into uncultivated lands requires mechanization to modify the surface, and supplements in the form of fertilizers, herbicides, pesticides and irrigation water. Excessive use of machinery and chemical supplements, however, breaks up soil structure, increases erosion,

chemically pollutes soil, contaminates groundwater and surface water, changes greenhouse gas fluxes, destroys habitat and builds genetic resistance to chemical supplements.” (UNEP, 2012:69)

The conversion of “natural” land (in the majority case forest but also grasslands, savannah and drylands) to other uses, primarily for food (e.g. crops and meat)¹¹ or fuel (e.g. bio-fuel)¹², has been responsible for the large scale deforestation patterns we have seen rapidly expand across the tropics since the Great Acceleration began in the 1950s, as Sachs (2015:330-331) quite pointedly notes:

“Many of the forests today are being threatened with deforestation, especially equatorial rain forests. Populations are encroaching on these areas for a variety of reasons, including to make way for pastureland and cropland or to get fuel wood and other goods and services. The pace of deforestation is currently unsustainable in all of the great equatorial rain forests. Some of the forests are overlogged for tropical hardwoods, which are highly valued but used in an unsustainable manner around the world. Rain forests are also being cut down and replaced by massive tree plantations; for example, to grow high-demand products like palm oil, a problem that is particularly intense in Indonesia, Malaysia, and Papua New Guinea.”

Inland and coastal wetlands, particularly in low and middle income countries (e.g. mangroves and seagrasses), have also continued to undergo widespread exploitation and conversion primarily driven by rising population pressures, economic development and growth and an expansion in urban infrastructure. Wetlands are also negatively impacted by broader deforestation activities, landscape fragmentation, freshwater withdrawal, fertiliser runoff and overexploitation (UNEP, 2012). Peatlands are suffering a similar fate to that of wetlands, especially through agricultural conversion and drainage: of the 400 million ha of peatland 50 million ha are currently undergoing drainage and degradation – whilst this loss of habitat is in itself concerning such widespread peatland destruction also has significant implications for climate change: degrading 50 million ha of peatland is equivalent to 6% of global carbon dioxide emissions (UNEP, 2012).

1.4 Ocean And Riverine Systems

Earth is a blue planet, the only one in our solar system to contain such obvious signs of liquid water. Without liquid water life, as we know it, would not be possible. Most of the world’s water can be found in the oceans, which cover 70% of our planet; making them home to the greatest show of life anywhere on Earth. Due to their vastness they play crucial roles in global weather patterns, climatic processes, biogeochemical cycling and they also massively affect terrestrial ecosystems through which they are inextricably connected. If oceans represent the broad circulatory system of the planet then rivers are the fine network of capillaries innervating the Earth System: they are the world’s irrigators. Containing barely a fraction of the planet’s water (approx. 0.0001%) they are nevertheless responsible for draining

three quarters of the Earth's land surface, and as such play a fundamental role in the hydrological cycle and in the transportation of sediment and distributions of nutrients; in addition, they connect different terrestrial biomes (from mountain tops to tropical lowlands) and act as conduits between terrestrial and marine systems. Oceans and rivers are also fundamental to our own human history, they are indelibly written into the fabric of our cultural and religious identities, they are routes for trade and transport¹³, and they have borne witness too many conflicts and battles. More importantly, they are reservoirs of water for drinking, sources of food and energy, waste disposal areas, places of settlement and essential to billions of livelihoods: most of the world's cities and urban areas were born on the banks of rivers (Vince, 2014). However, over the last several decades we have begun to seriously transform, affect and undermine both ocean and riverine systems, as Vince (2014) explains with regards to river systems:

“In the Anthropocene, humanity is draining the world's rivers and other sources of freshwater [...] Greater water extractions by humans for agriculture, industry and energy mean that many rivers have dried up, while others are now too polluted to use [...] Humans now control more than two thirds of the world's freshwater. We've captured so much water that we've redistributed its weight around the world and the globe now spins a fraction slower [...] In the past century, we have drained half of the world's wetlands, built 48,000 large dams and diverted most of the world's large rivers – only 12% still run freely now from source to sea.” (pg. 72)

She then goes on to spell out a similarly momentous myriad of impacts we have had on ocean systems:

“In the Anthropocene, humanity is transforming oceans. Where the oceans separated landmasses, we have joined them with bridges and tunnels. In the process, endemic biodiversity has been diluted and dispersed by us as we introduced species from one island or continent to another [...] The oceans have become our sewer for chemical, biological and material waste. Our pollution has made them more acidic and altered the nutrient concentrations, especially near shorelines. Humanity has heated, swelled and overhunted the oceans.” (pg. 151-152)

Clearly we are having a tremendous effect on ocean and river systems in many different but equally disturbing ways. Changes to ocean systems from anthropogenic-induced climate change are having a number of noticeable impacts. For example, sea levels are rising – recent estimates suggest a rate approaching 0.5 cm a year¹⁴ – this has serious consequences for coral systems which are unable to adapt quickly enough to this rate of change. Sea level rise also imperils many inhabited island systems and low-lying inhabited continental areas. It is estimated that for every 1 degree rise in temperature sea level will rise by 2.3 metres, a situation catastrophic for those living in places like the Maldives which will be washed away (Vince, 2014). The majority of sea level rise (some also results from glacier melting¹⁵) is a

consequence of the thermal expansion of the oceans – as the greenhouse effect continues to accelerate so the oceans absorb more energy:

“The ocean’s large mass and high heat capacity allows it to store huge amounts of energy: more than 1000 times that found in the atmosphere for an equivalent increase in temperature. The earth is absorbing more heat than it is emitting back into space, and nearly all this excess heat is entering the ocean and being stored there.” (WOA, 2016: Chapter 5, pg. 6)

To a large extent the ‘greenhouse’ component of climate change is driven by the continual rise of atmospheric carbon dioxide, the result of historical and current domestic, industrial and agricultural metabolism of fossil fuels. This has a double-whammy effect on coral systems, for example, because not only does increased accumulation of atmospheric carbon dioxide help drive the thermal expansion of the oceans but by virtue of the fact that the oceans are the largest absorber of atmospheric carbon dioxide (absorbing about 25% of carbon dioxide emissions) it also makes them more acidic – combined: thermal expansion and acidification have resulted in major mass coral bleaching events (this is where corals expel their commensal algae inhabitants and as a result become white and undergo mass die-offs):

“The problem of coral reef destruction in the Anthropocene is acute and global. The 1998 mass bleaching event destroyed 16% of reefs world-wide. Since then, there have been at least six major bleaching events, and scientists estimate that at least 20% of Australia’s Great Barrier Reef – the world’s largest – has been destroyed, and up to 90% of coral has been lost in the Indian Ocean and Caribbean.” (Vince, pg. 169)

Ocean acidity has risen by 15% since 1990 alone (Vince, 2014; WOA, 2016). As we have just noted, however, rising ocean acidity spells clear and present danger for creatures whose very existence depends on creating calcium carbonate shells or skeletons, in particular, due to the influence lowering pH has on aragonite abundance and the bio-availability of other key minerals and nutrients, as the recent World Ocean Assessment states:

“These chemical changes in turn affect marine plankton via several mechanisms including the following: (1) decreases in the degree of aragonite saturation makes it harder for calcifying organisms (e.g., coccolithophores, foraminifera, and pteropods) to precipitate their mineral structures; (2) decreases in pH alters the bioavailability of essential algal nutrients such as iron and zinc; and (3) increases in CO₂ decrease the energy requirements for photosynthetic organisms to synthesize biomass. Such biological effects are likely to perturb marine biogeochemical cycles including carbon export to the deep sea via the biological pump which may have a positive feedback on the build-up of CO₂ in the upper ocean and atmosphere.” (WOA, 2016: Chapter 6, pg.24-25)

Acidification can also have more low level but equally insidious effects, such as affecting fish navigation and behaviour alongside increasing the loudness of the marine environment, as Vince (2014:168) spells out:

“The bones of fish and whales are also made of calcium carbonate, and researchers have found that acidification is altering bone formation, particularly in the delicate ear bones, which fish use to navigate and sense speed and direction. Fish are not just disoriented by acidification: biologists have discovered that their neurotransmitters are also disrupted, changing their behaviour. Acidification also changes the way sound waves travel through the water. A 0.3 decrease in pH allows sound to travel 70% further in water – the ocean is becoming louder in the Anthropocene, perhaps confusing animals that communicate through sound, like cetaceans.”

Just as sea creatures are being severely affected by human activity so too are marine and aquatic plants known as macrophytes. Macrophytes are significant primary producers of ocean biomass: they are both a habitat for marine life as well as a food source and significant ecosystems for climate regulation but increasingly they are under considerable threat, a threat which has the potential to destabilise the functioning of many coastal ecosystems:

“Macrophyte habitats are being lost and modified (e.g., fragmented) at alarming rates [...] i.e., 2 per cent for macrophytes as a group, with total areal losses to date of 29 per cent for seagrasses, 50 per cent for salt marshes and 35 per cent for mangrove forests [...] As a whole, the world is losing its macrophyte ecosystems in coastal waters four times faster than its rain forests and the rate of loss is accelerating.” (WOA, 2016: Chapter 6, pg. 7)

We continue to pollute rivers and oceans with a range of chemical and waste products, mainly in two ways: First via agriculture. Agricultural fertiliser run-off and animal waste into our oceans via river systems has caused widespread eutrophication and undermined the delicate balance of nitrogen and phosphorous global cycling:

“Anthropogenic N [nitrogen] and P [phosphorous] loading to estuarine and coastal marine ecosystems has more than doubled in the last 100 years [...] leading to a global spread of coastal eutrophication and associated increases in the number of oxygen-depleted “dead zones” [...], loss of sea grass beds [...], and increases in the occurrence of toxic phytoplankton blooms.” (WOA, 2016: Chapter 6, pg. 16)

Furthermore, irrigation activities associated with crop production is having huge impacts on freshwater sources, with many of the world’s major rivers completely over-used. The pumping of underground aquifers for irrigation purposes has been so extensive that they are being depleted on a vast scale because their natural recharged rate cannot keep up with extraction (Sachs, 2015).

Second, a more recent human pollutant which has garnered widespread attention is plastic (Vince, 2014; UNEP, 2016). As a recent report by UNEP *Marine Plastic Debris and Microplastics* (2016) makes clear:

“Plastic debris/litter and microplastics are ubiquitous in the ocean, occurring on remote shorelines, in coastal waters, the seabed of the deep ocean and floating on the sea surface; the quantity observed floating in the open ocean in mid-ocean gyres appears to represent a small fraction of the total input.” (pg. 1)

A deeply worrying situation further underscored by the observation of Vince (2014) that:

“Every square kilometre of ocean now contains an average of 18,500 pieces of floating plastic, and vast floating islands of garbage coalesce in the currents. In the major ocean gyres, the ratio of plastic to marine life is six to one by weight.”

The type, amount and global distribution of both macro- and micro-plastics can have serious ecological consequences, for example, through species becoming entangled in plastic debris or species ingesting plastic debris. In relation to entanglement:

“It is estimated that between 57 000 and 135 000 pinnipeds and baleen whales globally are entangled each year, in addition to the countless fish, seal, birds and turtles, affected by entanglement in ingestion of marine plastic.” (UNEP, 2016, pg. 81)

Whilst with regards to the ingestion of different plastics the same report notes that:

Seabirds appear to be particularly susceptible at mistaking plastics for their natural prey. Most dead Laysan albatross (*Phoebastria immutabilis*) chicks on Midway Atoll in the Pacific Ocean have been found to contain plastics in their guts, with items such as disposal cigarette lighters, toys and fishing gear.” (pg. 84)

And that overall:

“A review commissioned by the Scientific Technical and Advisory Panel (STAP) of the GEF, in collaboration with the Secretariat of the Convention on Biological Diversity (CBD 2012), concluded that 663 species had been reported as having been entangled in or ingested plastic debris, an increase of 40% in the number of species since the previous global estimate.” (pg. 85)

Metallic chemicals and elements exuded from ocean litter pose particular dangers because they can enter the food chain where they then accumulate through being ingested and amassed by different creatures eventually reaching toxic levels: levels of toxicity that can detrimentally impact species and food-webs alike (UNEP, 2012). Recent evidence, for example, has indicated that cigarette butts present a particularly serious marine environmental harm: Of the estimated 5 trillion cigarette butts that are discarded into the global environment on a daily basis (a startling statistic), many end-up degrading in the marine environment, with elements such as cadmium, iron, zinc, nickel, arsenic, copper and manganese leaching out in quantities that, when considered on an annual basis, pose an especially nasty threat to marine life (Dobaradaran et al., 2016). Other sources of marine pollutants include shipping cargo, and in particular discharges from shipping vessels as well as instances of accidental oil spills (UNEP, 2012).

The oceans have been plundered as a source of food for centuries, but since the 1950s in particular, our plundering of the oceans to meet rising demand, enhanced by technological innovations in tracking and catching fish, has reached astronomical proportions and lead to

the collapse of many fish stocks imperilling marine biodiversity and productivity (Vince, 2014; Sachs, 2015; WOA, 2016). For example, as Sachs (2015:460) states:

“The wild catch in 1950 was about 20 million tons. By 1990, that had become about 80 million metric tons, and it then levelled off at that rate.”

These figures reflect the population-growth-led drivers of fish consumption worldwide, and the fact that, for many millions of people, fish protein is the main source of dietary protein: particularly poor people in low and middle income countries. However, at the same time, fish is also consumed in large quantities in middle and high income countries in the “developed” world, and indeed here, per capita levels of consumption are higher than in the poorer countries which rely on fish as their primary protein:

“...fish and marine invertebrates provide 17 per cent of animal protein to the world population, and provide more than 20 per cent of the animal protein to over 3 million people, predominantly in parts of the world where hunger is most widespread. Asia accounts for 2/3 of the total consumption of fish. However, when population is taken into account, Oceania has the highest per capita consumption (approximately 25 kg per year), with North America, Europe, South America and Asia all consuming over 20 kg per capita, and Africa, Latin America and the Caribbean are around 10 kg per capita.” *WOA, 2016: Chapter 10, pg. 4)

As a consequence of such rapid increases in wild catch many fish stocks are nearing depletion, as Vince (2014:183) explains:

“Globally, 85% of stocks are over-exploited, depleted or fully exploited or in recovery. Stocks of large fish are down 90% [...] Vast areas of the seabed of the Mediterranean and North Sea already resemble a desert, expunged of fish by Europeans using increasingly efficient methods such as bottom trawling.”

The consequences of fish stock depletion are three-fold, and they are all examples of demand driving supply. First, the fishing industry has extended its global reach in capturing new fish stocks, so returning to the example of Europe once again, a significant proportion of the European catch is now made in African waters (Vince, 2014). The reality being that, over the course of the last 60 years, we have gone from fishing a few miles off the coast along a number of key coastal and river areas, to conducting fishing activities on the open ocean wherever populations of fish are able to be caught (Sachs, 2015). For example:

“Significant growth in marine capture fisheries has occurred in the eastern Indian Ocean, the eastern central Atlantic and the northwest, western central and eastern central Pacific over the last decade...” (WOA, 2016: Chapter 11, pg. 3)

The result has been devastating in some cases, as Vince (2014:183) highlights:

“All West African fisheries are now over-exploited, so that many local people can no longer support their families through artisanal fishing.”

A second consequence (as well a driver) of the repeated collapse in fish stocks is that humans have been ‘fishing down the food chain’. We have altered marine ecosystem structure

and functions, by altering their food-web structure: at each stage siphoning off the large fish species at the top of the food chain and then progressively, as those fish get eaten, moving down the food chain all the time to maintain a supply of fish that meets demand, and as a result degrading the whole food chain (Pauly et al., 1998; Sachs, 2015). In others cases it is argued that rather than starting at the top we may actually be “fishing up the food chain” (Branch et al., 2010) – whichever it is, up or down, in either case humans are severely perturbing the normal structure and functioning of marine systems.

The third consequence of reduced wild caught fish has been, particularly since the 1970s, the rapid growth in the aquaculture industry and an escalation in aquaculture production to the extent that farmed fish now forms a significant proportion of the diet of millions of people around the world:

“Globally aquaculture production has increased at approximately 8.6 per cent per year since 1980, to reach an estimated 67 million tons in 2012, although the rate of growth has slowed slightly in recent years. Of that total, however, more than 60 per cent is from freshwater aquaculture.” (WOA, 2016: Chapter 10, pg. 2)

Overall, consumption of fish from wild caught and farmed sources has continued to grow over the last 60 years:

“Production of fish from capture fisheries and aquaculture for human consumption and industrial purposes has grown at the rate of 3.2 per cent for the past half century from about 20 to nearly 160 million million tons by 2012.” (WOA, 2016: Chapter 11, pg. 1)

An important implication of humanity’s intensive fishing activities is the amount of unintentional species we catch, so-called by-catch, much of which is then subsequently discarded:

“All species caught or damaged that are not the target are known as by-catch; these include, inter alia, marine mammals, seabirds, fish, kelp, sharks, molluscs, etc. Part of the by-catch might be used, consumed or processed (incidental catch) but a significant amount is simply discarded (discards) at sea. Global discard levels are estimated to have declined since the early 1990s, but at 7.3 million tons are still high.” (WOA, 2016: Chapter 10, pg. 8)

In many cases we are overharvesting exotic species, in many cases for lucrative high consumption markets in East Asia, for example sharks and turtles:

“Around 100 million sharks are killed each year – their numbers have declined by 80% worldwide, with one-third of shark species now at risk of extinction.” (Vince, 2014, pg. 185)

In another form of plunder, human demand for energy has also had a massive impact on river systems and the marine environment (Vince, 2014). Hydropower for electricity generation, which involves the construction of colossal dam infrastructure developments, have

had an enormous impact on the world's major river systems: changing their character, composition, course, and in some cases creating entirely new river systems – and these hydrodam projects are set to continue at the global scale, as Vince (2014:77) explains:

“Over the last century, humans have built the equivalent of a dam a day – the vast majority since 1950. Two-thirds of the world's major rivers have now been disrupted with more than 50,000 large dams – there are more than 85,000 dams in the US alone [...] With a 40% increase in global hydropower predicted by 2050, humanity in the Anthropocene has designs on most major rivers [...] In Europe and North America, most of the hydropower potential has now been exploited [...] In Africa, Asia and South America, though, hundreds of hydrodams are being planned to provide essential electricity for some of the world's poorest people, and in some of the most ecologically important environments from Patagonia to the Amazon to the Congo.”

A third of the world's oil and gas extraction now occurs offshore¹⁶, and the scale of these operations has risen rapidly over recent decades to meet escalating global demand, with the result that there are now around 900 large-scale oil and gas offshore platforms in operation around the world – many of them pushing the envelope of extraction depth limits in order to find new reserves as others dry-up – some drilling up to depths of almost 3000 metres (WOR 3, 2014). As the World Ocean Review No.3 (2014:18) goes on to state in relation to oil:

“According to recent studies, 481 larger fields were found in deep and ultra-deep waters between 2007 and 2012. They account for more than 50 per cent of the newly discovered larger offshore fields, i.e. fields with an estimated minimum 170 billion barrels of recoverable reserves, corresponding to around 23,800 million tonnes of oil equivalent. The deepwater and ultra-deepwater sectors are thus becoming ever more important.”

Many of these new and important oil deposit regions are located off the Atlantic coast of South America and the west coast of Africa, so future exploration and extraction activities look set to continue, develop and grow in these regions (WOR 3, 2014).

Seabed mining is still in its infancy but as onshore mining deposits for various mineral ores and precious metals become exhausted and extraction and production become too expensive to meet global demand so attention will turn, buoyed by technological developments, to the wealth of resources beneath the seabed and in particular to the extraction of manganese, cobalt and sulphide deposits. The consequences for the marine environment could be particularly risky in the early stages of mining developments as well as long lasting in some cases because mining activities can profoundly alter the local physical environment (WOR 3, 2014). Dredging of the seabed for other materials such as sand and gravel is an activity with a long history, with vast quantities of sand for example removed on an annual basis, usually for use in construction and manufacturing. For instance, in 2012, 93.5 million cubic metres of sand were removed from European waters alone. Dredging on this

scale has profound effects not only on the physical structure of the seabed but also marine biodiversity (WOR 3, 2014).

Overall, humans are changing (and have changed) the marine environment in a number of different ways¹⁷, and our influence can be seen throughout the marine and river systems (Vince, 2014).

1.5 Planetary Boundaries

Over the last 12,000 years humanity has flourished in a period of relative climatic stability called the Holocene. Achieving this position has not been without costs however: the driving forces behind Modernity – economic growth and globalisation – have begun to nudge us beyond those favourable Holocene conditions (Rockström et al., 2009; Vince, 2014; Sachs, 2015; Steffen et al., 2015), as Steffen et al., (2015:736) remark:

“There is increasing evidence that human activities are affecting ES [Earth System] functioning to a degree that threatens the resilience of the ES – its ability to persist in a Holocene-like state in the face of increasing human pressures and shocks.”

Providing a global lens through which to examine changes in Holocene conditions and indicate “safe operating zones” for conditions like climate change and biosphere integrity presently imperilled by human activities, the Planetary Boundaries Framework (PBF) offers a means to assess the risks of further boundary transgressions¹⁸: highlighting those crossed (e.g. genetic diversity and biogeochemical flows) and others on the brink (Figure 1.2; Rockström et al., 2009; Steffen et al., 2015)

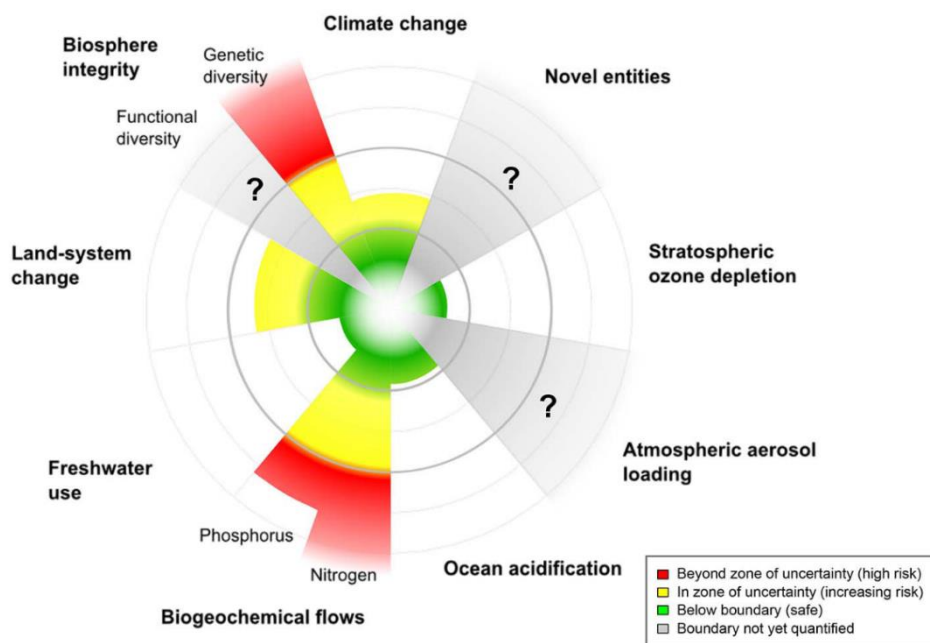


Figure 1.2 The PBF and the current nine planetary boundaries (PB) (source: Steffen et al., 2015)

Clearly, PBs operate not only at the global scale but also at multiple sub-global scales, and as such may have different and not necessarily singular thresholds thus, it is argued, that providing a boundary represents an important step in ensuring these Earth System processes remain within “acceptable” zones of operation and don’t creep towards zones of uncertainty where they could tip over into high risk areas moving way beyond our Holocene-like conditions (Rockström et al., 2009; Steffen et al., 2015).

Human activities have caused a number of PBs to enter zones of uncertainty and some to cross-over into high risk zones, for example: Values for key climate change variables (i.e. radiative forcing and atmospheric carbon dioxide concentration) both exceed what are regarded as acceptable PB limits: atmospheric carbon dioxide is already at 399 ppm, above the PB of 350 ppm, whilst radiative forcing stands at 2.3 W m^{-2} beyond the recommended limit of 1.0 W m^{-2} , with observations of increased disruption to global circulation patterns, instances of drought, duration of heat waves and alteration in rainfall patterns increasing (Sachs, 2015; Steffen et al., 2015). Biogeochemical flows have witnessed increasing degrees of transgression; rates of fertilizer applications have expanded enormously, with current estimates suggesting that the global application of phosphorous to croplands (approx. $14.2 \text{ Tg P yr}^{-1}$) is twice the level of the recommended PB. Similarly, industrial and biological fixation of nitrogen is currently estimated to be 150 Tg N yr^{-1} ; again, twice as large as the recommended PB value (Sachs, 2015; Steffen et al., 2015). Actions such as deforestation, pollution, land use change, ocean acidification, freshwater depletion and climate change have all impacted upon biosphere integrity, as measured in terms of genetic and functional diversity, with species extinction rates orders of magnitude higher than background levels (Sachs, 2015; Steffen et al., 2015).

1.6 Final Remarks

It is evident from this chapter that the “Natural Garden” we have inherited, that we live in, that we have shaped and transformed, in many cases unwittingly so, is in a challenging state. We have not covered all aspects of the Garden here partly because the scope is vast, but we have selected and highlighted those aspects of the Garden that do relate more explicitly to the broader theme of ecosystem services covered in subsequent chapters. However, the parlous state of these facets we have discussed is likely indicative of those dimensions we have not mentioned. While the state of the global environment might be imperilled in many respects, what this chapter has also shown is the sheer dependence of humanity on those resources, particularly in terms of food and livelihoods, and therefore the stake we have in, moving forwards, halting and reversing those deleterious environmental consequences of our actions.

Notes

1. According to the Convention of Biological Diversity (CBD) biodiversity refers to:

“...the variability among living organisms from all sources including, among others, 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” (UN 1992 Article 2).

2. As UNEP (2012:144) reports:

“Declines in freshwater populations are steeper, at 35 per cent since 1970, than those for terrestrial populations, which have fallen by 25 per cent and marine populations by 24 per cent; those in the tropics are steeper than those in temperate latitudes.”

3. According to the IUCN Red List website (www.iucnredlist.org) the Red List is:

“...a critical indicator of the health of the world’s biodiversity. Far more than a list of species and their status, it is a powerful tool to inform and catalyze action for biodiversity conservation and policy change, critical to protecting the natural resources we need to survive. It provides information about range, population size, habitat and ecology, use and/or trade, threats, and conservation actions that will help inform necessary conservation decisions. The IUCN Red List is used by government agencies, wildlife departments, conservation-related non-governmental organizations (NGOs), natural resource planners, educational organizations, students, and the business community.”

4. The implications of wildlife exploitation are quite startling:

“Globally, utilized vertebrate populations have declined by 15 per cent since 1970 as indicated by the Living Planet Index. Similarly, the extinction risk of utilized bird species increased during 1988–2008, driven in part by overexploitation.” (UNEP, 2012:142)

And in terms of the impact of overexploitation on marine capture fisheries, the trend is equally as disturbing:

“In the marine realm, capture fisheries more than quadrupled their catch from the early 1950s to the mid-1990s. Since then, catches have stabilized or diminished, despite increased fishing effort. The proportion of marine fish stocks that are overexploited, depleted or recovering from depletion rose from 10 per cent in 1974 to 32 per cent in 2008. Of the 133 local, regional and global extinctions of marine species documented worldwide over the last 200 years, 55 per cent were caused by overexploitation, while the remainder were driven by habitat loss and other threats.” (UNEP, 2012:142)

5. As detailed in Miura et al., (2015) the soil protection and formation function of forests is especially important:

“It is generally accepted that forests and trees, in undisturbed form, provide the greatest vegetative protection against erosion from rain, wind, and coastal waves [...] Accordingly, they also significantly contribute to the reduction of downstream sedimentation [...] The root system of the trees creates increased soil strength [...] Forests and trees contribute to the preservation of a good soil structure thanks to the protection against splash erosion (provided the litter layer and the understory vegetation are maintained) and maintenance of robust biological activity in the soil [...] In this context, forests and trees also contribute to the mitigation of risks of shallow landslides. However, deep-rooted mass movements triggered by tectonic events cannot be prevented by forests and trees...” (pg. 36)

6. In addition: roughly 0.83% of global forest area is also designated for coastal stabilization; 3.4% for ensuring clean water; 0.36% for avalanche control; 5.1% for flood and erosion control; 25.4% for other ecosystem services; 4.3% separately for public recreation; 1.3% for carbon storage; 1.9% for cultural services and additional 2% for unspecified services (Miura et al., 2015).

7. Forests are central to the livelihoods of millions of people as well as the global economy as Köhl et al., (2015:22) explain:

“It is estimated that about 410 million people are highly dependent on forests for subsistence and income, and 1.6 billion people depend on forest goods and services for some part of their livelihoods [...] Wood and manufactured forest products add more than \$450 billion to the world market economy annually, and the annual value of internationally traded forest products is between \$150 billion and \$200 billion.”

8. As further evidenced by Sloan and Sayer (2015), infrastructural developments can be highly deleterious to forests for both acute and chronic reasons:

“Massive infrastructure investments are planned for many tropical regions and will soon open most of the world’s remaining remote and pristine forests to commercial interests seeking land for estate crops, including industrial forest plantations [...] The effects of such investment on forests are difficult to anticipate and have arguably not been fully accounted for in national economic and forest management strategies [...] Agricultural expansion along new and improved roadways may concentrate populations and enable agricultural transformation and intensification; where this occurs in agriculturally favourable areas a depopulation of hinterlands may reduce pressures on forests [...] In countries with weak governance, new infrastructure may pave the way for opportunistic land development, with negative consequences for forests and the people dependent on them [...] Whereas infrastructure development would ideally be directed towards regions with high agricultural potential and little forest cover, the contrary is often the case when infrastructure expansion targets mineral resources or estate crops and, to a lesser degree, industrial timber plantations [...] New mineral infrastructure poses significant threats to the major tropical forests in the Amazon and Congo Basins as well as the islands of Borneo and New Guinea – collectively accounting for most of the world’s intact tropical forests.” (pg. 135)

9. Expanding a little more the geographical heterogeneities in natural forest contraction and planted forest expansion at the national level, based on the Global Forest Resource Assessment, Keenam et al., (2015) make the following observations:

“At the national scale, net loss of forest area between 2010 and 2015 for countries in South America was dominated by Brazil (984 K ha y⁻¹), but there were also significant net losses in Paraguay (325 K ha y⁻¹), Argentina (297 K ha y⁻¹), Bolivia (289 K ha y⁻¹) and Peru (187 K ha y⁻¹). In South and Southeast Asia, the rate of net forest loss was greatest in Indonesia (684 K ha y⁻¹), followed by Myanmar, where the loss rate of 546 K ha y⁻¹ between 2010 and 2015 was 25% higher than in the 1990s. In Africa, the greatest net losses in forest area between 2010 and 2015 were in Nigeria (410 K ha y⁻¹), Tanzania (372 K ha y⁻¹), Zimbabwe (312 K ha y⁻¹) and Democratic Republic of Congo (311 K ha y⁻¹) [...] Other countries have reported a net rise in forest area between 2010 and 2015. China has the highest rate of expansion (1.5 M ha y⁻¹), though this is only 63% of the corresponding rate in the 2000s. Forest area increased rapidly in the last five years in Chile (301 K ha y⁻¹), the USA (275 K ha y⁻¹), the Philippines (240 K ha y⁻¹), Lao People’s Democratic Republic (189 K ha y⁻¹), India (178 K ha y⁻¹), Vietnam (129 K ha y⁻¹) and France (113 K ha y⁻¹). There was a net increase in forest area of 308 K ha y⁻¹ in Australia between 2010 and 2015 but, reflecting the variability of climate in this country, this followed a net loss of 563 K ha y⁻¹ in the 2000s, caused by a mixture of drought, fire and human clearance.” (pg. 12)

10. Soil erosion is an especially widespread and problematic ecological issue caused by extensive land conversion, either for the purpose of agriculture or the result of settlement developments, as Vince (2014:128) outlines:

“Soil is running out around the world. Every year, 75 billion tonnes – more than 100,000 square kilometres of arable land – is lost. Around 80% of global farmland is now moderately or severely degraded and, in the past forty years, one third of cropland has been abandoned. In Europe, soil is being lost seventeen times faster than it can be replenished naturally; in China, it’s fifty-seven times. Every year, more land is sealed beneath buildings and roads. And because cities have usually grown from settlements on the most fertile land, vast urban centres are now squatting on some of our best soils. Soil

erosion has brought down empires in the past, and is a major concern in the Anthropocene.”

11. If we take the example of meat consumption in particular recent trends suggest a global rise in production and consumption patterns, although these patterns are not homogenous and differ geographically, probably fuelled by the expansion of middle income groups, the environmental consequences can be stark:

“Meat production has increased significantly during the past two decades, outpacing the rate of population growth over the same period. Large differences in meat consumption exist both within and between countries, ranging from an average of 83 kg per person per year in North America and Europe to 11 kg per person per year in Africa. Population growth, urbanization and increasing incomes are expected to continue to raise demand for meat, particularly in developing countries. Considering the entire commodity chain, including deforestation for grazing and forage production, meat production accounts for 18–25 per cent of the world’s greenhouse gas emissions, which is more than global transport.” (UNEP, 2012:81-82)

The growth in meat production also has knock on effects on crop cultivation, as more crops are needed to provide feed for livestock (UNEP, 2012:82):

“As meat production has grown, so has the area harvested for soybean crops, which expanded to 98.8 million hectares in 2009 from 74.3 million hectares in 2000, and 50.4 million hectares 30 years ago. An increasing demand for meat has the potential to compound rangeland degradation.”

12. As detailed in UNEP (2012), the rise of biofuels over the last two decades has been especially significant in relation to the conversion of primary and secondary forests as well as in relation to the conversion of standard crop varieties to fuel. There has been a consequent rise in biofuel related crop cultivation, for example, palm oil, sugar cane, soy and maize. These changes link to wider issues associated with food and feed production, which combined have significant ecological and social implications. Massive levels of land conversion to biofuel crops have occurred in the United States, where in 2007 24% of its corn was converted to ethanol, and similar trends have also been witnessed across Europe. Of major environmental significance biofuels have also been linked to widespread deforesting activities in tropical regions such as Indonesia, for example:

“The expansion of oil palm plantations, both for food and fuel, is one of the most significant causes of rainforest destruction in South East Asia, where the area under oil palm increased from 4.2 million to 7.1 million hectares between 2000 and 2009. In Indonesia, two-thirds of oil palm expansion has occurred by converting rainforest. Clearing tropical forests produces a carbon debt that lasts from decades to centuries, contradicting one of the main reasons for pursuing biofuels in the first place. It also compromises vital ecosystem functions provided by rainforests that cannot be replaced by plantations.” (UNEP, 2012:84)

13. In relation to shipping and trade, according to World Ocean Review No.4 (2015:39):

“...the quantity of goods transported by ships since the mid-1980s has constantly grown – from around 3.3 billion tonnes in 1985 to around 9.6 billion tonnes in 2013.”

14. As Vince (2014) points out, sea level rise is not homogenous, but instead rises at different rates and levels at different points around the globe:

“The oceans are not rising uniformly – the Pacific, for example, will grow 20% higher than average. And, while tropical islands disappear, new ones are showing up in the artic. Uunartoq Qeqertag (meaning Warming Island) was born in 2006, the result of the melting of an ice bridge that formerly cloaked it and mainland Greenland as one.” (pg. 178)

15. Projections state that should the Greenland Ice Sheet melt global sea level will rise by several metres as a consequence (Vince, 2014).

16. As reported in World Ocean Review No.3 (2014:17):

“Offshore oil extraction currently accounts for 37 per cent of global production. At present, 28 per cent of global gas production takes place offshore – and this is increasing.”

17. The World Ocean Assessment (WOA, 2016: Chapter 54, pg. 1-2) notes seven distinct human pressures on the marine environment that require future attention:

“(a) Climate change (and ocean acidification, including the resulting changes in salinity, sea-level, ocean heat content and sea-ice coverage, reduction in oxygen content, changes in ultra-violet radiation);

(b) Human-induced mortality and physical disturbance of marine biota (such as capture fisheries, including by-catch), other forms of harvesting, accidental deaths such as through collisions and entanglement in discarded nets, disturbance of critical habitat, including breeding and nursery areas);

(c) Inputs to the ocean (these can be broken down according to the nature of their effects: toxic substances and endocrine disruptors, waterborne pathogens, radioactive substances, plastics, explosives, excessive nutrient loads, hydrocarbons). Remobilization of past inputs also needs to be considered;

(d) Demand for ocean space and alteration, or increase in use, of coasts and seabed (conflicting demands lead to both changes in human use of the ocean and changes to marine habitats);

(e) Underwater noise (from shipping, sonar and seismic surveys);

(f) Interference with migration from structures in the sea or other changes in routes along coasts or between parts of the sea and/or inland waters (for example, wind-farms, causeways, barrages, major canals, coast reinforcement, etc.);

(g) Introduction of non-native species.”

18. Human-biosphere interactions are dynamic, operate across multiple spatial and temporal scales and are mediated by feedback loops – by definition they are hard to predict (Steffen et al., 2015)

Chapter 2: Connections In The Living Garden: Biodiversity And Ecosystem Services

“As well as the broad and complex nature of biodiversity and ecosystem services, the links between them are many and varied. Biodiversity may underpin some ecosystem services but not others; it may provide few improvements to ecosystem services in the short term but aid sustainable, long-term provision.” (Science for Environment Policy, 2015, pg. 7)

“Ecosystem processes result from the life processes of multi-species assemblages of organisms and their interactions with the biotic environment, as well as the abiotic environment itself.” (TEEB, 2012, pg. 46)

In Chapter 1 we briefly described the present state of our planetary Garden. In Chapter 2 we take the opportunity to explore recent evidence that sheds light on the connections and interactions between “life” in the Garden. In particular, we cast our gaze on the relational connections between ecosystem services and biodiversity, a lynch-pin of the ecosystem services narrative, as well as between climate and biodiversity. In so doing, we touch upon one of the central research pillars in the application of the ecosystem services framework, whilst also discussing some recent science-policy developments in operation (at different scales) that seek to understand, provide and translate ecosystem service science into environmental policy and practice.

2.1 Biodiversity And The Ecosystem Services Framework

The relationship between ecosystem services and biodiversity has undergone a radical transformation over recent years (Jacobs et al., 2014). From being considered just one of many services biodiversity is now recognized as a linchpin, an overarching support to all ecosystem services (Figure 2.1; TEEB, 2012; Jacobs et al., 2014). Whilst this newly integrated recognition of the importance of biodiversity is welcomed, supporting services are often regarded as intangible, and generally beyond quantification and monetization (UK NEA, 2011). This has created problems when priceable ecosystem services benefits are used as proxies to value ‘biodiversity’ (Jacobs et al., 2014). In Part 5 we discuss in detail the issues and challenges presented by the valuation component of ecosystem services, here we suggest that another significant issue lies in the way biodiversity is used to convey multiple concepts in the language of ecosystem services, and beyond that how management goals concerned with optimizing conservation outcomes (i.e. biodiversity gains) can conflict with managing for a broader array of ecosystem services (Bennett et al., 2009; Jacobs et al., 2014). Disentangling the fundamental relationships between biodiversity and ecosystem services is therefore fundamental to the

ecosystem service paradigm and biodiversity science-policy (Mace et al., 2011; Fu et al., 2013; Balvanera et al., 2014).

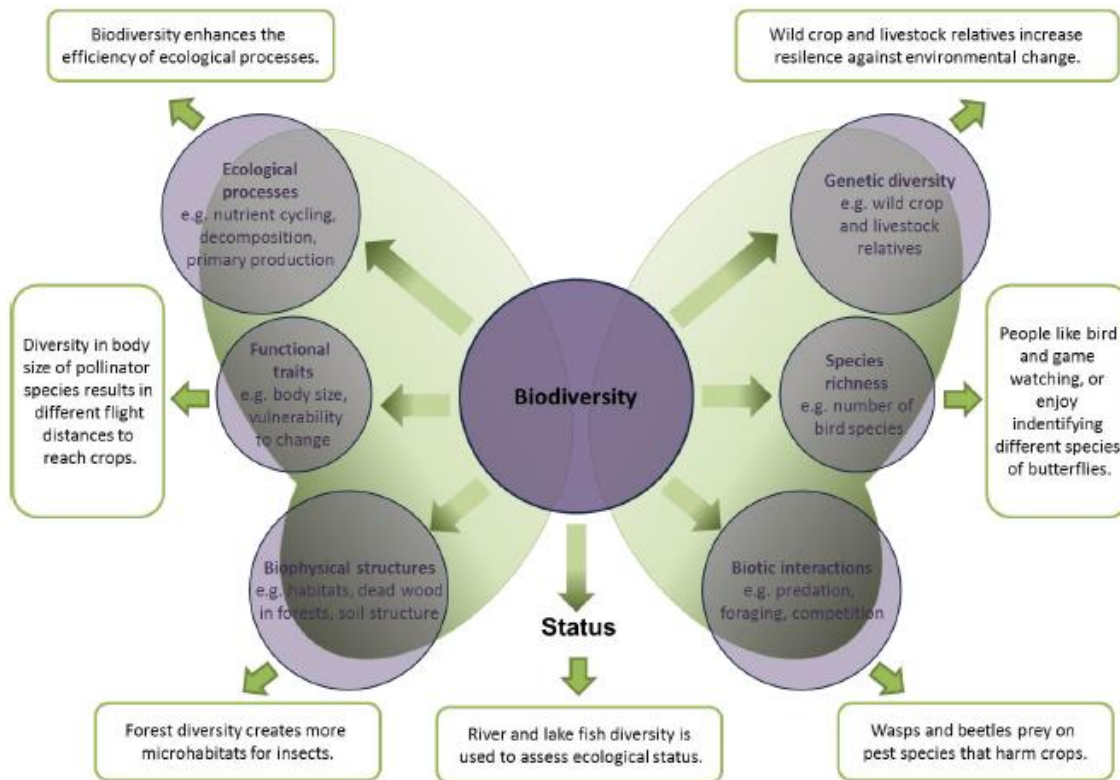


Figure 2.1 The integrative and multifaceted functions of biodiversity in support of ecosystem services (source: Figure 3 from European Union (2013) Mapping and assessment of ecosystems and their services: an analytical framework for ecosystem assessments under action 5 of the EU biodiversity strategy to 2020, discussion paper. (<http://ec.europa.eu>))

2.2 Ecosystem Integrity, Biodiversity And Ecosystem Services

One concept that has gained recent ascendancy in its attempts to grapple with the complex issues of distilling the connections between biodiversity and ecosystem services, as well as addressing underlying questions regarding the provision of ecosystem services, is ecosystem integrity (Fu et al., 2013a). According to Fu et al., (2013a) ecosystem integrity comprises three dimensions, namely: structure¹, composition² and process³, and collectively, these components and their interactions, it is argued, determine the regulation and provision of ecosystem services. In particular, Fu et al., (2013a) singles out water⁴, soil⁵ and carbon⁶ as the principal ecological factors regulating ecosystem processes due to their significant spatial and temporal dependence.

On a different front, Mace et al., (2011) tackle the idea that biodiversity does not just have a single straightforward relationship with ecosystem services, as is normally presented, but a more complex one and they sketch out three facets of biodiversity that have important

implications for the relationship between biodiversity and the ecosystem services framework, namely: biodiversity as a regulator of ecosystem processes; biodiversity as a final ecosystem services (e.g. functional aspects of biodiversity linked to provisioning services) and biodiversity as a good (e.g. aesthetic value of biodiversity). Disentangling the different roles of biodiversity in this way has served as a positive means of moving beyond conceptual confusion and ESV difficulties (Jacobs et al., 2014) because as Balvanera et al., (2014:) state:

“...we need to know whether, how, and when the maintenance of biodiversity is key to sustaining the flow of services to societies.”

2.3 Biodiversity And Ecosystem Functioning

Significant progress has been made in the field of biodiversity and ecosystem functioning over recent decades (Cardinale et al., 2012). Experimental and theoretical research has vacillated between the importance given to taxonomic diversity, species richness and functional trait diversity (Mori et al., 2013). Recent meta-analyses and syntheses have been published trying to pull together and draw out key general themes (e.g. Hooper et al., 2005; Balvanera et al., 2006; Hillebrand and Matthiessen, 2009; Reiss et al., 2009; Cardinale et al., 2012; Elmqvist and Maltby et al., 2012; Naeem et al., 2012; Mori et al., 2013; Snelgrove et al., 2014).

In summary these analyses have found that biodiversity supports, diversifies and stabilizes ecosystem functioning (Jacobs et al., 2014), as Elmqvist and Maltby et al., (2012:50) remark:

“...an increasing body of scientific evidence indicates that functional diversity, rather than species diversity *per se*, enhances ecosystem functions such as productivity, resilience to perturbations or invasion and regulation of the flux of matter.”

Conversely, loss of biodiversity is associated with poorer ecological functioning (e.g. nutrient cycling) and a weakening of basic ecosystem processes (Hillebrand and Matthiessen, 2009; Cardinale et al., 2012). In some cases, there are species which have a disproportionate impact on how a community and ecosystem functions relative to their abundance and biomass, sometimes referred to as keystone species or in more specific contexts ecosystem engineers, their loss can have a dramatic impact on community dynamics through their effects on say, for example, bio-invasions or pollination (Wright and Jones, 2006; Elmqvist and Maltby et al., 2012). From this perspective functional trait diversity acts as an insurance policy for ecosystems (Elmqvist and Maltby et al., 2012):

“Redundancy or contingency (i.e., more than one species performing the same process role) in functional traits and responses in ecosystems may act as an ‘insurance’ against the loss of individual species. This is enhanced if the diversity

of species in the ecosystem encompasses a variety of functional responses types [...] Response diversity [...] has been argued to be critical in ecosystem resilience.” (pg. 54)

Overall the temporal stability of ecosystem functions is related to higher levels of biodiversity (Cardinale et al., 2012; Elmqvist and Maltby et al., 2012), and “stability” is regarded as a key ecosystem attribute at a time when multiple anthropogenic forcing factors are undermining ecosystem functioning:

“Stability may be critical as ecosystems come under increasing pressure from myriad anthropogenic drivers, from climate change to invasive alien species. Furthermore, these drivers may have a dual effect: a direct impact on ecosystem services, and an impact on biodiversity, which in turn can affect ecosystem services.” (Science for Environmental Policy, 2015, pg. 8)

Ecosystem functionality in particular has been shown to have strong connections with individual aspects of biodiversity (e.g. genetic, trophic and resource diversity), with the composition of functional traits acting as a reasonable predictor of ecosystem processes (Jacobs et al., 2014). It has been suggested that land management activities (e.g., agrobiodiversity) impact ecosystem processes and ecosystem service provision through their influence on functional trait diversity and composition, and that taking a functional trait approach to management regimes will improve agro-ecological conditions (Woods et al., 2015). In this regard the spatial-temporal maintenance of multiple ecosystem processes requires higher levels of biodiversity but, it appears, not just any type of biodiversity rather there is evidence to recommend that inter-trophic loss of diversity is likely to have a greater impact on ecosystem functions than intra-trophic loss of diversity (Cardinale et al., 2012). And, taking it one stage further, it has been argued that overall biodiversity loss – particularly in relation to primary productivity – is as significant a global driver of environmental change as factors such as rising greenhouse gas concentrations and ocean acidification (Cardinale et al., 2012; Hooper et al., 2012; Naeem et al., 2012; Balvanera et al., 2014).

Nevertheless, even with such extensive progress questions and debates remain, with many researchers arguing that investigations continue to focus too heavily on the relationship between species richness and ecosystem functioning, and that in fact much greater attention to response diversity⁷ would provide:

“...a deeper, process-orientated understanding and recognition of the functional consequences of biodiversity that goes beyond a focus on maximizing species richness and diversity as the ultimate insurance against environmental change” (Mori et al., 2013:361)

Supporting this view Norris (2012) argues that the relationship between biodiversity and ecosystem functioning needs to be appraised within a far more dynamic framework, with a

move away from a purely species-level oriented focus and a greater level of inquiry directed towards other aspects of biodiversity and indeed other variables:

“...response variables of interest may not necessarily be measures of biodiversity. It might be more appropriate to consider chemical, physical, economic or social endpoints rather than biodiversity per se” (Norris, 2012:195)

Moving forwards, Cardinale et al., (2012) propose that more attention needs to focus on exploring scenarios of biodiversity change that are more realistic and account for the effect of human actions. In other words, greater experimental realism and complexity is necessary across spatial-temporal scales, for example, modelling the impact on food webs of species invasions and species extinctions arising from anthropogenic activities (e.g. MacDougill et al., 2013).

2.4 Biodiversity And Ecosystem Services

Developments in uncovering the relationships between biodiversity and ecosystem services have gained pace over recent years, even though the connections between specific aspects of biodiversity like species richness or functional trait diversity and ecosystem services and subsequently on through to human-wellbeing remain elusive and sparse (Cardinale et al., 2012; Balvanera et al., 2014). This elusiveness can be, in many ways, traced to three principal reasons that have been posited to explain why relating biodiversity measures to ecosystem services is challenging: The first of these complications is the multidimensional nature of biodiversity with measurements of biodiversity taking a myriad of different forms, the issue then becomes a question of which face of biodiversity actually relates to ecosystem service generation and provision. The second complication arises from the fact that the wide varieties of measures used to assess specific facets of biodiversity have been designed for particular purposes, some measures may therefore not be appropriate for particular needs. The third complication boils down to determining the strength of the relationship between biodiversity and ecosystem service provision, and which measures of biodiversity are best suited to predicting ecosystem service provision quality and quantity (Elmqvist and Maltby et al., 2012).

Some recent empirical evidence (i.e. observational, experimental and theoretical) supports connections between species richness and six ecosystem services, namely: forage, timber, fishery stability, climate regulation, pest regulation and water quality (Balvanera et al., 2014). In another robust and wide ranging analysis examining a total of 530 studies, biodiversity attributes were, overall, established to be positively related to ecosystem services provision (Harrison et al., 2014). The authors of this study noted in particular that regulating services (e.g. water quality, water flow and mass flow) and cultural services (e.g. landscape aesthetics) were enhanced by habitat and community area. Ecosystem services such as

pollination, pest regulation and atmospheric regulation were positively correlated with species richness and diversity. Furthermore, species abundance was also linked to pest regulation, pollination and recreation services whilst richness was associated with timber production and freshwater fishing. Importantly, few negative associations between biodiversity and ecosystem services were identified, in some cases these related to freshwater provision, for example, it was noted that there were instances in which increases in afforestation reduced average water yield, however, in most cases negative associations between biodiversity and ecosystem services related to the impacts of invasive species (Harrison et al., 2014).

Surveying particular ecosystems, most of the work assessing linkages between biodiversity and ecosystem services has focused on forests (e.g. Myers, 1997; Foley et al., 2007; Chazdon et al., 2008; Guariguata and Balvanera, 2009; Lara et al., 2009; Martinez et al., 2009; Hector et al., 2011; Conti and Diaz, 2013; Edwards et al., 2014; Balthazar et al., 2015), grasslands and plants (e.g. Sala and Paruelo, 1997; Phoenix et al., 2008; Egoh et al., 2009; Maestre et al., 2012; Midgley, 2012; Grigulis et al., 2013; Lavorel, 2013; Storkey et al., 2013), and to lesser extents aquatic and marine systems (e.g. Worm et al., 2006; Beaumont et al., 2007; Mumby et al., 2008; Mora et al., 2011; Samhuri et al., 2012; Liqueste et al., 2013; Thuber et al., 2014) and soils (e.g. Dominati, 2010; Pulleman et al., 2012; Blouin et al., 2013; Robinson et al., 2013).

What this body of research reveals is the sheer variety and types of ecosystem services supplied by these systems, as well as the crucial role of biodiversity in supporting the generation and provision of those services (Jacobs et al., 2014). It is also clear that when these systems are degraded and exploited ecosystem disservices such as flooding, loss of soil fertility, soil erosion, irregular water supply, lack of nutrient cycling can arise all of which negatively impact upon biodiversity and human wellbeing (e.g. Foley et al., 2007; Worm et al., 2006).

Not unsurprisingly, a great of recent work has focused on the links between restoration, biodiversity and ecosystem functioning (e.g. Aerts and Honnay, 2011; Trabucchi et al., 2012), as Blignaut et al., (2014a:35) explain:

“...there is growing recognition that hands-on ecological restoration (ER) and rehabilitation are required as part of the suite of responses society must make to address and reverse widespread ecosystem degradation, desertification, anthropogenic climate change, and the unprecedented loss of biodiversity for which humans are responsible.”

There is a distinction to be made between ecological restoration on the one hand and the restoration of natural capital, where the latter is often regarded as being a much broader concept, again as Blignaut et al., (2014a:40) relay:

“...the concept of restoring natural capital (RNC) is broader than that of ecological restoration, namely “any activity that integrates investment in, and replenishment of, natural capital stocks to improve the flows of ecosystem goods and services, while enhancing all aspects of human wellbeing.” Thus, ecological amelioration and ecological and economic revamping of production systems, resource extraction systems, and transport systems can also contribute to RNC. Furthermore, environmental and ecological education, outreach, and capacity building also contribute.”

Restoration understood in this much broader sense has been used to argue for an economic development pathway to sustainability (Blignaut et al., 2014b):

“RNC combined with on-going land use and resource management adjustments after restoration has begun, aims to increase stocks of natural capital and, therefore, the flow in resources and ecosystem services that exist if, and only if, we maintain natural capital. This requires recognising that something has gone wrong, and then making the choice – and the investment – to bring it right” (pg. 55)

As Blignaut et al., (2013:1) have argued elsewhere:

“Long-term sustainability requires society to invest in restoring natural capital to increase the supply of ecosystem goods and services and to maintain biodiversity that is vital to ecosystem functionality.”

Certainly, restoration programmes directed towards the replenishment of natural capital and the preservation of biodiversity in order to maintain ecosystem functioning demonstrate a generally positive impact on ecosystem service provision, human-wellbeing and socio-economic conditions (e.g. Blignaut et al., 2013; 2014a; 2014b; Montoya et al., 2012; Meli et al., 2014).

Restoration efforts in this respect are not homogenous across scales or geographies, however, with some biases in favour of particular environments and ecosystem services compared to others, for example, as Blignaut et al., (2013:6) note:

“The evidence provided seems to indicate a strong ‘biome bias’ towards restoration in rivers and wetlands, probably because of the rapidly increasing need for clean water, and towards productive terrestrial systems such as forests, woodlands and moist savannas. Less productive and more remote systems (e.g., deep sea, arid savannas, true desert) received less attention [...] The emphasis on livelihood-linked provisioning services in the papers studied decreases from poor to rich host countries, and conversely, the importance of a social and cultural focus increases [...] In high-income countries, the focus of restoration research on cultural services suggests that biodiversity issues, or recreational, aesthetic and amenity values of environments are the major drivers of restoration.”

Restoration projects are quite often severely impeded by a lack of appropriate funding (Montoya et al., 2012). Spelling out the costs and benefits of restoration is therefore especially important (Montoya et al., 2012; Blignaut et al., 2014b). Untangling the economic costs and benefits of restoration, however, is less straightforward and the evidence base has generally been rather sparse in terms of detailed studies, yet recent analysis suggests that restoration can

yield medium to long-term economic gains even when upfront costs are still viewed as prohibitive (Blignaut et al., 2014b):

“...there is almost no information available on the cost-effectiveness of ecological restoration [...] and until now, restoration programmes have been predominantly viewed as an expense (cost) with few tangible financial and economic benefits. Often this is because of erroneous accounting practices and a tendency for conventional cost-benefit analyses to exclude the impact of human activities on ecosystem goods and services affected [...] However, ecological restoration also yields excellent returns on investment, provided a mid- to long-term perspective is adopted, and that the full range of known benefits is considered [...] With regards the broader aim of RNC, the economic argument is much stronger still.” (pg. 58)

Moving forwards, it has been argued that the theory and practice of restoration need to adopt more straightforward assessments of restoration outcomes; focus on delivering bundles of ecosystem services; act at scales that better accommodate ecological populations, communities and meta-dynamics, and have a long-term rather than a short-term view of restoration exercises and programmes (Montoya et al., 2012). In a similar fashion, Menz et al., (2013) argue for a much more holistic and policy-oriented approach to restoration practices that yields the greatest set of benefits:

“...we propose a four-point plan to ensure that restoration sustains and enhances ecological values: (i) identify focal regions with high restoration demands, (ii) identify knowledge gaps and prioritize research needs to focus resources on building capacity, (iii) create restoration knowledge hubs to aggregate and disseminate knowledge at the science-practice interface, and (iv) ensure political viability by establishing economic and social values of functioning restored ecosystems.” (pg. 526)

There is a growing trend in research and practice that is moving away from assessing and concentrating on individual ecosystem services and their provision to the production of multiple ecosystem services (e.g. Hector and Bagchi, 2007; Bennett et al., 2009, Jacobs et al., 2014). This focus on the multi-functionality of ecosystems and landscapes suggests a greater importance for higher levels of biodiversity because species influence functions differently (Elmqvist and Maltby et al., 2012). The inference being that for the production of ecosystem service bundles the more biodiversity the better, at least until the system reaches a saturation point after which each additional species matters less (Jacobs et al., 2014).

Nevertheless, as Balvanera et al., (2014) point out; there remains a great deal of uncertainty regarding the relationships between biodiversity and ecosystem services. The main reasons put forward to explain the still persistent levels of uncertainty are that: (i) different components of biodiversity have different impacts on ecosystem services; (ii) changes in the final ecosystem services consumed by society are rarely measured in relation to, for example, species richness; (iii) only a few of the ways biodiversity can influence ecosystem service provision have been considered and, (iv) most investigations are highly controlled and not

realistic in terms of the factors they consider. Consequently, there are calls for greater realism and a much more focused approach to understanding the wider context in which the interactions between biodiversity and ecosystem services play out (e.g. habitat connectivity, alien species, landscape management and socio-economic decision-making, governance), in order to provide more extensive, robust, accurate and policy-relevant findings (Balvanera et al., 2014; Harrison et al., 2014; Ruckelhaus et al., 2015). These recommendations are also supported by Bennett et al., (2015) who, in surveying various knowledge gaps in our understanding of the connections between biodiversity and ecosystem services, argue that future investigations need to focus on: (i) how biodiversity and other forms of environmental heterogeneity (e.g., landscape and seascape) impact on the production of multiple ecosystem services and (ii) the role of path-dependency and legacy effects in structuring the production and provision of multiple ecosystems services.

2.5 Climate Change, Biodiversity And Ecosystem Services

The evidence for climate change is considerable and its impact on the functionality of the biosphere through various feedback relationships (Field et al, 2007; Chapin III et al., 2008), given the modelled predictions concerning atmospheric and ocean temperature increases, sea-level rise, alterations in ocean salinity and acidity and changes in the composition of atmospheric greenhouse gas concentrations, is thought to be profound (Thomas et al. 2004; MA, 2005, Pimm, 2009; Montoya and Raffaelli, 2010; IPCC, 2014), as the IPCC Fifth Assessment on Climate Change Synthesis Report (2014) remarks:

“Warming of the climate system is unequivocal, and since the 1950s, many of the observed changes are unprecedented over decades to millennia. The atmosphere and ocean have warmed, the amounts of snow and ice have diminished, and sea level has risen.”

The potential implications of an increasingly warm climate for biodiversity are stark:

“...up to one-sixth of all species may go extinct if we follow “business as usual” trajectories of carbon emissions.” (Lambers, 2015:501)

The effects of climate change and related global change issues on the biosphere and human-wellbeing, mediated in part through their influence on the capacity of ecosystem service stocks and flows to be maintained, is expected to be substantial (MA, 2005; Lobell et al., 2008; Montoya and Raffaelli, 2010), as Lafortezza and Chen (2016:576) state:

“Global change issues, including climate change, natural disasters, air and water pollution, urban expansion, and water resource shortages are placing mounting pressures on a range of ecosystem services such as biodiversity, carbon storage, nutrient and water recycling, flood protection, soil quality, and other services.”

This view is also supported in an editorial to a Special Issue of *Current Opinion in Environmental Sustainability* written by Fu et al., (2013b:1) in which the authors state that:

“Climate change studies indicate large changes in mean air temperatures and precipitation patterns during this century with regional variations. Changes in seasonal patterns as well as in the frequency and intensity of episodes are also projected. These changes are affecting ecosystem structure and spatial patterns, driving changes in species distributions and numerous processes in both terrestrial and aquatic ecosystems.”

In addition, as Mooney et al. (2009) note, climate changes effects are expected to augment pre-existing negative anthropogenic impacts on the sustainability of natural resources, and the rise of ecosystem disservices (Munang et al., 2013:49):

“Ecosystem degradation is a process which will eventually lead to the collapse of the ecosystem. The degradation process reduces the capacity of the ecosystem to buffer the impacts of climate change, for example, more frequent heavy rains, droughts, melting glaciers and sea level rise. Biodiversity loss from ecosystem degradation could cause the breakdown of food chains and eventually the collapse of the ecosystem, leading to biological disasters such as the invasion of new species.”

However, it is also clear that the extent of these impacts, whilst acknowledged, is far from certain (Vihervaara et al., 2013:1):

“Climate change poses a serious threat to biodiversity, ecosystem services, and the future of coupled human–environment systems. Biodiversity loss and changes in ecosystem functioning are often local or regional while climate change has a global influence. This makes it difficult to understand the global impact of biodiversity loss and changes in ecosystem function, and to determine their potential negative effects on society.”

Current research indicates that climate change will have serious ecological impacts through, for example, changes in species distributions i.e. range expansions and contractions (Thomas et al., 2004) and the associated problem of increased species invasions (Montoya and Raffaelli, 2010) and temporal alterations in phenology (Cleland et al., 2006; Parmesan, 2006; Cleland et al., 2007; Vihervaara et al., 2013). For example, alteration in plant flowering could have profound effects on pollinators and overall pollination services:

“Shifts in flowering phenology could be wide-ranging, as they may cause temporal mismatch with pollinator activity and modify gene flow among plant populations. Warming can also modify flower quantity, thereby affecting the resources available to pollinators and, potentially, seed production. The flowering responses to climate change have the potential to profoundly affect plant community composition, ecosystem services, and livelihood of many people all over the world.” (Vihervaara et al., 2013:58)

Alterations in phenology are not just associated with chronological shifts in when plants flower but also when species breed and when they decide to migrate, and these changes can have quite profound outcomes:

“...shifts in the timing of climatic events can lead to populations altering the timing of seasonal activities such as migration, flowering, or breeding, with potential consequences for the demography and population dynamics of species and communities. Such phenological shifts are commonly observed in response to climate change and have been linked to the seasonality of climate, but also to trends in climatic averages and extremes. Delays of snowmelt date, for example, have led to delays in the hibernation emergence date of ground squirrels in Canada.” (Garcia et al., 2014: 1247579-5)

In addition, it is expected that species responses to climate drivers should produce heterogeneous range shifts due to intrinsic species differences, affecting the relationship between above-ground and below-ground terrestrial assemblages (Mooney et al., 2009; Raffaelli and Montoya, 2010). For instance, experimental investigations of precipitation patterns on above and below-ground species interactions in a dryland ecosystem in Spain demonstrated that climatological factors do indeed affect the functionality of those interactions (González-Megías and Menéndez, 2012:3115):

“Our results show that rain intensity changes the effect of below-ground detritivores on both plant traits and above-ground herbivore abundance. Enhanced rain altered the interaction between detritivores and plants affecting flower and fruit production, and also had a direct effect on fruit and seed set.”

Effects of climate change have also been observed in aquatic and marine environments, notably in this respect are the changes in ocean temperatures and also acidification (via absorption of atmospheric carbon dioxide) leading to mass coral bleaching events and reduced mineral availability for coral formation (Hoegh-Guldberg, 1999; De'ath et al., 2009). Impacts thought likely to become annual events by 2030-2050. The consequences of which are substantial for ecosystem services, particularly in the so-called Coral Triangle, where over 100 million people who depend on reef ecosystems for their livelihoods will face increasingly severe food security situations (Mooney et al., 2009). Higher ocean temperatures and atmospheric carbon dioxide levels have also been shown to negatively affect nutrient release (Bulling et al., 2010).

Recent marine modelling analyses have indicated that climate change may induce significant shifts in species ranges and distributions as well as cause local extinction events, as Jones and Cheung (2015:741) explain:

“...indices of change across a set of 802 species of exploited marine fish and invertebrates. Results indicate an average poleward latitudinal shift across species and SDMs [species distribution models] at a rate of 15.5 and 25.6 km decade⁻¹ for a low and high emissions climate change scenario, respectively. Predicted distribution shifts resulted in hotspots of local invasion intensity in high latitude regions, while local extinctions were concentrated near the equator. Specifically, between 10°N and 10°S, we predicted that, on average, 6.5 species would become locally extinct per 0.5° latitude under the climate change emissions scenario Representative Concentration Pathway 8.5. Average invasions were predicted to

be 2.0 species per 0.5° latitude in the Arctic Ocean and 1.5 species per 0.5° latitude in the Southern Ocean.”

There has been a consistent research trend modelling the interactions between climate change and biodiversity, in particular, examining the possible ramifications climate change may have in store for biotic networks, refugia, and regional pools of species (Garcia et al., 2014). Some of this work has suggested that alterations in climate will have especially notable effects in tropical and sub-tropical regions:

“Local climate anomalies are projected to affect the tropics, subtropics, and northern high latitudes. More than half the global area currently covered by tropical climates faces large changes in average climate in relation to historical inter-annual variability.” (Garcia et al., 2014: 1247579-1)

The extent of these forecast changes could be quite dramatic, as Montoya and Raffaelli (2010) point out, effects on biodiversity will likely produce an entirely different biodiversity landscape and, as a consequence, ecosystems will likely change and function somewhat differently from those of the present affecting management practices and future delivery of ecosystem services. For example, where climate change interacts with land cover change, another major global stressor, impacts on the biodiversity landscape can be severe (Mantyka-Pringle et al., 2015:103):

“Risk analysis was used to estimate the risk of biodiversity loss due to alternative future land-cover change scenarios and to quantify how climate change mediates this risk. We demonstrate that the interaction of climate change with land-cover change could increase the impact of land-cover change on birds and mammals by up to 43% and 24% respectively and alter the spatial distribution of threats.”

The vision this research alludes to is further supported by a more recent assessment of the potential synergistic interaction between habitat loss and fragmentation and climate change in relation to future levels of biodiversity (Segan et al., 2016). In a nutshell, Segan et al., (2016) found that habitat loss and fragmentation act synergistically with climate change to increase biodiversity loss; in particular the authors noted that:

“...recent climate change is likely (probability>66%) to have exacerbated the impacts of HLF [habitat loss and fragmentation] in 120 (18.5%) ecoregions. Impacted ecoregions are disproportionately biodiverse, containing over half (54.1%) of all known terrestrial amphibian, bird, mammal, and reptile species.” (pg. 12)

The magnitude of such changes are thought to be easier to predict for ecosystem services because the same services can be potentially derived from different types of ecosystem, and ecosystem service provision is also dependent upon inherent aspects of ecosystem resilience (Côté and Darling, 2010; Montoya and Raffaelli, 2010). However, if ecosystems are dramatically affected delivery of services is unlikely to be sustained, even more so, ecosystems subject to high abiotic stress may fair less well (due to an increased sensitivity

to climate change) than those systems under less biotic stress (Montoya and Raffaelli, 2010). The degree to which specific biodiversity attributes are likely to be responsible for regulating ecosystem services supply is somewhat mixed (Montoya and Raffaelli, 2010). Some voices contend that community properties, such as species richness, are particularly important (Maestre et al., 2010), whilst others suggest that it is the biotic interactions that operate and affect ecosystem processes independently of climate change which are of most significance (Sarmiento et al., 2010; Yvon-Durocher et al., 2010). In reality, what occurs and how is likely to depend upon the nature of the ecosystem; this will dictate which over-arching components of the system will be ultimately responsible for sustaining service flows rather than some absolute universal rule.

In respect of ecosystem resilience and climate change, Côté and Darling (2010) argue against the current status quo – that ecosystem resilience to climate change can be increased by management strategies that reduce local stressors – instead, with explicit reference to coral reefs, they argue that such management actions could in fact result in increased ecosystem vulnerability to climate impacts by reducing disturbance-tolerant species – shifting the tipping point towards a threshold event (or so-called regime shift). Understanding concerning how multiple drivers will interact is limited and so defining potential future environmental scenarios resulting from multiple interacting drivers is not straightforward, particularly as driver effects are unlikely to be additive, and may therefore result in positive and negative feedbacks (Mooney et al., 2009; Montoya and Raffaelli, 2010), as Garcia et al., (2014: 1247579-7) point out:

“The actual effects of climate change on biodiversity are extremely difficult to predict owing to the complexity of species and community dynamics, in addition to the interaction with other stressors.”

It is also likely that certain drivers will have more significant and lasting impacts than others; for example, the effects of some human activities on ecosystem services may be greater than specific climate variables. Disentangling the degree of influence of particular drivers will be both necessary and difficult (Montoya and Raffaelli, 2010).

A number of future research agendas have been proposed to tackle the ambiguities that surround the current climate change-biodiversity-ecosystem services literature, we highlight some here:

- i. Synthesize the diverse range of observations concerning climate effects on biodiversity and ecosystem services into a coherent theory that enables accurate predictions concerning future scenarios to be made, preferably within a multilinked framework woven from metabolic theory, food web theory and biodiversity-ecosystem functioning theory (Mooney et al., 2009; Montoya and Raffaelli, 2010).

- ii. On a related theme, there needs to be an increase in the number of long-term ecological monitoring programmes, for example, building on the large-scale collaborative projects like the Global Observation Research Initiative in Alpine Environment (GLORIA) and the Role of Biodiversity in Climate Change and Mitigation (ROBIN) programme operating across Meso- and Latin America (Vihervaara et al., 2013).
- iii. Greater attention needs to be paid towards integrating mapping of stocks and flows at multiple scales, and in a marine context, exploring the sensitivity of projected changes in biodiversity along coastal regions to regional climate models (Montoya and Raffaelli, 2010; Jones and Cheung, 2015).
- iv. Models assessing the effects of climate change on biodiversity need to use, include, and accommodate more local-based environmental changes, for instance, climatic extremes or the timing of particular climatic events (Garcia et al., 2014).
- v. Focus greater attention towards adaptation strategies to cope with alterations in ecosystem service provisions, particularly through abatement of habitat loss, enhancement of ecosystem quality and heterogeneity, promoting landscape connectivity and restoration activities (Montoya and Raffaelli, 2010; Garcia et al., 2014).
- vi. Initiate a range of conservation, restoration and natural resource management programmes for the purposes of maximizing ecosystem services delivery, which also consider service trade-offs and build resilience to global change drivers (Montoya and Raffaelli, 2010).

2.6 Biodiversity And Ecosystem Service Research And Policy Developments

In addressing some of the issues we have covered in Sections 2.1 to 2.5 a number of recent national and international research and policy programmes have been developed. We highlight some examples of current initiatives below.

2.6.1 UK Level

As a research focused programme The National Environment Research Council (NERC) launched the Biodiversity and Ecosystem Services Sustainability (BESS)⁸ programme in 2011. A six year programme, the goal of BESS is to tease apart the fundamental interactions between biodiversity, ecosystem functions and the landscape scale delivery of ecosystem services across wetland, farmland, uplands and urban areas. The way BESS approaches this over-arching objective is to fund collaborative and cross-disciplinary consortia. Four programmes are currently funded: (i) F3UES⁹; (ii) CBESS¹⁰; (iii) Wessex BESS¹¹ and, (iv) DURESS¹² (NERC-BESS, 2015).

From a science-policy perspective, established in 2012 in the wake of the 2011 Natural Environment White Paper the Natural Capital Committee (NCC)¹³ was convened to provide independent expert advice to the UK Government on the state of England's natural capital. According to the NCC website it has a trio of foci:

“(i) provide advice on when, where and how natural assets are being used unsustainably; (ii) advise the Government on how it should prioritize action to protect and improve natural capital and, (iii) advise the Government on research priorities to improve future advice and decisions on protecting and enhancing natural capital” (NCC, 2015)

2.6.2 *EU Level*

The European Union (EU) contributes to a number of research and policy-oriented programmes. For example, OpenNESS¹⁴ established in 2012 as a five year programme proposes to capture and operationalize the concepts of natural capital and ecosystem services in a way that can provide testable practical solutions for “integrating ecosystem services into land, water and urban management and decision-making”, linking to wider European social, economic and environmental policy (OpenNESS, 2015). In a similar way, the OPERAs¹⁵ project aims to span the divide between ecosystem service science and practice by, for example:

“Exploring and validating mechanisms, instruments and best practices to maintain a sustainable flow of ecosystem services, while preserving ecological value and biological diversity” and “Improve existing decision-support tools and instruments to better capture and represent the concepts of ecosystem services” (OPERAs, 2015).

ALTER-Net¹⁶ is another example of a multi-partner pan-EU collaborative focused programme concerned with assimilating research capacity across Europe towards:

“...assessing changes in biodiversity, analyzing the effect of those changes on ecosystem services and informing policymakers and the public about this at a European scale” (ALTER-Net, 2015).

2.6.3 *Global Level*

At the global scale three initiatives stand out. The first is the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES)¹⁷. It has long been argued that an effective mechanism was required to periodically assess the state of the world's biodiversity and ecosystems in a full systematic, scientific, collaborative and independent manner, similar to (but different from) the Intergovernmental Panel on Climate Change (Larigauderie and Mooney, 2010; Vohland et al., 2011; Brooks et al., 2014). Established in 2012 IPBES is an independent intergovernmental body open to all member countries of the United Nations. The central purpose of IPBES is to assess the present (and potential future) state of the planet's biodiversity, ecosystems and the collective services they provide, by providing a robust mechanism agreed by science and policy communities to synthesis, review and

disseminate knowledge and information through expert assessments. Furthermore, it aims to strengthen the capacity of evidence-based policy and decision-making across all levels, and link to biodiversity-related multilateral agreements (Vohland and Nadim, 2015).

The second initiative is the Group on Earth Observations Biodiversity Observation Network (GEO BON)¹⁸. This represents one arm of nine GEO areas that will provide data and meta-data on biodiversity to the Global Earth Observation System of Systems (GEOSS), which will then act as a decision-support tool to academics, practitioners and policy makers etc. The mission statement of GEO BON is to acquire, coordinate and provide biodiversity information to key decision-makers and users, in particular, as a programme it is focused on:

“...the development of more integrated, efficient and interoperable biodiversity observation networks that can produce more reliable, accessible and timely observations to serve those needs [...] GEO BON is focusing its efforts on the implementation and adoption of the essential biodiversity variables and related monitoring guidelines and interoperable data management systems.” (GEO BON, 2016)

GEO BON has spawned a European level sub-initiative called EU BON¹⁹, comprising 18 countries and 30 partners, its purpose is to create an online open access platform for sharing biodiversity data and information integrating “social networks of science and policy and technological networks”, collating on-the-ground and remote sensing data to address policy and information needs in some cases in real-time. In particular EU BON has the following objectives:

“Advancing the technological/informatics infrastructures for GEO BON, by moving existing biodiversity networks towards standards-based, service-oriented approaches and cloud computing, enabling full interoperability through the GEOSS Common Infrastructure [...] Improving the range and quality of methods and tools for assessment, analysis, and visualization of biodiversity and ecosystem information, particularly focusing on predictive modelling, identification of drivers of change, and biodiversity indicators, and to support priority setting.” (EU BON, 2016)

The third recent development is Future Earth²⁰. Less about formal assessments and more about research developments, innovation and cross-collaborations Future Earth builds on pre-existing collaborative scientific global environmental change programmes²¹, and seeks to be an ‘international hub’ to promote and coordinate three central research themes, namely: dynamic planet²², global sustainable development²³ and transformations towards sustainability²⁴. Each of these themes is underpinned by a series of diverse research projects. Open to all disciplines Future Earth seeks to be ‘a platform for international engagement’ and innovation (Future Earth, 2015).

2.7 Final Remarks

Over the last two decades substantial progress has been made in elucidating the linkages between biodiversity, ecosystem functioning and ecosystem services, and the connections between these aspects and global environmental change issues such as climate change. These developments have gathered pace in recent years, delivering new levels of knowledge and understanding regarding the living connections in the Garden, and in particular, how those linkages connect with the notion of ecosystem services and the implementation of the ESF. These developments, in particular, can be seen with respect to national and international collaborative science-policy research projects and enterprises that in different ways, through transdisciplinary and multi-technological modalities, are helping to shed further light on biodiversity and ecosystem processes and connections. Ultimately improving our theoretical and practical understanding as well as supplying necessary information to key policy and decision-makers to facilitate the development of robust environmental policy strategies.

Notes

1. “Ecosystem structure refers to the distribution and arrangement of ecosystem components, which is subject to both physical and anthropogenic influences” (Fu et al., 2013a:4)
2. “Ecosystem composition refers to the comprising types and abundance of biotic and abiotic elements in a defined ecosystem” (Fu et al., 2013a:4)
3. “Ecosystem processes are the means to deliver ecosystem services; for example, pollination, soil formation and water regulation provide the services of food and potable water” (Fu et al., 2013a:4)
4. “Water movement is critical in delivering the services of potable water, irrigation and industrial production, as well as indirectly influencing food production, microclimate regulation, the esthetic view and more” (Fu et al., 2013a:6)
5. “Rational soil movement helps to consolidate soil fertility and improve water availability; moreover, it helps to eliminate silting in rivers and reservoirs off-site” (Fu et al., 2013a:6)
6. “Carbon cycling plays an important role in buffering the global temperature within a range tolerable for human beings. In addition, it indirectly influences other services such as biodiversity conservation” (Fu et al., 2013a:6)
7. According to Elmqvist et al., (2003:488) response diversity is:

“...the range of reactions to environmental change among species contributing to the same ecosystem function, and is critical to resilience, particularly during periods of ecosystem reorganization.”
8. www.nerc-bess.net
9. F3UES – Fragments, functions, flows and Urban Ecosystem Services has five main objectives, namely:

“(i) characterize the spatial ecological structure of urban areas; (ii) determine the influence of connectivity on biodiversity-ecosystem relationships; (iii) determine the flows of biodiversity and service delivery in selected cases; (iv) determine the impact of these flows on ecosystem delivery and, (v) Integrate these findings in the form of a spatially explicit model (NERC-BESS, 2015)

10. CBESS - A hierarchical approach to the examination of the relationship between biodiversity and ecosystem service flows across coastal margins (e.g., mudflats, saltmarshes) focusing on supporting, regulating, provisioning and cultural services (NERC-BESS, 2015)

11. Wessex BESS - Biodiversity and the provision of multiple ecosystem services in current and future lowland multifunctional landscapes. Focusing on four key service areas, namely: climate change mitigated by greenhouse gas regulation; water-related services of fisheries and clean water; crop production enhanced by pollination and pest control and cultural services relating to recreation and aesthetics, the project seeks to integrate “experiments with large-scale biodiversity and environmental gradients existing in the Wessex Chalk landscape” (NERC-BESS, 2015)

12. DURESS - Diversity in Upland Rivers for Ecosystem Service Sustainability. The purpose of the project is to:

“...focus on four examples of river ecosystem services: The regulation of water quality, the regulation of decomposition, fisheries and recreational fishing and river birds as culturally valued biodiversity. Using a range of spatial and temporal scales, the project will test the overarching hypothesis that 'biodiversity is central to the sustainable delivery of upland river ecosystem services under changing land-use and climate’” (NERC-BESS, 2015)

13. www.naturalcapitalcommittee.org

14. Operationalisation of natural capital and ecosystem services (www.openness-project.eu)

15. Operational Potential of Ecosystem Research Applications (www.operas-project.eu)

16. A Long-Term Biodiversity, Ecosystem and Awareness Research Network (www.alternet.info)

17. www.ipbes.net

18. <http://geobon.org/>

19. www.eubon.eu

20. www.futureearth.org

21. DIVERSITAS, the International Geosphere-Biosphere Programme (IGBP), the International Human Dimensions Programme (IHDP) and the World Climate Research Programme (WCRP).

22. “Research under the Dynamic Planet theme provides the knowledge required to understand observed and projected trends in the Earth system. This includes both natural and social components, interactions between them, and variations and extremes, both globally and regionally. It encompasses research that seeks to observe, monitor, explain and model the state of the planet and its societies.” (Future Earth, 2015)

23. “Research under the Global Sustainable Development theme provides the knowledge needed to understand the links between global environmental change and human well-being and development.” (Future Earth, 2015)

24. “The Transformations towards Sustainability theme goes beyond assessing and implementing current responses to global change and meeting gaps in development needs. It will consider the more fundamental and innovative long-term transformations that are needed to move towards a sustainable future.” (Future Earth, 2015)

Part 2: The Human Garden

In Part 2 we consider the social-ecological characteristics of our Post-Edenic Garden. We highlight, in particular, the “artificial” urban landscapes we have created for ourselves in our Human Genesis project, environments in which most of humanity now resides, by exploring and reflecting on the existence of *Homo urbanus*. We then go on to discuss how resilience and regime shift theory, and other associated concepts from social-ecological systems thought, is being applied to the social-ecological appraisal of ecosystem services and our Post-Edenic Garden.

Chapter 3: The Garden As A Social-Ecological System

“Mythologies across the globe have always pointed to the close connection in both origin, and continuity, between animals in nature and us; between the plants in the landscape and our spiritual and material lives [...] One of the challenges before us is to re-conceptualize the interactions between people and nature [...] and to think of human agents as organic parts of nature.” (Emilio F Moran, *People and Nature: An Introduction to Human Ecological Relations*, 2006, pg. 8)

A central challenge facing humanity is how to achieve sustainable outcomes that benefit both people and nature (1). Using a social-ecological systems (SESs) approach in the generation of knowledge and the formulation of sustainable governance solutions is critical, as it explicitly recognizes the connections and feedbacks linking human and natural systems. Understanding how the potential for social-ecological sustainability varies with context is vital to solving this dilemma.” (Leslie et al., 2015, pg. 5979)

In Chapter’s 1 and 2 we discussed the Garden from a largely “natural” perspective, describing its present state and condition as well as highlighting the interactions occurring between different aspects of life (i.e. biodiversity) and the generation and provision of ecosystem services. In Chapter 3 we recognize the fact that the Garden is not simply a ‘natural entity’ in which humanity resides in some distance and detached sense, where we enter and exit the stage leaving nothing behind; instead we recognize the opposite truth. The social systems we have created are enmeshed and embedded within the broader environment and biosphere at local through to global scales. That symbiosis is essential. Everywhere around us, our interactions are facilitated and mediated through social-ecological connections. In Chapter 3 we flesh out in a little more detail some of the latest thinking concerning the reality of the social-ecological connections that everywhere abound in the Garden. In particular we make a point of highlighting the Urban Gardens that humanity is creating for itself, and how, in our ‘anthropocene adventure’ more of us are living in these ‘artificial’ gardens than in the gardens from which we originated.

3.1 Describing The Whole

The vision of social-ecological systems (SES) has changed (Audouin et al., 2013). Increasingly SES are now regarded as dynamic and complex coupled human-nature arrangements composed of multiple subsystems interacting through multi-scalar linkages and feedbacks, which operate in response to various internal and external drivers producing new forms of behaviour (Lui et al., 2007; Carpenter et al., 2009; Ostrom, 2009). The recognition of SESs as dynamic complex adaptive systems (CAS) with non-linear responses and emergent behaviour has had important consequences for ecosystem management (Dawson et al., 2010),

inclusive decision-making processes (Ostrom, 2009; Dawson et al., 2010), policy formulation (Carpenter et al., 2012), the bridging of disciplinary divides to foster collaborative research and the creation of multi-layered adaptive institutions and governance arrangements (Burkhard et al., 2010).

SES frameworks addressing ecosystem services and natural resource management have undergone a marked proliferation over recent years. The hallmark of this radiative expansion has been a diversity in the design and focus of these frameworks, with some concerned to highlight the connections between ecosystem services, human-wellbeing and poverty alleviation (e.g. Fisher et al., 2013), whilst others have concentrated on characterizing the drivers and pressures that influence SES dynamics (e.g. Rounsevell et al., 2010; Villamanga et al., 2013) as well as shedding light on their network properties (e.g. Bodin and Tengo, 2012). In other cases, the purpose behind these frameworks has been to address the institutional and governance arrangements and contexts that dictate natural resource use and management (e.g. Duraiappah et al., 2013; McGinnis and Ostrom, 2014). Explicitly connecting governance and policy to environmental management strategy implementation, ecosystem service assessment and supply and demand functions has also been a productive area of development (e.g. Turner and Daily, 2008; Balmford et al., 2011; Bastian et al., 2013; Chapman, 2014). Against this backdrop, the pace of SES development has advanced rapidly in its application to marine, terrestrial, urban and social systems with respect to governance and management matters (Ostrom, 2009).

3.2 Dissecting The Whole

3.2.1 SES And Marine Systems

Some authors have adopted a driver-pressure perspective in how they approach the SES framework, in other words, linking socio-economic and socio-cultural driving forces that spur human activities with specific environmental pressures, for example, in modelling system responses to natural and human-induced disturbances in the Gulf of California (Leslie et al., 2009). Looking to solve food consumption and distribution problems, for instance in India, social-ecological perspectives have been employed to identify trade-offs between human security and fishery-resource allocation (Iwaski and Sahw, 2008). In some cases an historical framing has been used in the application of SES thinking, for example, in an analysis of the developments and transformations of the southern Benguela fishery in South Africa (Jarre et al., 2013). By teasing out the complex interrelations between the fishing industry, legislation and coastal demographics the authors revealed how these factors influenced local livelihoods and the composition of fish stocks over time (Jarre et al., 2013). Employing a similar strategy Whalley et al., (2011) demonstrated how the changing ecological conditions of two wetland

systems in the Murray-Darling Basin, Australia, was the result of waves of settlement, infrastructure and agricultural developments occurring in the region since the arrival of 19th Century European settlers.

Based on the notion that SES function through the complex interlinkages that occur between system components, Glaser et al., (2012) developed indicators to measure social-ecological connections that enhance sustainability processes between ecosystem services and resource-use in tropical marine systems. Similarly, Lopez-Angarita et al., (2014) argued that to manage and improve the ecological health and social conditions of two marine protected areas in Columbia required viewing them as linked social-ecological systems. Delivering improved ecological conditions in marine environments, particularly in the case of fisheries management, also requires a focus on promoting social-ecological resilience (Villasante, 2012). Emphasizing the social dimension of SES has provided avenues to improve the sustainability of coral reef resource-use and management (Kittinger et al., 2012).

Recently, Glaser and Glaeser (2014) proposed a global framework for the analysis and management of coastal marine social-ecological systems (CM-SES) based on an integrative approach. The authors argued that CM-SES assessments and policy formulations should proceed on the basis of multi-scale, multi-knowledge and multi-governance considerations (Glaser and Glaeser, 2014). At the same time they also identified the need to consider issues such as negotiating potential social divides, place-based cultural conditions, local rights and participation, social learning and knowledge systems (Glaser and Glaeser, 2014).

A practical invocation of these sentiments, the development of KnowSeas highlights the importance of collaborative research programmes, in this case involving 16 European partner countries and 30-plus institutions, as a means of providing a coherent evidence-base to inform the design and implementation of effective policy solutions concerning large-scale jurisdictional marine management issues (Mee et al., 2015). However, implementing a social-ecological approach to the management of marine resources is not always straightforward, undertaken robustly or guaranteed to produce only positive net gains. In their recent analysis, Stevenson and Tissot (2014) reviewed a number of studies that employed Ostrom's SES framework (see McGinnis and Ostrom, 2014) in a marine co-management context, establishing that whilst the framework provided useful insights it could also produce mixed outcomes depending upon how the framework was interpreted and implemented.

3.2.2 SES And Terrestrial Systems

In terrestrial systems SES thinking has been employed to scrutinize the problems of land degradation and sustainable community development in dry-land systems (Huber-Sannwald et al., 2012). Extending the sustainability narrative, in Tanzania, SES assessments

identified the development potential of small-scale water system innovations to positively influence agro-ecosystem dynamics (Enfors, 2013). In particular these assessments established how the complex relationships between climate, people and institutions determine the capacity of farming systems to adapt and utilize new methods of innovation (Enfors, 2013). Complex relationships have also been discovered between agriculture and aquaculture systems in rural Malawi (Blythe, 2013). All three cases point to the capacity of SES to draw on and tap into the multi-dimensionality of system dynamics, illustrating the power of SES to uncover “hidden” relationships by probing these systems from different angles.

From a biodiversity perspective, SES has been used in conjunction with scenario analysis to investigate how approaches to governance can filter-down and produce a range of landscape-scale ecological outcomes (Mitchell et al., 2015). The comprehensive picture that SES promotes has provided the theoretical basis for the establishment and evaluation of long-term socio-ecological research (LTSER) programmes in biodiversity conservation (Ohl et al., 2010). Tackling invasive species is also another feather in the cap of SES. In Hawaii, for example, an SES approach was used to evaluate the human and ecological factors contributing to the appearance of the Coqui frog and develop control strategies based on public involvement (Kalnicky et al., 2014). At the opposite end of the spectrum, social-ecological approaches have been used to identify both shortcomings and improvements in conservation interventions by assessing feedbacks between programmes and social outcomes (Miller et al., 2012).

In relation to forested environments, Filotas et al., (2014) have demonstrated that understanding the ecology of five global forest biomes is illuminated and enriched by acknowledging that the forests within these regions function as SES. By viewing forests through the CAS lens the authors argue that this provides a basis to develop:

“...holistic management approaches that improve the resilience and adaptive capacity of forests in uncertain times” (Filotas et al., 2014:17)

This view is further supported by Messier et al., (2015) who have recently proposed a set of seven principles¹ rooted in SES thinking to improve the adaptive capacity of forests in the face of future uncertainty.

3.2.3 SES And The Social Sphere

Just as extensive are the applications of SES to the social sphere. Prominent in this respect is the use of SES to assess and develop governance systems for managing common pool resources (e.g. Armitage 2008; Carlsson and Sandstrom, 2008; Cox, 2014; Frey and Berkes, 2014; Hinkel et al., 2015), engender social capital creation across multi-level institutions (e.g. Brondizio et al., 2009; Menzel and Buchecker, 2013), build and manage

resilience throughout the system (e.g. Booher and Innes, 2010; Stokols et al., 2013; Goulden et al., 2013) and address human environmental attitudes and perceptions (Fischer, 2010). A focus on eco-health, reducing inequities and hazards whilst maintaining resource flows has also been a feature of SES when adapted to watershed management (Bunch et al., 2010). Generic evaluations of wellbeing and governance matters related to the conceptual foundations of social theory and its interaction with SES have also emerged recently (Armitage et al., 2012). Finally, the impact of environmental education, individual learning and adaptive capacities on improved environmental management and institutional functioning has been viewed through the SES lens (e.g., Krasny and Roth, 2010; Krasny et al., 2010a; Krasny et al., 2010b; Plummer 2010).

3.2.4 Technical Developments And SES

From a technical perspective spatial mapping procedures have enlisted SES to identify similarities between human perceived landscape values and directly measured ecological values in order to generate so-called socio-ecological hotspots (Alessa et al., 2008). Mapping has highlighted spatial relationships between biodiversity and watershed-specific ecosystem services provision (Bai et al., 2011). Further modelling and decision-support tools have been developed that integrate SES with software that can represent the dynamics between humans, ecological processes and governance systems, enabling the quantification of trade-offs and facilitating local and regional planning and management decisions (e.g. Summers et al., 2015).

Agent-based models (ABMs) have been increasingly employed to reveal the intricacies and dynamics of SES through altered stakeholder behavioural responses to differing scenarios (e.g. Milner-Gulland, 2012; Rounsevell et al., 2012). Others have developed ABMs to understand and examine the complex interactions governing indigenous community relationships between subsistence agriculture and hunting as a way of assessing the future sustainability of the system (Iwamura et al., 2014). Progressively ABMs are also being employed to examine how agents learn in response to system dynamics, for example, Bohensky (2014) explored 'learning dilemmas' in the water sector in South Africa: Combining human perception with social-ecological elements he showed that the capacity of agents to learn is affected by system predictability, and importantly, when systems are variable agents are more likely to use novel water management strategies leading to an enrichment of their learning experience. Progress in the development and application of ABMs has been substantial over recent years, nevertheless, as some authors point out further improvements are required for ABM to reach its potential (Filatova et al., (2013:1):

“(1) design and parameterizing of agent decision models, (2) verification, validation and sensitivity analysis, (3) integration of socio-demographic, ecological, and biophysical models, and (4) spatial representation.”

3.3 A Digression Into The Life Of *Homo Urbanus*

“The understanding of how urban ecosystems function, provide goods and services for urban dwellers; and how they change and what allows and limits their performance can add to the understanding of ecosystem change and governance in general in an ever more human-dominated world.” (Hasse et al., 2014a, pg. 407)

The majority of the world’s population now lives in urban³ areas – 54% according to the latest estimates (UN-HABITAT, 2016). This is a radical step change in the way we live and conduct our lives and the relationship we have with the rest of the biosphere, it is certainly an experience that is light years away from our hunter-gather existence, as Vince (2014:340) captures with typically elegant bravura:

“The Anthropocene is the urban age. Our species is undergoing the biggest migration in human history – already more than half of us live in cities; by 2050, around 7 billion of us will. We have become *Homo urbanus* – a different creature, a faster-thinking, more reactive, more genetically diverse human. Human history is increasingly urban history.”

The rate at which urban centres have grown and developed in recent decades, in very different patterns across the world, is on an unprecedented scale⁴ (Andersson et al., 2014). By mid-century, for example, it is expected that almost three quarters of China’s population will be urban residents with India following close behind (UNEP, 2012). Urbanization is a multifaceted process⁵ and parallels the increasing size and scale of urban population growth (McDonald et al., 2013; Fragkias et al., 2013). In a relatively short space of time urban decision-making and urban perspectives on the world have come to determine how humanity affects the Earth System:

“The Earth System has become urbanized in the sense that decisions by the majority of the human population now living in cities affect the resilience of the entire planet.” (Andersson et al., 2014, pg. 445)

Whilst this point of view is mirrored by Elmqvist et al., (2015), they also go on to suggest that this also presents an important set of new opportunities:

“We are entering a new urban era in which the ecology of the planet as a whole is increasingly influenced by human activities, with cities as crucial centers of demand for ecosystem services and sources of environmental impacts. Approximately 60% of the urban land expected to exist 2030 is forecast to be built in 2000–2030. Urbanization therefore presents fundamental challenges but also unprecedented opportunities to enhance the resilience and ecological functioning of urban systems.” (pg. 101)

Taking advantage of these new opportunities, however, means overcoming a number of hurdles along the way: The escalation in urban developments and cities across the globe presents significant social, economic, cultural, political and environmental challenges, particularly in terms of governance, equality of access to public services, poverty, security,

adequacy of human settlements and housing, climate change and the consumption of resources such as water (Sachs, 2015; UN-HABITAT, 2016:5):

“Environmentally, the current model of urbanization engenders low-density suburbanization— largely steered by private, rather than public interest, and partly facilitated by dependence on car ownership; it is energy-intensive and contributes dangerously to climate change. Socially, the model of urbanization generates multiple forms of inequality, exclusion and deprivation, which creates spatial inequalities and divided cities, often characterized by gated communities and slum areas.”

Poverty, for example, is at its starkest in cities, the very nature of how cities are constructed and governed seems to precipitate those extreme aspects of human existence:

“Often poverty is most pronounced in urban centres that are experiencing the most dramatic improvements in prosperity, such as Nairobi and Mumbai. Mumbai, which is on rack to become the world’s biggest city, is home to around 9 million slim-dwellers – more than half the population – many of whom live in the shadow of some of the world’s most expensive new apartments. By 2025, the World Bank estimates that 22.5 million people will be living in Mumbai slums. Even in the heart of Europe slums exist, such as Canada Real Galiana on Madrid’s borders, where poverty is rampant and infrastructure as poor as in any developing-world shanty town. Nowhere on Earth is wealth disparity so obvious as in a city.” (Vince, 2014:344)

Although no simple relationship exists between the state of the environment and urbanization, the evidence indicates that since the 1950s the continued and unparalleled advance of urbanization has been concomitant with widespread environmental degradation and increased demand and consumption of natural resources (MacDonald et al., 2013). It is also clear that urban ecological systems, those environments that are internal to the urban enterprise in particular, face a unique set of pressures determined by their situation within a socially constructed arena (Haase et al., 2014a). Most current and future urbanization is thought likely to occur in low to middle-income regions where there is limited ‘economic’ and ‘industrial’ development, and where constraints on investments in environmental protection, conservation and provision of ecosystem services will be greatest⁶ (Seto et al., 2013).

Indeed, the lion’s share of the most recent urban growth has occurred in regions closes to biodiversity hotspots and biodiversity rich coastal areas (Seto, et al., 2013). The rise of the city and urban centres drives demand for ecosystem services and parallels increasing impacts on biophysical and ecological processes (McDonald et al., 2013). The biodiversity impacts of urbanization are not the same everywhere, they differ geographically and across scales according to the types of urban developments, but most of the severe effects on biodiversity result from land-use changes (e.g. habitat fragmentation), effects of pollution (e.g. soil, air and water), climatic changes (e.g. heat island effect and changes in precipitation patterns), biological invasions, and resource consumption (Fragkias et al., 2013; MacDonald et al., 2013;

Müller et al., 2013). Urbanization and associated urban sprawl, for example, poses significant threats to amphibians (Kruger et al., 2015); urban developments can impact on arthropod communities in nearby mosaic landscapes (Rocha-Ortega and Castaño-Meneses, 2015) and they can also affect fundamental nutrient cycling processes (Enloe et al., 2015). Cities also present a substantial climate change challenge as they account for the majority of global energy consumption and greenhouse gas emissions (Ziter, 2015).

The resilience of cities⁷ are also affected by where and how they develop, for example, this can determine their vulnerability to natural disasters⁸, the resource required to provide municipal services (e.g., water supply and basic infrastructure), how they are able to feed themselves and secure food supplies both from domestic agricultural production but also through international trade networks, their social cohesion and the way they are able to deal with combined food-energy-water issues (Deutsch et al., 2013; Fragkias et al., 2013; Güneralp et al., 2013).

On the flip-side urbanization has brought a number of important and profound benefits, particularly from a human perspective, for example, in terms of increased economic growth and development, employment, production, access to essential services and in many cases enhanced quality of life (UN-HABITAT, 2016:27)⁹:

“Urban areas offer significant opportunities for both formal and informal employment, generating a sizeable share of new private sector jobs [...] Urbanization has helped millions escape poverty through increased productivity, employment opportunities, improved quality of life and large-scale investment in infrastructure and services.”

Vince (2014) offers us a personal perspective from her travels in South America on why cities are a powerful draw and why they also need to be recognized for the benefits they provide:

“Already, 75% of Latin Americans live in cities, by 2050 92% of South Americans will do, driven there by environmental degradation of their rural lands, conflict and lack of employment. And hope. However grim it looks here for these shantytown dwellers, in many ways it is far better than what they have left. Employment opportunities, income levels and access to services from markets to health care are all better than in rural areas.” (pg. 349)

Cities offer a platform for innovation, they can provide sustainable mobility which reduces social, economic and environmental stresses on infrastructure and services, increasing the efficiency with which people are able to connect, access goods and services as well as jobs. In this way cities are also able to catalyze innovative technological developments that improve their overall environmental sustainability such as in renewable energy, energy storage and distribution and decarbonization (UN-HABITAT, 2016). Often these rich developments are the result of the density-dependent effects cities have, which improve their productivity,

efficiency and influence¹⁰ (Vince, 2014). Developed urban areas can also produce significant levels of ecosystem service benefits for their residents, for instance, in terms of micro-climate regulation, health benefits, water regulation, pollution reduction, habitat and cultural services (Elmqvist et al., 2015). Some of these services may derive from novel systems, as Andersson et al., (2014:446) explain:

“Cities are rife with “novel ecosystems”, which deserve to be acknowledged for the values they possess in terms of biodiversity and ecosystem services. Comprehensive analyses of urban green spaces have shown that land uses such as private and public gardens, cemeteries, old brown-fields, and golf courses may contribute significantly to ecosystem services provided by the urban landscape.”

The provision of these services is regarded as having prominent human-wellbeing outcomes, as Ziter (2015:EV1) describes:

“Natural spaces and green infrastructure within urban areas provide citizens with places to recreate, and increase aesthetics, for example, while urban agriculture provides residents with local food. This local provision of ES to urban occupants is an important factor in how functional and enjoyable a city is to live in.”

The environmental challenges that urban areas need to continue to meet, to deal with and to solve revolve around how to provide and supply a sustainable flow of a full range of public services in an equitable way¹¹, how to mitigate environmental risks, how to reduce the deleterious effects of land transformation on biodiversity and ecosystem services, and how best to continue to invest in and pursue energy efficiency and decarbonization technologies (UN-HABITAT, 2016). A powerful case for helping to achieve some of these aims via the restoration of urban ecosystem services is made by Elmqvist et al., (2015:101):

“Investing in urban green and blue infrastructure constitutes a tangible contribution that cities can make to the United Nations’ agenda on a Green Economy for the 21st century and the Sustainable Development Goals (SDGs).”

As the day to day experience of most people is the “urban jungle” SES and CAS have been increasingly co-opted as tools to deliver smart sustainable cities, urban green spaces¹² and a built environment that works for people and nature (Seto et al., 2013). How has this been achieved? Primarily, by catalyzing the alignment of conservation and urban development, and focusing on incorporating novelty and innovation in the creation of green infrastructure and the delivery of urban services (Anderson and Elmqvist, 2012; Müller et al., 2013). The application of SES to urban settings spans spatial scales (e.g. Colding, 2013; Kronenberg et al., 2013; Nagendra et al., 2013; Seto, 2013) and much of it has focused on how to: (i) include and maximize biodiversity; (ii) generate ecosystem service bundles through multi-functional green infrastructure and minimize trade-offs; (iii) enhance service provision and distribution to foster human-wellbeing; (iv) restore and regenerate urban ecosystem services and associated infrastructure; (v) incorporate green thinking into the built environment to create more

“natural” settings and improve energy efficiency; (vi) incorporate adaptive design that can respond to specific local challenge; (vii) involve local stakeholders in decision-making, planning and policy and; (viii) evaluate the influence of political and institutional governance settings on urban functionality (Ahern, 2013; Lovell and Taylor, 2013; Elmqvist et al., 2013; Schewenius et al., 2014; Haase et al., 2014b, Hansen and Pauliet, 2014; Elmqvist et al., 2015)¹³.

Central to the challenges and issues facing urban social-ecological systems in achieving sustainable, resilient and equitable cities reside are governance and planning arrangements and how they interact, operate and are implemented (UN-HABITAT, 2016). Where there is poor governance then problems follow, for example, as noted in a recent urban case study in Poland social institutional failures, particularly in terms of empowerment, mobilization and funding can be detrimental to achieving urban greening and the development of an urban ecosystem service infrastructure (Kronenberg, 2015). Remedies for these kinds of failures may be found by confronting the seven urban governance challenges outlined by Wilkinson et al., (2013), namely: political and intellectual legitimacy; integrating environment equity and justice; improving institutional capacity and governance effectiveness; navigating competing urban priorities; reducing the mismatches between the scale of governance and the scale at which urban ecosystem service provision and delivery occur; negotiating trade-offs between urban development, biodiversity and ecosystem services and enhancing cooperation and collaboration in urban ecosystem management. Meeting these challenges they conclude will be best served by adopting several tools and approaches, in particular: urban design; the regulation of land-use; employing appropriate planning tools; using the correct financial and economic instruments and applying a broad range of integrative management principles (Wilkinson et al., 2013). In addition, increasing cross-sector partnerships and technological driven initiatives within an open and inclusive system of governance have also been advocated as a mechanism to improve urban social-ecological governance regimes (UN-HABITAT, 2016).

Ultimately, the way *Homo urbanus* progresses, the way our cities and urban lives develop and flourish will take a coordinated effort across-sectors and across governments, involving all people in decision-making processes, so that we all have a stake in the decisions that will affect the places in which we live, so that we all have the capacity to contribute to the future destination of our urban existences, so that we recognize them as social-ecological systems: and therefore understand that we need to incorporate nature within our cities, that it is fundamental for the full flourishing of our urban selves, and that we recognize and acknowledge both the debt we owe to the biosphere outside the borders of our cities in providing the resource-base and the platform necessary for them to grow, but that simultaneously we take into account the very real and potential negative environmental

impacts cities can have as they grow and develop, and that as a consequence we must strive for a future of smart and sustainable urban biomes, as Vince (2014:345) speculates:

“The urban revolution of the Anthropocene could prove the solution to many of the environmental and social problems of our age, allowing humans to inhabit the planet in vast numbers but in the most sustainable way. Or, it could finally prove our species’ undoing, the apocalyptic version of the dystopian megacity so often portrayed in science fiction.”

3.4 Final Remarks

What is evident from this brief survey is that by introducing the notion of social-ecological thinking, by providing a social-ecological perspective on our lives, that the Garden we inhabit is a far more complex one than we may have initially recognized. We are deeply integrated with the world and the environment around us, even as *Homo urbanus*, the “artificial” gardens we have created for ourselves, quite rapidly in recent decades, and in which the majority of humanity now resides are not detached and somehow separate. Even here, our social and cultural systems are fundamentally concatenated with the environmental resource-base that ultimately continues to support and sustain us. By providing an overview of social-ecological research we have highlighted how adopting such a perspective can illuminate our understanding of the complex problems we currently encounter, but more than that, that by doing so it also provides us with a mechanism, a remedy, for navigating those complex issues and ultimately providing a means to solve them.

Notes

1. According to Messier et al., (2015:373-375) the seven principles are as follows:

- Principle 1: “Replace the sustained single good or objective-yield paradigm with one that integrates risk/flexibility/adaptability into scenarios of sustained provision of various goods and services.”
- Principle 2 “Consider the taxonomic and functional diversity (i.e., range of ecological functions that organisms support in communities and ecosystems) of the tree species pool in terms of its ability to maintain a balance between diversity and redundancy and provide desired ecosystem goods and services in an ever-changing biological and social environment.”
- Principle 3 “Promote an optimal balance among modularity (i.e., the extent to which a system can be divided into independent units) and connectivity at multiple scales.”
- Principle 4 “Plan and assess interventions across a range of spatial and temporal scales, e.g., from plant neighbourhoods to landscapes.”
- Principle 5 “Plan and develop long-term scenarios using new analytical tools and models that specifically acknowledge the prevalence of highly uncertain social, economic, climatic, and ecological conditions.”
- Principle 6 “Increase involvement of local communities and other stakeholders to ensure that future forests are better aligned with the needs and preferences of local people.”
- Principle 7 “Allow social–environmental systems to self-organize and adapt to novel biological, environmental, and social conditions.”

2. Developed against the backdrop of complex system science, evolutionary programming and computational sociology agent-based models (ABM) are computational models designed to simulate the behaviour and interactions of autonomous agents (individuals or groups) for the purposes of

evaluating how they affect the properties of the whole system. Through the interactions and operations of individual agents ABMs seek to artificially recreate and forecast the emergence of complex phenomena. As Macal and North (2010:151) describe:

“By modelling systems from the ‘ground up’—agent-by-agent and interaction-by-interaction—self-organization can often be observed in such models. Patterns, structures, and behaviours emerge that were not explicitly programmed into the models, but arise through the agent interactions.”

3. What constitutes ‘urban’ is to some extent open to debate, and classifications of what is ‘urban’ and ‘rural’ (i.e. where does rural stop and urban begin and vice versa) differ quite considerably among countries and continents (Seto et al., 2013).

4. For example, according to the latest UN-HABITAT World Cities Report (2016:6):

“Since 1990, the world has seen an increased gathering of its population in urban areas. This trend is not new, but relentless and has been marked by a remarkable increase in the absolute numbers of urban dwellers—from a yearly average of 57 million between 1990-2000 to 77million between 2010-2015. In 1990, 43 per cent (2.3 billion) of the world’s population lived in urban areas; by 2015, this had grown to 54 per cent (4 billion). The increase in urban population has not been evenly spread throughout the world. Different regions have seen their urban populations grow more quickly, or less quickly, although virtually no region of the world can report a decrease in urbanization.”

Underlining the point that urban growth and development is set to continue at record pace; Vince (2014:340) charts this projected rise:

“A million-person city will be built every ten days over the next eighty years. There are currently around thirty mega-cities on the planet [i.e., cities of 10 million people or more], and by 2050 they are expected to merge into dozens of megaregions, like Honk Kong-Shenzhen-Guangshou in China, where more than 100 million people will live in a seemingly endless city. The Tokyo metropolis, Japan’s national capital region, already hosts 36.7 million people with a population density more than double that of Bangladesh, and the largest metropolitan economy in the world.”

5. Urbanization is regarded as a dynamic, multi-scaled process operating across time and space driven by technological innovations in communication and transport, efficiency gains and economies of scale from high density urban populations. From this perspective urbanization is clearly associated with changes in human population structure (e.g., reduction in fertility as people delay getting married and having families and also have fewer children), economic development and environmental transformation (MacDonald et al., 2013; Seto et al., 2013)

6. For example, according to Vince (2014:340):

“The majority urbanization in the coming decades will consist of poor people in Africa and Asia migrating from rural areas for paid work. Almost all of them will live in slums at densities as great as 2,500 people per hectare, sharing as many toilets as the average American family home.”

Picking up on Vince’s point regarding the destination of many of these rural migrants, the growth in slum areas remains a serious and challenging issue, particularly for poor and rapidly urbanizing regions, but at the same time the absolute number of people living in slum areas has decreased, although much progress remains to be made:

“Recent estimates provided by UN-Habitat show that the proportion of the urban population living in slums in the developing world decreased from 46.2 per cent in 1990, 39.4 per cent in 2000, to 32.6 per cent in 2010 and to 29.7 per cent in 2014. However, estimates also show that the number of slum dwellers in the developing world is on the increase given that over 880 million residents lived in slums in 2014, compared to 791 million in 2000, and 689 million in 1990.⁵⁸ This implies that there is still a long way to go in many countries, in order to reduce the large gap between slum dwellers and the rest of the urban population living in adequate shelter with access to basic services.” (UN-HABIAT, 2016:14)

7. Recommendations such as linking climate change adaptation and disaster risk reduction strategies in an integrated framework within a planning setting have been advanced as a means to improve the resilience of cities (UN-HABITAT, 2016).

8. According to UN-HABITAT (2016:87):

“Globally, the number of natural disasters is increasing in both intensity and frequency (4,000 between 2003 and 2012, compared with 82 in 1901-1910). Natural disasters are particularly detrimental to the urban poor and their recognized human rights to decent living conditions, since unplanned urbanization and inadequate infrastructure leave them more exposed than the rest of the population.”

9. As another section of the report substantiates:

“The transformative role urbanization can play in environmental sustainability has been increasingly recognized. When well-planned and managed, urbanization, together with building design and transport modalities, provides a welcome opportunity to devise resilience strategies, in the process reducing resource use, entrenching incremental development gains and managing vulnerability vis-a-vis all plausible hazards.” (pg. 87)

10. As Vince (2014:345) explains:

“If the population of a city is doubled, average wages go up by 15%, as do other measures of productivity, like patents per capita. Economic output of a city of 10 million people will be 15-20% higher than that of two cities of 5 million people. Incomes are on average five times higher in urbanized areas in countries with a largely rural population. And at the same time, resource use and carbon emissions plummet by 15% for every doubling in density, because of more efficient use of infrastructure and better use of public transportation.”

11. As the UN-HABITAT (2016:89) report makes clear:

“Cities must ensure universal access to basic services like water, sanitation, waste management, energy, food, and mobility, which are crucial to socioeconomic welfare, public health and the urban environment.”

Yet in many regions gross inequalities in services persist, many of which have substantial and detrimental human welfare implications, as the report goes on to illustrate:

“In Africa as a whole, the average urban sanitation rate stood at 54 per cent in 2010, with diseases like cholera still plaguing urban areas. Similarly, in Sub-Saharan Africa electricity was available to only 32 per cent of the urban population in 2011, with power shortages in at least 30 countries. In the Latin America-Caribbean (LA C) region, overall proportions are comparatively higher but access to basic services remains inequitable: in 2010, over 20 per cent of the urban population still had no access to improved sanitation, 6 per cent lacked access to safe water and 7 per cent to electricity.”

12. In their systematic review of the urban green space literature Hunter and Luck (2015) make the point that although there is general agreement over what is considered urban green space (e.g. parks, gardens, campuses, green roofs etc.) there is less agreement on how urban green spaces should be conceptualised and defined. Principally, this stems from the fact that they are not homogeneous spaces in size, composition, or in their capacity to provide ecosystem services or indeed to the extent that they function fully as social-ecological systems. Thus Hunter and Luck make the point that urban green spaces should be characterised in terms of a set of metrics that assess their qualities, both social and ecological, in an explicit way. However, Müller et al., (2013) make the point that there is a much richer urban biodiversity typology that perhaps goes beyond discussions of urban green space per se, but includes agricultural landscapes, urban-industrial landscapes and remnant vegetation as well as more traditionally considered urban green spaces such as ornamental gardens and landscapes.

13. Urban ecosystem service research has largely focused on issues of modelling (e.g., urban ecosystem health, system pressures and their influence on ecosystem service provision), governance (e.g. organizational structures, management and social networks and their influence

on the processes underpinning the governance of urban green infrastructure and services), tools (e.g. practical, physical and modelling tools to create, inform and examine the maintenance and development of urban green space), economics (e.g. valuation studies of urban ecosystem service demand, supply and provision) and social aspects (e.g. importance of social perception, behaviours and psychology in the interactions with urban green space, infrastructure and ecosystem services) (Hubacek and Kronenberg, 2013).

In a recent review of the literature (Luderitz et al., 2015) it was found that most studies (50%) concerning urban ecosystem services were conducted in China and the USA, with the rest predominantly from European countries. In the main these studies took an ecological (35%) and planning (20%) perspective, with less than 10% focusing on governance. When viewed through the lens of the ecosystem services cascade model almost a third of studies considered only one component of that model with an additional 10% failing to assess any component. Equally, however, some 45% of studies did examine connections between two cascade components, but only 14% assessed three or more components. The majority of studies (48%) focused on a single ecological structure, primarily forests, rivers and agricultural land, and most examined ecosystem services were assessed in relation to these ecological structures. Where governance issues were investigated the predominant perspective taken was one associated with a traditional top-down and centralized form. Overall, most studies lacked full engagement with all aspects of the ecosystem services “production chain”, and thus in many cases were unable to develop a full appreciation of how best to sustainably manage urban ecosystem services within an integrated urban social-ecological framework.

Chapter 4: The Stresses And Strains In Social-Ecological Systems

“Tipping points are so dangerous because if you pass them, the climate is out of humanity's control: if an ice sheet disintegrates and starts to slide into the ocean there's nothing we can do about that.” (James Hansen, Newsweek, 2009)

“Some experts look at global warming, increased world temperature, as the critical tipping point that is causing a crash in coral reef health around the world. And there's no question that it is a factor, but it's preceded by the loss of resilience and degradation.” (Sylvia Earle)

In the previous chapter we introduced the idea – and explicitly recognized – that the Garden(s) in which we live are complex social-ecological systems. The first reason for introducing this perspective is that throughout the history of our species whatever biomes, ecosystems and habitats we have resided in and been a part of we have always engaged with nature, with how those natural systems function and operate, and we have always been, in our own right, a keystone species and an ecosystem engineer. The second reason is to acknowledge that, at the same time, since we transitioned from hunter-gatherer to agricultural farmer and moved out of our “natural” home humanity is, nevertheless, and very ironically, having a greater impact on those environments, on those complex systems, than we ever did when we were – to our minds – more integrated within them.

Yet this perspective somewhat belies the reality that, although on the surface we appeared to distance ourselves from the natural world when we developed settled cultures, societies, cities and nations, and for most of us became *Homo urbanus*, in essence, we never really extricated ourselves from nature: in many ways we have become more reliant on it just at the moments where we feel more distant from it. We fashioned it [nature that is] and honed it to such an extent that we took for granted, we forgot, that it supported our own species' development and our achievements. It is only latterly that we have begun to recognize that we live in a complex and interconnected set of social-ecological systems that operate across scales, across space and across time.

In Chapter 4 we recognize that we operate within multiple overlapping social-ecological systems, and in some sense we focus our view more narrowly, delving deeper into how our actions in the Anthropocene are affecting those social-ecological systems and what this means for us and the biosphere moving forwards. We do so by focusing attention on two key ideas that have become, in many respects, the dominant framings of how we think about social-ecological systems, namely: resilience and regime shifts. These two concepts have been

integral to how we have investigated social-ecological systems and how we understand them to operate, and in this chapter we briefly foray into this research.

4.1 The Resilience Of Social-Ecological Systems

The Resilience¹ concept maybe 40 years old but it remains a diverse and hotly debated concept bound by discipline and context (Biggs et al., 2012). Often two broad types of resilience are distinguished: ecological resilience with its focus on system recovery times and the maintenance of function following disturbance in ecosystems (e.g. Folke, 2006; Webb, 2007; Brand, 2009; Fleischman et al., 2010; Miller et al., 2010) and socio-cultural and community resilience with their emphasis on governance, institutions, social learning, social memory, perceptions and reflexivity (e.g. Brondizio et al., 2009; Crane, 2010; Davidson, 2010; Magis, 2010; Plummer, 2010; Michel-Kerjan, 2015). With respect to ecological resilience in particular, but also to some extent social resilience too, there is an important adjunct to this debate which concerns a number of related concepts such as resistance, vulnerability and adaptive capacity – and it is perhaps just worth pondering on these for a moment (e.g. Folke et al., 2005; Eakin and Luers, 2006; Young et al., 2006; Miller et al., 2010; Connell and Ghedini, 2015; Hodgson et al., 2015; Nimmo et al., 2015).

It has recently been argued that a better comprehension and improved application of resilience theory to ecological, environmental and conservation issues would be achieved by recognizing and differentiating resilience into resistance and recovery components (Connell and Ghedini, 2015; Hodgson et al., 2015; Nimmo et al., 2015). For example, Connell and Ghedini (2015) advance that there needs to be a refocusing on the theory of “ecological compensation” or “resistance” and a greater assessment of the stabilizing processes at work in mitigating perturbations:

“Emerging research suggests that there are processes acting well in advance of processes of recovery; such early responses adjust the dynamics of a system to counter the otherwise unchecked effects of disturbance. Such compensatory effects powerfully underpin the resistance of communities to change and maintain overall stability by gaining in strength with increasing intensity of disturbance.” (pg. 513)

Strengthening their initial argument, they go on to say that:

“We propose broadening the ideas of compensatory dynamics to include adjustments in the strength of pre-existing processes to absorb the effects of disturbance. If we move beyond the current focus on changes in biodiversity, then there is scope to understand how compensatory dynamics can maintain community stability without major loss or change in biodiversity, in other words without community restructure. Such stability, therefore, requires pre-existing processes to change in strength in proportion to the effect of disturbance. Thereby, compensatory processes act as inconspicuous mechanisms that counter

the effects of disturbance, and can explain why observed community shifts are smaller than expected after disturbance events.” (pg. 514)

Extending this line of argument, and making the case for the resistance-recovery duality of resilience, Hodgson et al., (2015) argue that this paves the way for a more theoretically robust understanding of ecological systems based on a systematized and standardized set of measurements, where assessing a systems “change in state” and “return time” can be properly examined and compared by revealing a systems “resilience space”:

“We recommend the adoption of bivariate measurement and analysis of ‘change in state’ and ‘return time’. This approach will help to determine which natural systems are more resilient than others, but will force us to consider whether resilience is achieved via resistance or recovery. It will help to guide the management of natural systems.” (pg. 505)

In practical terms Nimmo et al., (2015) regard making the distinction between resistance and resilience as a progressive step because, they argue, combining these two qualities together into a resistance-resilience framework will, by reconfiguring the lens through which ecologists examine ecosystems, provide a number of biodiversity conservation benefits, for example: it will enable the determination of the internal properties of ecosystems and their state to perturbations and disturbances; allow longer-term ecological predictions and projections to be made regarding how those ecosystems will operate under different future scenarios, and therefore also be especially useful to policy makers, as they explain:

“By adopting a ‘resistance–resilience’ framework, important insights for conservation can be gained into: (i) the key role of resistance in response to persistent disturbance, (ii) the intrinsic attributes of an ecological unit associated with resistance and resilience, (iii) the extrinsic environmental factors that influence resistance and resilience, (iv) mechanisms that confer resistance and resilience, (v) the post-disturbance status of an ecological unit, (vi) the nature of long-term ecological changes, and (vii) policy-relevant ways of communicating the ecological impacts of disturbance processes.

As they go onto explain with reference to the example of invasive biology:

“Identifying the determinants of resistance in the face of a persistent disturbance is particularly valuable for conservation management. For instance, invasion ecology has provided many ideas about the ability of biotic communities to ‘resist’ ongoing species invasion. For example, biotic communities with diverse functional groups have been shown to be more resistant to the spread of invasive species.”

It would be wrong to suggest, however, that there is complete agreement across the spectrum on the terminology applied to notions of resistance. What constitutes resistance is not entirely fixed, in other words, there are nuances of meaning and of emphasis: For some, resistance is understood to refer to: “...the capacity of a system to reorganise and return to a prior state” (Connell and Ghedini, 2015:513), whilst Hodgson et al., (2015:503) decompose resilience into two distinct components – resistance and recovery – where resistance denotes:

“the instantaneous impact of exogenous disturbance on system state” and recovery: “captures the endogenous processes that pull the disturbed system back towards an equilibrium”, meanwhile for Nimmo et al., (2015:516) resistance: “is the ability to persist during the disturbance”. Taken together, however, whether regarded as a subcomponent of resilience or a property sitting alongside resilience, they all acknowledge that resistance is a stabilizing quality in the face of a disturbance or perturbation.

Whilst there has been a growing recognition that ecological resilience is a broad church composed of a number of different facets, so there has also been a growth in the articulation of resilience as a form of adaptive capacity, a notion particularly advanced in terms of social-ecological governance (Karpouzoglou et al., 2016:1):

“...the notion of adaptive governance brings attention to how social-ecological systems can adapt to constantly changing conditions, especially where decisions need to be taken under high uncertainty. Adaptive governance is in line with the emergence of new modes of governing in which multiple actors are involved, interactions within and across state, private sector and civil society are key and decisions require action across multiple scales and levels.”

The growth and emergence of this concept springs from the recognition that social capital and social memory are important properties of social-ecological systems, which having strong connections to institutional and organizational facets of governance can, if properly promoted, increase levels of resilience within the broader system and thereby link to notions of adaptability (e.g. Folke et al., 2005; Uzawa, 2005; Brondizio et al., 2009; Ishihara and Pascual, 2009; Nkhata et al., 2009). It has been argued for example that the ability to marshal social capital and social memory in response to disturbances allows transformability, that is, the capacity to generate a new form of governance with an enhanced capability to manage dynamic ecosystems (Dietz et al., 2003; Olsson et al., 2004; Walker et al., 2004; Folke et al., 2005; Folke, 2006).

Frequently, adaptive capacity and resilience are discussed in relation to vulnerability (Adger, 2006). In part this is because vulnerability has often been viewed through the lens of adaptation, in the sense that being able to build and enhance system resilience eschews or reduces vulnerabilities (Folke et al., 2005). Vulnerability is also frequently invoked because it is a core concept in disaster risk and hazard approaches, to the study of livelihoods and poverty, climate change and security – primarily from the standpoint of biophysical risk factors (Eakin and Luers, 2006; Miller et al., 2010). Political-economic and political-ecological examinations of vulnerability also emphasize important socio-economic and socio-political dimensions of risks and hazards, the differential impacts that these contextual conditions create as well as the extent to which they influence recuperation and coping mechanisms (Eakin and Luers, 2006). Whilst vulnerability shares some characteristics with resilience and when used in the same

context widens the applicability of the social resilience framing, it is also a quite a separate term that has increasingly been viewed as:

“...a condition encompassing characteristics of exposure, susceptibility, and coping capacity, shaped by dynamic historical processes, differential entitlements, political economy, and power relations, rather than as a direct outcome of a perturbation or stress.” (Miller et al., 2010, pg. 4)

This conceptual diversity is often credited with creating difficulties in generating consensus or enabling disciplinary integration (Webb, 2007; Fischer et al 2009; Plummer 2010). This is particularly so in the social sciences where suggestions are that it conflicts with ideas of power, democracy and self-determination (Duit et al., 2010). Nevertheless, the application of resilience to SESs, and specifically, how the biophysical and social components integrate and interact when viewed as CAS has gathered pace in recent years (Folke, 2006; Biggs et al., 2012). This is in no small way due to the heuristic function and measurable quality of resilience (Biggs et al., 2012). And it is perhaps this that helps counterbalance a concept that can sometimes feel too broad to be of use, unwieldy perhaps, yet at the same time it could be argued it is the fact that it has so many strands, so many dimensions to it, that it presents many possible routes to understand and investigate SESs, as Hodgson et al., (2015:503) put it:

“Resilience has come to mean so many different things that it must assume its broadest definition.”

The interaction between SES and resilience theory covers the spectrum of social-ecological system issues. Assessing the potential of environmental drivers and perturbations to impact ecosystem resilience has been a major stalwart of research. Modelling local disturbance effects on ecological resilience at the landscape scale, for example, Van De Leemput et al., (2015) demonstrated that resilience is not necessarily a gradually declining phenomenon, but in homogeneous ‘landscapes’ can remain consistent whilst subject to drivers of change until a critical threshold (i.e. the Maxwell point) when resilience dramatically decreases. Their analysis suggests that gauging impending resilience decline is difficult, and that undertaking environmental management interventions such as restoration in spatially homogeneous landscapes may ‘fail’ or lead to a ‘landscape-wide transition’, issues equally applicable to larger spatial scales (Van De Leemput et al., 2015).

As a major environmental driver of system instability climate change is often linked to ecological resilience. For example, Bernhardt and Leslie (2013) argue that the management of coastal and marine systems needs to focus on ecological diversity, ecosystem connectivity, and adaptive capacity in the form of phenotypic plasticity and microevolution, as these characteristics are fundamental to coastal and marine resilience. In support of this view, Basket et al., (2014) demonstrated that maintaining coral reef response diversity to system disturbance is necessary for sustaining community-level resilience.

Building and maintaining the resilience of ecological systems is therefore seen as a critical step in mitigating the worst effects of climate change, for example, Schippers et al., (2014) recently investigated the notion that landscape resiliency is a product of diversity. Their results suggest that landscape heterogeneity (i.e. in terms of spatial configuration) is associated with greater diversity (e.g. genetic, ecological and economic) and increased resiliency, in part, because diverse landscapes produce more ecosystem services (Schippers et al., 2014). In addition to more ‘natural’ systems, enhancing resiliency and adapting to environmental change is also an increasing concern for the development of urban ecosystems and ecosystem service provision in cities (McPhearson et al., 2015). However, to improve ecological resiliency also requires the ability to monitor and map the resilience characteristics of ecosystems, and yet the availability of spatially explicit ecosystem resilience assessments is poor, but recent progress has been made in the form of an indicator-based system for assessing forest ecosystem resilience from which large-scale resilience maps can be generated (Yan et al., 2014).

The resiliency of SES is also deeply interwoven with the way systems are governed and managed for sustainability. This is especially true in many developing countries, where the need to balance agricultural production against the very real pressures of water scarcity and climate change for example are highly prescient issues (Maleksaeidi and Karami, 2013). Work undertaken in the Asian Highlands has demonstrated the importance of linking upstream and downstream conservation measures with local climate adaptation strategies to enhance local livelihoods and ecosystem capacity (Xu and Grumbine, 2014). This example makes the argument that building resilient SES requires the involvement of local communities, local knowledge, and recognizing cross-scale governance dynamics (Garmestani and Benson, 2013; Ruiz-Mallén and Corbera, 2013). It also highlights the necessity for institutional social-ecological rules to be adaptable and sensitive to system changes (Duer-Balkind et al., 2013). Flawed governance regimes operating in Taiwan’s Danungdafu Forestation Area, a region known for its land-use, legal and environmental conflicts, have also shown how managing resilience can be incapacitated by serious erosion of the central elements of governance, namely: accountability, transparency and participation (Tai, 2015).

At their heart these narrative emphasize the importance of individual human “response diversity” as a fundamental element of the resilience of social-ecological systems. The suggestion is that this response diversity (i.e. the variety of actions and decision-making people undertake in response to similar challenges and opportunities) is central to the capacity of SES to adapt and transform to changing socio-economic, environmental and political conditions (Leslie and McCabe, 2013). Overall then these examples support the “policy-relevant” principles outlined by Biggs et al., (2012:421) for improving social-ecological resilience to enhance ecosystem services provision, namely:

“(P1) maintain diversity and redundancy, (P2) manage connectivity, (P3) manage slow variables and feedbacks, (P4) foster an understanding of SES as complex adaptive systems (CAS), (P5) encourage learning and experimentation, (P6) broaden participation, and (P7) promote polycentric governance systems.”

4.2 Regime Shifts: Tipping Points To New Thresholds

Resilience theory and the development of regime shift theory are deeply connected (Scheffer and Carpenter, 2003; Biggs et al., 2009). The weakening of a system’s resilience under pressure from a complex set of interacting drivers of change increases the likelihood of the system experiencing a regime shift² (Biggs et al., 2009; Biggs et al., 2012; Crépin et al., 2012). For instance, as Leadley et al., (2014:665) succinctly describe:

“Regime shifts can be driven by a variety of mechanisms that vary in their speed, their spatial extent, and the types of drivers involved.”

Regime shifts are therefore generally understood from the perspective of a:

“...substantial reorganization in system structure, functions and feedbacks that often occurs abruptly and persists over time” (Crépin et al., 2012:15)

A similar, perhaps fuller, complementary definition is described by Andersen et al., (2008:49):

“Ecological regime shifts can be defined as abrupt changes on several trophic levels leading to rapid ecosystem reconfiguration between alternative states. These shifts are generally thought to be driven by external perturbations (e.g., climatic fluctuations, overexploitation, eutrophication, and invasive species) or by the system’s internal dynamics.”

Notably, this definition stresses the fact that regime shifts can be large and sudden as well as persistent, resulting from internal dynamical changes in feedbacks that prevent the system from returning to a previous state (Biggs et al., 2009). It is the way these dynamics operate that provides the mechanism for regime shifts, as Crépin et al., (2012:16) summarize:

“All complex systems contain both damping (also known as negative or balancing) and amplifying (also known as positive or reinforcing) feedback loops. Over time, the many feedbacks within a system can evolve and combine in only a limited number of ways, leading the system to self-organize around one of several possible equilibrium points, attractors or stable states. A particular combination of dominant feedbacks that structure the system and lead it to evolve towards a specific attractor corresponds to a particular domain of attraction or regime.”

In general regime shifts occur as the result of gradual changes in underlying system variables, with the manifestation of these variable changes (i.e. a change in system condition) rarely showing until a critical threshold or “tipping point” is reached (Biggs et al., 2009; Crépin et al., 2012). As a result predicting or anticipating regime shifts is a difficult task (Brock and Carpenter, 2010). This explains why:

“Changes in system resilience associated with a weakening of dominant system feedbacks therefore usually goes unnoticed until an actual regime shift occurs. Once the system is close to a critical threshold, a shift in dominant feedbacks can be precipitated by even a small shock to the system” (Crépin et al., 2012:16)

Internal feedback mechanisms are responsible for the “hysteresis effect” of regime shifts, meaning that once a system has switched to an alternative regime it tends to remain in this ‘new’ state, even if the original drivers of system change are lessened or removed (Biggs et al., 2009; Crépin et al., 2012). Consequently, reversing regime shifts is exceptionally unlikely and is highly dependent upon the magnitude of the dominant feedbacks in operation (Crépin et al., 2012). This is reinforced by the reality that many systems have multiple tipping points and so-called “landscapes of stability” in which they can exist (Leadley et al., 2014). But why are regime shifts so important?

It is quite clear from the picture painted in Chapter 1 that in the world’s current condition we may expect numerous and deleterious regime shifts to occur. Evidence indicates that human activities (e.g. pollution, natural resource extraction, deforestation etc.) are increasing the probability of catastrophic regime shifts occurring, particularly under the influence of climate change (Leadley et al., 2014). In fact regime shifts have been identified across many terrestrial and aquatic systems at local and regional scales (Biggs et al., 2009; Biggs et al., 2012; Crépin et al., 2012; Leadley et al., 2014). The switch from oligotrophic lakes to eutrophic lakes, the collapse of fish stocks, macro-algae dominated coral reefs, and the shift from grassland to woodland systems are just a few examples (Polasky et al., 2011). But in SES financial crashes and large-scale social uprisings are also examples of regime shifts (Polasky et al., 2011). The implications of these alterations in the way SES function is severe, with strongly negative outcomes for ecosystem service provision and human-wellbeing (Crépin et al., 2012; Leadley et al., 2014). As Leadley et al., (2014:666) remark:

“...our analysis suggests that ecosystem, socioeconomic, and biophysical mechanisms could interact to produce widespread, difficult-to-reverse losses of biodiversity, degradation of ecosystem services, and net negative effects on human well-being at regional scales within the twenty-first century.”

These newly created regimes may also be highly resilient and not necessarily for the better, and their effects may be enhanced through aggregation³, synergy⁴ and spreading⁵ (Leadley et al., 2014). As an example, Howarth et al., (2014) have recently argued the simplification of ocean systems by human activity has often created newly impoverished (i.e., reduced biodiversity) but highly resilient systems. Indeed, regime shifts such as collapsing fisheries also represent significant livelihood and economic catastrophes (Crépin et al., 2012; Leadley et al., 2014). Recent work on marine regime shifts⁶ underscores the global extent of these occurrences and their negative ecological, social and economic impacts (e.g. Levin and Möllman, 2015; Möllman et al., 2015; Rocha et al., 2015).

Managing regime shifts therefore represents a considerable challenge to future sustainability and is highly dependent on the institutional context of particular SES (Horan et al., 2011; Crépin et al., 2012). Extensive work has focused on attempting to identify ecosystem, socio-economic and biophysical tipping points (Leadley et al., 2014; Dakos et al., 2015). Much of this work has been based on theoretical mathematical models and has revealed that although there may be little evidence of change in the mean condition of the system approaching a threshold other monitoring data (e.g. variability in time series signals, rate of recovery from disturbances, skewness) may be detectable (Biggs et al., 2009; Brock and Carpenter, 2010). Real world experimental systems also support this view (Carpenter et al., 2011). However, drawing upon Leadley et al., (2014:670) again large uncertainties remain in many areas:

“Ecosystem changes in which early warning signs have been detected and projections are relatively robust include snowfield and glacier melts, coral reef bleaching, coastal degradation due to sea level rise, the collapse of some fisheries, and migration of species due to climate change. There is only moderate confidence in mechanisms associated with the large-scale degradation of cloud forests of the Andes or the humid tropical forests of the Amazon. The lowest confidence is in the socioeconomic dynamics, because these are very difficult to predict.”

Furthermore, as Andersen et al., (2008) highlight, identifying environmental drivers of system change can be difficult due to the problems of teasing apart social and environmental factors which are strongly interrelated.

There is a strong case for management activities to build resilience into currently weak systems, enhancing diversity and ecological redundancy and ensuring vital system functions are maintained (Crépin et al., 2012). In terms of pressing next steps, work on regime shifts needs action across all scales and an integrative approach to research and governance systems if large scale changes in ecosystems and SES are to be avoided (Biggs et al., 2009; Leadley et al., 2014). As Crépin et al., (2012:21) neatly summarize:

“Improved understanding of system dynamics can be gained through a combination of monitoring programs, data analysis, modelling and testing by scientists with a wide range of scientific backgrounds. Improved understanding of how regime shifts impact on human well-being can be gained through integration of natural and social science that builds from understanding of likely changes in ecosystems linked to the changes in provision of ecosystem services, as well as linking to adaptation strategies that allow people to better cope with new circumstances.”

4.3 Final Remarks

It is clear from what we have discussed in this chapter that our impacts on social-ecological systems is profound, and has in many respects, been detrimental to their

functioning. It is also clear that both resilience theory and regime shift theory, despite certain misgivings and flaws, have provided, and continue to provide, useful theoretical, applied and policy-relevant insights into the functioning of social-ecological systems: How we are affecting them; what the likely implications of those actions are, the challenges we face in mitigating many of the potential negative consequences of those outcomes, and how best we can go about resolving them. So that the Garden of the future exhibits fewer, less abrupt and less serious regime shifts and is altogether, in the truest sense of the word, far more resilient for the right reasons.

Notes

1. Since its original application to environmental systems in 1973 (Holling, 1973) resilience has been consistently refined with present interpretations describing resilience as the capacity of a system to absorb disturbance and re-organise while undergoing change so as to retain essentially the same function, structure, identity and feedbacks (e.g. Peterson et al., 1998; Gunderson, 2000; Berkes et al., 2003; Elmqvist et al., 2003; Folke et al., 2004; Walker et al., 2004; Carpenter and Folke, 2006; Folke, 2006; Webb, 2007; Andersen et al., 2008).

2. Anderson et al., (2010) make the point that the term regime shift is just one of a number of different terms that are virtually used interchangeably to refer to the same phenomena:

“...we contend that terms such as regime shift, abrupt change, break- or change-point, structural change, ecological threshold, tipping point, and observational inhomogeneity basically address the same problem.” (pg. 50)

3. Aggregation is the phenomenon where:

“...regime shifts may co-occur in contiguous areas, which may lead to large areas being affected” (Leadley et al., 2014:667).

4. Synergy describes:

“...the processes underlying regime shifts can be synergistic, which can lead to greater degrees of degradation than would occur from a single process” ” (Leadley et al., 2014:667).

5. The term spreading applies to situations in which:

“...atmospheric transport, movements of organisms, or human migrations can increase the spatial extent or impact of regime shifts” (Leadley et al., 2014:667).

6. See the Special Issue ‘Marine regime shifts around the globe: theory, drivers and impacts’ in the Philosophical Transaction of the Royal Society B 2015, 370(1659)

Part 3: The Garden In The Age Of Sustainability

In Part 3 we discuss the increasing interconnections between environment and development issues, and the growing unification in practice and policy between the narratives of ecosystem services and sustainability. We also bring to the fore the links between ecosystem services and human-wellbeing, poverty and food, water and energy security. Finally, we assess the prospects for improving the sustainability trajectory of our Post-lapsarian and Post-Edenic world.

Chapter 5: The Call For Sustainability In The Garden: Aligning Development, Ecosystem Services And Wellbeing

“Despite these difficulties, the ideas of sustainability and sustainable development provide useful concepts for discussing the goals and outcomes of environmental and social interventions. Further, by speaking to how we should live in the world, sustainability and sustainable development become more than concepts or ideas. They become a sort of bridge connecting our thinking and planning about the future to actions and consequences embedded in material ecosystem and social processes.” (Lockie and Rasan-Cooper, 2015, pg. 124)

“The sustainability of ecosystem service provision is threatened by human impacts on the environment. While these impacts are necessary to provide a number of the provisioning services, e.g. agriculture for food and deforestation for timber, these interventions by a given beneficiary can negatively impact the same services available to other beneficiaries or different services provided by the same landscape.” (Mulligan and Clifford, 2015, pg. 179)

The preceding chapters have laid out the state of the planetary Garden, as well as Gardens at smaller scales and the context of human-nature relations. These chapters have also taken the opportunity to highlight connections between different aspects of the Garden and ecosystem services, and how ecosystem services have been impacted upon by humanities historical and contemporary activities. In Chapter 5 we attempt to make the case that, going forward, we need to secure a sustainable means of development in which ecosystem services and the connections between ecosystem services and human-wellbeing are not only regarded as essential, but also provide a particularly important mechanism for achieving future sustainability in socio-cultural, socio-economic and environmental terms. In the course of this chapter we survey the central issues in the global sustainability debate, and in particular, approach those discourses through an ecosystem service lens and with an emphasis on ecosystem service-human wellbeing relations.

5.1 A Note On Sustainable Development

Talking in detail and sketching out the history of sustainable development as well as critiquing it, though interesting and important, is both beyond the scope of this chapter and not central to its core discussions¹. That being said, it is nevertheless important to provide a brief synopsis of sustainable development and what it means, as this provides the background context to present sustainability discussions and efforts, as well as controversies and debates, and highlights the significant connections between ecosystem services and sustainability.

The history of sustainable development as both a discourse and a movement can be traced back, in a recognizable form, to a growing “environmentalism” that started to blossom towards the late 1960s and early 1970s and stimulated debates between academia, environmentalists, policy-makers, inter-governmental organizations and non-governmental organizations alike through, for examples, publications such as Garrett Hardin’s *The Tragedy of the Commons* (1968), Erhlich’s *The Population Bomb* (1968), the Club of Rome’s *Limits to Growth Report* (1972), and international meetings such as the 1972 Stockholm Conference on Sustainable Development (Adams, 2009; Sachs, 2015). These landmark events set the tone for the next twenty years, and in particular influenced the first proper recognition of sustainable development outlined in *The World Conservation Strategy*² published in 1980, which itself laid the foundations for the World Commission on Environment and Development (WCED) report, often referred to as the Brundtland report, produced in 1987. This landmark report, entitled *Our Common Future*, for the first time, offered a succinct and lasting definition of sustainable development: one that has influenced environmental and development policy ever since (Adms, 2009; Sachs, 2015). The definition of sustainable development the report offered was as follows:

“Sustainable development is development that meets the needs of the present without compromising the ability of future generations to meet their own needs.”
(WCED, pg. 43)

Breaking new ground, the definition the report adopted implicitly recognized that there were developmental limits, at the same time it explicitly stated that development is concerned with meeting human “needs” and also, critically, it emphasized both intra- and inter-generational equity and justice as the framework to achieve those sustainable ends (Adams, 2009; Sachs, 2015).

In a fundamental way, what followed in the 1990s, in terms of the 1992 United Nations Conference on Environment and Development, the so-called Rio “Earth Summit”, merely built-up the outcomes and recommendations of this report, especially in terms of the adoption of Agenda 21³, the Rio Declaration⁴ and the establishment of the Commission on Sustainable Development⁵, and from a biodiversity and ecosystem services perspective both the Convention on Biological Diversity (CBD)⁶ and the Framework Convention on Climate Change (UNFCCC)⁷ were agreed and signed in Rio (Adams, 2009). The outcomes and substance of the Rio Conference in terms of its achievements is debated and contested, but it certainly opened the way for the mainstreaming of sustainable development thinking, as well as paving the way for developments such as the Millennium Summit (New York, 2000) which produced the Millennium Declaration⁸ and the Millennium Development Goals⁹, and the World Summit on Sustainable Development (Johannesburg, 2002) that drew-up the Johannesburg Plan of Implementation (Adams, 2009).

Whatever the merits, machinations, positives or negatives of these global meetings in terms of whether they entrenched conservative positions and inequalities or were forward looking and open and inclusive, whether they achieved, underachieved or did not achieve at all their aims or ambitions, all of which is important in terms of their capacity to deal with the real issues of poverty and environmental degradation, they nevertheless increased the currency and popularization of ‘sustainable development’ and the issues with which it is associated (Adams, 2009)¹⁰. And notably, from a biodiversity and ecosystem service perspective, because they started to embed and entwine the concepts of environment and development they formed the background out of which the Millennium Ecosystem Assessment (2002 – 2005) was born, a global project which crystallized the idea that poverty and the environment are inextricably linked (MA, 2005).

Returning to the definition of sustainable development articulated by the Brundtland Report, there are serious issues regarding the ambiguity of meaning that its phraseology conveys: about how it portrays “development”. This has produced a narrative surrounding sustainable development that is not just contested but which has been coopted and employed in a range of contexts, each of which has spun it in a particular direction to further a specific agenda: it has shown itself capable of “restructuring” and “reorganizing” the practice of development (Adams, 2009)¹¹. Out of this contestation some would argue that the dominant discourse of sustainable development to emerge has been structured to promote Northern (i.e., Western) hegemony, co-linear with this point of view some would also charge that the radical aspects of sustainable development have been down-played through the actions of powerful interest groups and business-as-usual approaches that co-align themselves with neo-liberal forms of capitalism and globalization; and yet there are also on the fringes counter-currents that push for radical interpretations of sustainable development (Adams, 2009; Springett and Redclift, 2015). The result is that sustainable development has borne a range of different flavours: spanning the spectrum from radical to conservative interpretations (Adams, 2009; Springett and Redclift, 2015).

Yet it remains an important and powerful concept – on university campuses and in international policy arenas – and its wide interpretation, its multidimensionality, far from being an Achilles heel in many respects has enabled it to host a smorgasbord of values and ideas, it has become a rich tapestry of different meanings, and over time it has grown to encompass the acute problems of poverty and environmental degradation: the global crises that define our age (Adams, 2009). In fact, over the last thirty years, the normative facets of sustainable development have been honed, and in many circles, it is seen as a vehicle that recommends goals and pathways to aspire to; a practical holistic approach (Sachs, 2015:4):

“Thus the normative side of sustainable development envisions four basic objectives of a good society: economic prosperity; social inclusion and cohesion; environmental sustainability; and good governance by major social actors, including governments and business.”

Here the concepts of intra-and inter-generational justice, so explicitly conveyed in the Brundtland report, have been relegated to a secondary position in favour of this tripartite vision of social, economic and environmental goals (Sachs, 2015). In the most recent incarnation of sustainable development, articulated at the Rio+20 Summit (‘The Future We Want’) in 2012, the following definition was adopted by the UN General Assembly (UN, 2012, para. 4):

“We also reaffirm the need to achieve sustainable development by: promoting sustained, inclusive and equitable economic growth, creating greater opportunities for all, reducing inequalities, raising basic standards of living; fostering equitable social development and inclusion; and promoting integrated and sustainable management of natural resources and ecosystems that support inter alia economic, social and human development while facilitating ecosystem conservation, regeneration and restoration and resilience in the face of new and merging challenges.”

Conceived in this way sustainable development deals with concerns about the “global economy”, the processes that underpin it and the implications economic behaviour has for social and environmental outcomes, it concerns itself with “social interactions”, “Earth systems” and the problems of “governance” (Sachs, 2015). In essence then, by and large, contemporary understandings and interpretations of sustainable development would find general agreement in the views expressed by Sachs (2015), namely that sustainable development is:

“...a way to understand the world as a complex interaction of economic, social, environmental and political systems. Yet it is also a normative or ethical view of the world, a way to define the objectives of a well-functioning society, one that delivers wellbeing for its citizens today and for future generations. The basic point of sustainable development in that normative sense is that it urges us to have a holistic vision of what a good society should be.” (pg. 11)

At this juncture it is important to mention the sustainable development-sustainability dichotomy. The essence of this debate centres on the emphasis each concept places on the economy: it is argued that sustainable development explicitly argues that economic growth is achievable and also necessary for human-wellbeing, whilst sustainability makes none of those same claims. Sustainability, it has also been argued, is seen as a less overtly “political”. On the flip-side, sustainability is also regarded as a normative laden and complex concept just like sustainable development, and is often seen as being subsumed within the wider sustainable development discourse (Springett and Redclift, 2015). So is it sustainable development we are after or sustainability? A good summary of this dialectic is given by Springett and Redclift (2015:17):

“Despite the calls for sustainability to be extricated from the sustainable development discourse – or to replace it – there is also evidence that a number of writers have in mind an all-embracing concept that eschews neo-classical economics, calls for better understanding and treatment of nature, demands social equity and eco-justice based on a less instrumental understanding of democracy, and that this overall conception of ‘the good life’ is sometimes referred to as ‘sustainability’, and sometimes as ‘sustainable development’.”

But there is also another interpretation of sustainability, one that is often associated with economists, which views “sustainability” or “sustainable development” or even “sustainable economic development” from either a “systems” perspective or from a “capital approach” (Barbier, 2011). The “systems” perspective preferentially uses the term sustainable development and sees the concept very much in the same way we have already discussed, as residing at the intersection of ecological, economic and social systems – with the objective of sustainable development being to maximize the goals of each system whilst simultaneously minimizing their trade-offs. On the other hand, the capital approach, generally talks more in terms of ‘sustainability’ and sees this concept as comprising different capital assets (e.g. social capital, human capital, financial capital, natural capital etc.). From this perspective sustainability, or indeed sustainable development, is about managing those assets as a portfolio (as a collective capital stock) for human-welfare benefits (Barbier, 2011).

The capital approach to sustainability has particular significance for ecosystems, biodiversity and the services they supply because within this framing there are two possible positions – strong and weak sustainability. The strong sustainability position – promoted by ecological economists – proposes that natural capital stock has a special status: that other forms of capital (e.g. human or physical) cannot substitute for all types of environmental resources that make up natural capital, or indeed, substitute for all the ecosystem services supplied by nature. This position, at its heart, questions whether in fact there is a “homogeneous total capital stock”, and claims that some forms of natural capital are inimitable and essential to human welfare (Barbier, 2011; Balaceanu and Apostol, 2014; Pelenc and Ballet, 2015). On the other hand, the weak sustainability view – promoted by environmental economists – argues that capital stocks can be shuffled around, that the stocks in some sense are like for like, that natural capital does not have a special status, as long as depleted natural capital can be replaced with equally or more valuable forms of other asset stocks then the aggregate value of the portfolio remains intact, is not diminished and in some cases may even be enhanced – and in delivering this outcome sustainability is achieved (Barbier, 2011, Balaceanu and Apostol, 2014; Pelenc and Ballet, 2015).

The sustainable development or sustainability picture is a complex one, but whatever the position taken, this has become the dominant discourse in debates regarding environment and development issues over the last four decades, and within the last 15 years the field of

ecosystem services has aligned itself with this discourse by explicitly connecting poverty and environment and placing human-wellbeing at the centre of that inextricable connection.

5.2 Environmental And Sustainability

Mounting evidence indicates human activities are adversely affecting planetary systems: exceeding fundamental biophysical thresholds (i.e. “planetary boundaries”), changing climatic conditions and seriously undermining the long-term sustainability of human societies (Folke et al., 2004; Rockström, 2009; IPCC, 2014; Steffen et al., 2015a). This is occurring at such an unprecedented speed and scale that some believe we have entered a new human-induced geological era, the so-called “Anthropocene”: characterized by regime shifts in biogeophysical systems and the potential exit from our Holocene conditions (Steffen et al., 2007; Steffen et al., 2011; Hughes et al., 2013; Steffen et al., 2015b). The drivers of these system level changes include climate change, land-use conversion, pollution, invasive species introductions and disruption of biogeochemical cycling propagated by human material consumption (Steffen et al., 2011; Oldekop et al., 2016). International assessments (e.g. Millennium Ecosystem Assessment and the Stern Review) have highlighted the connections between these prescient environmental “health” issues and human welfare and the state of the economy (Oldekop et al., 2016). In the Modern-era it is the consumption patterns, resource demands and trade policies of developed capitalist economies that have destabilized the functional capacity of ecosystems; many located in least-developed and middle income countries, leading to the widespread erosion of ecosystem health in those regions (Matutinovic, 2007). By the year 2000, for example, 75% of global terrestrial habitats had been significantly transformed (to some extent) by human activity (Hughes et al., 2013), and according to the Millennium Ecosystem Assessment 60% of the ecosystem services evaluated were suffering from degradation or overexploitation (MA, 2005).

In 2010 a set of international targets to halt biodiversity loss were missed, but in the same year, in Nagoya, Japan, Parties to the CBD signed-up to a new Strategic Plan for Biodiversity (2011-2020) for the purposes of achieving, in part, some of the goals that had been missed the previous decade but also to recognize a new set of priorities, progress in scientific knowledge and new policy options with regards not just to halting biodiversity loss but doing so in a way that was coupled to human-wellbeing – and as part of this new strategic plan five principal goals and 20 associated targets were adopted, more generally known as the Aichi 2020 Targets (Perrings et al., 2010; Rands et al., 2010; CBD, 2014; Marques et al., 2014; Hill et al., 2015). Recent evidence suggests that progress towards achieving these targets has in general, with some exceptions and acknowledging achievements in some sub-targets, been

relatively slow, on occasion very poor and in other instances the reality has worsened (Table S5.1 Appendix A on CD; CBD, 2014; Hill et al., 2015), as Hill et al., (2015:22) remark:

“15 of the Aichi Targets are unlikely to be delivered; 3 are likely to be delivered in part; and 2 in full.”

The reasons for such woeful progress span a host of social-ecological explanations ranging across economic, social, political and environmental factors and their pair-wise interactions (Hill et al., 2015). Take Target 3 for instance and the economic-political interactions that hamper it, as Hill et al., (2015:30) spell out:

“Target 3 encounters the multiple economic distortions in the global political arrangements for trade, typically market rules that protect subsidies for developed-world farmers, fishers and foresters and urban consumers. New market mechanisms that “enclose” land and natural resources in many cases end up dispossessing the poor rather than preventing biodiversity loss, as lack of political recognition of the rights of the poor prevents feedbacks, again a form of partial decoupling.”

Meeting target ambition is also hindered by the variety of trade-offs, synergies and interactions that occur between the different targets: sometimes helping each other, other times not (Marques et al., 2014:5):

“For example, protecting areas with high number of threatened species may not overlap with areas where habitat loss (Target 5) is occurring at faster rates. The adoption of some approaches to sustainable agriculture practices (Target 7) may reduce agricultural yields, which may make more difficult halving the rate of loss of natural habitats (Target 5).”

Despite these impediments, Hill et al., (2015) identify a series of “leverage points” through which improvements in achieving targets can be made. For example, by concentrating on the social-environmental interaction and directing attention to achieving more effective knowledge co-production as well as more equitable food systems governance then the barriers to making headway with Targets 1, 2, 3, 5, 7 8 , and 9 can be lifted (Hill et al., 2015).

The importance of the Aichi 2020 Targets does not reside simply in their benefits for biodiversity, ecosystem services and natural resources if achieved, but it also the influence that meeting these targets will have on the wider sustainability agenda and in particular the Post-2015 sustainable development settlement, as the CBD’s *Global Biodiversity Outlook 4* Report (2014:10) puts it:

“Meeting the Aichi Biodiversity Targets would contribute significantly to broader global priorities addressed by the post-2015 development agenda; namely, reducing hunger and poverty, improving human health, and ensuring a sustainable supply of energy, food and clean water.”

5.3 Social And Economic Sustainability

However, this is not simply an issue of environmental transformation. It is also about the significant upheaval in the lived-experiences of people and society facing considerable social, economic and political inequalities exacerbated by the burdens of uneven development within and across nations affecting all aspects of human-wellbeing, including: income, health, education, poverty, governance, and livelihoods (Wilkinson and Pickett, 2010). Significant global improvements in life expectancy, health, sanitation, gains against infectious disease, universal primary education and wealth have been achieved in the last two decades, for example, headcount poverty rate measures (the proportion of people living under a specific poverty threshold) have shown continual declines from 43 percent in 1981 to 21 percent in 2010 (UN, 2013; Sachs, 2015). Yet despite these impressive gains, 850 million people still suffer from hunger; 740 million lack access to clean drinking water; 2.4 billion lack basic sanitation; 383 million live on less than US\$1.25/day and almost half of humanity resides in degraded coastal areas (UN, 2013). In South Asia, for example, in 2010 31 percent of the population (or 507 million people) lived in extreme poverty¹², and in East Asia, an area that has witnessed significant economic growth and rapid declines in poverty, 250 million people still experience extreme poverty (Sachs, 2015).

According to the most recent UNICEF report *The State of The World's Children* (2016) the number of non-attending primary-aged school children has risen since 2011 with 40% leaving primary school without the ability to read or write, and although child survival has improved dramatically children in sub-Saharan Africa are still 12 times more likely to die before their fifth birthday than their counter-parts in high-income countries, and even more saddening around 250 million children continue to live in regions experiencing armed conflict. In a recent appraisal of children and armed conflict produced for the Secretary General of the UN the serious brutality of that reality was vividly expressed:

“In the Syrian Arab Republic, the five-year conflict has caused the deaths of more than 250,000 people, including thousands of children. In Afghanistan in 2015, the highest number of child casualties was recorded since the United Nations began systematically documenting civilian casualties in 2009. In Somalia, the situation continued to be perilous, with an increase of 50 per cent in the number of recorded violations against children compared with 2014, with many hundreds of children recruited, used, killed and maimed.” (UN, 2016a, pg. 2)

Since 2000 significant progress and improvements have been made in the global state of children living in extreme conditions; however, the rate of transformation has also been too slow and if that trend continues then it is estimated that by 2030 167 million children – predominantly in sub-Saharan Africa – will still be living in extreme poverty, almost 4 million will die annually (many from preventable deaths), and roughly a population the size of the UK will still be outside the education system (UNICEF, 2016). Making the investment in

children's early wellbeing, their health, security and cognitive development is central to ensuring bright future prospects (Sachs, 2015; UNICEF, 2016). Promoting an inclusive agenda for education along the education chain, from primary school onwards, that focuses not just on enrollment but also staying within the education system is critical for future employment, livelihood prospects and female empowerment (Sachs, 2015; Oldekop et al., 2016; UNICEF, 2016)¹³.

“While there has been great progress made at the primary school level, the progress in educational enrolment and attainment is much less at the secondary level and above [...] in tropical Africa and in parts of Asia where extreme poverty persists, secondary education remains inadequate.” (Sachs, 2015, pg. 257)

As stated in the opening chapter of *The State of the World's Children* (2016:1):

“If the soul of a society can be judged by the way it treats its most vulnerable members, then by a similar measure, a society's future – its long-term prospects for sustainable growth, stability and shared prosperity – can be predicted by the degree to which it provides every child with a fair chance in life. Providing every child with that fair chance is the essence of equitable development.”

Social inclusion (e.g. issues that concern prosperity, discrimination, equality and social mobility) is a growing problem, across the world there are great divisions between and within societies: Gini coefficients indicate widespread income inequalities within and between rich nations as well as between rich and poor nations (Sachs, 2015). For instance, take the United States, the most economically and politically powerful nation on Earth, yet it has as a Gini coefficient of 0.45, as Sachs (2015:56) observes:

“What is remarkable is that the top 0.01 percent of US households, just 12,000 households out of a total of 120 million households, now takes home around 5 percent of the total income of American households, up from 1-2 percent in the 1970s.”

The significance of these income inequalities is that they cut across core human rights, such as the right to life and good health, as well as across ethnic groups – they lay bare the cracks in society, staying with the example of the United States, Sachs (2015) goes on to outlining:

“The United States, already noted as having the highest income inequality among the high-income countries, also has significant disparities in life expectancy. [...] The northeastern seaboard of the United States, including Boston and New York City, has high life expectancies. However, counties of the Deep South of the United States, in states such as Alabama or Georgia, have several years fewer in life expectancy. African Americans have significantly fewer years of life expectancy compared with white, non-Hispanic Americans.”

Virtually 75% of the world's population now lives in societies where the distribution of income is more unequal than it was just two decades ago:

“In most countries, the gap between rich and poor is at its highest level since 30 years. Today, in OECD countries, the richest 10% of the population earn 9.6 times the income of the poorest 10% [...] In several emerging economies, particularly Latin America, income inequalities have narrowed, but gaps remain generally higher than in OECD countries.” (OECD, 2015 (Summary), pg. 1)

The insidious nature of income inequality means that it associates and feeds into other forms of inequality like health as we have just seen with the life expectancy differences in the United States, but also gender, education, identity as well as affecting long-term economic growth (OECD, 2015; Oldekop et al., 2016)¹⁴. For example, poverty rates for indigenous groups and marginalized communities are high around the world, whether that is in South America, Sub-Saharan Africa, or parts of Southeast Asia. Issues of race, ethnicity, and power are particularly potent and incendiary forces that precipitate and percolate through the inequality tide and undermine social cohesion (Sachs, 2015). Regional variation in gender inequality is substantial but is still massively higher in Africa and South Asia compared to many other parts of the world (here the political and social status of women still lags behind), as evidenced by the UNDP’s Gender Inequality Index (Sachs, 2015). Yet recent findings from the World Economic Forum’s Global Gender Gap Index demonstrate that gender inequalities are broadly-speaking improving, even though many barriers persist (WEF, 2015:7):

“In some countries, progress is occurring rapidly regardless of starting point and income level, but in others, change is much slower or negligible. A decade of data has revealed that the Economic Participation and Opportunity gender gap has been closed by 59%, with slow improvements of 3% over the past ten years. In Educational Attainment, the gender gap has decreased compared to 2006 and now stands at 95%. Health and Survival is the subindex that is closest to parity, at 96%, but the gap has widened slightly compared to 2006. While the most relative improvement over the last decade has been made in Political Empowerment, the gender gap in this area remains the widest, with only 23% being closed.”

With predictable certainty these report reveals that Scandinavian countries make the top of the list, with Iceland placed in number 1 spot, whilst languishing at the bottom are the more impoverished and socially unsettled countries of Syria, Pakistan and Yemen (WEF, 2015). Confronting these persistent inequalities is essential going forwards for a number of reasons, as Oldekop et al, (2016:70) remarks:

“Tackling inequality in its different forms can have major benefits for individual and social wellbeing including health, education and nutrition, poverty reduction, as well as the stability of public institutions and political dynamics.”

Moving out of extreme poverty requires proper political processes and governance: good governance, exercised through various means and mechanism (e.g. norms, laws etc.) at formal and informal levels, underpins the foundation of a stable society and economic development (Sachs, 2015; Oldekop et al., 2016). Public institutions and organizations create space for political and economic processes and decision-making that enables government to

function and connects the citizenry to the State (Oldekop et al., 2016). Effective government is central to building capacity, providing basic infrastructure and public services, ensuring the rule of law, framing and uphold rights, regulating the economy and enhancing social and economic mobility – where these areas go awry, are undermined or fail to function altogether then the outcomes for the vast majority are poor – high rates of unemployment, poor health conditions, civil unrest (Sachs, 2015). Fostering good governance is central to inclusive and responsive development strategies, for reducing inequalities, conflict and enhancing the political participation of social groups, particularly those often located on the fringes of society (Oldekop et al., 2016) Putting the right policies in place to overcome the challenges¹⁵ that retain parts of world in extreme poverty is achievable, for example, though far from ideal important and momentous progress has been made in sub-Saharan Africa to the extent that recent evidence indicates that growth rates have risen 6 percent per annum (Sachs, 2015:149-150):

“The most poverty-stricken region of the world is tropical sub-Saharan Africa. In 2010, an estimated 48.5 percent of the population of tropical sub-Saharan Africa remained below the poverty line. Fortunately the rate is declining now and has been declining since the start of the new millennium. Some estimates put the poverty rate even lower today [...] There is definitely good news in Africa [...] The average growth rate in sub-Saharan Africa picked up significantly after the year 2000. Indeed, sub-Saharan Africa has been growing faster than the average of the world economies, at around 5 percent per year even faster in certain years.”

Governance is also fundamental to environmental sustainability: good governance supports and legitimizes the establishment of property rights and rules of ownership regarding the use of natural resources, it provides institutions and decision-making arenas for the development of natural resource policy, and it can affect whether transformative routes are taken to deliver an equitable and sustainable flow of ecosystem services (Bennett et al., 2015).

Many attribute the realities we have been discussing and the lack of more rapid progress to an ‘economic growth’ paradigm that fails to provide a basis for full human prosperity; promotes environmental degradation; entrenches inequalities that serve only to increase consumption and conflict whilst prohibiting cooperation on sustainability, and diminishes the social foundation upon which human-wellbeing depends (Kosoy et al., 2012; Raworth, 2012; Sachs and Rockström, 2013). A new economic development path will need to be inclusive, to balance a reforming agenda of the global financial system with the imperatives of globalization, as well as develop transparent and accountable institutions and legal arrangements that tackle issues such as labour rights and exploitation head-on (Oldekop et al., 2016). Consequently, those that regard themselves on the strong sustainability arm of the sustainable development debate have argued for a new approach to future economic

development based on an ecological economic ethos (Costanza et al., 2015). A sentiment further echoed by (Raworth, 2012:7):

“...central to pursuing sustainable development is the imperative of eradicating poverty [...] this depends in good part on ensuring humanity’s collective use of natural resources remains within sustainable limits.”

5.4 Conflict And Sustainability

Conflict and political instability have detrimental impacts on social and economic development and the natural asset bases of the poorest nations (e.g. Collier 2009; 2010; 2012). Inter- and intra-state conflict can be damaging to biodiversity as a consequence of altering natural resource use behaviours and consumption patterns (Fjeldså et al., 2005; de Merode, 2007; Hanson et al., 2009; Sutherland et al., 2009; Beyers et al., 2011; Gandiwa et al., 2013). Conflicts also severely undermine peoples’ livelihoods and security (Justino, 2011), as Oldekop et al., (2016:68) explain:

“Both new and old conflicts are generating particularly fragile scenarios within and beyond country borders, potentially increasing the numbers of displaced people, refugees and asylum seekers”

Moreover, deteriorating social conditions, rising civil disturbance and violence dramatically alter household activities; increase family mortality rates; restrict access to basic services; reduce incomes and employment opportunities; undermine infrastructure and strengthen local community dependence on natural resource exploitation profoundly affecting social-ecological sustainability (de Waal and Whiteside, 2003; de Sherbinin et al., 2008; World Bank, 2011; Rus, 2012).

The myriad ways conflicts degrade broader ecological sustainability include enhanced forest and natural resource exploitation (Gorsevski et al., 2012; 2013); land conversion (Alvarez, 2003); the creation of poaching opportunities and nascent wildlife trade and timber economies (Barber and Talbott, 2003) and the transformation of local livelihood strategies in ways that continue to have post-conflict environmental legacies (Loucks et al., 2009). Conversely, adequately managing natural resources can provide a route to effective peacebuilding (i.e. social reintegration and reconciliation) by supporting economic growth, creating employment opportunities and securing a pathway for livelihood recovery (UNEP, 2015).

5.5 Poverty, Wellbeing, Biodiversity And Ecosystem Services

It is obvious that the poor rely disproportionately on ecosystem services for their basic needs and well-being (UNDP-UNEP, 2009). As a joint ICSU-UNESCO-UNU report (2008:19) stated:

“...human-wellbeing and poverty are intrinsically linked on a continuum.”

Indeed, as far back as the UNDP's Human Development Report (1992) and (1994) the crucial linkages between poverty, environmental sustainability and human development were recognized. The contribution of the environment to the development prospects of the poor can be seen in relation to livelihoods, health, economic development and environmental resilience (UNDP-UNEP, 2009).

5.5.1 Poverty, Health And Disease

The links between health and health systems, poverty and economic development are clearly established (SDSN, 2014). Since 2000, there has been as massive and worldwide increase in health and healthcare-related investment and spending, with a significant amount flowing from development assistance, in 2013 total health spending reach US\$13 trillion (WHO, 2015a). Health is essential for productivity; for example, poor health reduces per capita incomes by diminishing labour productivity as well as the size of the labour force (Bloom et al., 2004), and is also directly associated with low educational attainment (WHO, 2015a). Health therefore substantially affects levels of employment and wages (Todaro and Smith, 2009). Two fundamental measures of progress in human development (i.e. childhood mortality rates and life expectancy at birth) are critically dependent on socio-economic conditions and associated individual, household and community factors – dimensions closely linked to social inequalities and relative poverty (Cerrellati and Sunde, 2005; 2011; Boco, 2010). For example, take runder-5 mortality rates, as Sachs (2015:282) describes:

“The world average for the 5-year period 2010-2015 is 52 per 1,000, according to the World Bank estimates. For the developed countries it is 7/1,000, and for developing countries it is 57/1,000. For the LDCs [Least Developed Countries], it is 99/1,000. Among the world's regions, under-5 mortality rate is the highest in sub-Saharan Africa (110/1,000), followed by South Asia (55/1,000). These two locations are the epicenters of the challenge of extreme poverty and health.”

Where poverty, and especially its pattern of distribution, is widespread disease can be acute and severely undermine public health (Glassman, 2013). For example, poor sanitation has severe implications for population morbidity and, by extension, numerous social and economic repercussions, all of which can be reinforced by pre-existing inequalities particularly in relation to accessing goods and services (Mara et al., 2010; WHO-UNICEF, 2014).

5.5.2 Biodiversity, Ecosystem Services And Health

As Keune et al., (2014:181) state:

“The plea for human health as a sustainability indicator exemplifies the strategic importance of human health in terms of both ecosystem services and biodiversity conservation.”

The links between environmental condition (e.g. air quality, pollution etc.) and human health are generally well known, however, the underlying connections between development, health and wellbeing are much more sketchy (Summer et al., 2012; Smith et al., 2013; Oosterbroek et al., 2016). Recent evidence indicates that biodiversity has a supporting role in delivering ecosystem services which are fundamental to human health and wellbeing, and thus loss of biodiversity can impair and erode aspects of health and wellbeing (Summers et al., 2012; Sandifer et al., 2015). Contact with nature (e.g. urban green space, parks, forests etc.) and to a lesser extent particular aspects of biodiversity is increasingly associated with improvements in psychological, cognitive and physical aspects of human health (Smith et al., 2012; Summers et al., 2012; Keune et al., 2014; Sandifer et al., 2015).

The emergence of infectious zoonotic diseases (e.g. malaria¹⁶, TB¹⁷, HIV¹⁸ etc.) has been shown to have well recognized negative human, social and economic costs (Keune et al., 2014). Climate change is regarded as a particularly important factor driving a rise in growth and distribution of vector-borne diseases (WHO, 2015a). At the same time the spread of these vectors and pathogens also negatively impacts ecosystem functions (Sandifer et al., 2015). Some recent studies have suggested that biodiversity can assuage the risks associated with disease transmission and consequently promote ecosystem health, whilst biodiversity loss is suggested to have the opposite effect. The overall picture connecting ecosystems to human health is not entirely clear due to a range of complexities, for instance, multi-scalar interactions, the effect of socio-economic co-variates, and the range and difference in ecosystem services (Oosterbroek et al., 2016).

However, land use changes and ecosystem degradation are correlated with disease emergence as well as the spread of invasive species (Keune et al., 2014; Sandifer et al., 2015). For example, in sub-Saharan Africa, illegal and unsustainable harvesting has led to extensive forest degradation, decimating biodiversity and encouraging the spread of zoonotic diseases, realities that hold serious implications for future human health and wellbeing (Lele et al. 2013). Ecosystem disservices such as these can affect food security and the capacity to cope with natural disasters (Jouanjan et al. 2014). On a more positive note, a recent study by Bauch et al., (2015) looking at the effects of conservation policy interventions on public health in the Brazilian Amazon revealed that malaria, diarrhea and acute respiratory infection were all significantly negatively correlated with the establishment of environmental protected areas. In other words, protecting natural capital had a (small) positive effect on human health:

“...we analyze a rich dataset on disease, climate, demography, land uses, and conservation policies in the Brazilian Amazon. Unsurprisingly, we find that the health dividends vary across conservation policies and are small relative to the overall burden of these diseases. However, interventions targeted specifically at preserving biodiversity (strict protected areas) generate health co-benefits. Thus,

given a chance, nature does its part for human (health) capital, especially for the poor and politically voiceless.” (Bauch, et al., 2015:7414).

Rapid declines in biodiversity are also coinciding with a global rise in the prevalence of human allergies and chronic inflammatory diseases, particularly in urban communities, as a result of reduced exposure to microbes (Sandifer et al., 2015). Similarly, potential new drugs and medicines to treat human illnesses are being lost through the widespread decimation of biodiversity, particular loss of plant diversity, which impacts upon ecosystem services such as genetic resources and natural products that underpin the development of new medicines (Alves and Rosa, 2007; Keune et al., 2014). Overall, more in depth, focused coordinated research programmes concentrating on major diseases to establish testable hypothesis and collect and collate robust data are required (Sandifer et al., 2015).

5.5.3 Food Security

Agriculture represents a crucial life support system intimately connected to experiences of poverty and economic development (Sachs, 2015). Agriculture generates 10% of GDP in low and middle-income countries (FAO, 2015). In low income countries 80% of rural households depend on agriculture as their primary source of revenue, and in sub-Saharan Africa, for example, the agricultural sector employs 65% of the labour force making annual contributions of over a third of GDP in two-thirds of countries (Diao et al. 2010; Chauvin et al. 2012; Dethier and Effenberger, 2012; Jouanjean, 2013). Women are central to the performance of the agricultural sector; across low and middle-income countries they represent 43% of the agricultural workforce: though in comparison to their male counterparts they often face a number of social, economic and market constraints (FAO, 2015). Most of the agricultural activities undertaken in low and middle income countries occur on small family run farms (frequently less than one hectare) who produce food mainly for themselves: in fact the poorest of these farming households are net buyers of food (FAO, 2015). This hints at the fact that many agricultural communities in low and middle-income countries across the world regularly face food security issues particularly in the form of undernourishment (UNEP, 2012:80):

“Although estimates vary [...] to reduce the proportion of developing countries’ populations that are chronically undernourished to 4 per cent in the year 2050, world food production will need to increase by 70 per cent from 2005 levels.”

This is why a thriving agricultural sector can have far reaching socio-economic impacts, for example, in terms of: increasing food production, incomes, and profitability; expanding the local farming economy; stimulating on-farm and off-farm job creation; contributing to wider economic growth via links to upstream and downstream non-farm sectors, and generating development trajectories that favour poorer sectors of society (Daio et al. 2010; Cervantes-Godoy and Dewbre 2010; Dethier and Effenberger, 2012; Jouanjean,

2013). The available evidence also indicates that a functioning agriculture sector is much more effective at tackling poverty than non-agricultural sectors in the poorest communities, reducing poverty by up to 52% (Cervantes-Godoy and Dewbre 2010; Chrisiaensen et al. 2010).

However, as demand for agriculture has grown, year on year for decades, the consequences of intensification and extensification has been a number of significant environmental harms (FAO, 2012; 2014). The impacts on environmental sustainability have been severe, with substantial land conversions and changes in land use, extensive use of fertilisers and pesticides, and excessive groundwater extraction. The result has been widespread deforestation and forest degradation with concomitant loss of biodiversity and increasing soil erosion, alongside declines in water quality and more frequent water shortages, increases in greenhouse gas emissions and changes in biogeochemical cycles (Gibbs et al. 2010; Quinton et al. 2010; Lambin and Meyfroidt, 2011; Lenzen et al. 2012; Mills Busa, 2013; WWAP, 2014). Yet biodiversity is crucial in supporting food production and ensuring food quality. Firstly, agro-biodiversity provides a wide array of food sources and types and underlies a healthy diet. Secondly, genetic diversity and crop diversity are essential for promoting a resilient food system particular in the face of extreme climate change. Thirdly, biodiversity underpins essential pollination; nutrient cycling and pest regulation services all of which are under threat (Keune et al., 2014).

5.5.4 Water Security

Water is an essential resource, essential for life, yet access to basic water infrastructure, to water that is clean and readily available is for many a luxury and not a right (WaterAid, 2016:2):

“...more than 650 million of the world’s poorest people are living without access to an ‘improved’ source of drinking water.”

Further evidence in support of this global reality is also offered by Grey et al., (2013) highlighting the fact that:

“...too many people do not have adequate services: about 800 million people are without improved water supply, expending time and labour transporting water and risking health; about 2.5 billion people are without sanitary toilets, many defecating in the open, risking health and dignity; and about 900 million people are malnourished, with multiple associated risks.” (pg. 2)

The countries experiencing the most extreme water security-related issues are mainly in Sub-Saharan Africa (e.g. Chad, Nigeria, Angola) Central Asia (e.g. Pakistan, India, Bangladesh) and South and East Asia (e.g. China, Indonesia, Papua New Guinea) (WaterAid, 2016). Transforming this reality represents a pathway out of poverty and towards a sustainable and

secure future, because the lack of access to secure water sources is a major source of ill-health (particularly for women and children), a loss of income and detrimental to long-term economic sustainability (WaterAid, 2016:2):

“People from impoverished, marginalised communities have no choice but to collect dirty water from open ponds and rivers, or spend large chunks of their income buying water from vendors [...] This water is always a health risk; in many cases, it proves deadly. Globally, diarrhoeal diseases caused by dirty water and poor sanitation are the second biggest child killer after pneumonia, taking 315,000 young lives every year.”

Growing demand for water (both domestic and industrial), rising population and demographic changes (especially in low and middle-income countries, but also in particular China and India), increased consumption (especially from agriculture), widening inequalities (such as social exclusion), environmental drivers of change (especially climate change and deforestation), alterations to the world’s hydrological cycle, overexploitation of ground water sources, regional lack in government funding; conflict and weak governance means that water security¹⁹ is and will be a major global challenge in the years and decades ahead (Bakker, 2012; Grey et al., 2013; WaterAid, 2016):

“While water-related risks continue to threaten society at the local, national and international scales, they now increasingly do so at global scales owing to rapid economic, demographic and climate change. While we are not ‘running out of water’, we urgently need to understand better how larger global changes will affect freshwater availability.” (Grey et al., 2013, pg. 3)

This major global challenge and the concerns it has stimulated has led to a rise in research and policy activities, in particular focusing on issues relating to water supply threats (e.g. lack of access, contamination), water-related hazards and their impact on economic development and livelihoods (e.g. floods, droughts), threats to water-related ecosystem services (e.g. point-source pollution, water consumption) and impacts on the hydrological cycle in a context of uncertainty and climate change (e.g. the frequency and magnitude of droughts and floods) (Bakker, 2012). Over the last two decades significant progress has been made not just in these areas and their practical application but also in the real world circumstances of people living in water insecure regions:

“This is an era of unprecedented progress in reaching the world’s poorest people with safe water. The Millennium Development Goal target for halving the proportion of the planet’s population without safe drinking water was met in 2010, well ahead of the 2015 deadline.⁶¹ Over 90% of people now have access to improved sources of drinking water.” (WaterAid, 2016, pg. 13)

Nevertheless, there is still much to do, still many improvements and steps to be taken to improve millions of lives around the globe, as WaterAid’s State of the World’s Water (2016) report goes on to state:

Yet, even in countries that have made the most impressive progress in reaching people over the past 15 years, tens of millions of people are still unserved with their basic human right to safe water. The stories here show that, while there is much to celebrate, the stark inequality between the haves and have-nots in these nations urgently needs to be addressed.” (pg. 13)

According to Grey et al., (2013) two principal challenges will define the future of global water security over the next few decades: how to ensure that by mid-century 9 billion people will be able to have access to efficient water-related services (e.g, water conservation, water recycling, optimizing water production, focusing on the water-energy-food nexus); and how best to manage and mitigate the variety of water-related threats to society (e.g. focusing on water quality and quantity, identifying tipping points). Providing solutions to these broad challenges will require interdisciplinary research and cross-sector collaboration in the formulation of policy, a focus on the social-environmental and socio-economic impacts of an altered water cycle, and a multi-scalar approach to water security issues from the local to the global in terms of geography and geopolitics (Bakker, 2012; Grey et al., 2013). But, on a more practical note, improving the water security problems of the planet and for those living with these issues on a day-to-day basis requires funding, investment, the political will and sound and effective government policies and the involvement of the private and voluntary sectors, for example, as the WaterAid (2016:18) recommend:

“Governments must bring about a dramatic and long-term increase in public and private financing for water, sanitation and hygiene, building the strong national systems needed to achieve universal access to sustainable services.”

And furthermore,

“Private and public sectors need to cooperate more effectively to achieve universal access to water, sanitation and hygiene in workplaces, communities, and throughout supply chains. The emerging UNICEF initiative ‘WASH4Work’ is a key opportunity to bring together businesses, governments and multilateral agencies in service of this goal.”

And finally,

“Governments must take an integrated approach, ensuring that improving access to water, sanitation and hygiene services is embedded in plans, policies and programmes on health, nutrition, education, gender equality and employment.”

5.5.5 Energy Security

Demand for energy is increasing, at local, regional and global scales energy consumption is increasing and alongside that so are demands for natural resources (Oldekop et al., 2016). Some projections suggest that energy usage will grow by 33% up to 2040 driven in particular by the domestic production and consumption patterns in China, India, Southeast Asia and parts of the Middle East and Africa; whilst due to increased energy efficiency, reduced overall consumption associated with demographic and economic changes energy

consumption by OECD countries is set to decline over the same period (IEA, 2015). The overall rise in energy demand and supply, and its regional variation, has significant implication of poverty reduction, social cohesion, long-term economic development and environmental sustainability, for example, as Oldekop et al., (2016:67) state:

“Unequal access to natural resources and to the revenues generated by their exploitation, combined with the socio-environmental impacts of extractive industries, are among the main causes of social conflicts in the Global South.”

Indeed, as if to underline this point the International Energy Agency (IEA) in their most recent *World Energy Outlook* (2015:3) report state:

“Despite the serious efforts already made, today an estimated 1.2 billion people – 17% of the global population – remain without electricity, and 2.7 billion people – 38% of the global population – put their health at risk through reliance on the traditional use of solid biomass for cooking.”

It still remains the case that universal access to commercial energy is still largely an aspiration of the future, and in many parts of Africa and Asia the lack of appropriate electrification is having significant social and economic impacts particular in terms of enabling the provision of basic services such as health and education (WEC, 2013). Energy security will be dictated over the next couple of decades by developments in China and India in particular, but also across low and middle-income countries more broadly, as their changing energy demands and infrastructure requirements influence production and consumption in key energy sectors (e.g. coal, gas, oil, renewables) (IEA, 2015).

Take coal, for example, a significant player in global carbon emissions and at present responsible for 40% of global power supply, and though global reserves of coal decreased by 14% between 1993-2011 production rose by 68% during the same period – despite coal’s poor environmental credentials it is nevertheless relatively cheap and reliable and as a consequence has improved access developing regions and countries have to a consistent energy source (WEC, 2013). In the long-term the global fraction of energy derived from coal is set to decrease (in part replaced by gas²⁰ and other renewables) even though in the short absolute consumption will increase; countries such as China are attempting to curb their coal consumption through new emission trading schemes and alongside reduce their impact on global climate change:

“China is set to introduce an emissions trading scheme in 2017 covering the power sector and heavy industry, helping to curb the appetite for coal. From a mere 3% in 2005, half of China’s energy use today is already subject to mandatory efficiency standards, and continued improvements in efficiency, alongside large-scale deployment of wind, solar, hydro and nuclear power, lead to a flattening and then a peak in China’s CO₂ emissions around 2030.” (IEA, 2015, pg. 2)

Important renewables such as hydro-power and wind are both set to grow. For instance, hydro-power already exists in more than 100 countries (especially in North America, Russia, China and Brazil) and is responsible for supplying 15% of global energy. China is by far the world's most important hydro-power generator, responsible for 24% of global hydro-power capacity. At the same time the potential to turn this capacity into energy generation (and not just in China) has been undermined to some extent by water security issues, and in particular shortages (WEC, 2013).

Many developing countries are net importers of energy – some countries spend almost half their export earning on importing energy – with such high important dependency they are also open to the capricious and volatile nature of energy prices: securing long-term affordable, clean and reliable energy sources will be central to achieving development trajectories (GNESD, 2010). This will be especially necessary at the household level, because domestic energy consumption represents a far higher fraction of total energy consumption in developing countries, and at this scale energy insecurity is a far more tangible and noticeable reality, particular in terms of its impact on household income:

“For example, less than 8% of the Kenyan population has access to electricity, with 42% and 56% in Senegal and India respectively. Even in areas where households are connected to the grid, supply interruptions are a common occurrence in many developing countries [...] In Kenya, low income families spend more than 20% of their total income on energy commodities.” (GNESD, 2010, pg. 4)

Balancing future energy demands with environmental sustainability, economic growth and socio-economic realities to deliver a sustainable trajectory will require strong political processes, good governance, good public and private sector funding initiatives, investment in new low carbon technologies and technology transfer between developed and developing nations as we attempt to transition to a low carbon and decarbonized future (Oldekop et al., 2016)

5.6 Tracking Sustainable Development

Delivering a global sustainability agenda requires identifying the factors reciprocally influencing ecological, socio-economic and human health dimensions (Campbell et al., 2011; Griggs et al., 2013) in order to halt the degradation and exploitation of ecosystems and their corresponding services (Chapin III et al., 2010) and provide poverty alleviation measures and improve livelihoods (Adams et al., 2004). Achieving that overarching objective is not easy and Bartlett et al., (2011) identified four principal mechanisms to try and bring forth the links between poverty traps and biodiversity: (i) dependence on limited natural resources; (ii) shared vulnerabilities; (iii) failure of social institutions and, (iv) unintended consequences and lack of

adaptive management. In many cases these factors are both context-specific and contingent making the identification of causal connections fraught with difficulty: a situation that has led to calls for integrated approaches geared towards problem-oriented and solution-focused endpoints (Campbell et al., 2011; Leemans and Solecki, 2013; Mauser et al., 2013).

Understanding how socio-economic factors and political dynamics combine and interact to influence the joint health status of society and ecosystems requires adequate monitoring. Composite (or multidimensional) indicator systems gauging socio-economic and political impacts on social and environmental sustainability have been used, with increasing urgency in recent years, to make and inform policy (Mayer, 2008). These indicator systems have been developed to chart the influences of macroeconomic policies on sustainability e.g. Genuine Progress Indicator, Indicator of Sustainable Economic Welfare and Sustainable Net Benefit Index²¹ (Lawn, 2005); quantify human-wellbeing and poverty e.g. Wellbeing Composite Index²² (Reig-Martinez, 2013), Multidimensional Poverty Index²³ (Alkire and Santos, 2014) and Human Wellbeing Index²⁴ (Yang et al., 2013); track social and environmental sustainability e.g. DURAMAZ²⁵ (Le Tourneau et al., 2013) and Ocean Health Index²⁶ (Halpern et al., 2012; Elfes et al., 2014) and articulate the state of global ecological integrity e.g. the Living Planet Index²⁷ (Collen et al., 2009). Many more examples of human-wellbeing indicator systems are discussed in Smith et al., (2012). What many of these indexes describe is the detrimental impact human actions have had on the environment, on our own societies and on our relationship with the environment as well as providing a narrative that highlights our failure to properly enact the principles of sustainable development:

“Sustainability will require us to confront the damage to our ecosystems caused by our choice of development modality. Slavery, genocide, colonialism, industrialism, and exploitation of limited natural resources are part of those development choices. Sustainability will require us to address disparities that exist because of this development model and its history, and revisit deliberate public policy decisions to sacrifice certain groups and communities for development in the quest for a better life for the majority.” (Collin and Collin, 2015, pg. 210)

5.7 Onwards: The 2030 Agenda For Sustainability And The Sustainable Development Goals

There have been renewed calls to tackle human-environment problems in a holistic manner under the auspices of a new “global sustainability” agenda (Griggs et al., 2013). At the United Nations Sustainable Development Summit in September 2015 this new framework emerged as the “2030 Agenda for Sustainability”: in effect it is both a renewal of the Millennium Declaration but also an attempt to forge a newer sustainability path based on the successes and the failures of the previous 15 years in which the global community was

engaged (to varying degrees) with the implementation of the Millennium Development Goals (UN, 2016b):

“In implementing the Agenda, countries and stakeholders will have to make choices on where, when and how to act. In that process, they have pledged to endeavour to reach the furthest behind first. Fifteen years from now, when the current and the next generations together assess the implementation of the 2030 Agenda, a key measure of success will be the extent to which it has allowed improvement in the lives of the poorest and most vulnerable, regardless of gender, race, age, religion, place of residence or any other factor.” (pg. x)

Integral to achieving this aim and the 2030 Agenda are the 17 goals and 169 targets that make up the Sustainable Development Goals (SDGs) (Figure 5.1, ICSU-ISSC, 2015; Sachs, 2015; UN, 2016b). The SDGs are not simply a rehash of the MDGs, as Stevens and Kanie (2016:394) remark:

“While many of these goals look quite similar to the eight Millennium Development Goals, much of the content expands on those in a variety of different ways. Environmental dimensions, the interconnection between different problems, interrelated aspects of poverty and marginalization, and others are much more pronounced in the SDGs than they were in earlier efforts.”

The success of the SDGs will depend on the extent to which they align with existing international agreements and processes (e.g. Post-2015 Framework for Disaster Risk Reduction), how effectively they are implemented (e.g. need to optimize synergies between goals and targets and reduce potential trade-offs), and the degree to which progress in each of the goals and targets is measurable and verifiable (e.g. adequacy of indicators) (ICSU-ISSC, 2015; Costanza et al., 2016):

“To achieve the SDGs, policy makers, scientists, and practitioners will have to clarify how the goals and targets interconnect, including trade-offs and synergies, and develop three additional elements: (1) an aggregation of metrics of human and ecosystem well-being, (2) dynamic models of the integrated system of humans and the natural world, and (3) innovative ways to build broad public consensus on the future we want.” (Costanza et al., 2016, pg. 58)

At present only 29% of targets are considered well developed, 54% are considered to be in the “could be strengthened” bracket, with a further 17% requiring “significant work” (ICSU-ISSC, 2015; Hak et al., 2016). Nevertheless, the SDGs do have the potential to transform economic, social and environmental sustainability (Stevens and Kanie, 2016). Through their construction the SDGs provide a route that supports the idea that to reduce poverty and improve environmental sustainability (e.g. sustaining the provision of ecosystem services) crystallizing poverty-environment linkages in policy making, budgeting and implementation processes at multiple scales, from local to international levels, is critical (UNDP-UNEP, 2009). Many argue that one of the best means of achieving this is through the harmonization of the SDGs with the Aichi 2020 targets, integrating ecosystem services,

biodiversity and poverty in a coordinated way (Langlois et al., 2012; Lucas et al., 2014; ICSU-ISSC, 2015). Focusing on integration between land-use, agriculture and food security, adopting a nexus approach, has been argued as one effective pathway (Lucas et al., 2014; ICSU-ISSC, 2015, UN, 2016b):

“...promoting sustainable use of natural resources for hunger eradication and addresses the underlying causes of biodiversity loss in an integrated manner.”
(Lucas et al., 2014:204)

Ultimately, as Costanza et al., (2016:58) make clear, The 2030 Agenda and SDGs offer a pathway for hope, a pathway for encouraging a transformation to sustainability:

“The SDGs represent a major potential turning point in the future of humanity. For the first time in recorded history we have a set of goals and targets agreed upon by all UN countries, which include the full range of factors that contribute to equitable and sustainable well-being. We must not squander this opportunity to change the trajectory of humanity toward a more sustainable future.”

5.8 Final Remarks

In this chapter we have surveyed the principal issues concerning the sustainability of today's garden. We have, albeit briefly, sketched out the salient background to the sustainability debate and thus provided a necessary contextual hook for the rest of the chapter. Subsequently, we characterized the main sustainability challenges that we face spanning the core constituents of sustainability, namely, environment, social and economic spheres. In so doing we tried to highlight the linkages between these three dimensions, to illustrate where progress has been made but also where we have fell short and thus where improvements need to be made. In pursuing this path we have also sought to emphasize the challenges and barriers along the way, in particular, by trying to make the connection between ecosystem services, biodiversity and human-wellbeing. This is evident in discussions concerning health and disease, food security, water security and energy security, and especially in relation to the development of the 2030 Agenda and the SDGs. Over the course of the last two decades much progress has been made and achieved; yet, the Garden still remains some way off a sustainable trajectory, one where those pillars of sustainability are being fully met. However, with the SDGs there is a ray of hope for achieving that elusive path to sustainability over the next two decades and beyond. The Garden may still yet have a sustainable future.

Sustainable Development Goals

Goal 1. End poverty in all its forms everywhere

Goal 2. End hunger, achieve food security and improved nutrition and promote sustainable agriculture

Goal 3. Ensure healthy lives and promote well-being for all at all ages

Goal 4. Ensure inclusive and equitable quality education and promote lifelong learning opportunities for all

Goal 5. Achieve gender equality and empower all women and girls

Goal 6. Ensure availability and sustainable management of water and sanitation for all

Goal 7. Ensure access to affordable, reliable, sustainable and modern energy for all

Goal 8. Promote sustained, inclusive and sustainable economic growth, full and productive employment and decent work for all

Goal 9. Build resilient infrastructure, promote inclusive and sustainable industrialization and foster innovation

Goal 10. Reduce inequality within and among countries

Goal 11. Make cities and human settlements inclusive, safe, resilient and sustainable

Goal 12. Ensure sustainable consumption and production patterns

Goal 13. Take urgent action to combat climate change and its impacts*

Goal 14. Conserve and sustainably use the oceans, seas and marine resources for sustainable development

Goal 15. Protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and halt biodiversity loss

Goal 16. Promote peaceful and inclusive societies for sustainable development, provide access to justice for all and build effective, accountable and inclusive institutions at all levels

Goal 17. Strengthen the means of implementation and revitalize the Global Partnership for Sustainable Development

* Acknowledging that the United Nations Framework Convention on Climate Change is the primary international, intergovernmental forum for negotiating the global response to climate change.

Figure 5.1 The SDGs adopted at the United Nations Sustainable Development Summit in September 2015 (source: UN Sustainable Development Knowledge Platform, <https://sustainabledevelopment.un.org>)

Notes

1. For thorough discussion regarding the history and development concerning sustainable development, both in thought and in practice, see Bill Adams' book *Green Development: Environment and Sustainability in a Developing World* (2009) and also Jeffery Sachs' recent book *The Age of Sustainable Development* (2015)

2. According to Adams (2015:199):

“the World Conservation Strategy argued that conservation was essential to human survival, and that development should be seen as ‘a major means of achieving conservation, rather than an obstruction to it’”.

3. A 600-page document, Agenda 21 laid out, in a none binding manner, a series of detailed actions to promote ‘sustainability’ covering four themes: social and economic dimensions; conservation and

management of resources for development; strengthening the role of major groups (e.g. women, indigenous groups, business) and means of implementation (e.g. how to pay for sustainable development, technology transfer etc.). The document itself therefore covered a myriad of issues from biodiversity, to gender equality to labour rights (Adams, 2009).

4. A declaration of 27 principles outlining: the relationship between environment and development; the position of human beings within that context; how sustainable development should best be achieved; the important role and responsibilities that governments and countries in the Global North and Global South need to play, as well as citizens and different sectors. It is regarded, to some degree however, as being somewhat watered down and bland in comparison to the envisaged Earth Charter (Adams, 2009).

5. Setup to oversee the implementation of the outcomes of the 1992 United Nations Conference on Environment and Development it has, since 2013, been replaced by the High Level Political Forum on Sustainable Development (Wikipedia, 2016)

6. According to Adams (2009):

“The aim of the CBD was to conserve biological diversity and to promote the sustainable use of species and ecosystems, and the equitable sharing of the economic benefits of genetic resources.” (pg. 97)

It came into force in 1993 and by 1997 had been ratified by 162 countries. The convention has its antecedents in the 1980s and in particular the World Conservation Strategy document of 1980 (Adams, 2009).

7. The convention was signed by over 150 states and the European Community and came into force in 1994. The commitments it laid out, in respect of country’s obligations in reducing greenhouse gas emissions were none binding (Adams, 2009).

8. The Millennium Declaration was launched by Kofi Annan, the then Secretary General of the UN, in September 2000 as a ‘path breaking’ manifesto to forge a new kind of future for all mankind based on the principles of sustainable development. In it the Millennium Declaration (2000) states:

“We recognize that, in addition to our separate responsibilities to our individual societies, we have a collective responsibility to uphold the principles of human dignity, equality and equity at the global level. As leaders we have a duty therefore to all the world’s people, especially the most vulnerable and, in particular, the children of the world, to whom the future belongs [...] We believe that the central challenge we face today is to ensure that globalization becomes a positive force for all the world’s people. For while globalization offers great opportunities, at present its benefits are very unevenly shared, while its costs are unevenly distributed. We recognize that developing countries and countries with economies in transition face special difficulties in responding to this central challenge. Thus, only through broad and sustained efforts to create a shared future, based upon our common humanity in all its diversity, can globalization be made fully inclusive and equitable. These efforts must include policies and measures, at the global level, which correspond to the needs of developing countries and economies in transition and are formulated and implemented with their effective participation.”

In particular the Declaration makes specific pledges on issues of: Peace, security and disarmament; development and poverty eradication; protecting our common environment; human rights, democracy and good governance; protecting the vulnerable; meeting the special needs of Africa; and Strengthening the United Nations.

9. In order to meet the pledges and aspirations described in the Millennium Declaration, world leaders adopted 8 specific goals each comprising a number of targets to be met over a fifteen year time horizon, which came to be known as the Millennium Development Goals (MDGs): Goal 1 – Eradicate extreme poverty; Goal 2 – Achieve universal primary education; Goal 3 – Promote gender equality and empower women; Goal 4 – Reduce child mortality; Goal 5 – Improve maternal health; Goal 6 - Combat HIV/AIDS, malaria and other diseases; Goal 7 – Ensure environmental sustainability and Goal 8 – Develop and global partnership for development (<http://www.un.org/millenniumgoals/>)

10. Adams (2009) argues that in many respects sustainable development has failed to live up to its promise of reconciling and improving development and environment outcomes, as he candidly remarks:

“My own understanding of both sustainability and sustainable development has continued to grow and evolve. I have been distressed by the way in which the radical potential of debates about poverty and environment has been dissipated, and the ease with which key words and phrases have been taken up and incorporated as a ‘greenwash’ over corporate, governmental and individual ‘business as usual’. Truly the path to sustainable development is paved with good intentions, but rhetorical vagueness of that master phrase has made it too easy for hard questions to be ignored, stifled in a quilt of smoothly crafted and well-meaning platitudes [...] It seems to me that, if the ‘sustainable development’ debate is to have any value, it must address the challenge of relationships between people in their use of nature, and between humans and the rest of the biosphere.” (pg. xvii)

11. As Springett and Redclift (2015:15) explain:

“The international literature reflects the ‘stakes in the ground’ of specific groups: economics, ecology, environmental management, environmental philosophy, the claims and contestations of academic disciplines, views from the South and political and corporate positions all reveal the political, ideological, epistemological, discipline-based and philosophical approaches that compete for legitimacy. Broadly speaking, these fall into three major camps: ecology-centred, market-based and neo-Marxist approaches. From a critical perspective, sustainable development is perceived, not only as a social construct, but a multi-constructed and strongly contested concept that is political and radical. The dismissive charge of ‘vacuousness’ that has been made needs to be explored to discover whether such ‘vacuity’ is used as an obfuscatory gag on the radical aspects of the concept – a way of excluding competing views in the struggle for ownership – or whether the concept is, indeed, vapid jargon.”

12. We can define extreme poverty in a number of ways but perhaps the most useful and common sense way is to say that it concerns peoples’ individual prospects of accessing and meeting their basic needs which they need to survive on a day to day basis. Here basic needs refers to clean water, food, sanitation, shelter, clothing, access to healthcare, energy and in broader terms beyond that access to education, transport and communication networks (Sachs, 2015).

13. Education also has significant health benefits as a recent World Health Organization (WHO) Report *Health in 2015: From MDGs to SDGs* (2015) acknowledges:

“Education is strongly linked to health and other determinants of health, contributing directly and indirectly to better health. For example, education has an independent and substantial causal effect on adult mortality and morbidity, and also affects health indirectly through proximate determinants such as nutrition, sanitation and prevention and treatment practices.” (pg. 31)

14. This view is further supported by Sachs (2015:134) who regards inequalities as:

“...a kind of scourge [...] Highly unequal societies are both unfair and inefficient, squandering the potential of the poor by a failure to help the poor invest in their skills and health in order to achieve high lifetime productivity. As a result, the economic pie is both smaller and unfairly divided.”

15. Sachs (2015) identifies seven principal categories that have been attributed and associated with extreme poverty, and which left unchecked and unresolved maintain the status quo of extreme poverty, these are:

“...poverty trap, economic policy framework, fiscal framework, physical geography, governance patterns and failures, cultural barriers, and geopolitics.” (pg. 151)

16. According to the World Malaria Report (WHO, 2016) there has been a huge reduction in the global malaria burden over the last 15 years, and during that period 57 countries have managed to reduce the number of malaria cases by 75%. As the report states:

“The number of malaria cases fell from an estimated 262 million globally in 2000 (range: 205–316 million), to 214 million in 2015 (range: 149–303 million), a decline of 18%. The number of malaria deaths globally fell from an estimated 839 000 in 2000 (range: 653 000–1.1 million), to 438 000 in 2015 (range: 236 000–635 000), a decline of 48%. Most cases and deaths in 2015 are estimated to have occurred in the WHO African Region (88%), followed by the WHO South-East Asia Region.” (pg. 5)

Particularly important is the fact that the proportion of children infected with the malaria parasite has halved in endemic areas of Africa since 2000, and during the same period there has been significant scientific and technological progress in prevention and diagnostic tools (WHO, 2016)

17. Tuberculosis (TB) mortality has fallen dramatically since 1990 and the Millennium Development Goal to halt and reverse the incidence of TB has been achieved worldwide, as the World Tuberculosis Report (WHO, 2015b:2) states:

“TB mortality has fallen 47% since 1990, with nearly all of that improvement taking place since 2000 [...] Globally, TB incidence has fallen by an average of 1.5% per year since 2000 and is now 18% lower than the level of 2000.”

It is now a treatable and curable disease, but despite this, as the report goes on to detail it still remains a global killer:

“In 2014, TB killed 1.5 million people (1.1 million HIV-negative and 0.4 million HIV-positive). The toll comprised 890 000 men, 480 000 women and 140 000 children.” (pg. 2)

18. In 2014 the number of people newly infected with HIV was judged to be down 40% on 1990 figures, with AIDs-related deaths reduced by 42% from a peak just over a decade ago, and at present 17 million people are living with HIV due to anti-retroviral therapy (3.4 million of those are in South Africa alone). In Africa treatment coverage increased from 24% in 2010 to 54% in 2015, in Latin America and the Caribbean it reached 55% in 2015; and 42% in the Asia and Pacific region, however, it was only at 22% coverage in Eastern Europe and Central Asia (UNAIDS, 2016). However, despite these advances, AIDs remains one of the leading causes of ill-health and mortality:

“While investments in the HIV response have achieved unprecedented results, globally, in 2014, there were 36.9 million people living with HIV, 2.0 million new infections and 1.2 million deaths.⁸ Seven out of ten people living with HIV are in sub-Saharan Africa, where HIV is a leading cause of death among adults, women of child-bearing age and children.” (WHO, 2015a)

19. According to Bakker (2012:914) water security is:

“...defined as an acceptable level of water-related risks to humans and ecosystems, coupled with the availability of water of sufficient quantity and quality to support livelihoods, national security, human health, and ecosystem services”

This definition of water security divides water security into two segments – one about water-related risks and the other about water-related services. Grey et al., (2013:4) on the other hand offer a more succinct version of water security that focuses more on societal risk and is less explicit about the two components of water security previously defined:

“water security is a tolerable level of water-related risk to society.”

20. The IEA (2015:4) report makes clear:

“Where it replaces more carbon-intensive fuels or backs up the integration of renewables, natural gas is a good fit for a gradually decarbonising energy system: a consumption increase of almost 50% makes it the fastest-growing of the fossil fuels.”

This is also supported by the World Energy Council’s (WEC) report (2013:14) that indicates the growth in gas production:

“The reserves of conventional natural gas have grown by 36% over the past two decades and its production by 61%. Compared to the 2010 survey, the proved natural gas reserves have grown by 3% and production by 15%.”

21. The Genuine Progress Indicator and Indicator of Sustainable Economic Welfare are attempts to compute the sustainable progress or welfare of the citizens of a nation at a specific moment in time in economic terms. Private consumption expenditure is the foundation of both indexes from whence national account transactions pertaining directly to human wellbeing are added or subtracted in accordance to whether they are designated as costs or benefits. Following, modifications are made based on non-market valuations of social and environmental benefit and costs. The final indexes are constructed from all the included items represented in monetary terms and conveyed in ‘real’ rather than ‘nominal’ values. In the case of the Sustainable Net Benefit Index the presentation of the items used to calculate its value is different, and as a consequence it directly compares the costs and benefits of a developing macro-economy. Two dimensions are computed (i.e. uncanceled benefit and the uncanceled cost) and it is the difference between these components that provides the final index value. The uncanceled benefit is essentially the sum total of all income generating economic activities minus outgoings (i.e. net psychic income), and the uncanceled cost represents the natural capital costs incurred in supporting the economic process (Lawn, 2005).

22. The Wellbeing Composite Index is designed to overcome the flaws of GDP and the Human Development Index as measures of the socio-economic development status of individual nations, and is constructed from seven variables: income per capita, environmental disease burden, income inequality, gender-gap, education, life expectancy at birth and government effectiveness. Weightings are applied to the different components using data envelopment analysis to overcome ‘subjectivity’ and the final index value is derived from these values (Reig-Martinez, 2013).

23. The Multidimensional Poverty Index was developed to gauge the magnitude of acute poverty in the developing world. It consists of three dimensions composed of a total of 10 indicators: health (i.e. nutrition and childhood mortality); education (i.e. years of schooling and school attendance) and living standards (i.e. cooking fuel, sanitation, water, electricity, household flooring and assets). Indicator values are derived at the household level, which is represented by a single individual. Deprivation scores are computed for each individual based on the proportion of indicators for which they meet the ‘deprivation threshold’. Deprivation scores are measured against a ‘poverty cut-off’ to judge whether they meet the criteria of being multi-dimensionally poor. Subsequently, a head-count-ratio (i.e. the proportion of the population considered multi-dimensionally poor) and the intensity of multi-dimensional poverty (i.e. the average deprivation score) are calculated. These two values are then multiplied together to obtain the final index score (Alkire and Santos, 2014).

24. The Human Wellbeing Index is a quantitative attempt to capture the links between ecosystem services and human wellbeing. The index is made up of five dimensions each of which is composed of multiple indicators: basic material for a good life e.g. affordability of food; security e.g. property safety; health e.g. physical health; good social relations e.g. cohesion and freedom of choice and action e.g. affordability of quality healthcare. The values of individual indicators and by extension each dimension are derived from stakeholder responses to social survey questionnaires (Yang et al., 2013).

25. DURAMAZ is designed to capture a holistic conception of sustainable development at different sub-national scales and provide a comparative assessment across research sites. It is a hierarchical composite indicator composed of four over-arching dimensions: life conditions; environmental conditions; present needs and future perspectives and governance. In turn these are supported by fourteen sub-dimensions (e.g. health, schooling, pressure on the environment, general changes and future implications for the community and local governance and inner social ties), which are underpinned by 44 basic indicators. Information concerning the basic indicators is derived from social and ecological fieldwork and laboratory analysis. The four over-arching dimensions and fourteen sub-dimensions act as two tiers of thematic aggregation for communication of the underlying indicator variables (Le Tourneau et al., 2013).

26. The Ocean Health Index seeks to measure the health benefits of the ocean to society by monitoring the current and probable future trends in ten public goals associated with the ocean system. These goals are: food provision; artisanal opportunities; natural products; carbon storage; coastal protection; coastal livelihoods and economies; tourism and recreation; sense of place; clean waters and biodiversity – some of which have associated sub-goals. Each goal receives a score based on four criteria: current status; recent trend; existing pressures and expected resilience. The value of the index

is then computed as the cumulative linear weighted sum of those scores modified according to their individual weightings. The index is not a measure of the 'pristine' nature of the ocean but rather attempts to measure the amount of benefit relative to an optimum and is therefore helpful in guiding management policy (Halpern et al., 2012; Elfes et al., 2014).

27. The Living Planet Index (LPI) measures the present state of global biodiversity based on the trends observed in vertebrate species. LPI is calculated from time series data for 9000+ populations relating to over 2700 vertebrate species. This data is derived from various sources including peer-reviewed articles, reports and databases. Trends are determined for each population, averaged, and then aggregated for each species. A second level of aggregation occurs as species trends are aggregated to derive index values for terrestrial, marine and freshwater systems. Finally, these values can be combined to produce a single 'global' index value accounting for tropical and temperate conditions (Collen et al., 2009).

Part 4: Assessing The Garden

In Part 4 we highlight the importance of assessing changes in the Garden, specifically, in terms of ecosystem services as an essential means of informing environmental management decision-making and policy formulation. Evaluating changes in the social-ecological conditions of the Garden are a necessary precursor to understanding how ecosystem generation, provision and distribution relates to, and impacts, aspects of human-wellbeing; and moreover, whether policy initiatives and management programmes are effectively managing ecosystem services in a sustainable manner. In this way, we are better able to document the positive and negative effects of our continued transformation of our Post-Edenic Garden.

Chapter 6: Tools To Quantify The Garden's Ecosystem Services: Trade-Offs, Mapping And Indicators

“The creation of general tools capable of integrating ecological production functions with valuation for multiple services would allow a major advance towards integrated decision-making in diverse contexts across scales.” (Tallis and Polasky, 2012, pg.37)

“Explicit mapping of ecosystem services is recognized as a key step for the implementation of the ecosystem services framework in decision-making.” (Nahuelhual et al., 2015, pg. 162)

In the previous chapter we covered in some detail the issues and challenges of ensuring a pathway to sustainability for our Garden. Judging and measuring sustainability is therefore an essential element in determining and delivering this pathway. This subject lies at the heart of this chapter, with the narrative told from the perspective of ecosystem services. Evaluating the spatial generation and distribution of ecosystem services is critical in determining the costs and benefits arising from the provision of those services, the values placed upon them (which forms the basis of Part 4), and ensuring their future sustainability through key environmental management intervention measures (the basis of Part 5).

Why? Because this information is central to apprising the decision-making and policy processes concerned with the management of ecosystem services. For example, imagine a simple scenario, where a manager of a patch of land wants to guarantee the provision of at least two ecosystem services (A and B) at a particular scale and magnitude and choose the most effective means of delivering that outcome. Crucial to making an informed and robust decision will be to have information derived from modelling the various alternative processes that could produce the stated outcome. To know what ecological and social factors are key to delivering ES provision at that specified scale and assess them over time, and also to understand if there are any potential trade-offs that may arise by focusing purely on services A and B, such as, will that course of action adversely affect the provision of services C, D and E (Burkhard et al., 2013).

In Chapter 6 we briefly survey a number of developments in the spatial and temporal assessment of ecosystem services in terms of trade-offs, spatial mapping and indicators. Our purpose is not to be exhaustive with our coverage, but rather to provide a sufficient level of detail across these three areas to afford a basic understanding regarding current trends in research (e.g. directions, issues, challenges etc.), and how presently ecosystem services within the Garden are being appraised.

6.1 Ecosystem Service Bundles And Trade-offs

Delivering sustainable management of landscapes for multiple functions means having to balance competing needs, in particular, ensuring communities have enough living space and infrastructure, that food production is sufficient to meet local and national demands, and that ecosystem processes and biodiversity are maintained (Seppelt et al., 2013). Crucial to achieving this degree of “sustainability” is an adequate assessment of the relationships between ecosystem service generation and provision, which means that both supply and demand components must be considered and that management strategies should be underpinned by an understanding of the underlying associations between ecosystem services (Mouchet et al., 2014). For example, as Howe et al., (2014:264) succinctly remark:

“One principal challenge in managing ES is that they are not independent of each other and relationships may be highly non-linear, with unintentional trade-offs resulting when we are ignorant of the interactions among them.”

This means that co-production processes underlying ecosystem service generation, as well as the drivers that influence the provision of multiple ecosystem services, need to be explicitly acknowledged in order to afford an understanding of how so-called “ecosystem bundles” arise and are regulated (Bennett et al., 2009; Mouchet et al., 2014). Evaluating the relationships between ecosystem services through the processes of identification and quantification is central to the ability of land management planning to ensure the provision and distribution of ecosystem services under environmental change, and just as importantly account for potential trade-offs and synergies between ecosystem services (Mouchet et al., 2014).

Recent evidence suggests that trade-offs are three times more likely to be identified compared to synergies, and that their occurrence is particularly associated with stakeholder private interests in the resource-base, provisioning ecosystem services and localism (Howe et al., 2014). In an editorial for a Special Feature in *Ecology & Society*, Cavender-Bares et al., (2015) highlighted the variety of biophysical and social-economic trade-offs that can occur across scales and contexts (e.g. from temperate regions in the US to tropical regions in Kenya) in cases where multiple competing objectives are pursued. At the same time, however, their research provided some promise that if constraints and human needs are explicitly acknowledged then ways can be found to navigate or reduce these trade-offs (Cavender-Bares et al., 2015).

Although, for some, the term may be inappropriately used or even confused trade-off analysis is a particularly important tool in dissecting ecosystem service interactions in multi-functional landscapes (Mouchet et al., 2014). For example, Raudsepp-Hearne et al., (2010) successfully demonstrated how the concept of ecosystem service bundles can be employed to

assess trade-offs in multi-functional landscapes by linking specific service bundles to the social-ecological dynamics of particular landscape units. Relatedly, trade-off analysis has also been applied to uncover how landscape multi-functionality can differentially affect human-wellbeing and conservation outcomes by modulating the generation of particular landscape services (Campbell et al., 2010). Evidence from Spain suggests that differences in social preferences amongst communities (i.e. educational background, gender and environmental behaviour) can significantly influence their perception of an ecosystem's capacity to provide services. At the landscape scale these social preference difference can manifest in effects on ecosystem service bundles and prospective ecosystem service trade-offs (Martín-López et al., 2012).

Employing the trade-off approach has also been important for looking at the relationships between biodiversity and ecosystem service provision, as Nagendra et al., (2013:504) state:

“Trade-offs between biodiversity and ES have thus been acknowledged as one of the key risks associated with a strictly ES approach to landscape planning.”

This clearly suggests the need to view biodiversity and ecosystem service provision within a landscape frame, especially because of the deeply interconnected relations between biodiversity and the processes that underpin ecosystem service provision (Nagendra et al., 2013). From this standpoint, unravelling the relationships between land-use/land cover change, biodiversity functional traits and the subsequent provision of ecosystem service bundles is especially important (Nagendra et al., 2013). An example that makes some progress in this direction is a recent European-wide assessment of the interactions (i.e. synergies and trade-offs) between biodiversity, ecosystem services and conservation status (Maes et al., 2012a). In this paper the authors established a positive correlation between indicators of biodiversity and aggregated ecosystem service provision, further showing that this relationship was subject to spatial trade-offs occurring between provisioning and regulating services. Finally, they were also able to show that habitats with more favourable conservation conditions exhibited higher levels of biodiversity and a greater supply of cultural and regulating services (Maes et al., 2012a).

Notably, to aid landscape management decision-making processes literature discussions regarding ecosystem service trade-off assessments have increasingly called for a greater focus on ecosystem service supply and social demand (Castro et al., 2014; Mouchet et al., 2014). For instance, Ruijs et al., (2013) recently produced a spatially explicit trade-off analysis for ecosystem services in Eastern Europe. Capitalising on the availability of spatial data regarding agricultural revenues, cultural ecosystem services, carbon sequestration and biodiversity across 18 Eastern European countries, the authors focused on the supply-side and attempted to

estimate the opportunity costs resulting from the trade-offs occurring between ecosystem services as a consequence of marginal land-use changes (Ruijs et al., 2013). Maximising any ecosystem service was found to be cost-effective in situations where opportunity costs were minimal, and with careful targeting several ecosystem services could be jointly enhanced (Ruijs et al., 2013). However, improving carbon sequestration required focusing specifically on areas where sequestration levels were already prominent, and enhancing biodiversity alone was more cost-effective in regions of high biodiversity due to the complex relationships between agricultural revenues, biodiversity and cultural ecosystem services (Ruijs et al., 2013).

Extending the example of Ruijs et al.(2013) one stage further and addressing both supply and demand perspectives at a landscape scale, Castro et al., (2014) have recently produced a spatial assessment of ecosystem service trade-offs across biophysical, socio-cultural and economic “value dimensions” across different “landscape units” in southeast Spain. Their results clearly emphasise the complementarity between using different “value dimensions”, whilst also highlight the importance of identifying supply-side ecosystem service trade-offs and demand-side “value dimension” mismatches (Castro et al., 2014). This information is crucial to landscape planners seeking to maximise ecosystem service benefits and negotiate potential conflicts (Castro et al., 2014).

The take-home messages from these studies are neatly captured by the recent framework developed by Mouchet et al., (2014), which seeks to advance the argument that articulating and evaluating ecosystem service ‘associations’ should proceed according to the following categorisations: supply-supply (i.e. trade-offs and synergies simultaneously provided by ecosystem services); supply-demand (i.e. spatial and/or temporal lag between ecosystem service supply and social benefits), and demand-demand (i.e. negotiation between different stakeholders and stakeholder views). In this context landscape acts as the glue that coherently binds these characterisations together (Mouchet et al., 2014). Viewing ecosystem service trade-offs from an ‘association’ and ‘bundle’ perspective, as part of a supply and demand narrative, it is argued improves our understanding of the temporal and spatial dynamics of ecosystem services associations, provision and use (Mouchet et al., 2014). In addition, it is further advanced that it allows feedbacks that may arise through management, financial or off-site actions to be accounted for, and thereby enables the management of ecosystem service bundles in a way that better improves alignment with political and economic decision-making (Mouchet et al., 2014). Overall, adopting a trade-off approach is more likely to lead to win-win outcomes in planning by allowing decision-makers to maximise synergies (Howe et al., 2014).

However, what constitutes a trade-off is not always evident; the costs and benefits of where and how much (in financial and biophysical terms) specific services are delivered will not fall equally on all stakeholders (Campbell et al., 2010). As McShane et al., (2011) note, the

importance of understanding and addressing trade-offs is the demystification of the win-win framing of conservation and sustainable natural resource management discourse. In other words, choices concerning landscape options need to be made, and even the selection of an “optimal” landscape solution incurs trade-offs, with benefits and costs, generally, unevenly distributed between and within stakeholders and community groups with regards to social, economic and ecological outcomes as a consequence of inherent inequalities. The work of Howe et al., (2014) further supports this view, emphasising that trade-offs amongst stakeholders can produce diverging human-wellbeing outcomes that stem from differences in wealth, status, influence, and geography. Addressing institutional, structural and actor agency power in relation to trade-offs is therefore particularly important (Howe et al., 2014). This perspective is eloquently summed up by McShane et al., (2011:4):

“A focus on trade-offs allows multiple actors to recognize the hard choices involved in conservation and development, the outcomes of which will change the diversity, functioning and services provided by ecosystems and the range of opportunities available to people over space and time”.

6.2 Mapping Ecosystem Services

Mapping approaches, methodologies and tools are increasingly being used to capture ecosystem service provision, flows and demands across spatial and temporal scales (Bagstad et al., 2013; Burkhard et al., 2013; Crossman et al., 2013). For decision-makers and institutions ecosystem service maps represent important reference and explanatory tools, specifically for facilitating the development and implementation of policies to maintain high level ecosystem service areas (Martinez-Harms and Balvanera, 2012). Policy-makers need to understand the ecosystem services make-up of specific landscapes as well as their distribution (i.e. their spatial configuration), in order to successfully allocate resources and manage the complex supply and demand dynamics of ecosystem service provision (Burkhard et al., 2013; Schägner et al., 2013). Investigating the concept of ecosystem services demand Wolff et al., (2015) identified four “demand types”: (i) risk reduction; (ii) preferences and values; (iii) direct use and (iv) consumption of goods and services each of which have differing associations with ecosystem services, arguing that there needs to be a more coherent and clarified understanding of “demand” in management and policy as well as in terms of the factors that underpin its regulation. This is particularly important in complex multi-functional landscapes where different sectors such as agriculture, forestry and water management overlap (Malinga et al., 2015). For example, as Polasky et al., (2012:250) argue in relation to conservation planning:

“Conservation managers typically face a situation in which they have a large number of worthwhile conservation projects but only have resources sufficient to fund a small fraction of these projects [...] spatially explicit models of ecosystem

services and biodiversity can expand the type of information available to conservation managers and improve conservation decision-making.”

This perspective is further supported and underlined by Willemen et al., (2015:1) who remark:

“...we found that the best ES mapping practices to support decision making should be robust, transparent and stakeholder-relevant. These mapping practices include robust modeling, measurement, and stakeholder-based methods for quantification of ES supply, demand and/or flow, as well as measures of uncertainty and heterogeneity across spatial and temporal scales and resolution. Best ES mapping practices are also transparent to contribute to clear information-sharing and the creation of linkages with decision support processes. Lastly, best ES mapping practices are people-central, in which stakeholders are engaged at different stages of the mapping process and match the expectations and needs of end-users.”

From this standpoint, mapping and modelling ecosystems services is particularly important for assessing trade-offs, identifying overlaps with biodiversity, charting trends and patterns in provision and distribution, evaluating the costs and benefits associated with delivery, and deriving monetary valuations from biophysical quantification (Maes et al., 2012b; Malinga et al., 2015).

A variety of different mapping and modelling approaches exist and many of these have been recently evaluated (e.g. Maes et al., 2012b; Martinez-Harms and Balvanera, 2012; Schägner et al., 2013; Malinga et al., 2015). In many instances mapping investigations continue to be based on secondary data, proxies, unit values, and non-validated models (Martinez-Harms and Balvanera, 2012; Schägner et al., 2013). Moreover, most mapping investigations disproportionately focus on regulating services (e.g. water quantity regulation, climate regulation) with many fewer assessing supporting and cultural services (Malinga et al., 2015). These observations support the claims for more robust (i.e. validated) and sensitive approaches using multi-dimensional indicators capable of capturing the dynamic changes in ecosystem service provision, and more thorough mapping of the connections between underlying processes and spatial delivery (Maes et al., 2012b; Burkhard et al., 2013; Willemen et al., 2015). To some extent this is happening with validated models, and in particular those connecting ecological and social variables (i.e. integrated dynamic process-based models), but they still remain the minority of studies (Martinez-Harms and Balvanera, 2012; Schägner et al., 2013).

Scale represents another crucial element fundamental to the mapping and modelling of ecosystem services (Burkhard et al., 2013). Most mapping occurs at intermediate scales (i.e. the regional level), from 10,000Km² to 100, 000Km², with fewer studies conducted at smaller spatial scales (Martinez-Harms and Balvanera, 2012; Schägner et al., 2013; Malinga et al., 2015). The scale of spatial mapping has particular significance for policy, for example,

ecosystem service trade-off assessments need to be conducted at a scale that is meaningful, applicable and useful to decision-makers and stakeholders (Burkhard et al., 2013). As Malinga et al., (2015:5) remark:

“An important strength of presenting mapping studies with an extent at the intermediate scale is that land use policies are often developed at municipal, regional and national levels, because studies at these scales are better situated to inform their development.”

In particular, the resolution quality of mapping approaches is central to the usefulness of larger spatial scales outputs, many it seems are fine grained enough to provide meaningful information for land management decision-making (Malinga et al., 2015). Even so, there is a need for more fine scale analyses to inform local decision-making in a manner that links to wider national scale policies (Martinez-Harms and Balvanera, 2012). Moreover, reflecting previous observations, there remains a need to exploit land science, planning and ecosystem service models in conjunction with mapping and modelling approaches across spatiotemporal scales, to provide a richer tapestry of understanding regarding the complex social-ecological system dynamics that affect ecosystem service provision and distribution (Burkhard et al., 2013). As Burkhard et al., (2013:e2) remark:

“Better understanding of the interactions between landscape management and ecosystem service supply and demand across multiple scales is needed.”

In this respect, over the last few years there has been an increasing number of mapping and modelling software tools (e.g. ARIES¹, EnviroAtlas², InVEST³) that are available, some freely so, which can enable researchers, decision-makers and stakeholders to collaborate and produce multi-tiered and dynamic ecosystem provision and distribution models that explicitly consider supply-demand dynamics (Karieva et al., 2012; Bagstad et al., 2013). Take InVEST, this tool has been applied to a number of different contexts, for instance, to model water supply and hydropower issues (Mendoza et al., 2012); crop pollination services (Lonsdorf et al., 2012); the value of provisioning and regulatory services in agriculture (Nelson et al., 2012); terrestrial carbon sequestration and storage (Conte et al., 2012), and cultural services and non-use values (Chan et al., 2012) to list but a few examples.

On a related front, there is often a correlation between the spatial scale of ecosystem service delivery and stakeholders, primarily because the benefits stakeholders derive are determined by the scale at which the service is supplied (Hein et al., 2006; Burkhard et al., 2012). Consequently, stakeholder involvement in mapping and modelling ecosystem service flows, in particular, by eliciting their values and combining them with landscape service functions is especially important (Burkhard et al., 2013), as Willemen et al., (2015:2) point out:

“Best ES mapping practices meet the expectations and needs of map users and engage with stakeholders at different stages of the mapping process to best capture what ES are all about: the link between ecosystems and people.”

Participatory mapping of this kind is useful for uncovering hotspots of multiple landscape functions that identify “multiple win locations” (Gomina and van der Horst, 2007; Alessa et al., 2008; Raymond et al., 2009). Cognitive mapping, for example, is progressively being employed to modify ecosystem service assessments and valuations by providing shared understandings between providers and beneficiaries leading to place-specific solutions (Moreno et al., 2014). Overall, as Raymond et al., (2009:1302) note:

“In order to both enhance the robustness of local social-ecological systems and to solve environmental management problems, it may be necessary to implement policy instruments and management programs which recognise local values and empower local knowledge and expertise [...] Integrating landscape values [...] with the concept of natural capital and ecosystem services may provide a potential framework for enabling the detailed understanding of the broad range of values (called community values) that can shape planning for targeted conservation and environmental management.”

Ascribing values to ecosystem services and subsequently targeting management activities requires an understanding of ecosystem service flows (Bagstad et al., 2012). Dissecting spatial flows has been argued to be central to understanding ecosystem processes and functions that are particularly mobile, such as species migrations and pollination (Bagstad et al., 2014). Overlaying ecosystem service sources or “provisioning” regions with cognate benefitting areas provides a way to understand how ecosystem flows underpin service delivery (Serna-Chavez et al., 2014), as Tallis and Polasky (2012:268) put it:

“Ecosystem services are not delivered or valued uniformly across landscapes [...] This means that mapped predictions of the relative value of different locations in terms of ecosystem service provision can be very useful when implementing investments, incentives or regulations.”

Although some regional scale modelling of ecosystem service flows has been done it still remains an understudied area (Bagstad et al., 2012; Serna-Chavez et al., 2014), with the majority of studies predominantly conceptual in nature and lacking a temporal perspective (Syrbe and Walz, 2012; Serna-Chavez et al., 2014). In part this is because current frameworks are limited by their capacity to account for the different temporal dimensions of multiple ecosystem services (Serna-Chavez et al., 2014). Yet, if flows are not or only partially accounted for the values attributed to specific services can start to make little sense (Bagstad et al., 2012). This is especially problematic for the management of multi-functional landscapes, and can also limit the extent to which mapping contributes to planning and management decisions, as Bagstad et al., (2012:118) remark:

“The inability to consistently describe, quantify, and map ecosystem service flows limits the application of ecosystem service concepts to decision-making.”

Some recent work (e.g. Bagstad et al., 2012; 2014; Palomo et al., 2012; Serna-Chavez et al., 2014) does go some way to redressing the deficits in ecosystem service flow modelling. For example, Serna-Chavez et al., (2014) recently produced a spatial flow framework using an indicator system to identify benefitting areas supported by the spatial distribution of ecosystem services emanating from provisioning regions. As the author's remark about their framework:

“The study of ecosystem service flows requires guidelines for assessments that are appropriate for all services as well as all spatial scales. The generic framework for analysis of spatial flows of ecosystem services, as presented in this paper, can be a first basis to support such assessments.” (Serna-Chavez et al., 2014:32)

Overall, we agree with Malinga et al., (2015) that mapping approaches are extremely useful for managing multi-functional landscapes by securely embedding ecosystem dynamics within a useful decision-making scale. Ultimately, the mapping and modelling methodologies adopted will depend upon the particular conditions and circumstances to which these methods and tools are to be applied, and so will be affected by a diversity of factors such as data availability, study location (i.e. ecosystem services assessed and geographical features and characteristics), resource availability and the policy situation (Schägner et al., 2013). However, to improve the application of these tools and enhance their credibility it is also important to reduce the errors and uncertainties that affect their accuracy and precision, without properly accounting for these limitations ecosystem service mapping risks raising doubts over the validity of modelling outputs and their usefulness as a decision-making aid (Grêt-Regamey et al., 2013; Hou et al., 2013; Schägner et al., 2013). Quantifying ES through mapping and modelling also has other ancillary benefits such as improving social outcomes, for example, learning and engagement and awareness raising, leading to an improvement in the overall relevance and quality of decision-making, as Willemen et al., (2015:4) explain:

“The social outcomes of ES mapping processes, such as social learning and the creation of social capital, are important drivers of sustainable land use. Also, the resulting ES maps can serve a purpose to indirectly affect decision-making such as initiating discussions about the relevance of ES and biodiversity. Other social outcomes [...] listed as an important contribution towards sustainable future land use are: (i) awareness-raising and community engagement; (ii) empowerment effects and; (iii) the transfer of ecological knowledge within (and among) the communities and across generations.”

6.3 Indicating Ecosystem Services

Making robust environmental decisions that account for competing as well as different priorities such as food security, land use planning, and conservation etc. requires validated information regarding the quantification of ecosystem service stock-flow dynamics (Alkemade et al., 2014). Over the last few years, authors such as Feld et al., (2009), Layke (2009),

Vihervaara et al., (2010) and Reyers et al., (2013) have argued for the need to adequately capture the dynamics of ecosystem service delivery, the linkages between service generation and underlying biophysical processes, and the connections between service provision and human-wellbeing to inform decision-making processes. Consequently, indicator development has is a prominent area of research (Müller and Burkhard, 2012), and has been central to the development of important biodiversity strategies such as the European biodiversity 2020 strategy (Maes et al., 2016).

Indicator systems are necessary to monitor and manage the supply-demand dynamics of ecosystem service bundles (Alkemade et al., 2014). Moreover, as the primary influence of ecosystem service dynamics is land-use change and landscape management indicators need to relay and capture these relationships (Müller and Burkhard, 2012). Indicators are therefore critical communication devices of system changes (Reyers et al., 2013). As Müller and Burkhard, (2012:26) outline:

“Ecological indicators are communication tools that facilitate a simplification of the high complexity in human-environmental systems.”

A perspective supported by Alkemade et al., (2014:161):

“Indicators represent real conditions and as such are helpful to communicate inherently complex processes.”

A central role for indicators therefore is to be policy-relevant, and to communicate coherently system trends in a manner that can inform sustainable management practices (Layke, 2009; Müller and Burkhard, 2012). Indicators are avatars: vehicles to depict properties, states or interactions that cannot be directly evidenced. Comprehensively describing these properties usually requires multiple indicators, and for the chosen indicators to be complementary, scalable and capable of being integrated (Müller and Burkhard, 2012). Consequently, there are a number of scientific, practical and policy-related criteria that potential indicators ought to meet (Box 6.1).

Clearly, these criteria are challenging to meet, in whole or even in part, and particularly so for less tangible ecosystem services (Alkemade et al., 2014). However, as Van Berkel and Verburg (2014) demonstrate, with respect to cultural services, making the intangible tangible is possible if personal preferences can, for example, be linked to specific landscape attributes by using indicators capable of adequately capturing that information. Martín-Lopez et al., (2014) have also demonstrated that indicators can adequately capture trade-offs and synergies at the landscape scale, whilst others have developed frameworks to picture these trade-offs in ways that can be utilised by landscape management (Grunewald et al., 2014).

Box 6.1 Indicator selection criteria

Indicator criteria:

1. Policy relevant
2. Scientifically sound
3. Simple and easy to understand
4. Practical and affordable
5. Sensitive to relevant changes
6. Suitable for aggregation and disaggregation
7. Suitable for projections of future scenarios
8. Scale of indicator measurement should match the problem associated with the ecosystem service.
9. Establishment of baselines and thresholds required for indicators to be predictive
10. Indicators should be tied to specific mechanisms and a production function
11. Indicators should consider trade-offs with other services in the bundle of landscape service
12. Indicators should be amenable to interpretation and translatable into action

Selection criteria derived from Feld et al., (2009), Layke (2009), Vihervaara et al., (2010) and Reyers et al., (2013)

Landscape metrics have also become an increasingly widespread and important tool for representing and quantifying landscape patterns (i.e. heterogeneity) by characterising landscape structural qualities (Syrbe and Walz, 2012). For instance, as Uuemaa et al., (2013) illustrate, landscape metrics (designed to evaluate land cover/land-use patterns) have addressed issues concerning the landscape impacts of protected areas through to restoration and mining. Assessing “landscape functions” (*sensu stricto* De Groot (2006)), habitat functions have primarily been evaluated from the perspective of mammalian biodiversity and landscape composition (Uuemaa et al., 2013). The focus on regulation functions has tended to concentrate on water quality regulation as well as fire risk in relation to forest management planning, with fewer studies considering flood and erosion control (Uuemaa et al., 2013). It appears that with regards to information functions, though the number of studies identifying important connections between perception and landscape composition and configuration (e.g. aesthetics) has increased, quantitative assessments of landscape quality remain challenging due to the changing dynamic and scalable properties of landscape (Uuemaa et al., 2013). Overall as Uuemaa et al., (2013:105) comment:

“The main advantage of landscape metrics is their simplicity and speed of calculation, as rapid environmental changes demand easily obtainable indicators. Landscape metrics as a part of geospatial data analysis provide background information as well as scenario testing of environmental policies and monitoring goals set by international conventions and agreements.”

Nevertheless, we agree with Müller and Burkhard (2012) that there remain a number of challenges and barriers for indicators to overcome in order to improve their “suitability” and “quality”, issues that if resolved will allow them to better capture the essential connections

between ecosystem services, landscape change and management and human-wellbeing, and as a result improve decision-making and environmental policy (Box 6.2).

Box 6.2 Necessary improvements for ecosystem service indicators

1. Better connecting the indicator with the property of interest (comprehensive indicator suite)
2. Relationships between indicator and indicandum have to be clearer and more explicit (i.e. demonstrate relevant cause-effect interactions)
3. Need indicators to recognise the interactions between specific system components, but also the interrelations between the suite of indicators used
4. Improve the rationale and reasoning for indicator selection (i.e. transparency)
5. Balance the need to employ many indicators to assess a system with the need of policy-makers to identify with and convey a small set of quantifications
6. The normative aspects of indicators must be recognised
7. Recognise the inherent uncertainties in indicators and assess these to ensure their reliability

List adapted from Müller and Burkhard (2012)

6.4 Final Remarks

This chapter has sought to provide both an overview and demonstration of the increasing and widespread importance placed on quantifying, assessing, measuring and indicating changes in ecosystem services provision (supply and demand) across spatial and temporal scales and differing geographic contexts. Specifically, as a route to better inform and thus optimize decision-making processes and policy formulation pertaining to ecosystem service and natural resource management.

In our brief overview we have highlighted areas in which new and more sophisticated mapping and modelling tools and software developments have progressed, and through that progression have enhanced our knowledge of how the spatial generation and distribution of ecosystem services overlaps with important social and economic determinants of human wellbeing. In parallel we have also discussed some of the important barriers and challenges to the further development of ecosystem service quantification methods and applications. Together, what we have described demonstrates the continuing appraisal of how the social-ecological interactions within the Garden both determine and influence not only ecosystem service provision and distribution but also the relationships between ecosystem services and human-wellbeing.

The continued level of progress made in ES quantification provides an important route to ensure that our actions within the Garden are positioned along a path towards greater sustainability, specifically: by being aware of trade-offs; by understanding how our actions may affect service supply and distribution; and by being able to track those changes in relevant ways. By proceeding in this way we provide ourselves with opportunities to improve the way

we think about, and manage, the complex world around in a fashion that is both relevant and effective.

Notes

1. Artificial Intelligence for Ecosystem Services (ARIES) has been in development since 2007 and in use since 2010. The general purpose of ARIES is to quantifying ecosystem services in a way that captures ecosystem service complexity and dynamics, but in such a manner that it remains intelligible, straightforward and simple so that its findings are both generalizable and scalable across differing levels of data availability and detail (Villa et al., 2014). As Villa et al. (2014:2) the way ARIES functions is based on two particular developments:

“An extension of ES science intended to enrich the dominant MEA [Millennium Ecosystem Assessment] narrative with a renewed focus on beneficiaries, probabilistic analysis, and spatio-temporal dynamics of flows and scale. The result can heighten awareness of important distinctions such as that between potential and actual benefits [...] The capability to automatically assemble the most appropriate ES models based on a library of modular components, driven by context-specific data and machine-processed ES knowledge. A model structure fitting the goals, the context and the available data can thus be used for each situation, avoiding the pitfalls of the common “one model fits all” paradigm.”

(For further details of ARIES see (Villa et al., 2014))

2. EnvironAtlas is an online web-based platform that acts as a repository of data and tools for use in varying contexts by bringing together environmental, social, economic and demographic information within an ecosystem services context. Developed in the United States, EnvironAtlas is essentially a geospatial instrument that enables examination of ecosystem service supply and demand budgets, and provides information users can use to assess and view ecosystem services benefits relating to different elements of human health and wellbeing across the US (see Pickard et al., 2015 for further information).

3. Integrated Valuation of Ecosystem Services and Trade-offs (InVEST) was developed by the Natural Capital Project (www.naturalcapitalproject.org) as a means to improve, mediate and facilitate more robust, credible and improved decision-making through combining models on ecological processes (production functions) with economic valuation methods. It therefore combines biophysical and economic information together in a spatially explicit manner to aid conservation and natural resource management and conservation, as Tallis and Polasky (2012) summarize:

“InVEST is a set of computer-based models that: Focuses on ecosystem services themselves, rather than on the underlying biophysical processes alone; is spatially explicit; provides output in both biophysical and monetary terms; is scenario driven; clearly reveals relationships among multiple services; and has a modular, tiered approach to deal flexibly with data availability and the state of system knowledge.”

(For further information see Natural Capital: Theory and Practice of Mapping Ecosystem Services (2012). Eds. Kareiva, P., Tallis, H., Ricketts, T.H., Daily, G.C., Polasky, S. Oxford University Press. Oxford, UK.)

Part 5: Valuing The Garden: Economics And Ecosystem Services

In Part 5 we tackle another central pillar of the ESP, namely, the relationships between ecosystems (and their services), economics and values. In order to manage our metaphorical garden more effectively, sustainably and appropriately we first need to be able to capture how we, as human beings, perceive and value the natural world: in terms of its ecosystems, its biodiversity and particularly the services these systems provide that sustain our wellbeing. It is only through being able to capture, holistically, the way in which we value these services that we can engage in more fundamental value-based decision-making. And more specifically; where, rather than being an afterthought natural capital, and the flows of services that emanate from this ecosystem-base, are explicitly included, front-and-centre, in decision-making processes that openly acknowledge the potential multiple trade-offs that may ensue following the implementation of any policy initiatives.

Chapter 7: The Nature Of Value

“Over the years the concept of ‘value’ has gained a specific meaning in neoclassical economics - as the ability of a good to satisfy one’s innate desires or wants. It is individual and subjective [...] this is not the only understanding of the word. It is rather a very particular one. Instead, one could define value to cover views about what is a good life and a good society, principles concerning what is important and right to do.” (Arild Vatn, *Institutions and the Environment*, 2005 pg.146)

“If we are to view ecosystems as economic assets, then it is helpful to be able to measure this form of ‘ecological wealth’.” (Edward B Barbier, *Capitalizing on Nature: Ecosystems as Natural Assets*, 2011 pg.26)

7.1 Introduction

The environmental landscape is subject to constant change but, how we perceive, use, exploit and manage “nature” remains one of the fundamental challenges of developing sustainable societies and a flourishing Earth System (Arias-Maldonado, 2016; Folke et al., 2016). From economists to conservationists, many have championed the assimilation of economics and ecology as a means to align environmental interests with those of the broader political economy. The purpose of this co-alignment has been to “mainstream” nature into general policy-making processes. In recent versions of this process attempts have been made to delineate the connections between ecosystem goods and services and human-wellbeing (e.g. Polasky and Segerson, 2009; Liu et al., 2010; Turner et al., 2010; Barbier, 2011; TEEB, 2012). However, with reference to the latter development, such fruitful integration remains contingent on the challenges of identifying ESs, attributing values to ES, and resolving and validating the issues of societal dependence on ES all of which have yet to be fully resolved (Salles, 2011; TEEB, 2012).

These difficulties are exemplified by the continuing exchanges occurring between environmental and ecological economists regarding the nature of ES, the role of valuation in the implementation of the ES paradigm and the translation of ES into practical decision-making processes and policy applications (Beder, 2011; Salles, 2011; Farley, 2012). Largely, these debates centre on key differences related to ideas of sustainability, the use of cost-benefit analysis (CBA)¹, monetary valuation and marginal analysis, and the relationship between distributive justice and (economic) efficiency (Beder, 2011; Farley, 2012). Whilst at first glance these deliberations may seem overly esoteric, some argue (e.g. Farley, 2012) that they should be considered as enlivening, enriching and informing our understanding of ES rather than regarded as fostering discontent.

Our purpose therefore is to present a codified and synthesized elaboration of these important strands of discourse. In the chapters that unfold, we first sketch out the current trends and debates in the ecosystem service-value dialectic: where progress has been made

(theoretically and methodologically) but also where sticking points remain (Chapter 7). Second, in Chapter 8, following these discussions, we then go on to give a brief account of how valuation has been applied more recently to various terrestrial, coastal and marine ecosystems and their services at global and local scales. Then finally, in Chapter 9, we identify three core issues, two of which have philosophical and moral aspects and implications, namely, uncertainty and discounting, and a third, benefits transfer, which although it has normative qualities is more suitably considered for its practical ramifications. Collectively, all three need to be acknowledged and negotiated when applying the value lens to ecosystem services. Following, we conclude by highlighting future areas of progress.

7.2 What Is Value?

The concept of value underwrites the process of environmental valuation (Spangenberg and Settele, 2016). This statement forces us to grapple with the nature of value, it asks us to examine what environmental valuation is and why it is important, and invites us to wrestle with the difficulties environmental valuation poses. These questions and their answers are central to the ecosystem services paradigm (Vatn, 2005; Barbier, 2011). Starting by outlining what environmental valuation is and addressing its connection with value, in a nutshell:

“Valuation is about assessing trade-offs towards achieving a goal. All decisions that involve trade-offs involve valuation, either implicitly or explicitly [...] The value of ecosystem services is therefore the relative contribution of ecosystems to that goal.” (Costanza et al., 2014, pg.153)

This quote makes two clear points: that the purpose of valuation is to inform decision-making processes and choices regarding different environmental management and protection options; and secondly, it asserts that values are the mediators of that decision-making – in other words, values act as a necessary bridge linking beliefs to behaviours, or motivations to actions, as O’Brien and Wolf (2010:234) explain:

“Values have been closely associated with worldviews, which describe the basic assumptions and beliefs that influence much of an individual or group’s perceptions of the world, their behavior, and their decision-making criteria.”

This perspective builds on the value-belief-norm (VBN) theory detailed by (Stern et al., 1999) in relation to the support for social movements. In the elaboration of VBN, the core features are the causal connections made between an individual’s values with a set of broad and core beliefs, concerning consequences and responsibility, which inform and co-create a personal set of norms that lead to a mobilisation of action, as Stern et al., (1999:83/84) make clear:

“We propose that norm-based actions flow from three factors: acceptance of particular personal values, beliefs that things important to those values are under threat, and beliefs that actions initiated by the individual can help alleviate the threat and restore values [...] each social movement seeking collective good develops its positions based on certain basic human values and that each movement’s ideology contains specific beliefs about consequences and responsibilities that, in conjunction with its chosen values, activate personal norms that obligate individuals to support the movement’s goals.”

It should be rather obvious then that notions of value are broad and rich, as Spangenberg and Settele (2016:107) express:

“When talking about “value” the term should be specified or be understood as an “umbrella concept” comprising several incommensurable kinds of value.”

Delving beneath this “umbrella” we can distinguish as number of values, from overarching categories such as Ideal values and Real/Objective values to Subjective values² – and it is this latter classification of values that, by and large, forms the basis of the value typology adopted in environmental valuation: a category which itself can be partitioned into a subset of values each of which that can be expressed in relation to ecosystem goods and services (Table 7.1, Figure 7.1, Dendoncker et al., 2014; Spangenberg and Settele, 2016).

Table 7.1 Value typologies underlying ESV

Value Category	Definition	Value Type	Definition
Instrumental (Use Value)	Value is derived from an object’s capacity to achieve a given purpose – to be functional.	Market	The value of a commodity/good or service garnered in an open market. So-called ‘exchange’ value. To be exchanged goods must be regarded as scarce and considered useful. Value is derived from transactions, thus market price reflects utility. Exchange value provides a measure of the utility of a flow of goods from a stock, so-called marginal utility, not the utility of a stock of goods.
	Instrumental values are not absolute as values can change depending upon on how an object is used in a given context. The use value of an object is therefore subject to moral precepts (e.g. deontological, consequentialist, utilitarian).	Direct Use Value	The value attached to products and services provided by nature for direct consumptive (e.g., timber and food) or non-consumptive use (e.g., recreation and esthetic experiences)
		Indirect Use Value	The value attached to indirect utilization of ecosystem services, through the positive externalities that ecosystems provide (e.g., flood protection and carbon sequestration)
Inherent	Value is linked to the utility of particular objects (e.g., species, ecosystems etc.) that derives from a good not being substitutable, having a value for its own sake, but also providing end values.	Intrinsic (use)	Commodity values with little market recognition, but still recognized as having use-value.

Table 7.1 *Contd.*

Value Category	Definition	Value Type	Definition
Intrinsic (Non-use)	Value is ‘non-derivative’ and subjective, arising because it has been attributed value based purely for its own sake (e.g., in the words of Spangenberg and Settele (2016) either a ‘non-negotiable transcendental’, an ‘identity value’ or an attribution to ‘moral subjects’.	Intrinsic (non-use)	The inherent value of a naturally existing environment or life form irrespective of its market worth.
		Existence	Value attached to the knowledge that ecosystem services (including biodiversity and environments) exist irrespective of whether they are utilized.
		Bequest	Follows the sustainability criteria of the Bruntland commission, in that it concerns the willingness to pay to maintain the good condition of the environment for present and future generations.
		Option	Value based on the present willingness to pay for the utilization of a particular asset in the future, current likelihood of using it is highly unlikely.

Adapted from Chee (2004), Häyhä and Franzese (2014) and Sapangenberg and Settele (2016)

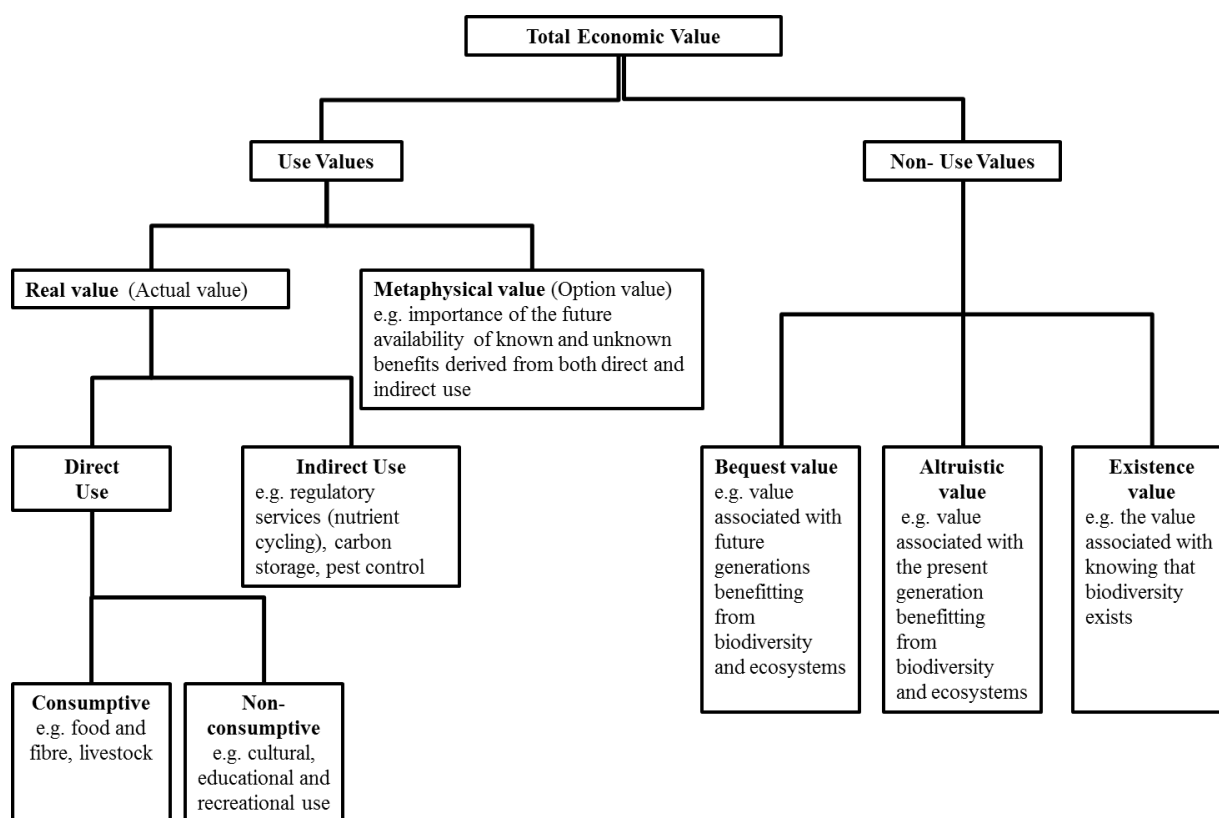


Figure 7.1 Value typology diagram showing the relationships between value types (source: adapted from Vo et al., 2012; TEEB, 2012)

7.3 Valuation – Some Criticisms: Money Isn’t Everything

The process of valuation, and in particular monetary valuation (i.e. monetization), has a long history and is not as recent as some have previously claimed (Baveye et al., 2013). Yet, despite its historical pedigree, valuing biodiversity and ecosystem services for policy and

decision-making purposes remains a contested issue (Spangenberg and Settele, 2010; Salles, 2011). As Spangenberg and Settele (2016:100-101) clearly argue:

“...the trend of the last 30 years is to value almost everything in terms of money, applying concepts of mainstream neoclassical economics to ecological systems [...] Despite the broadness of the debate, it is still disputed what is the niche where monetisation can play a positive role, based on an axiological typology of values, on basic economic theory, and empirical evidence regarding its past performance.”

Yet the over-arching logic behind ecosystem service valuation (ESV) seems to be reasonably clear; as Ed Barbier (2011:29) neatly synthesises:

“The idea that ecosystems provide a range of “services” that have value to humans is an important step in characterising these systems as “natural capital”. In order to view ecosystems as a special type of capital asset – a form of “ecological wealth” – then just like any other asset or investment in the economy, ecosystems must be capable of generating current and future flows of income or benefits. It follows that, in principle, ecosystems can be valued just like any other asset in an economy. Regardless of whether or not there exists a market for goods and services produced by ecosystems, their social value must equal the discounted net present value (NPV) of these flows.”

Nevertheless, Baveye et al., (2013) argue - based on an historical examination of monetary valuation techniques (especially in an environmental context over the last 50 years) – that there are a number of prominent inadequacies in the dominant monetary valuation paradigm, concluding that alternatives ways of accommodating nature into economic decision-making processes are required.

Criticisms of ESV cover a broad array of issues, including: (i) the suitability and validity of valuation typologies and methods (Spangenberg and Settle, 2010; Salles, 2011; Tisdell, 2011; Chan et al., 2012); (ii) the perception of valuation as simplifying ecological complexity (Norgaard, 2010); (iii) the lack of psychological and cultural parameters in models generating preference data (Parks and Gowdy, 2013); (iv) the ability of valuation to aid land-use policy and biodiversity conservation (Vira and Adams, 2009; Viglizzo et al., 2012) and, (v) conflicts of interest and power dynamics in its management application (Barnaud and Antona, 2014).

The first two criticisms are especially significant. Those cautioning against excessive ESV stress the inconsistencies and subjectivities of valuation methods (Spangenberg and Settle, 2010; Salles, 2011; Tisdell, 2011); whilst also highlighting the inherent biases towards monetarisation and the continued lack of interest in non-use values by economists in valuation exercises and the wider literature (Spangenberg and Settle, 2010; Salles, 2011; Chan et al., 2012):

“Non-market valuation, and other economic techniques that emphasize exchange values over cultural and ecological values, have been subject to criticism regarding the inability of exchange values to represent the total value of an ecosystem.” (Allen and Moore, 2016 pg.78)

The assertion of a continued lack of interest in non-monetary appraisals of ecosystem services is perhaps increasingly open to debate, as there is much movement in this direction, but that this view persists is, for some, indicative of a continued absence of methodological formalization in this area (Kelemen et al., 2016). Indeed, this view may be symptomatic of the widespread use of stated preference methods in ESV exercises which is regarded by some as a particular problem, primarily because peoples’ preferences are neither “pre-formed” nor “stable” (both of which remain standard neoclassical economic assumptions), but also, perhaps more fundamentally runs the argument, is the inability of monetary valuation to capture the diversity of values ecosystems have (e.g. intrinsic), which through the application of a single metric fails to account for the incommensurability³ of those values (Bunse et al., 2015). Commensurate with this perspective is commentary that finds fault with the application of economic valuation to the discernment of ideal values, such as equality or justice, as well as non-instrumental values. Making the case that monetary valuation has a limited application to environmental resources as a consequence, Spangenberg and Settele (2016:106) state:

“Economic valuation methods fail in cases where their basic model is in contradiction with the characteristics of the object to be valued. In particular, the calculus is not applicable to stocks of biological resources such as ecosystems and species since the economic definition of stock values (aggregate value of the flows generated by the stock) fails for renewable resources with their potentially unlimited lifetime.”

However, it is not obvious from this latter comment why flows from a resource that has generative properties should not be subject to economic valuation. Renewable resources in a theoretically abstract sense may have an unlimited lifetime, but they are not fixed either in terms of their stock or their flows which mean the patterns of change they exhibit are amenable to economic valuation. Nevertheless, clearly the central thread to these narratives is that values are multiple – and there are many expression systems or ‘languages’ of valuation in addition to monetary conceptions (Kallis et al., 2013).

In the same vein, others bemoan attempts to equalizes ESs (Salles, 2011) and simplify ecological complexity by reducing valuations to single dimensions (Norgaard, 2010). These ‘reductionist’ critiques are highlighted by Neuteleers and Engelen (2015:254) in their overview of the main criticisms of monetary valuation, where they remark:

“...it has been argued that it is simply impossible to value different environmental goods using a single monetary scale. Comparing goods is always done with regard to a specific comparative value (such as beauty, healthiness or economic benefit).

Since there is no overall comparative value ('betterness'), converting all goods to a single scale can only be done by favoring one comparative value. Asking people to express what they are willing to pay for environmental goods, is thus inevitably reductionist with regard to the spectrum of values."

7.4 Valuation – A Broadening Field

It is fair to say that, like the parent values from which they are borne, ESV methods represent a diverse smorgasbord of instruments; equally however, it is easy to argue that they are predominantly fixated on producing quantitative and monetary-oriented valuations of ES – as the critics have observed (Table 7.2, Lui et al., 2010). But whilst this may be generally the case, it is also true to say that in response to these criticisms there have been considerable movements toward expanding the role of qualitative and non-monetary valuation methodologies (Christie et al., 2012). Developments in deliberative methodologies such as deliberative monetary valuation (DMV)⁴ are testament to this (e.g. Spash, 2008; Brondizio and Gatzweiler et al., 2012; Lo and Spash, 2013; Bunse et al., 2015; Lienhoop et al., 2015). There have also been increasing attempts towards trying to define the total economic value (TEV)⁵ of ecosystems as exemplified by TEEB⁶ (e.g. Braat and de Groot, 2012; TEEB, 2012).

Nevertheless, some suggest TEV simply sums the main function-based values and therefore fails to account for the total value of an ecosystem (Jones-Walters and Mulder, 2009; Bunse et al., 2015), whilst critics of DMV point to the unsubstantiated nature of many of its hypotheses (Turner et al., 2010). Even those sympathetic to DMV acknowledge that depending on the mode employed the approach faces a number of theoretical, conceptual and procedural challenges (Lo and Spash, 2013:780):

"...the preoccupations of the researcher/practitioner can result in excessive intervention in the deliberative processes or even manipulation of outcomes (whether intended or unintended). Under preference economisation approaches [emphasise the elicitation of values at the level of the individual] individuals are required to strengthen their economic beliefs, while under preference moralisation approaches [social value elicitation supplants the individual perspective] they should convert their perspective toward a particular moral end. The proposed DMV processes appear to violate the requirements of value pluralism and multiple value expression." Square bracket insertions are mine.

However, even acknowledging some of the technical difficulties underlying DMV Bunse et al., (2015:95) still argue that:

"DMV can support a better understanding of beliefs, motivations and socio-demographic aspects that influence choices and actions by local people in relation to the environment. Consequently, it can provide a different and innovative approach that does not only facilitate shared understanding of the human-landscape relationships but also fosters collective management of common values."

Table 7.2 Economic valuation approaches and techniques

Valuation Approach	Valuation Methodology	Advantages and Disadvantages of Valuation Techniques
Market Cost	<i>Avoided Cost:</i> ES valued on the basis of costs avoided i.e. prohibiting the degradation or damage of environmental benefits	Mismatches can arise between the likely benefits of intervention compared to original benefits leading to misleading WTP results. Applies the precautionary principle. Can estimate indirect-use benefits
	<i>Replacement Cost:</i> valuation is based on the cost of replacing lost natural system services with artificial substitutes	Risk of over-estimation, and cannot estimate non-use values. Few available studies to verify the validity of the approach. On the other hand, it is useful for the estimation of indirect use benefits in the absence of available ecological data
	<i>Production Function:</i> value of ecological function with regards to economic output effects (i.e. productivity) or enhancement of income. Changes in ES quality and quantity on human-wellbeing	Not able to assess non-use values. Difficult to derive data about changes in ES. Widely employed in the contexts of coastal and wetland ecosystems
Market Price	<i>Market:</i> based on Willingness to Pay (WTP). Frequently used for provisioning services.	Requires market data (questionable reliability), and policies may distort market prices. However, market prices reflect personal WTP and market price data is relatively easy to obtain.
Revealed Preference (observations of individual choices in current markets related to the ES that is the subject of valuation)	<i>Travel Cost:</i> survey method valuing site-based facilities. WTP for environmental benefits at particular locations	<i>Travel cost:</i> data intensive, it does not estimate non-use values and complex journeys are problematical. However, it is widely used and used in developing countries for assessing ecotourism
	<i>Hedonic Pricing:</i> valuations based on implied WTP via purchases in related markets – mainly labour and property	<i>Hedonic pricing:</i> data intensive, it does not estimate non-use values, and income-level restricts choices whilst surrogate markets must be a good reflection of values. However it can value the impact of some ES on land values
Stated Preference (survey-based presenting hypothetical scenarios asking participants to place a monetary value on the achievement or acceptance of environmental change)	<i>Contingent Valuation:</i> WTP or willingness to accept (WTA) compensation for alterations in ES. Respondents can name an amount they would pay (classical CV), or are asked to say whether they would pay a specific amount (di/polychotomous choice) or select an amount from several options (Choice Modelling).	<i>Contingent Valuation:</i> suffers from several sources of bias, inconsistent preferences, is costly and labour intensive to develop and can miss non-trivial information. However, it is able to estimate option and existence values.
	<i>Choice modelling:</i> involves more elaborate sets of scenarios (or choices) from which participant select their preferred alternatives based on a set of choice attributes. Choices constructed to reveal the marginal rate of substitution between a specific attribute and the trade-off item.	<i>Choice Modelling:</i> hypothetical bias and the choices can be complex where attribute numbers are high. However, compared to standard CV the experimenter has much more control, the statistics are more robust, attribute range is greater and the method suffers less from respondent strategic behaviour.

Table 7.2 *Contd.*

Valuation Approach	Valuation Methodology	Advantages and Disadvantages of Valuation Techniques
Value Transfer	<i>Benefit Transfer</i> : transference of values at one location (study site) to another location (policy site) of which there are four types: unit BT, adjusted BT, value function transfer and meta-analytic transfer	Large number of uncertainties not wholly accounted for between study and policy locations. Transfer of values from one context to another is difficult. Nevertheless, it is a quick and cheap method.
Participatory Valuation	<i>Deliberative monetary valuation (DMV)</i> : combines states preference methods with deliberative processes from political science, involving small groups of participants in reflective iterative dialogues. May use surveys or non-econometric approaches such as citizen juries.	<i>DMV</i> : Can elicit cultural/societal and communal contextual values. Transcendental and other-regarding values may be elicited if prompted through the deliberative process. Less bias, values are constructed in a social process. Inclusive of all stakeholder groups, but depending on the power-relations of stakeholders involved some value preferences may be articulated more forcefully than others. This can affect the ability of DMV to address value incommensurability and aggregation. Quite resource and time intensive as well as requiring a large sample size.
	<i>Deliberative Multi-criteria Analysis (DMCA)</i> : involves stakeholder groups coming together to generate formalised criteria against which to evaluate the non-monetary (as well as sometimes monetary) costs and benefits of particular management options as a platform for decision-making.	<i>DMCA</i> : Can elicit cultural, societal and communal contextual values, more transcendental and altruistic values are unlikely to be elicited unless prompted during the process of deliberation. Provides both individual and group values. The process of DMCA can vary in complexity and can thus require (in some cases) considerable expertise in facilitation, experimental design and statistical analysis.
	<i>Participatory modelling</i> : stakeholders are involved in designing and contributing to the content of analytical models that relate ES and their benefits to different spatial and temporal scenarios.	<i>Participatory modelling</i> : Can elicit both cultural/societal and communal contextual values. The structured process has the potential to restrict the values elicited, more altruistic and transcendental values may be revealed via additional deliberative exercises. Values produced reflect the relative importance of the parameters identified and their relationship to and with each other
	<i>Participatory mapping/GIS</i> : stakeholders generate physical and/or digital maps to highlight particular landscape features that they consider to be of specific value/significance or problematical in some way. Maps are usually constituted from a number of layers which can include photos, videos, drawings etc.	<i>Participatory mapping/GIS</i> : In particular, this process elicits communal contextual values. If the particular landscape features identified have wider significance then the process may also generate cultural/societal values. The group-based approach means that the features identified as important need not be made commensurable across a single metric or aggregated through an arithmetic process. Resources required are scalar dependent, but a level of expertise in GIS is necessary.

Table 7.2 *Contd.*

Valuation Approach	Valuation Methodology	Advantages and Disadvantages of Valuation Techniques
Non-monetary Deliberative and Participatory Approaches	Focus groups, Participatory Action Research (PAR), Health-based, Q-methodology: These are a set of group-based methods that are both participatory and deliberative, and seek to obtain information regarding human-nature relationships. PARs were developed specifically for use in developing countries to elicit local knowledge and enable local people to participate in decision-making. Health-based measures relate valuations to factors that affect quality of life and human-wellbeing. Q-methodology is a means of assessing the subjectivity of people's views and values.	Overall, able to provide values regarding biodiversity, provisioning, regulating and cultural services, and they enrich the qualitative components of value. Although they require literate participants, new data collection, trained individuals and can be affected by local nuances. Protocols can be adjusted to illiterate individuals; values can be aggregated to the scale required and in some cases they can be relatively straightforward to undertake. Engage a wide-range of stakeholders and are conveyable to policy makers.
Psychometric	Psychometric (deliberative) – <i>Value Compass</i> : this method relies on participants ranking or rating the importance of their individual transcendental values, and then discussing the degree to which these individual ratings/rankings might reflect and be of importance to the community, society and wider culture. In other words, how does one's individual value compass relate to, reflect and compare to a more 'society-wide' value compass. <i>Subjective wellbeing indicators</i> : particular useful for addressing the extent to which, and assessing how, specific places contribute to individual wellbeing and thus highly relevant for developing the scope of cultural ecosystem services. Uses a quantitative non-monetary metric.	<i>Value Compass</i> : Can elicit transcendental, individual, communal as well as cultural/societal values. The values generated are considered separately and compared but not aggregated. <i>Subjective wellbeing indicator</i> : Can elicit communal, societal and cultural values. The values provided can be considered separately or averaged and aggregated. Can be time consuming and does require statistical expertise, as well as a large enough sample group.

Adapted from Mendelsohn and Olmstead (2009), Lui et al., (2010), Turner et al., (2010), Pascual and Muradian et al., (2012), Christie et al., (2012), Liekens et al., (2014), Bunse et al., (2015) and Kenter et al., (2015)

In arguing for a “choice democratisation” turn in DMV (i.e. a process that supports value pluralism, value articulation and inter-subjective understanding) Lo and Spash (2013:784) agree with this perspective, stating that:

“As an institution, DMV has the potential to contribute to the social importance given to the act of valuing and any money values articulated. Meaning is assigned to monetary values through a process of cooperative engagement and is part of what is being sought by the deliberative institution. There is no need to rigidly envisage the social act of paying as universally fixed or always being a trade-off [...] we argue for DMV to be reconceptualised as a mutual agreement resulting from an interactive process involving contestation of discourses.”

Despite these substantial criticisms, proponents of ESV underline its capability to manage trade-offs between ESs and alternative management regimes (Liu et al., 2010). From this perspective it is important to bear in mind that although the relevant ESs need to be accounted for, valuing bundles of ES does not mean attempting to value everything, but

instead the value of the marginal changes associated with management alternatives (Dendoncker et al., 2014). Others have suggested that ESV can be used to assess the different ‘capital’ contributions to human-wellbeing over time (Pascual and Muradian et al., 2012). Moreover, ESV is often cited as an essential element in the design of institutional and market-based instruments (e.g. payments for ecosystem services) for natural resource management (Pascual and Muradian et al., 2012).

Increasingly, ESV forms part of the evidence base that aids and informs policy and decision-making processes (Liu et al., 2010; de Groot et al., 2012; Pascual and Muradian et al., 2012), although the extent to which this is actually the case has recently been called into question (Laurans et al., 2013). Nevertheless, and particularly in times of austerity, an important function attributed to ESV is as a tool to identify where limited conservation funds, which are often derived from public monies, may be best targeted to achieve the most credible gains (de Groot et al., 2012). Finally, in more general terms, it has been pointed out that ESV is useful as a kind of marketing device highlighting the relevancy of ES to society (Liekens et al., 2014), with the potential to influence policy and planning (Bunse et al., 2015) whilst also making a wider contribution to the sustainability agenda (Dendoncker et al., 2014). This supposition, embodied in the TEEB process (Box 7.1), rests on the premise that as “economics” represents the dominant normative language of politics, articulating the economic valuation of ecosystems is more likely to feedback onto society to better regulate and inform human-nature relationships and decision-making⁷ (Kumar et al., 2013).

7.5 Valuations – We Still Have Some Way To Go

In some quarters these views have been given short shrift, for example, as the ecological economists Spangenberg and Settele (2016:100) advance:

“The three promises of economic valuation, raising awareness in polity, saving biodiversity by internalising external cost, and contributing to better decisions are assessed and turnout to be more than questionable.”

Critical reflections on ESV have also drawn attention to the fact that the values valuation exercises generate are neither neutral nor independent of their institutional and social context, in other words, they are not “value free” as it were, but instead evolve out of complex social processes or “value articulating institutions” to use Vatn’s (2005) phrase that frame their orientation (Kallis et al., 2013:100):

Box 7.1 TEEB Progressions

Purpose and Innovation: TEEB was established in 2007 at the Potsdam G8(+5) meeting in order to evaluate the economic consequences of continued biodiversity loss (at all scales); make the economic case for continued conservation efforts, and supply policy makers with the tools to make effective decisions (Ring et al. 2010; TEEB, 2012). TEEB propagated a major re-think regarding the current economic paradigm, whilst acknowledging the need for an economic framework and approach to understand and manage biodiversity loss and ecosystem services degradation (Ring et al., 2010; TEEB, 2012). More latterly, as Hedden-Dunkhorst et al., (2015:41) highlight TEEB can be seen to be an:

“...instrument for the implementation of the CBD's Strategic Plan for Biodiversity 2011–2020. Specifically, Target 2 of the Strategic Plan stipulates that “By 2020, at the latest, biodiversity values have been integrated in to national and local development and poverty reduction strategies and planning processes and are being incorporated into national accounting, as appropriate, and reporting systems.”

In this respect, although TEEB was not original conceived of as a national accounting strategy it nevertheless provides a useful framework, suite of valuation methodologies as well as a reservoir of information to enable national accounting of ecosystem services (Hedden-Dunkhorst et al., (2015).

Outcomes and impacts: The importance of TEEB lies in its wider goal of highlighting the economic aspects of biodiversity, emphasizing the increasing economic burden created by continued biodiversity loss through its impact on ecosystems and their supply of goods and service, and furthermore, expounding the necessity for an interdisciplinary and international cooperative enterprise (Ring et al., 2010). The TEEB initiative is a three phase programme (Hedden-Dunkhorst et al., 2015). Phase 1 completed in 2008 with a synthesis report to the Convention on Biological Diversity highlighting the continued global decline of ecosystem services that was documented in the MA. Phase 2 of TEEB culminated in 2011 with four principal policy outputs (TEEB, 2012; Hedden-Dunkhorst et al., 2015).

These documents were produced for: (i) national policy makers (focusing on decision-making strategies for national and supra-national biodiversity and ecosystem policies); (ii) local policy makers (presenting valuation estimates at the local level for key ESs with methods suitable to design and implement incentive mechanisms); (iii) business users (identifying modes of assessing and managing business impacts on biodiversity and ecosystems) and (iv) individual and consumer organizations (emphasizing supply-chains in regulating producer behavior), alongside TEEB Ecological and Economic foundations (TEEB, 2012). The current phase 3 of TEEB's programme is focusing on ‘facilitation’ and ‘implementation’ in relation to specific ecosystem/biome assessments and valuations, the mainstreaming of TEEB into other related initiatives (e.g., green economy) and tools (e.g. natural capital accounting), and supporting country-level TEEB assessments (Hedden-Dunkhorst et al., 2015).

Critiques: There have been a number of criticisms levied against the TEEB initiative over the course of its duration, some of these have questioned the validity and legitimacy of the valuation approaches it has adopted to assess ecosystem services, and in particular, the emphasis placed on monetary valuation – what some refer to as the increasing ‘financialization of the environment’ (Sullivan, 2013; Hedden-Dunkhorst et al., 2015). Along similar lines, some have argued that TEEB is a driver of neoliberalist capitalism that functions to translate the ‘natural’ into market realities and the ‘virtual’, as MacDonald and Corson (2012:159) argue:

“In reducing the complexity of ecological dynamics to idealized categories TEEB is driven by economic ideas and idealism, and, in claiming to be a quantitative force for morality, is engaged in the production of practices designed to conform the ‘real’ to the virtual [...] We argue that TEEB's rhetoric of crisis and value aligns capitalism with a new kind of ecological modernization in which ‘the market’ and market devices serve as key mechanisms to conform the real and the virtual.”

Finally, another criticism that has been put forward states that the focus of TEEB has been too overly concerned with conceptualizing ecosystems and biodiversity in terms of natural capital, with the consequence being that it has ignored other forms of ecosystems and genetic diversity that are produced or co-produced by and with humans (Tisdell, 2014). Tisdell (2014) also argues that TEEB does not sufficiently highlight the fact that not all natural ecosystems and genetic material may be beneficial human assets, and in this regard may actually act as economic liabilities. Nevertheless, there remains a general consensus that TEEB continues to represent an important step on the journey to a wider acceptance and recognition that the ecosystem service's story does require an economic narrative, but one that is sympathetic to and acknowledges the complexities and critiques widely aired in academic and policy circles (Ring et al., 2010).

“...monetary valuations are not isolated phenomena of methodological interest, but part of broader commodification processes, which involve symbolic, institutional, intellectual, discursive, and technological changes that reshape the ways humans conceive and relate to nature.”

Framings that suggest monetary valuation implies notions of commodification and privatization, leading to some of the criticisms we have already discussed, are for some entirely misguided and fundamentally misjudge the purpose of valuation exercises and their inclusion of monetary estimates⁸, as Constanza et al., (2014:154) contend:

“It is a misconception to assume that valuing ecosystem services in monetary units is the same as privatizing them or commodifying them for trade in private markets. Most ecosystem services are public goods (non-rival and non-excludable) or common pool resources (rival but non-excludable), which means that privatization and conventional markets work poorly, if at all. In addition, the non-market values estimated for these ecosystem services often relate more to use or non-use values rather than exchange values. Nevertheless, knowing the value of ecosystem services is helpful for their effective management, which in some cases can include economic incentives, such as those used in successful systems of payment for these services. In addition, it is important to note that valuation is unavoidable. We already value ecosystems and their services every time we make a decision involving trade-offs concerning them. The problem is that the valuation is implicit in the decision and hidden from view.”

The reality is likely to lie somewhere between these two positions, and the research seems to back this up. Recent work on commodification has suggested that there are “degrees” of commodification, so that rather than being a binary phenomenon as normally presented commodification is actually more complex, and that it would be better characterised as a dynamic continuum (Hahn et al., 2015)⁹. For others though it is a question of language (Neuteleers and Engelen, 2015). From this perspective the concern is less to do with the possibility of real (and perhaps inappropriate) commodification, and more to do with the potential consequences of how the language of monetary valuation can promote a ‘commodification in discourse’. Here the issue is how a strand of discourse, by gaining widespread ascendancy, essentially becomes (and is viewed as) the only conversation in-town: edging-out other equally valid forms of discussion and infusing the narrative in such a way that it potentially reproduces the same moral ambiguities as real commodification (Neuteleers and Engelen, 2015).

Quite naturally, the question of when and under what circumstances monetary valuation is appropriate gains currency. For example, Kallis et al., (2013) offer four selection criteria for consideration, namely: “environmental improvement”; “distributive justice and equity”; “maintenance of plural value articulating institutions” and “confronting commodification under neo-liberalism”; wherein, if selected monetary valuation should be able to demonstrate how each criterion is met. Harking from a similar ethos, Spangenberg and Settele (2016:107)

make the case that within limited and specified parameters economic valuation can be “defendable”¹⁰, specifically, in the instance where:

“...valuation would be helpful to distinguish the cost of options if and only if the different economic cost and benefits are the decisive criterion to choose one of several options for which no overriding priority has been identified based on other types of values – value pluralism, in particular in economics, is an essential element of democracy and requires each agent to reflect her role, interests and positions.”

A similar claim is made by the pro ESV camp (e.g. Turner et al., 2010); namely, that monetary valuation in particular has necessary and important heuristic functions in contexts where multiple trade-offs exist between different management regimes choices (Turner et al., 2010).

Misapplication can also arise from confusion. Davidson (2013), for example, cites that the problem of inappropriate economic valuation lies in the conflation of the relationship between ecosystem services, intrinsic value and existence value. The argument that Davidson advances revolves around the issue of compatibility, and specifically, that intrinsic value and existence value are mutually exclusive, and that whilst existence value may be compatible with any ES applying the same logic to intrinsic value remains conceptually (philosophically) flawed. Nevertheless, as Davidson is at pains to point out, intrinsic value, as benefits to nature, can still be captured economically although this depends upon the moral stance one takes¹¹ (Davidson, 2013).

Pricing is another thorny issue in ESV. As some of those interviewed by Bauler and Pipart (2014) expressed, the boundary between economic value¹² and market price¹³ is easily blurred when monetary valuation enters the fray, with the logic soon being advocated that environmental finance markets are the universal panacea to environmental problems. For Parks and Gowdy (2013) part of the problem has been the dominant use of shadow pricing¹⁴ to calculate the social price of environmental goods, as well as the general application of welfare theory¹⁵ to environmental valuation via CBA.

In the eyes of Parks and Gowdy (2013) CBA suffers two central flaws: Firstly, its value theory foundations and secondly the dubious nature of the numbers it generates. The authors highlight a litany of common problems ensuing from these weaknesses including: (i) the generation of economic values that underestimate the “true” value of ecosystems and fail to capture aspects of sustainability or distributive fairness; (ii) the production of price values that reflect the dominant structural power narratives of the day; (iii) value monism (i.e. a lack of value pluralism); (iv) a misguided addiction to assessing marginal values and, (v) in more

general terms its fundamental utilitarianism which can be at odds with the ethics of many individuals (Parks and Gowdy, 2013).

These issues have the potential to create an additional set of problems; specifically, the idea that monetary valuation, as an extrinsic motivator¹⁶, and by extension the incentive programmes that are often designed based on valuation exercises, can displace peoples' intrinsic motivations¹⁷ for conserving biodiversity and ecosystem goods and services leading to an overall decrease in the 'demand and support for environmental protection' (Neuteleers and Engelen, 2015; Rode et al., 2015). This so-called "crowding-out"¹⁸ effect has been identified, but so too, albeit to a lesser extent, has a "crowding-in"¹⁹ effect (Rode et al., 2015).

Dissenters of the 'standard' ESV approach advocate value pluralism arguing against a one-size-fits-all framework (Salles, 2011; Brondízio and Gatzweiler et al., 2012; Chan et al., 2012). For example, Parks and Gowdy (2013) have recently called for researchers to expend far more effort in pursuing social valuation, describing it as the next 'frontier' in ESV. Echoing this sentiment, Dendoncker et al., (2014) have advocated for a tripartite valuation system jointly focused on assessing ecological, social and monetary dimensions, whilst pressing for greater use of non-monetary social valuation approaches, a view fervently endorsed in Spash and Aslaksen (2015) who demand that the social ecological aspects of ecosystems to be explored far more richly and extensively. These clarion calls have begun to bear fruit, for instance, Kenter et al., (2015:97) have recently developed a shared and social values²⁰ approach to the appreciation of ecosystem services, noting that:

"Seven distinct but interrelated and non-mutually exclusive types of shared and social value have been identified [namely: transcendental, cultural/societal, communal, group, deliberated and other-regarding values, and value to society], and the relationship between individual and shared values conceived of as a dynamic interplay, where values can be considered at multiple levels (individual, community, society)." (Bracketed wording is the author's insertion)

Making clear why this development is important they say:

"In general, the elicitation of shared and social values goes beyond the narrow elicitation of individual monetary valuations to incorporate common notions of social goods and cultural importance through social processes that can incorporate a broad set of individual and shared meanings and concerns." (pg. 96)

7.6 Valuation – Moving Forwards

Clearly, progress is being made but there is still much to do and still room for improvement in how ESV is approached and used, particularly for non-marketed ecosystem services (Perrings, 2014). However, in all probability, a pluralistic approach to ESV is most

likely to deliver more valuable and realistic information concerning biodiversity and ecosystem goods and services as a means to inform environmental management and policy decisions, as Spangenberg and Settele (2016:108) remark:

“As monetary valuation of non-market goods is questionable at best and misguided at worst, our suggestion is to distinguish real (market) and virtual costs and values, and focus economics on measurement of the real thing. Then economic valuation can be helpful as an illustration of one element: economic cost of options, complementing other methods in deliberative approaches to ecosystem management based on value pluralism.”

Ultimately, when we think about valuation – valuing biodiversity and ecosystem services – we need to be clear why we are doing it and what it is we are trying to achieve. We ought not to be axiomatically opposed, on the grounds of idealism, over the integration of economics and biodiversity, but rather, be pragmatic enough to understand the benefits that such an integration can offer in helping to improve the way we manage our interactions with nature, as Charles Perrings (2014:78-79) remarks in his book *Our Uncommon Heritage*:

“...the concept of ecosystem services has nothing to do with market ideologies. Nor does it imply the commodification of nature. Some ecosystem services that are important today are certainly provided through markets. Many foods, fuels and fibers are in this category. Other services will surely be provided through markets in the future [...] But there are many ecosystem services that are not now, and never will be, provided through markets. What the concept of ecosystem services does is to give us a way of characterizing the interests people have in their environment. Whether an ecosystem service can be provided through the market depends on its properties, and not the fact that it is important to people [...] By distinguishing between the price and value of marketed ecosystem services, it explains where markets are likely to succeed in signalling the importance of biodiversity and where they are likely to fail. By identifying the value of services for which no price exists, it indicates what we lose when we allow those services to erode.”

A view shared by de Groot et al., (2012:60):

“Values in monetary units will never in themselves provide easy answers to difficult decisions, and should always be seen as additional information, complementing quantitative and qualitative assessments, to help decision makers by giving approximations of the value of ecosystem services involved in the trade-off analysis. However, even if we do not have a ‘precise’ value for, for example, water purification we can assess broadly how valuable it is as an ecosystem service relative to other services, or the costs of the absence of that service, in a particular decision making situation. Note that expressing values in monetary units can be a time.”

Notes

1. Cost-benefit analysis is an approach or technique that systematically assesses the strengths and weaknesses of alternative options, comparing the costs and benefits of a project or policy. Its purpose is to identify the most welfare maximising choices (i.e. those with the lowest cost-benefit ratio), a strategy that can also increase economic efficiency. In attempts to evaluate the potential positive and negative impacts of a project cost-benefit

analysis may look at effects on users, non-users, externalities and social benefits. In general, cost-benefit analysis seeks to place all 'relevant' benefits and costs on a standard temporal platform using time value of money estimates, in other words, applying discounting rates to judge present and future value (Boardman et al., 2013).

2. According to Spangenberg and Settele (2016):

Ideal values (e.g., ideas, notions such as justice and liberty) are Platonic in origin. These values relate to abstract existing objects or objective and real things that are independent, that are outside us and our thinking. In other words, they are non-physical, in some sense eternal, beyond human perception and causally-inert. What determines their value is the content of goodness imparted on them.

Real/Objective values are values emphasising rationality, they are embedded in scientific naturalism. These are values related to real world objects, in opposition to Platonism, scientific approaches suggest that such objects are the only relevant things to be considered. We come to know about these objects (and through that process of knowing understand their value) via empirical measurement based on sensory experience articulated through the prism of rational-logical mediation. Their values are given, being determined by their own particular character subject to natural laws.

Subjective values are values attributed based on individual and social perception of real world objects. Because values are attributed they acknowledge that there is a subject assigning value in a socially constructed context subject to specific social norms and value systems. These values are therefore not fixed nor are they absolute.

3. According to the online Stanford Encyclopaedia of Philosophy (2016):

“Values [...] are sometimes said to be incommensurable in the sense that their value cannot be reduced to a common measure.” There are potentially three reasons for this: “incommensurability in terms of restrictions on how the further realization of one value outranks realization of another value” or “values are incommensurable if and only if there is no true general overall ranking of the realization of one value against the realization of the other value”, and finally, “in some conflicts of values, there is no true ranking of values.”

4. Deliberative valuation processes are based on the organic emergence of values emanating from social interaction and communication; as such they do not assume that individuals hold fixed and pre-existing values for ES and biodiversity. According to Brondizio and Gatzweiler (Chapter 4 TEEB 2012:164) deliberative valuation approaches attempt to:

“...turn the value elicitation process into a preference-constructing process in order to deal with the issue that people do not hold pre-determined preferences towards the environment and that such preferences should be well informed and deliberately derived.”

5. According to Pascual and Muradian et al., (Chapter 5 TEEB 2012:188) TEV is defined as:

“...the sum of the values of all service flows that natural capital generates both now and into the future – appropriately discounted. These service flows are valued for marginal changes in their provision. TEV encompasses all components of (dis)utility derived from ecosystem services using a common unit of account: money or any market-based unit of measurement that allows comparisons of the benefits of various goods”.

6. TEEB standard for The Economics of Ecosystems and Biodiversity (TEEB, 2012)

7. As the TEEB (2010) Mainstreaming the Economics of Nature: A synthesis of the approach, conclusions and recommendations of TEEB states:

“The invisibility of many of nature’s services to the economy results in widespread neglect of →natural capital, leading to decisions that degrade →ecosystem services and →biodiversity. The destruction of nature has now reached levels where serious social and economic costs are being felt and will be felt at an accelerating pace if we continue with ‘business as usual’.” (pg.25)

8. de Groot et al., (2012) make a very similar point to Costanza et al., (2014) in their global analysis of ecosystem service valuation estimates:

“We also want to make clear that expressing the value of ecosystem services in monetary units does not suggest that the values should be used as a basis for establishing prices and does not

mean that they should be treated as private commodities that can be traded in private markets. Most ecosystem services are public goods that cannot (or should not) be privatized. Their value in monetary terms is an estimate of their benefits to society—benefits that would be lost if they were destroyed or gained if they were restored. Thus, monetary valuations of the importance of ecosystem services to society can serve as a powerful and arguably essential communication tool to inform better, more balanced decisions regarding trade-offs involved in land use options and resource use. Ecosystem service valuations are best seen as complementary to conventional decision-making frameworks, in which the positive and negative externalities of the use or loss of many environmental goods and services are still not, or insufficiently acknowledged. Monetary valuation can help to make these externalities visible and complement the insights on the role and importance of nature gleaned via other quantitative and qualitative measures.” (pg.57)

9. According to Hahn et al., (2015) commodification in this context has the following meaning:

“Commodification of biodiversity and ES means, broadly speaking, the expansion of market trade in to previously non-marketed areas of the environment. This is often described as a process related to the idea of commensurability underlying monetary valuation.” (pg. 75)

However, in contrast to the consensus opinion that commodification proceeds by a sequential series of stages, Hahn et al. argue that:

“...these stages need not be consecutive and the process is not necessarily unidirectional or irreversible [...] hence we use degrees rather than stages. Based on Muradian et al. (2010:1206), we define the degree of commodification as the extent to which the value of biodiversity or an ecosystem service has become a tradable commodity.” (pg. 75)

The 7 degrees of commodification identified (see article for full description) are based on the extent of policy integration.

10. Ultimately, to be considered defensible economic valuation according to Spangenberg and Settele (2016:107) should have the following characteristics:

“To be *relevant* to decision-making on biodiversity, ecosystems and the services they co-produce economic valuation must be purpose specific e.g., market-based prices for real or hypothetical transactions can be aggregated to calculate damage cost, avoidance cost, restoration cost etc. [...] They must also be *case-specific*, as the results are based on the subjective value attribution of (often local) stakeholders, without a universal market value as a reference. [...] To be *effective*, the valuation method chosen must be *relevant*, referring to budget relevant cost [...] or to the opinions and preferences of the decision-makers’ constituencies. They ought to be *adequate* for the decision to be made and the object to be valued [...] To be credible in the longer-term results have to be *cautiously interpreted*; all methods make assumptions that have to be reflected and [...] spelt out.”

11. If one adopts a deontological position, that is, concern for the moral status of other agents or beings (i.e. intrinsic value) then individual actions/decisions should be made on the basis of personal moral duty and concern for the rights of others. In neo-classical economics decisions and preferences based on deontological criteria are termed lexicographic. In other words, in this case moral status (or intrinsic value) will always be preferred over any other set of preference options. Therefore from a deontological perspective intrinsic value is incompatible with economic valuation as by definition other alternatives must have the potential of being favoured. However, from a consequentialist perspective, which observes morality not so much as de facto present in the agent but determined by the outcomes of the actions agents undertake, intrinsic value is perceived as that relating to ‘states of affairs’. From this standpoint, as alternative states of affairs might arise for which agents have greater preference economic valuation is entirely commensurate with intrinsic value (Davidson, 2013).

12. Economic value is the benefit an individual economic agent derives from a good or a service (Hanley et al., 2013).

13. Market price refers to the economic price a good or service is attributed when offered in a marketplace (Hanley et al., 2013).

14. The concept of shadow pricing is employed when the assumptions of the general equilibrium model (i.e. the theory that attempts to explain supply, demand and price allocations across the whole economy for multiple interacting markets by postulating that a set of prices exist that can produce an overall or ‘general’ equilibrium) are not met such as perfect information or complete markets. In this case market values differ from the

theoretically 'ideal' or 'socially efficient' set of prices. Shadow prices are based primarily on single markets (or partial equilibrium estimates) for specific programmes. Due to the fact that shadow prices for environmental externalities are not 'socially observable' they are calculated from distance function approaches, in other words, by how far an individual agent is away from the so-called efficiency frontier. Assumptions underlying shadow price determination are restrictive (Parks and Gowdy, 2013).

15. Welfare economics (theory) is concerned with the application of microeconomics (i.e. how individual agent behaviours (i.e. choices/decisions) affect supply and demand) to assess wellbeing (or welfare) at the economy (or aggregate) scale. Welfare economics assumes that competitive markets operate in a Pareto efficient manner, in other words, resources are maximally allocated to each individual such that any alternative distribution would lead to at least one individual being made worse-off. Furthermore, and in addition to demonstrating Pareto efficiency, that any outcome that is Pareto optimal can function as a competitive equilibrium such that redistribution could be achieved without affecting prices (Anderson, 2013).

16. Extrinsic motivations refer to those activities that people undertake purely for instrumental value, in other words, to achieve a separable outcome usually in terms of a material or monetary gain, or perhaps, through benefits derived in a non-material way (Rode et al., 2015).

17. Intrinsic motivations refer to those activities that people undertake for pure enjoyment and satisfaction – these can be of two non-mutually exclusive types: a pro-social (e.g., social relations with other individuals or communities) and pro-nature (e.g., instrumental (benefits of ES) and non-instrumental value of nature (existence value)) (Rode et al., 2015).

18. Crowding-out refers to how monetary valuation, discourse, regulations and incentives can effectively undermine or erode peoples' intrinsic motivations for action in terms of engaging with ecosystem and biodiversity conservation activities, perhaps as a consequence of reduced satisfaction, control aversion, frame shifting via focusing on economic imperatives etc. (Rode et al., 2015).

19. Crowding-in is essential the opposite process to crowding-out, here monetary valuation, regulations and incentives reinforce peoples' intrinsic motivations to engage in environmental protection, perhaps as a consequence of increased internal satisfaction, or social standing, or positive attitudes and trust (Rode et al., 2015).

20. According to Kenter et al (2015):

Shared values:

“...used to refer to guiding principles and normative values that are shared by groups or communities or to refer to cultural values more generally [...] the conception of shared values as implicit, communal or public values, and of shared values as values that are brought forward through deliberative social processes” (pg.87-88)

Social values:

“...can refer to the values of a particular community or the cultural values and norms of society at large, but can also be used to refer to the public interest, values for public goods, 'altruistic' values and feigned altruistic values, the values that people hold in social situations, contribution to welfare or well-being, the willingness-to-pay (WTP) of a group, the aggregated WTP of individuals, or values derived through a social process [...] or as non-monetary place-based values.” (pg.88)

Shared social values also have a diversity of meanings for example:

“‘shared social’ was used to indicate group deliberated values reflecting that societal context [...] Shared social values were regarded as the outcome of processes of effective social interaction, open dialogue and social learning. From this perspective, shared social values were closely allied to shared meanings...” (pg.88)

Chapter 8: The Value Of Nature

“The services of ecological systems and the natural capital stocks that produce them are critical to the functioning of the Earth’s life-support system. They contribute to human welfare, both directly and indirectly, and therefore represent part of the total economic value of the planet.” (Costanza et al., 1997, pg.253)

“Economists do agree that, in order to determine society’s willingness to pay for the benefits provided by ecosystem goods and services, one needs to measure and account for their various impacts on human welfare [...] As long as nature makes a contribution to human welfare, either entirely on its own or through joint use with other human inputs, then we can designate this contribution as an “ecosystem service” [...] Valuation should show how changes in these services affect human welfare, after controlling for the influence of any additional human provided goods and services.” (Edward B Barbier, *Capitalizing on Nature: Ecosystems as Natural Assets*, 2011, pg.32-33)

Following on from Chapter 7, the purpose of this section is not to present an in-depth coverage and review of valuation exercises, plainly the literature on that topic is too vast; but rather, to put forward a snapshot, an illustration, of more recent developments and directions in the appraisal of ecosystem services.

8.1 Recasting Old Debates

In their ground-breaking but highly controversial *Nature* paper, Costanza et al., (1997) suggested that the global economic value of ecosystem services, to the world economy, was worth (on average) US\$33 trillion yr⁻¹; more specifically the authors stated:

“We estimated that at the current margin, ecosystems provide at least US\$33 trillion dollars’ worth of services annually. The majority of the value of services we could identify is currently outside the market system, in services such as gas regulation (US\$1.3 trillion yr⁻¹), disturbance regulation (US\$1.8 trillion yr⁻¹), waste treatment (US\$2.3 trillion yr⁻¹) and nutrient cycling (US\$17 trillion yr⁻¹). About 63% of the estimated value is contributed by marine systems (US\$20.9 trillion yr⁻¹). Most of this comes from coastal systems (US\$10.6 trillion yr⁻¹). About 38% of the estimated value comes from terrestrial systems, mainly from forests (US\$4.7 trillion yr⁻¹) and wetlands (US\$4.9 trillion yr⁻¹).”

At the time, perhaps not unsurprisingly, the paper garnered a wide amount of press coverage as well as a substantial torrent of criticism, especially from mainstream economists (e.g., Toman, 1998 and Bockstael et al., 2000). Their reaction to the article questioned not only the underlying methodologies used to arrive at headline grabbing figures such as “US\$33 trillion dollars”, but also the intellectual foundation and practical purpose (i.e. policy benefits) of the work. For example, Michael Toman a prominent economist, who is himself not unsympathetic to the idea of ‘valuing’ nature’s services, remarked in a *Special Issue of Ecological Economics*:

“Leaving aside [...] technical quarrels about the estimates in the paper, the fundamental problem is that there is little that can usefully be done with a serious underestimate of infinity. The paper asserts in its first paragraph that it is seeking

to estimate the ‘marginal’ value of ecosystem services, but the now-famous figure of \$33 trillion/year does not reflect such an incremental calculation. Instead, what has been done is to estimate total social surplus by taking selected average values per unit (e.g. hectare) and multiplying by all the units in the biosphere [...] So long as priorities must be set among competing claims for ecosystem protection and/or amelioration, it is necessary to understand how specific changes in different ecosystem states are affecting social interests and values [...] A simple point aggregation of ‘everything’, or a comparison of this aggregate with something like GDP (which is problematic on other grounds in any event), give no insights into either the directions of current changes in ecosystems and their services or the relative urgency of different changes.” (Toman, 1998:58)

For those involved in the original article there still remains a sense in which those earlier criticisms continue, and continue to “misrepresent”, or at least miss the point, of what they were attempting to demonstrate, which according to Costanza et al., (2014:157) is to communicate:

“...the relative contribution of natural capital now, with the current balance of asset types. Some of this contribution is already included in GDP, embedded in the contribution of natural capital to marketed goods and services. But much of it is not captured in GDP because it is embedded in services that are not marketed or not fully captured in marketed products and services. Our estimate shows that these services (i.e. storm protection, climate regulation, etc.) are much larger in relative magnitude right now than the sum of marketed goods and services (GDP).”

However, even if we agree with the critics and admit that the article suffered from a serious of significant economic flaws, putting those to one side for a moment, and with an eye to the bigger picture, the substantial contribution Costanza et al. (1997) made was not, in fact, the ‘showy’ big dollar estimates widely reported in the news; but, the globally significant and hitherto largely ignored, hidden and woefully undervalued input the biosphere makes in sustaining the human enterprise. Less profound perhaps, but by no means less important, was the article’s secondary effects of which two aspects are of particular note: First, the widespread debate it promoted led to the development of better and more diverse valuation methodologies (as we have seen in Chapter 7), and second, it paved the way, indeed it cried out for, similar types of analyses to be conducted in the future; and, over the intervening years ES valuations across all scales and ecosystem types have multiplied sharply (de Groot et al., 2012). Table 8.1 makes this point, for example, by displaying a compilation of global value estimates for 22 ecosystem services across 12 biomes drawn from recent valuation syntheses.

In a recent reappraisal of their 1997 study, though this time with the benefit of more up-to-date ES values and land-use change projections, Costanza et al., (2014) reported an estimated global value of ES in 2011 of US\$125 trillion yr⁻¹ or US\$145 trillion yr⁻¹ (when considering changes in ES estimates and not biomes): an amount virtually twice the value of global gross domestic product (GDP) and much of it contributed by ocean, coastal and wetland-systems.

Table 8.1 The maximum values for twenty-two ecosystem services in thirteen biomes (in Int.\$/ha/year, 2007 price levels [2009 price levels for Li and Fang, 2014])

Ecosystem Services	Open oceans	Coral reefs	Coastal systems	Coastal wetlands	Inland wetlands	Lakes and Rivers (Freshwater)	Tropical forests	Temperate and boreal forests	Woodlands	Grasslands	Cropland	Urban
Provisioning	22	20,892	7,549	8,289	9,709	5,776	9,384	1,736	862	715	372	
	102	55,724	2,396	2,998	1,659	1,914	1,828	671	253	1,305	3,984	
	106	57,953	2,492	3,117	1,726	1,991	1,901	698	264	1,358		
1. Food	22	3,752	7,517	2,600	2,090	196	1,204	1,204	203	82	302	
	93	677	2384	1,111	614	106	200	299	52	1,192	2,323	
	97	704	2,479	1,155	639	110	208	311	54	1,240		
2. Freshwater supply				4,240	5,189	5,580	875	455		602	50	
				1,217	408	1,808	27	191		60	400	
				1,266	424	1,880	28	199		62		
3. Raw materials	8	16,792	32	1,414	2,430		3,273	54	659	31	21	
	8	21,528	12	358	425		84	181	170	53	219	
		22,389	12	372	442		87	188	177	55		
4. Genetic resources		33,048		10			1,799				1,042	
		34,370		10			13					
							14					
5. Medicinal resources				301	99		1,782	23		1		
				313	103		1,504			1		
							1,564					
6. Ornamental resources		348		35	114				32			
		472			119				33			
		491										
Regulating	62	33,640	30,451	135,361	23,018	4,978	7,135	456	1,088	2,067	713	2,331
	65	171,478	25,847	171,515	17,364	187	2,529	491	51	159	970	921
	68	178,337	26,881	178,376	18,058	194	2,630	512	53	165		
7. Influence on air quality							957				61	492
							12					
							12					

Table 8.1 *Contd.*

Ecosystem Services	Open oceans	Coral reefs	Coastal systems	Coastal wetlands	Inland wetlands	Lakes and Rivers (Freshwater)	Tropical forests	Temperate and boreal forests	Woodlands	Grasslands	Cropland	Urban
8. Climate regulation	55 65 68	1,188 1,236	479 498	4,677 65 68	351 488 508		761 2,044 2,126	376 152 158	387 7 7	1,661 40 42	95 411	1,561 905
9. Moderating extreme events		33,556 16,991 17,671		9,729 5,351 5,565	4,430 2986 3,105		340 66 69					
10. Regulation of water flows					9,369 5606 5830	194	36 342 356	3			56	278 16
11. Waste treatment/water purification		77 85 88		120,200 162,125 168,610	4,280 3015 3,316	4,978 187	665 6 6	77 7 7	701	358 75 78	202 397	
12. Erosion prevention		153,214 159,343	25,368 26,383	755 3,929 4,086	2607 2,711		3,211 15 16	5 5	13 14	47 44 46	128 107	
13. Nutrient cycling			30,451	45 47	4,588 1,713 1,782		1,067 3 3	93 97			119	
14. Pollination							99 30 31		31 32		33 22	
15. Biological control	7	7			948 986		11 11	235 244			20 33	
Habitat	5 5	56,137 16,210 16,858	164 375 389	68,795 17,138 17,824	3,471 2,455 2,553		5,277 39 41	2575 862 896	1,277 1,327	298 1,214 1,263	101	
16. Life cycle maintenance			164 194 202	59,645 10,648 11,074	917 1,287 1,338		16 17		1,273 1,324		36	

Table 8.1 *Contd.*

Ecosystem Services	Open oceans	Coral reefs	Coastal systems	Coastal wetlands	Inland wetlands	Lakes and Rivers (Freshwater)	Tropical forests	Temperate and boreal forests	Woodlands	Grasslands	Cropland	Urban
17. Gene pool protection	5	56,137	180	9,150	2,554		5,277	2575	3	298	64	
	5	16,210	187	6,490	1,168		23	862	3	1,214		
		16,858		6,750	1,215		24	896		1,263		
Cultural	319	1,084,809	41,416	2,904	8,399	2,733	1,426	96	7	11	19	2,469
	332	108,837	300	2,193	4,203	2,166	867	990	7	193	82	5,740
		113,190	311	2,281	4,371	2,253	902	1,030		27		
18. Aesthetic information		27,317			3,906					167	15	2,016
		11,390			1,292					174		
		11,846			1,344							
19. Recreation and tourism	319	1,057,492	41,416	2,904	3,700	2,733	1,426	96	7	11	4	453
	332	96,302	256	2,193	2,211	2,166	867	989	7	26	82	5,740
		100,154	266	2,281	2,299	2,253	902	1,029		27		
20. Inspiration (art, design)					793							
					700							
					728							
21. Spiritual			21									
			22									
22. Cognitive development (education and research)		1,145	22					1				
		1,191	23					1				
Total	84	1,195,478	79,580	215,349	44,597	13,487	23,222	4,863	1,950	3,091	1,205	4,800
Economic Value (Total of Service Mean Values)	491	352,249	28,918	193,844	25,681	4,267	5,264	3,013	1,588	2,871	5,036	6,661
	511	366,339	30,073	201,598	26,708	4,438	5,474	3,135	1,652	2,986		

Notes:

Black (source: Kumar, 2012[2010]), Red (source: de Groot et al., 2012), Blue (source: Li and Fang, 2014), Green (source: Costanza et al., 2014)

Worryingly, the study also illustrated the severe effect the continued exploitation of natural resources and infringements on geosphere-biosphere systems has had since 1997, with high levels of ES losses resulting from land-use change impacts on ecosystem service provision and flows equivalent to US\$4.3–20.2 trillion yr⁻¹ (Costanza et al., 2014).

Although these values may seem astronomical, in reality, they are neither extreme nor plucked from thin air. For instance, in their global assessment of “synthetic green GDP” (i.e. the aggregated monetary and non-monetary ecosystem service values for each nation) for the year 2009 Li and Fang (2014) produced a similarly large figure of US\$149.61 trillion (in actual fact this is probably a conservative estimate). They also noted that the overwhelming contribution (approx. 75% equivalent to US\$112 trillion yr⁻¹) was provided by marine systems, particularly by coastal systems (US\$85.7 trillion yr⁻¹), and although terrestrial systems supplied just a quarter of the ecosystem service values much of this was derived from forest ecosystems (US\$16.3 trillion yr⁻¹) (Table 8.2, Li and Fang, 2014).

Table 8.2 Summary of global ES flows (source: adapted from Li and Fang, 2014)

Biome	Total global value (Int.US\$/yr × 10⁹)
<i>Terrestrial</i>	37,178
Croplands	1,195
Mixed croplands (60% croplands, 40% vegetation)	1,729
Mixed croplands (40% croplands, 60% vegetation)	2,240
Tropical Forest	6,147
Temperate Forest	8,777
Mixed tropical and temperate forest	1,819
Woodlands	4,754
Grasslands	2,470
Mixed woodlands (60% woodlands, 40% grasslands)	1,345
Mixed grasslands (60% grasslands, 40% woodlands)	1,345
Inland Wetlands	3,193
Freshwater (lakes and rivers)	1,436
Urban areas	8
<i>Marine</i>	112,431
Marine/ Open Ocean	26,736
Open Ocean	16,385
Coral reefs	10,351
Coastal systems	85,595
Seagrass/Algal beds	1,504
Shelf sea	119,675
Estuarine/Shore	3,167
Coastal wetlands	3,060
Total	149,609

Finally, terrestrial and marine mapping also established Russia, Southeast Asia, Central and South America, Northern and Western Europe and central Africa to have the highest ecosystem service values, indeed as Li and Fang (2014:303) remark:

“The twenty countries with the largest ESVs represent almost 71.9% (US\$ 95.8 trillion) of the total worldwide ESVs (excluding the ESV for the “Open Ocean” biome). The countries with the largest ESVs tended to have large areas (including both the terrestrial and marine areas, especially the marine areas).”

8.2 Values And Scale

It is quite noticeable that much of the ecosystem service values used to derive global estimates largely focus on provisioning and regulating services (66%), and frequently those services provided by wetland (46%), tropical forest (14%) and coral reef (14%) systems: the reason for this is that this is where most value data points are found, as is evident from Table 8.1 (de Groot et al., 2012). Further evidence of bias towards provisioning and regulating services is provided by a meta-analysis of the value of wetlands to agricultural landscapes, in which only regulating services were considered; chiefly, flood control, water supply and nutrient recycling (Brander et al., 2013). Added to that, the majority of the studies included in Chaikumbung et al. (2016) meta-regression analysis commonly expressed value estimates for three main ecosystem services: food production (188 studies); recreation (153 studies) and habitat-biodiversity (140 studies). A recent global benefit transfer analysis (Schmidt et al., 2016) indicated that most studies and value estimates related to food provision, recreation, raw material provision, maintenance of genetic diversity, climate regulation, water provision and extreme events prevention. Interestingly, this analysis also pointed towards a significant bias in where valuation assessments were conducted:

“The majority of case studies were conducted in lower latitudes (63%-97% of all case studies). Most studies, moreover, were located in areas characterized by a high population density and market accessibility above global average (89%-97%) as well as high threats of degradation (59%-83%)” (Schmidt et al., 2016, pg.15)

It therefore seems strange that in global analyses, quite often, little distinction is made between socio-political and economic settings in the aggregation of value estimates; however, factors such as these can be particularly important in determining the values placed on ecosystem services. For example, covariates such as “social”, “socio-economic”, “economic”, “economic sectors”, “life domains of human-wellbeing”, and “agricultural subsidies” have been identified as influencing, to varying degrees, monetary value transferability (Schmidt et al., 2016). Furthermore, for instance, the broader social-cultural and political context of developing countries has been suggested to affect how wetlands and their services are perceived and hence valued (Chaikumbung et al., (2016).

One step down from global ES estimates is the pan-regional scale. These include, for example, valuation syntheses of the economic value of marine goods and services in the Wider Caribbean Region (Schuhmann and Mahon, 2015). Syntheses such as these are quite important because they helpfully include valuations for ecosystem services associated with reef, pelagic and continental shelf ecosystems, as well as highlighting where there has been both a concentration and dearth of valuation activity. Indications are that reef systems have been the primary focus of valuation exercises, particularly in relation to recreation and tourism within marine protected areas, whilst at the same time the continental shelf ecosystem has garnered relatively little attention. The reason for this seems to be, as Schuhmann and Mahon (2015) explain, because:

“...the economic value of coral reefs for recreation and tourism and coastal protection is considerable and [...] while the valuation of reefs for fishing suggests that many reef-dependent fisheries comprise only minor components of national income and most fishers can be characterized as marginally profitable at best, it is clear that small-scale fishers in the WCR are highly dependent on reefs for livelihoods and food security.” (pg. 59/62)

However, the consequence of this means that there are:

“...notable gaps in our understanding of resource values in the region and [...] efforts related to the pelagic and continental shelf ecosystems have been modest in comparison to the nearshore zones, and are limited to market values of capture fisheries and off-shore recreation opportunities.” (pg. 59/62)

The upshot of this asymmetry in the focus and concern of valuation exercises is:

“To date, economic valuations in the WCR have focused on the benefits from marine goods and services that are relatively easy to measure and convey to the public, such as those from reef-based, mainly tourism-related recreation and nearshore protected areas, and benefits that are ascribed to easily measured market indicators such as those derived from capture fisheries and protection of control real estate.” (pg. 59)

Whilst analyses estimating the global value of ecosystem services supplied by the world's major biomes are important and provide a measure of what is happening at a planetary scale, a global direction of travel as it were, for the most part we do not experience or value ecosystem services in this way and to that extent these global value aggregations, viewed in this specific way, carry little inherent meaning. It is at smaller national and regional geographic scales where meaning gains ascendancy. Against this backdrop, national assessments of the economics of ecosystem services and biodiversity as well as green accounting exercises have increased in recent years, as Hedden-Dunkhorst et al., (2015:39) relay:

“Inspired by the results and recognition of the global TEEB study and its potential to impact on economic as well as environmental policies, a number of countries became interested to carry out national studies on the economics of biodiversity and ecosystem services. The focus and scope of country initiatives

varies substantially, depending on countries' specific initial situation, intentions and aims, the initiator of the study (governmental, scientific or civil society organisations), available professional research capacity, approaches chosen, data availability and financial resources. Final, interim, or upcoming outputs range from spatially-restricted biome-specific case studies to comprehensive assessments and valuations that may cover an entire country and a multitude of ecosystem services, or reflect on specific issues (climate change, trade chains etc.).”

In the UK, for example, the National Ecosystem Assessment (NEA) presented the first consolidated analysis of the changing ecosystem service landscape ascribing social, health and monetary values to a broad diversity of services (UK NEA, 2011; 2014). The first NEA report released in 2011 painted an all too familiar picture of ecosystem decline, indicating that across eight aquatic and terrestrial habitats roughly 30% of the ecosystem services provided by these systems had waned, whilst others were either reduced or severely degraded such as the case of marine fisheries. Often these changes had negative effects on biodiversity with much of it stemming from changes in food and energy production, infrastructure development and public consumption patterns (UK NEA, 2011)¹. A follow-up report produced in 2014, apart from consolidating many of the initial findings described in 2011, offered important policy and management recommendations (based on mainstreaming ESV) to inform and improve the future sustainability of the UK's natural capital base (UK NEA, 2014)². Subsequently, a number of European countries (e.g. Denmark, Germany, Netherlands and Norway)³ as well as non-European countries (e.g. Bhutan, China, Ecuador and India)⁴ have followed suit in establishing and undertaking similar style assessments of their natural capital base (Hedden-Dunkhorst et al., 2015).

8.3 Green Accounting And Green GDP

The development of a system of natural capital accounts, frequently referred to as green accounting, represents a complementary process to TEEB-style country assessments and appends itself to notions of sustainable economic development (Guerry et al., 2015). Advocacy for natural capital accounting systems has been growing over the last few years, for example, as Schaefer et al., (2015:7384) detail:

“Nations throughout the world are recognizing the value of natural capital and are taking steps to account for and conserve it [...] Globally, business and financial institutions are examining the implications of natural capital accounting. More than 40 financial institutions have signed the Natural Capital Declaration, an initiative to integrate natural capital considerations into loans, equity, fixed income, and insurance products, as well as in accounting, disclosure and reporting frameworks.”

Examples of accounting frameworks include the World Bank's Wealth Accounting and Valuation of Ecosystem Services (WAVES) initiative as well as its Global Partnership for

Ecosystem and Ecosystem Services Valuation, and the Inter-America Development Bank's Biodiversity and Ecosystem Services Programme (Guerry et al., 2015; Schaefer et al., 2015). It is important to point out that green accounting is not a welfare-based approach, rather it is a means by which measures of ecosystem services and ecosystem assets can be formalised into an account system, specifically in a spatially explicit monetary and physical manner, as Hein et al., (2015:87) explain:

“Environmental-economic accounts complement the national accounts by providing a description of the interdependency of economic activity and the natural environment in the form of various types of accounts. The accounts include physical flow accounts that describe the supply and use of materials (e.g., water, timber) and energy, as well as the residuals and return flows generated (such as emissions) and asset accounting recording the stocks and changes in stocks of environmental assets.”

Advocates of green accounting, even acknowledging certain inherent limitations⁵, argue that it supports environmental sustainability by affording a number of important benefits, including: (i) the standardisation of definitions for key concepts; (ii) cross-comparisons between environmental and national account outputs; and (iii) robust modelling of environmental assets in an integrated and spatially explicit manner that enables accurate assessment of changes in, and judgements concerning the sustainability of, ecosystem assets and services over time: all of which have important consequences for landscape management, planning and strategies designed to encourage the provision of environmental goods (Hein et al., 2015).

Similarly, the last few years have also witnessed developments in so-called “green GDP”; these analyses make an explicit link between ecosystem services and national income (Li and Fang, 2014). Take Bhutan, as an example, Kubiszewski et al., (2013) estimated the country-wide value of ES to human-wellbeing to be US\$15.5 billion yr⁻¹, a figure almost four times greater than national GDP and roughly equal to US\$15,400 per capita. What is also of interest is that the authors also established that over half (approx. 53%) of the ES provided by Bhutan benefitted people outside the country (Kubiszewski et al., 2013). Nonetheless, Li and Fang (2014) make the point that a lot of green GDP estimates are not well formulated and are often reliant on shaky and ad-hoc accounting data, and so by using spatially explicit mapping approaches for ESV and GDP they argue that their (global) analysis provides a much more comprehensive assessment of the connections between socio-economic and ecological systems. Their headline figure suggests that global ESV is more than double global GDP, as they remark:

“The world GDP (PPP) in 2009 was approximately US\$71.75 trillion (for 225 countries or regions), resulting in a total ESV to GDP ratio of approximately 2.09-1” (Li and Fang, 2014:302)

The authors were also able to show that by and large economic development had negative consequences for ecosystems, and that countries with the greatest 'economic aggregate' had the lowest ESV index. Nevertheless, this pattern was not always the case, as Li and Fang (2014:308) discuss:

“...the relationship between the GDP and ESV is not always a fixed pattern. Some countries with a wealthier economy have a high %ESV, for example, Russia, Australia, Canada and Indonesia. Interactions between ESV and GDP are greatly affected by the stock of natural capital, economic development patterns and other social factors and natural characteristics. Furthermore, these interactions are dynamic, changing based on the socio-economic level and development phase”

8.4 A Focus On Ecosystems And Bundled Ecosystem Services

Adopting an ecosystems-eye view of value estimates, with respect to forest, coastal and marine ESs research indicates significant spatial, contextual and geographic differences as well as variations in socio-cultural and economic realities, all of which affect service valuations. For instance, the value of forest carbon storage is reportedly worth US\$378 ha⁻¹ in Paraguay but as much as US\$1,500 ha⁻¹ in Borneo (Ferraro et al., 2012), whilst averaged figures for saltmarshes and mangroves suggest a value of US\$30.50 ha⁻¹yr⁻¹ (Barbier et al., 2011; Barbier, 2012). Similarly, the value of forest ecotourism ranges from US\$20 to US\$140 per person, with lower values in Ugandan forests compared to Costa Rican forests (Ferraro et al., 2012). Forest ecosystem service values in China demonstrate similar levels of heterogeneity between particular ecosystem services, for example, values for hydrological services range from US\$12-4,938 ha⁻¹, carbon storage from US\$4-4,422 ha⁻¹, soil conservation from US\$3-1,302 ha⁻¹ and nutrient cycling from US\$56-505 ha⁻¹ (D'Amato et al., 2016). Likewise in marine systems, the coastal protection services offered by nearshore coral reefs in India are valued at US\$174 ha⁻¹yr⁻¹, whilst similar coastal protection services afforded by saltmarshes in the USA and mangroves in Thailand are worth about 50 times as much, US\$8,236 ha⁻¹yr⁻¹ and US\$8,966-10,821 ha⁻¹ respectively (Barbier et al., 2011; Barbier, 2012). Fisheries maintenance values are likewise diverse, for example, estimates for nearshore coral reefs in the Philippines range from US\$15-45,000 km²yr⁻¹, yet for mangroves in Thailand the values are much narrower and smaller ranging from US\$708-987 ha⁻¹ (Barbier et al., 2011). These reports also argue for much needed valuation assessments of forest health and hydrological services as well as more detail assessments of the ESs provided by seagrass beds, sand dunes and beaches (Barbier et al., 2011; Barbier, 2012; Ferraro et al., 2012).

Finally, historically speaking, economists have tended to focus on single service valuations of ecosystems, however, recognizing this to be overly simplistic more recent efforts have started to consider the broader bundles of ESs provided by ecosystems and landscapes,

thereby capturing multiple service values (e.g. Nelson et al., 2009; Farley, 2012; Rudd et al., 2016). Two recent examples suffice to make the case. Firstly, in their valuation of ecosystem service bundles for the Oku Aizu forest reserve in Japan Ninan and Inoue (2013) estimated the value of seven ecosystem services: carbon fixing; nutrient cycling; soil protection; water conservation; water purification; air pollution and absorption and recreation. Overall, for this bundle of ES the authors estimated the forest reserve value to be approximately US\$17,016–17,671 ha⁻¹ (Ninan and Inoue, 2013). Secondly, remaining in East Asia, Li et al., (2015) applied a variety of monetary evaluation methods to estimate the value of ecosystem services supplied by the Napahai Wetland in China, namely: climate regulation; flood control; water supply; nutrient conservation; environmental purification; habitat; tourism and scientific research. Their estimates demonstrated a total monetary value for the wetland of ¥237.96 million Yuan (at 2008 prices) or ¥76,000 Yuan ha⁻¹ (Li et al., 2015).

8.5 Final Remarks

The take-home message from this brief coverage of recent valuation appraisals of ES is that the valuation literature is broadening: developing from a background of mainly single ES driven valuations towards a more inclusive set of assessments of ecosystem service bundles (across scales and ecosystem types), alongside a drive to connect ecosystem service valuations with human-wellbeing (socio-economic factors), natural capital accounting and green GDP. It is important to realise that the values presented in this chapter are not hard and fast estimates, certain and unchanging, absolute in their capturing of the flow of ecosystem services, instead they are decision-making aids, as Pascual and Muradian et al., (2012:187) spell out these values are:

“...a reflection of what we, as a society, are willing to trade-off to conserve these natural resources [...] and these values provide information that can guide policy making.”

Notes

1. In fact the NEA Synthesis Report (2011) makes the following point:

“The UK population will continue to grow, and its demands and expectations continue to evolve. This is likely to increase pressures on ecosystem services in a future where climate change will have an accelerating impact both here and in the world at large. The UK’s population is predicted to grow by nearly 10 million in the next 20 years. Climate change is expected to lead to more frequent severe weather events and alter rainfall patterns, with implications for agriculture, flood control and many other services. One major challenge is sustainable intensification of agriculture: increasing food production while decreasing the environmental footprint.” (pg.5)

It is perhaps worth quoting at length from the NEA regarding the origins of the changing fortunes of ecosystem services in the UK from the post-1945 era onwards, because this gives an insight into how different management and policy strategies driven by changing social, economic and political

circumstances have affected the provision and distribution of ecosystem services and influenced associated human welfare gains and losses over that period, and thus affected our perception of the importance and value of biodiversity and ecosystems (natural capital) and the services they provide:

“The late 1940s saw the UK enter a phase of national reconstruction, with priorities focused on increasing production and building homes and infrastructure. Much activity in these areas was in direct response to market forces, but government policy and subsidies promoting production and infrastructure development also played an important part. Agricultural production began a period of rapid expansion that continued for several decades. In England the area of land under crops increased by 40% from 1940 to 1980. Thanks to plant breeding, increasing chemical inputs and technological innovations, yields per hectare of most crops also increased – more than threefold in the case of wheat. Similar productivity gains have been seen in livestock, with average milk yields doubling between 1960 and 2009. Timber production also rose, almost entirely as an increase in softwood production, which now accounts for over 95% of timber harvest in the UK. Not all production increased. Most notably, landings of fish and other seafood declined steadily, from 1.0 million tonnes in 1970 to 0.5 million tonnes in 2000 (although this figure has remained roughly constant since then) (Figure 2). By the early 1990s, 10% or fewer of the fish stocks in UK waters were sustainably harvested. The gains in production had impacts on other ecosystems and ecosystem services. Extensive areas of semi-natural vegetation were converted or modified – it is estimated, for example, that 97% of enclosed semi-natural grasslands in England and Wales were lost between 1930 and 1984 through intensification or conversion to arable land. Major increases in fertiliser use, particularly nitrogen and phosphorus, have adversely affected aquatic ecosystems through runoff. The Farmland Bird Index – a measure of the state of biodiversity on agricultural lands – declined by 43% between 1970 and 1998. The push to increase timber production – which dates from the early years of the 20th Century – resulted, particularly in Scotland, in the creation of extensive areas of coniferous plantation at the expense of other habitats. Two-thirds of the UK’s current woodland area of around 3 million hectares is productive plantation, mostly less than 100 years old and much of it comprising non-native species. Other sectors, including energy, industry, housing and transport, also had major impacts on ecosystems and the delivery of ecosystem services. For example, through deposition of atmospheric nitrogen and sulphur, loss of habitats through construction, and disruption of flood regimes in river basins and coastal wetlands. Changes in the urban environment have had a direct impact on the very high proportion of the population living in cities and towns. There has been a marked decline in the condition and accessibility of urban greenspace: around 10,000 playing fields were sold between 1979 and 1997, while allotments are now down to 10% of their peak level, with an estimated total area of around 10,000 hectares, compared with over 100,000 hectares in the late 1940s.” (pg. 8-7)

2. With regards to the overall purpose of the follow-up NEA as a document to aid policy and practical decision-making regarding environmental management strategies for the sustainable provision of the UK’s ecosystem services the NEA Synthesis Report (2014) makes the follow points:

“The UK NEAFO confirms that the ecosystem services derived from natural capital contribute to the economic performance of the nation by supporting economic sectors, regional and national wealth creation and employment. But the relationship between our ‘natural capital’ and the wider economy is complex. By mapping the relationships between ecosystem services and major sectors of the economy, such as agriculture or food manufacture, we can begin to understand the economic impacts arising from any changes in our ecosystem services. The UK NEAFO has developed a Natural Capital Asset Check (NCAC) to help this process. It can be used to consider thresholds, trade-offs and the performance and resilience of our ecosystems. It can be used to gain further insights into the properties of different ecosystem services and contribute to our understanding of how best to manage the natural world for the long-term benefit of society. Building on the UK NEA, the UK NEAFO quantitatively values a number of additional ecosystem services, relating them to changes in land use, as well as marine and coastal ecosystems. The assessment concludes that spatially targeted policies deliver more economically efficient outcomes. It also shows that before decisions are made it is

important to fully appraise the widest possible range of policy options that take into consideration our natural capital stocks and flows. The UK NEAFO uses an updated land use change model to quantify the benefits of different forest planting strategies. The model includes changes in agricultural outputs and farm incomes, net greenhouse gas emissions, recreational visits, water quality and biodiversity. A suite of models were identified that can be used to address the different components of the marine shelf ecosystem, and a number of options for linking land use change models to coastal waters in order to assess the consequences for coastal ecosystem services. A range of methods were used to calculate a monetary expression of both marine ecosystem stocks and the marginal economic values for changes in the ecosystem service flows over time.” (pg. 5)

3. As an example of a European country also applying the TEEB process in a manner similar to the UK it is informative to highlight the example of the Netherlands as Hedden-Dunkhorst et al., (2015:39) describe:

“In 2011 three National Ministries in the Netherland jointly started a TEEB process, aiming to raise awareness of nature's values among major stakeholders (government, citizens and the business community) and to main- stream ecosystem and biodiversity values into economic and political decision-making. Ecosystems and biodiversity are assessed and valued in six separate cases: (1) TEEB for cities, including development of planning support tools, (2) TEEB for business, with a focus on agriculture, (3) TEEB for regions, including an analysis of trade- offs, (4) TEEB for health, (5) cultural ecosystem services in the Caribbean overseas territories, and (6) TEEB for trade chains. The valuation of trade chains moves far beyond the national level and includes an impact analysis of human consumption in the Netherlands on ecosystem services elsewhere, a unique focus in the landscape of TCSs [TEEB County Studies].”

4. As an example of a non-European country, Hedden-Dunkhorst et al., (2015) cite the TEEB-like ecosystem service assessments being undertaken by India:

“India's TCS now focuses on coastal and marine ecosystems, forests and inland wetlands and specifically aims to raise awareness among decision-makers and the public on the contribution of ecosystem services and biodiversity for human welfare. Policy recommendations are expected to provide guidance for sustainable development and conservation at national, state and local levels and to propose suitable tools for improved biodiversity-related business practice.”

5. The main limitations sketched out by Hein et al., (2015) include: the resource intensiveness of conducting an account assessment; the fact that the accounting process does not accommodate a social welfare function (thus providing only a partial picture of TEV); that it cannot be used to inform the design and development of instruments to deal with the impacts of long-term changes on ecosystems; and that the information accounting provides vis-à-vis tracking changes in environmental assets and services does not imply it will lead to improved governance regimes of the natural capital base.

Chapter 9: Valuation - Problems Come In Threes

“Ecosystem services provide multiple benefits to human wellbeing and are increasingly considered by policymakers in environmental management. However, the uncertainty related with the monetary valuation of these benefits is not yet adequately defined or integrated by policy-makers.” (Boithias et al., 2016 pg.683)

“There will be very few occasions when you are absolutely certain about anything. You will consistently be called upon to make decisions with limited information. That being the case, your goal should not be to eliminate uncertainty. Instead, you must develop the art of being clear in the face of uncertainty.” (Andy Stanely, American Clergyman)

The two preceding chapters have discussed the central role of environmental valuation in delivering and underpinning the ecosystem services paradigm. They have described the debates and developments in how valuation methodologies are chosen and applied, as well as how they are able to capture, increasingly so, a broad array of non-monetary values alongside the more traditional, and still dominant, monetary valuation assessments of ecosystem services. In addition we have highlighted (albeit briefly) the range of valuation assessments that have taken place in recent years, primarily to emphasize their commonality as a standard research and policy, thereby also demonstrating that their use regularly extends across scales, biomes and specific ecosystems. At each point, we have alluded to the fact that often what characterises the debates and controversies surrounding environmental valuation is the specific assumptions that valuations make, what they are able to capture (or not) and the degree of confidence we can have that elicited values accurately reflect reality. This chapter seeks to take on this latter issue by concentrating on three areas that go to the heart of the matter, namely: uncertainty, discounting and benefit transfer.

9.1 Uncertainty

Uncertainty looms large in policy, in decision-making, in choices about future actions and consequences, in research and the conveyance of information (Tuckett et al., 2015; Kwakkel et al., 2016). These are often connected to a broader repertoire of uncertainties sometime described as “deep uncertainty” and linked to so-called “Wicked Problems” (Kwakkel et al., 2016). Uncertainty is, however, a central property of complex systems, such as coupled social-ecological systems (Haila and Henle, 2014; Molbus and Kalton, 2015). But, in the context of this chapter, what do we mean by uncertainty? Specifically, we mean two things: First, doubt the probabilities concerning the possible outcomes of decision-making processes, and second, doubt over the possible outcomes that may ensue from taking particular decisions (Pascual and Muradian et al., 2012).

Pascual and Muradian et al., (2012) flag three sources of valuation uncertainty, namely: supply uncertainty (i.e. in the provision of ES, so-called “biophysical uncertainty”); preference uncertainty (i.e. the elicitation of individual values) and technical uncertainty (i.e. the ability to accurately measure values). Preference and technical uncertainty are dual aspects of the “process” of monetary valuation, and have elsewhere been termed “structural uncertainty” and “parametric uncertainty” (Boithias et al., 2016)¹. Similarly, supply uncertainty relates to what Haila and Henle (2014) identify, in relation to biodiversity, as uncertainties concerning “data” and “proxies”, while preference and technical uncertainty variously links to what they define as uncertainties in “concepts”, “policy and management” and “normative goals”.

Honing in on supply uncertainty, the degree of doubt regarding biophysical uncertainty is dictated by the extent of our knowledge (“epistemic uncertainty”) regarding the relationships between ecosystem services and biodiversity, management actions and the provision of ecosystem services and; ultimately, their transformation into human-wellbeing – although the evidence base is increasingly more comprehensive about these connections it is still far from robust (Pascual and Muradian et al., 2012; Haila and Henle, 2014; Boithias et al., 2016). However, in some instances, expected values for certain biophysical variables may be used where the probability distributions have been assigned to particular, so-called, “states of nature”. The benefit of this approach is that it resolves some of the ambiguity by “weighting” potential outcomes according to their likelihood of occurrence. Subsequent valuations are then based on the “weighted outcomes” of alternative states, for example, as in the case of expected damage function methodology (Pascual and Muradian et al., 2012).

The main challenges underpinning supply uncertainty have been classified as comprising a triumvirate of issues, specifically relating to: (i) the difficulties involved in disentangling the relationships between ES generation and ecosystem functioning (e.g. spatial heterogeneity that is the supply of ecosystem services is not generally on a uniform per hectare basis); (ii) the requirement that, for valuation purposes, ESs be regarded as independent when in reality they are interrelated and co-produced and finally, (iii) the difficulty of pinpointing, and accounting for, ES thresholds (de Groot et al., 2012; Kumar et al., 2013; Liekens and De Nocker, 2014). The latter issue is particularly interesting because as ecosystems continue to be degraded they will likely produce fewer ecosystem services, and those that they do produce will also likely be at reduced magnitudes. Economic valuation, however, rests on the principle of marginality. Thus, if we imagine for a moment that an ecosystem is made up of discrete units then, according to the principle of marginality, the next unit of an ecosystem used should not cause the system to pass beyond a threshold whereby service provision is continually declining, this of course is exactly the opposite of what we are likely to observe in degraded ecosystems (Liekens and De Nocker, 2014).

When we talk about preference uncertainty essentially we are referring to the capricious nature of people's preferences – that individuals do not hold (or have) a fixed set of preferences for the amount they would be willing to pay for particular ecosystem services: obtaining consistent and reliable valuable estimates then is like dealing with shifting sands, they are forever moving and hard to pin down (Pascual and Muradian et al., 2012; Tuckett et al., 2015). This is because, as Haila and Henle (2014:35) express:

“...uncertainty is inherently contextual, making sense of uncertainty in specific situations requires that we take into account several aspects of cognitive work and social reality.”

This has obvious and important ramifications for conducting valuations. Regularly used valuation methods, such as contingent, have attempted to identify peoples' uncertainties upfront by allowing individuals to express a value range rather than a specific value. Although adopting a value range approach is regarded as highlight promising, the problem of what value range ought to be chosen to elicit the most “truthful” response remains (Pascual and Muradian et al., 2012). Clearly, valuation metrics and the type of metric used to elicit particular information about a specific service or services is crucially important, as Boithias et al., (2016:684) remark:

“The choice of valuation metric has been reported to be relevant for the valuation, as different valuation metrics might be based on the same set of economic assumptions but approach the ecosystem services from different perspectives, with results varying widely depending on the choice of valuation metric rather than on the object under analysis.”

Finally, technical uncertainty relates to the reliability of stated preference values and the degree of ‘truthfulness’ they represent. Extending this further and drawing on the definition of parametric uncertainty, technical uncertainty is therefore also associated with uncertainties in the valuation metrics included in market prices for example (Boithias et al., 2016). Consequently, the drive for accuracy in valuation methods represents a constant battle against the odds, particularly for revealed preference and price-based approaches, where data availability and lack of being able to capture non-use values are problematic (Pascual and Muradian et al., 2012). Nevertheless, two so-called “data enrichment” methods have evolved to help deal with these issues, namely: data fusion and preference calibration approaches. Data fusion combines revealed and stated preference methods in a manner that allows the behavioural history of individuals to be allied to proposed behavioural changes whilst being firmly rooted on observed behaviour. On the other hand, preference calibration uses a combination of valuation methods to generate multiple ES values; these values are then subsequently condensed into a single preference function thus bypassing many of the

restrictions associated with employing just one type of valuation methodology (Pascual and Muradian et al., 2012).

9.2 Discounting

Directly linked to uncertainty and intimately related to the concept of sustainability, and like sustainability is hard to pin down, discounting neatly encapsulates the core feature and difficulty of sustainability, namely, our relationship and responsibility with and to future generations, as Gollier and Weitzmann (2010:350) explain:

“The concept of discounting is central to economics, since it allows effects occurring at different future times to be compared by converting each future dollar into a common currency of equivalent present dollars. Because of this centrality, the choice of an appropriate discount rate is one of the most critical issues in economics. It is an especially acute issue for projects involving long time horizons because in such situations the results of cost–benefit analysis (CBA) can be incredibly sensitive to even tiny changes in the discount rate.”

Discounting, in general as well as in relation to the environment, has been widely discussed extensively elsewhere (e.g. Hepburn, 2006; Howarth, 2009; Dhami, 2016) I therefore do not intend to provide an expansive theoretical coverage of the topic here; instead, highlighted are some of the core challenges associated with discounting environmental assets and services. The primary concern in providing valuation estimates is how these will be used in policy-making and management contexts to influence the continued provision of ecosystem services into the future. In other words, how can valuations be applied to inform our choices and underline our responsibility to use and conserve natural resources in a manner that is consistent with sustaining present generations, whilst at the same time ensuring future societies will be in a position to flourish?

“Evaluating the impacts of present activities on those living in the future is one of the most critical areas of uncertainty in environmental policy. The debate surrounding discounting is not only important to the numerical valuation of the costs and benefits of environmental policies (social benefits/costs and optimal path calculations), it is also central to designing policies that are incentive compatible with observed human behaviour and evolved neurological structures and pathways.” (Gowdy et al., 2013:S94)

As the quote above implies, discounting is a highly contested area of (environmental) economics, primarily because the rate adopted can produce two strongly contrasting outcomes: One that is highly conservative and favours fewer restrictions on current natural resource use so as to maximize present human welfare and a second, more radical stance, that calls for substantial curtailments in present natural resource use – generally through aggressive environmental policies – in order to sustain human welfare in the future (Gowdy et al., 2012;

2013). These difficulties are starkly expressed by Gollier and Weitzmann (2010:350-351) in relation to climate change:

“We think it is important to begin by recognizing that there is no deep reason of principle that allows us to extrapolate past rates of return on capital into the distant future [...] Even leaving aside the question of how to project future interest rates, additional issues for climate change involve which interest rate to choose out of a multitude of different average rates of return that are out there in the past and present real world. Furthermore, there is an ethical dimension to discounting climate change across many future generations that is difficult to evaluate and incorporate into standard CBA [...] The fundamental point is that there is enormous uncertainty and controversy about choosing an appropriate rate of return for discounting distant-future events, like climate change. Moreover, the great uncertainty about discounting the distant future is not just an academic curiosity, but it has critically important implications for climate-change policy. This disturbing ambiguity has given rise to a great deal of controversy and a variety of proposed solutions.”

The continuing challenge presented by discounting remains its standard neo-classical economic foundations which generally conceive of individuals as being entirely rational and utility maximising in their behaviour; and moreover, characterises the decisions individuals make and the actions they take being underpinned by complete and stable preferences and perfect information regarding the choices available to them. Thinking, however, has started to roll-back this orthodoxy and progress beyond these idealistic and flawed caricatures of human decision-making, for example, insights from behavioural, neuro and evolutionary economics are providing much richer pictures of peoples' internal machinations, especially in terms of emphasising its complexity and fuzziness (Kahneman, 2010; Parks and Gowdy, 2013; Gowdy et al., 2013; Tuckett et al., 2015; Dhimi, 2016). Some of these insights have been especially revealing, particularly in relation to the idea that people quite readily display loss aversion², risk aversion³, hyperbolic discounting⁴ and inconsistent discounting⁵, and moreover, that discount rates themselves have to acknowledge both uncertainty⁶ and price⁷ issues.

Based on these insights and applying them to the Ramsey discount equation⁸ (the standard means of calculating a discount rate) Gowdy et al., (2012) reach the stark conclusion that there exists no set of adequate economic-only guidelines for selecting a particular discount rate. In their view, whatever forms the discount rate takes, that choice, is a normative judgement – a matter of ethics and a moral act. The authors go on to argue that a variety of discount rates ought to be applied, including zero and negative values, though they are careful to emphasize that this needs to be set within the context of the particular conditions of the valuation exercise (Gowdy et al., 2012). This position is also supported by Kumar et al., (2013). Overall, a higher discount rate is considered likely to produce poorer future outcomes for ecosystems and biodiversity, particularly on a project by project basis. Conversely, Gowdy et al., (2012) also acknowledge the possibility that a lower discount rate, if applied across the

entire economy, may encourage greater levels of investment and result in environmental damage as a consequence of economic growth. Picking up on this thread, in reviewing the recent literature regarding the appropriateness and application of dual discount rates to manufactured consumption goods and environmental (impacts) services, as a means to account for the divergence between the growth in global GDP alongside the decline in global ecosystem services, Baumgartner et al., (2015:274) comment that:

“From this analysis it has emerged that dual-rate discounting is warranted if relative scarcities between different goods are changing over time, yet, future consumption is valued in constant relative prices or future prices for environmental goods are unavailable. Differing discount rates then serve to account for changing relative scarcities between the different goods. In contrast, if future consumption is valued in prices that change over time to properly reflect changing relative scarcities, then a uniform discount rate (reflecting pure time preference only) is appropriate.”

Against this background, Baumgartner et al., (2015) performed an assessment of 10 ecosystem services across a number of different countries to estimate the difference between discount rates for ecosystem services and manufactured consumption goods, finding that:

“...ecosystem services in all countries should be discounted at rates that are significantly lower than the ones for manufactured consumption goods. On global average, ecosystem services should be discounted at a rate that is $0.9 \pm 0.3\%$ -points lower than the one for manufactured consumption goods. The difference is larger in less developed countries and smaller in more developed countries. This result supports and substantiates the suggestion that public cost-benefit-analyses should use country-specific dual discount rates—one for manufactured consumption goods and one for ecosystem services.” (pg. 273)

Ultimately, producing a discount rate is about expectations of future human welfare and wealth. Therefore how much should be consumed now and how much should be left for future generations to consume are predominately moral judgements. Such decisions need to carefully balance the likely potential impacts on natural capital of business-as-usual approaches to consumption, which is fuelling global increases in GDP, with the potential negative consequences for doing so on long-term economic growth and human welfare in circumstances in which the natural capital base has become irrevocably eroded. However, as Gowdy et al., (2013:S102) acknowledge:

“Especially for long-term threats like climate change and biodiversity losses, environmental valuations to be discounted suffer from our current lack of knowledge, high uncertainty and our weaknesses to act as regent of future generations’ needs.”

Thus, the authors call for a social valuation approach to discounting, ultimately arguing that:

“Types of behaviour conducive to cooperation, doing with fewer material possessions, and recognizing the necessity of shared sacrifice, are also part of the human experience and these behaviours should certainly be taken into account in any intergenerational policy decisions.” (Gowdy et al., 2013:S102)

9.3 Benefit Transfer

Over the last 20 years or so benefit transfer (BT) has become an increasingly popular valuation and policy tool, primarily because it is seen as relatively cheap and easy – big pluses in an era where the costs and time associated with undertaking primary valuation studies are regarded as prohibitive (Plummer, 2009; Johnston and Rosenberger, 2010; Richardson et al., 2014):

“Benefit transfer is increasingly being used to meet the demand for increased information on nonmarket ecosystem service values in a manner relevant to the time frame and budget within which decisions often have to be made.” (Richardson et al., 2014:2)

Benefit transfer describes the process whereby monetary environmental values pertaining to particularly ecosystem services at a specific locale or “study site” are transferred to a different but relatively similar “policy site” (Plummer, 2009; Johnston and Rosenberger, 2010; Pascual and Muradian et al., 2012; Liekens et al., 2014; Richardson et al., 2014). The point has been made that, if carefully crafted, BT valuations can provide good approximations to areas that have not been previously studied (Richardson et al., 2014). However, issuing a word of caution, Johnston and Rosenberger (2010) note that there is considerable confusion and controversy in the academic literature regarding the overall effectiveness of BT, as well as over what the “optimal” BT methods to employ are. These disputes also relate to a long-standing divergence between how academics and policy-makers perceive BT, with the former often appraising it from an idealist perspective (i.e. primary valuation studies are preferential) and the latter taking a more pragmatic stance (i.e. what works best in the ‘real’ world). Debates such as these, though they may seem anodyne, are actually quite significant because the values BT valuations condone can have important policy and management implications for biodiversity conservation and the provision of ES: persistent issues concerning valuation transfer practices should not therefore be ignored but instead acknowledged and addressed (Johnston and Rosenberger, 2010; Richardson et al., 2014):

“Otherwise, if violation of the basic principles and methodological requirements for valuing ecosystem services through benefit transfer remains widespread, this may ultimately undermine the integration of ecosystem service values into policymaking.” (Richardson et al., 2014:2)

A significant proportion of BT debates concern the types of BT that are or should be employed, and four broad categories of BT approaches are recognized: (i) unit BT⁹; (ii)

adjusted BT¹⁰; (iii) value or demand function transfer¹¹ and, (iv) meta-analytic function transfer¹² (Pascual and Muradian et al., 2012; Liekens et al., 2014; Richardson et al., 2014). To varying degrees these BT permutations are challenged by a number of issues, and Pascual and Muradian et al., (2012) outline eight: (i) transfer errors¹³; (ii) aggregation¹⁴; (iii) spatial scale¹⁵; (iv) variations in ecosystem properties and context¹⁶; (v) non-constant marginal values¹⁷; (vi) distance decay and spatial discounting¹⁸; (vii) equity weighting¹⁹ and; (viii) primary valuation availability²⁰ (many of these are also discussed in Johnston and Rosenberger (2010), Richardson et al., (2014) and Schmidt et al., (2016)).

How would we sum-up the BT landscape? Overall, meta-analytic functions are regarded as the most robust and advanced form of BT with respect to dealing with the challenges outlined above; however, they are also the most complex and time consuming. Importantly, recent modelling developments have helped to improve function transfer estimates, and moreover, the development of web-based resources looks set to continue to improve BT methods going forwards, by enabling best practice exchange and the provision of accessible primary valuation databases (Richardson et al., 2014).

9.4 Final Remarks: Future Research And Progress

Drawing together the evidence we have laid out over the last three chapters, it is clear that progress in ecosystem services valuation has been substantial and many insights have been gained through endeavours such as TEEB and associated processes. For example, as Kumar et al., (2013) relate, these collaborative and transdisciplinary research and policy assessments have demonstrated the importance of using integrated knowledges and methodologies to undertake economic valuations. Nevertheless, there are still many areas of ESV that need to be improved. For example, as they go on to emphasize, ESV needs to be conducted at relevant policy scales that acknowledge the context and temporal dependent nature of ecosystem processes and human values. In this sense, they argue it is essential that the socio-cultural milieu that pervades and informs value articulating institutions are recognized by those individuals involved in economic assessments (Kumar et al., 2013).

Collectively, significant avenues for further progress in producing more realistic, credible and useful ESV remain, and in particular, action needs to be directed towards the following areas: (i) improving data availability, reliability and heterogeneity (e.g. increase the number of ecosystem services and estimates per biome as well as value estimates per biome); (ii) providing more consistent and coherent terminology and methods across studies to enable thorough in-depth systematic reviews and meta-analysis assessments; (iii) focusing more heavily on valuing and accounting for supporting and cultural services as well as neglected

biomes such as deserts and polar regions; (iv) further developing tools for cross-scale valuation; (v) expanding valuation methodologies to integrate non-use and use-values; (vi) focusing on mainstreaming social valuation methods (e.g. deliberative approaches especially group-based) as part of standard ESV assessments and further investigating how different elicitation processes construct and frame values alongside expanding the evidence base to illustrate the importance of shared and social values across sectors and spheres; (vii) improving valuation techniques to account for marginality, double counting and benefit transfer, selection bias of value estimates, aggregation of monetary values, the spatial explicitness of ES provision and distribution, ecosystem disservices and threshold effects in ecosystem states, differences in local socio-economic conditions and the scale at which services are provided to beneficiaries; (viii) finding ways to integrate and scale-up micro-economic outcomes to connect with a broader macro-economic framework for ecosystem accounting and finally; (ix) developing inter- and cross-sectoral analysis of the individual impacts of policies for ecosystem management (Mendelsohn and Olmstead, 2009; Plummer, 2009; de Groot et al., 2010; de Groot et al., 2012; Farley, 2012; Kumar et al., 2013; Parks and Gowdy, 2013; Kenter et al., 2015; Rode et al., 2015).

Notes

1. According to Boithias et al., (2016:684):

Structural uncertainty refers to:

“...the structure of the valuation process (i.e., selection of services, benefits, and valuation metrics)”

Whereas parametric uncertainty relates to:

“...the uncertainty in the parameters used in each of the valuation metrics (i.e., valuation methods).”

2. Loss aversion suggests that people feel losses (i.e. negatives) more than equivalent gains (i.e. positives). Consequently, people place a higher value on a loss than they do on a gain of the same magnitude (Kahneman, 2010). This provides one explanation for why WTP and willingness to accept (WTA) values are often different (Gowdy et al., 2012). This means that compensation for ecological losses is likely to be higher than the potential market value of that loss (Gowdy et al., 2012).

3. Risk aversion relates to future events and their uncertainty and thus the likelihood of, in essence, obtaining a reward or “pay-off”: if future events are (or perceived to be) highly uncertain, even if the “pay-off” is known, people tend to display conservative behaviour in the present. In other words, people stick with what they know – the business-as-usual-approach – rather than making substantial changes to their day-to-day behaviours (Gowdy et al., 2012).

4. Gowdy et al., (2012) provide some evidence to indicate that when people look to the future they discount in a hyperbolic manner. In other words, discount rate decline and then flatten over time from the present instead of being the straight line phenomena of standard economic analysis. In other words, “people discount the value of delayed consumption more in the immediate future as opposed to the distant future.” (Gowdy et al., 2013:S97). Using the standard paradigm in cases where hyperbolic discounting is a reality would, for example, considerably underestimate the future benefits of conserving present biodiversity.

5. There is some evidence to suggest that depending upon the particular issue at hand people display different discounting behaviours. In other words, people may have different discount rates for different outcomes. The potential for conflicting discount rates is therefore considerable (Gowdy et al., 2012).
6. Potentially uncertainty can impact discount rates. For example, high uncertainty about the future is likely to produce lower 'certainty-equivalent' discount rates as people invest in safe assets to guarantee a future level of economic welfare (Gowdy et al., 2012).
7. It is generally assumed that prices for consumption goods change at the same rate; however, for non-consumption goods (in the standard sense) estimating that rate of change in their value can present a not inconsiderable problem, as in the case of biodiversity. Thus a situation could arise where the estimated damages resulting from the continued loss of biodiversity may be of such a magnitude that it would actually offset a positive discount rate (Gowdy et al., 2012).
8. The Ramsey discount equation is given by $r = \rho + \eta * g$ (where r = discount rate, ρ = rate of pure time preference, η = elasticity of the marginal utility of consumption, and g = the growth rate of per capita income). The rate of pure time preference gives an indication of individual time and social preferences for the human-wellbeing of future generations. If ρ is positive then *certis paribus* the wellbeing of future generations becomes increasingly less important. Moreover, a larger value indicates that the potential negative impacts of current actions on future generations matters less. The second component of the equation ($\eta * g$) regards the degree to which we are concerned with the level of welfare (i.e. in wealth terms) experienced by future generations. In general η is normally assumed to equal 1, but a higher value for η requires that a future reward be considerable in comparison to a sacrifice today. Assuming a non-declining (g) and the importance of maintaining a stock of capital assets (inclusive of natural capital) then (g) can be considered as the growth in per capita income adjusted for externalities. In other models, and under certain assumptions, g can also be considered to represent subjective wellbeing. Under these circumstances some argue that this strongly suggests that g ought to have a negative value. In other words, the present generation should consume less (Gowdy et al., 2012).
9. In Unit BT values for an ecosystem service at the policy site are calculated from a mean monetary value for ES at the study site multiplied by the estimated quantity of ES at the policy site. These values are usually defined in terms of either per household/individual or per unit area (Pascual and Muradian et al., 2012; Liekens et al., 2014).
10. Adjusted BT also takes account of the differences in characteristics (e.g. income, prices, population) between the study site and the policy site, and factors these parameters into the calculation of transferred values (Pascual and Muradian et al., 2012; Liekens et al., 2014).
11. These methods use a value function derived from valuation techniques such as hedonic pricing or choice modelling at a study site alongside value parameters for the policy site to transfer values (Pascual and Muradian et al., 2012; Liekens et al., 2014).
12. In the case of meta-analytic functions values are derived from several valuation studies alongside value parameters for the policy site to transfer values ((Pascual and Muradian et al., 2012; Liekens et al., 2014).
13. Transfer errors arise when the values transferred differ from the actual values of the ecosystem at the policy site and they can occur for two reasons. Firstly, as a result of 'measurement errors' (i.e. value estimation errors at the study site) due to methodological limitations, and secondly, from the transfer process itself so-called 'generalization errors'. The evidence seems to indicate that meta-analytic transfer functions suffer from fewer generalization-associated errors (Johnston and Rosenberger, 2010; Pascual and Muradian et al., 2012).
14. The challenge of aggregation is to accurately take account of supply and demand in order to assess the total value of a service without producing spurious results. For example, in the case of demand where values are expressed as per beneficiary aggregation is an estimate of total WTP, and so the size of the market for that ES needs to be accurately determined. In relation to supply, values are aggregated over the extent of the ecosystem and so in performing this aggregation it is critical to estimate service delivery rather than potential service supply. Moreover, when aggregating across

multiple ecosystem services the potential for double-counting needs to be negotiated (Johnston and Rosenberger, 2010; Pascual and Muradian et al., 2012; Liekens and De Nocker, 2014).

15. Spatial scale can affect BT in three main ways. Firstly, scale is an important determinant of the provision of ES and the location of beneficiaries. Secondly, scale has significant implications for the proximity as well as availability of substitute or complementary ES for beneficiaries. Thirdly, spatial scale can affect distance decay and also how values are discounted (Pascual and Muradian et al., 2012; Liekens and De Nocker, 2014).

16. Ecosystem service values will be affected by ecosystem type and condition, the geographic and socio-economic characteristics of beneficiaries and the context in terms of other available sites or services (Pascual and Muradian et al., 2012; Richardson et al., 2014).

17. Frequently the evidence indicates that ecosystem services values display non-constant returns to scale. That is to say, ecosystem service values may have diminishing returns to scale perhaps as a consequence of fundamental ecological relationships, such that less ES is produced per unit area at the margin for a large ecosystem compared to a smaller ecosystem. Equally, ecosystem service values may show increasing returns to scale such as habitat provision. At the very least both the size and change in size of the ecosystem must be factored in to transfer value determination (Johnston and Rosenberger, 2010; Pascual and Muradian et al., 2012).

18. Distance decay refers to the pattern of decreasing ES values witnessed as a consequence of the distance beneficiaries are away from the supply of ES (i.e. the lower the value attributed to an ES the further the beneficiary is from the site of service provision), a property that is also related to the type of service being valued (e.g. whether the service is a provisioning services or a cultural service). Directly related to this, the rate of decline in ES values can be accounted for by spatial discounting. In other words, weightings are applied to values whereby lower weights are given to those ecosystem services further from beneficiaries. Failing to account for distance decay effects is likely to produce an overestimation of total ES values (Johnston and Rosenberger, 2010; Pascual and Muradian et al., 2012).

19. Equity weighting proposes to account for the different level of dependence poorer households and communities have on ES compared to wealthier households and communities. Equity weighting can also be used when comparing developed and developing countries (Pascual and Muradian et al., 2012).

20. Without primary evaluation data the capacity of BT to produce realistic values is questionable. Lack of primary data represents a rate limiting step in terms of having information about relevant ecosystems, ecosystem services as well as socio-economic and socio-cultural conditions ((Johnston and Rosenberger, 2010; Pascual and Muradian et al., 2012).

Part 6: Managing The Garden

An important focus of the ecosystem service paradigm is a concern with the management and provision of environment public goods. In particular, ways of securing ecosystem service provision, ways of balancing and harmonising potentially competing environment and development objectives, and ways of negotiating trade-offs in production activities versus conservation objectives. In our Post-Edenic world, these are the concerns we have to deal with if we are to ensure our human-wellbeing needs are met whilst at the same time preventing further degradation and unsustainable exploitation of the biosphere.

Chapter 10: Providing Public Goods – Ecosystem Services And Externalities

“Externality and the public good nature of many ecosystem services may lie at the heart of the biodiversity change problem, but it is worth underlying the fact that they are not the only drivers of biodiversity change. Nor do they exist in a vacuum. They are the product of a set of feedbacks that at once reflects an evolving set of property rights, technological change, the homogenization of managed/impacted ecosystems, and the fact that the global economic and ecological systems are both becoming increasingly integrated.” (Charles Perrings, *Our Uncommon Heritage*, pg. 180)

10.1 A Short Introduction To Public Goods Provision And Ecosystem Services

Drivers of change in biodiversity and ecosystem services occur across multi-spatial and temporal scales, and as we have seen from earlier chapters (identify chapters) are driven by a combination of different anthropogenic forcing factors including: infrastructure and urban developments; land conversion and agricultural expansion; over-harvesting; pollution and water appropriation. Many of these forcing factors can be framed as externalities¹, or for our purposes “ecosystem or biodiversity externalities”. They are externalities in the sense that they are outside (or external to) the purview of market transactions – the unplanned consequences of production and consumption activities related to marketed goods and services (Perrings, 2014). In this respect, externalities can be considered to be either positive or negative, or as Caplan (2008) describes:

“Positive externalities are benefits that are infeasible to charge to provide; negative externalities are costs that are infeasible to charge to not provide”

Why is the consideration of externalities important from an ecosystem services and biodiversity perspective? Well, as Perrings (2014:148) highlights these externalities affect:

“...the wellbeing of either consumers or producers by altering the ecological functioning on which consumption or production depends.”

Indeed, the fact that the twentieth century witnessed such a dramatic decline in ecosystem services, up to 60% according to the MA is, in the eyes of Perrings (2014:148), referring to the earlier work of Kinzig et al., (2011), due in large part to the fact that:

“We do not pay for them, and they generate no return to the landholders whose actions affect their supply. Since we get what we pay for, we should expect such services to be neglected”

Crucially, not all positive or negative by-products of a decision are regarded as an externality, in fact, whether something is considered to be an externality or otherwise depends upon whether or not its scarcity is appropriately signalled (Dixit, 2014). As Dixit (2014:70) explains:

“When you buy something, you use up the labour, materials, and other resources that went into making it, leaving less for others. But the price you pay for your purchase in a well-functioning competitive market equals the marginal cost of production. Therefore you face the correct scarcity price for your action, and have the correct incentive to economize on the use of society’s scarce resources. Only when you do not face the correct scarcity price, as in cases like clean air and roads, will your actions create externalities. What is an externality therefore depends on whether a market puts the correct price on that action.”

This is a particularly important conclusion then because it implies, according to standard neoclassical economic mantra, that prices reflect scarcity: ergo without the existence of sufficient price levels appropriate allocation of resources reflecting their underlying scarcity is (bar some kind of intervention and adjustment mechanism) unlikely to occur, primarily because of the reality that, as Dixit (2014:70) observes:

“Unfortunately many such markets are missing or malfunctioning and externalities are ubiquitous.”²

The implications of this statement are that externalities arise as a direct consequence of market failures, in other words, the failure of markets to properly capture and account for particular goods and services in their transaction behaviours – this is one of the main arguments put forward to account for why ecosystem services and biodiversity have been at the mercy of historical exploitation and degradation (Perrings, 2014). In the case of ecosystem services and biodiversity, the argument goes, market failures arise for two main reasons: one is the result of the structuring of property rights³ and the other is a consequence of the fact that ecosystem services primarily take the form of public goods⁴ (Perrings, 2014). The effect that a poor structuring of property rights may have is to cause, landholders for example, to ignore the impact that their management regimes may have on other people, whilst the public goods nature of many ecosystem services means markets for them will not likely arise naturally (Perrings, 2014).

Provisioning services such as food, fibre etc. (we could term these agri-environmental goods) primarily display properties associated with traditional private goods (i.e. they are rival and excludable), whereas most of the other types of ecosystem services have features of non-rivalry and non-excludability, with some also being both public and international⁵. To the extent that ecosystem services are non-rival and non-excludable, then they are also considered to be either pure public goods or impure public goods (e.g. common pool resources), where in the latter case they are only partially non-rival or non-exclusive. For international public goods, like biodiversity, they may also be considered in terms of their “technology of supply” (Perrings, 2014)⁶. It is argued that both the purity of public goods and their technology of supply are important because it relates directly to the incentives individuals, communities or countries have to provide them, for example, with respect to ecosystem services that are considered international public goods Perrings (2014:175) points out that:

“...what we know about incentives to provide international environmental public goods, the incentive to countries to free-ride on others is greatest in the cases of additive supply technologies such as the conservation of endangered species, harvested wild-living species in areas beyond national jurisdiction, flood or coastal protection. It is least in the case of weakest-link technologies such as the management of infectious disease, quarantine and port inspections, pest control, or the eradication of invasive species. The incentive to individual countries to act unilaterally is greatest in the case of best-shot supply technologies such as vaccine development, or the provision of information about pest and pathogen risk. It is least in the case of additive technologies...”

The question then becomes how best to manage these externalities in order to ensure the maintenance and provision of environmental public goods, or applying theory to the real world of competing land-use priorities (e.g. production for consumption, or conservation for biodiversity-related ecosystem services), how best to managing the land for multiple land-uses or multiple ecosystem services – this necessarily means that three aspects need to be navigated: (i) internalising the externalities via regulatory means such as taxes or the assignment of property rights requires an authority to assign rights (which is particularly difficult at the transnational level); (ii) free-riding⁷; and (iii) developing market-based instruments or cooperative agreements to increase investment in the provisioning of ecosystem services (Perrings, 2014).

These issues, whilst problematic at an international-level – where agreements between countries have to be struck (activities that ultimately require the negotiating strategic behaviours) – are far less problematic at the national level where governments can authorise and allocate property rights, develop national incentive programmes, and create national agencies responsible for overseeing programmes that are designed to deliver environmental public goods (Perrings, 2014). There are generally two classifications of instruments available to governments to internalise externalities: instruments designed to address negative externalities (e.g. taxes, access charges, user fees, non-compliance penalties) and instruments designed to address positive externalities (e.g. subsidies, grants, payments for ecosystem services) (Perrings, 2014).

10.2 An Introduction To Market-Based Instruments

The chapters that follow represent case-study examples of instruments designed to internalise positive externalities, and in that respect whilst chapters 11 and 12 focus on payments for ecosystem services and chapter 12 concentrates on agri-environment schemes, they are united in the fact that both types of instrument effectively pay landholders, farmers, or communities to undertake land-use management activities designed to increase the supply of environmental public goods. In many respects these instruments can be loosely grouped, albeit somewhat imperfectly, under the collective heading of market-based incentive mechanisms (MBIs) (see Figure 10.1)

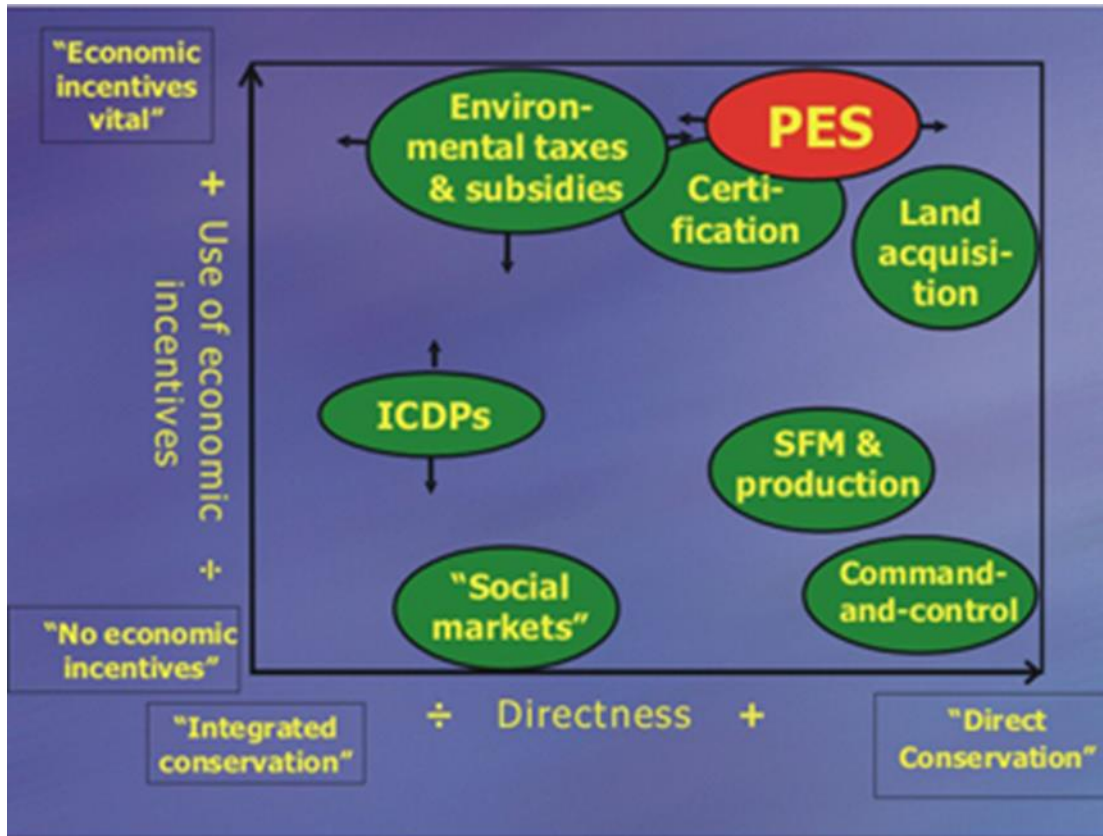


Figure 10.1 Representation of different instruments for supplying public goods (source: Wunder (2006) *Ecology and Society*, 11(2):23)

This is a useful point at which to introduce briefly the concept of MBIs. The application of MBI mechanisms to deal with the challenges of landscape and environmental protection, climate mitigation, wetland restoration and biodiversity conservation is growing (Gómez-Baggethun et al., 2010; Muradian and Rival, 2012; Pirard, 2012). This signals an underlying shift in national and international natural resource use policy (Farley and Costanza, 2010; Pokorny et al., 2012). The emergence of MBIs have been justified on the grounds that they correct market failures, reduce information asymmetry, provide price signals for decision makers, and bridge the conservation funding gap (Gomez-Baggethun and Ruíz-Perez, 2011; Pirard, 2012).

Despite these endorsements concerns remain. For some, MBIs represent a plurality of “hybrid governance” instruments that conflate conceptually different philosophies and mechanisms (i.e. rewards, incentives, markets), often addressing social–environmental problems not externalities arising from market failures (Muradian and Gómez-Baggethun, 2013; Muradian, 2013). There are also doubts over the ability of MBIs to adequately secure the provision of public goods and common pool resources (Muradian and Rival, 2012; Van Hecken and Bastiaensen, 2010; Kinzig et al., 2011; Lockie, 2013) whilst providing cost-effective policy (Kemkes et al., 2010). Other challenges include potential misapplication of

MBIs (Lockie, 2013); the propensity to commoditize nature (Kosoy and Corbera, 2010), which could lead to reductions in ecological complexity and a “commodity fiction” (Gomez-Baggethun and Ruíz-Perez, 2011; Muradian and Rival, 2012; Robertson, 2012); and the perception that MBIs represent encroaching neo-liberalist interventions (McAfee and Shapiro, 2010; McElwee, 2012; Arsel and Büscher, 2012; McAfee, 2012; Shapiro-Garza, 2013).

10.3 Final Remarks

The case study chapters that follow each take a different perspective of MBIs. The first two case study chapters look at payments for ecosystem services. In particular Chapter 11 assesses the effectiveness of these programmes (at the global scale) in delivering their stated outcomes: in achieving their goals across environmental, social and economic dimensions. Chapter 12 looks at the possibility of developing payments for ecosystem service programmes for a globally important and yet threatened marine coastal ecosystem, specifically seagrasses, and the prospect of developing these types of programme alongside other carbon management (i.e. so-called Blue Carbon) interventions. Finally, Chapter 13 looks at the case of agri-environment schemes in a UK context, and in particular a two tiered programme called Environmental Stewardship implemented in England under the auspices of Natural England. The chapter examines Environmental Stewardship schemes from the perspective of farm advisors – agents that act as intermediaries, working with farmers to develop scheme agreements whilst also in the process liaising with Natural England. Collectively, from what these chapters present, we get a sense of the positive social-economic and environmental benefits these programmes can generate, but also, the range of challenges and barriers they face (and need to consider) at different stages of the policy intervention (i.e. implementation, operationalisation and institutional matters) in terms of achieving their overall goals.

Notes

1. Dixit (2014: 69-70) makes the point that externalities are about side-effects, the result of choices and outcomes, and so can be considered in the following way:

“Many actions of consumers or firms have side-effects, beneficial or harmful to others [...] In many such situations, people and firms lack the incentives to take into account the by-product effects when making their choices. Alas, most of us are not sufficiently other-regarding to include the harm or benefit to others automatically in our calculations. When we ignore the harm our action imposes on others, we carry the action beyond the level that would be best for aggregate social efficiency; when we ignore the benefits to others, we do too little. That is why we see too much congestion on our roads, and sometimes dangerously low vaccination coverage of the population. Economists call such effects *externalities*, positive when they are beneficial to others and negative when they are harmful.”

Dasgupta (2007:53) expresses a similar definitional understanding of what constitutes an externality, but explicitly frames the meaning in terms of private choice:

“The private provision of public goods confers an extreme form of an effect known as *externalities*. By an externality, we mean the effects that decisions have on people who have not been party to the decisions. In some cases the effects can be beneficial (they are known as *positive externalities*); in other cases they are detrimental (*negative externalities*).”

Perrings (2014:155-156) supports this position, but also emphasises in the interdependencies that create externalities:

“Externalities arise because of interdependencies in either production or consumption. If one production or consumption activity affects another and those effects are not registered in any market transaction between parties, then they are said to be externalities of the activity.”

2. As Caplan (2008) notes in his summary essay introducing externalities:

“While the concept of externalities is not very controversial in economics, its application is. Defenders of free markets usually argue that externalities are manageably small; critics of free markets see externalities as widespread, even ubiquitous.”

3. According to Dasgupta (2008:47) property rights are relational, the linkage of access and ownership to specific commodities:

“Property rights to a commodity are the rights, restrictions, and privileges regarding its use.”

Alchian (2008) sketches this view out in a little more detail:

“A property right is the exclusive authority to determine how a resource is used, whether that resource is owned by government or by individuals. Society approves the uses selected by the holder of the property right with governmental administered force and with social ostracism.”

4. Public goods are those commodities that are both non-rival (in terms of consumption) and non-excludable (in terms of access). Public goods are non-rival in the sense that an individual’s consumption of a particular good does not diminish the availability of that same good to be consumed by another individual, and they are non-excludable in the sense that it is not possible for one individual to exclude another individual from consuming the same good (Dasgupta, 2008).

As Perrings (2014:158-159) remarks in with a particular ecosystem services framing of public goods:

“Ecosystem services that provide non-exclusive, non-rival benefits to people typically do so at many different scales. Pollination services, for example, tend to be quite localized. Watershed protection, on the other hand, can extend from extremely small scales to regions involving several countries. In all cases, the benefits of the service are non-exclusive in the sense that once the good is provided, none can be excluded from the benefits it confers. Some are also non-rival or indivisible, in the sense that consumption by one country or one group does not diminish the amount available for others.”

5. In many cases ecosystem services are jointly produced by various management regimes at different spatial and temporal scales, sometimes with different supply technologies, as Perrings (2014:175) states:

“Equally important is the fact that many environmental public goods are jointly produced both with other public goods and services, and that many of these public goods are produced at different scales. For example, tropical forests are the source of a set of private benefits (e.g., timber, medicinal plants, hunting, fishing, recreation, and tourism), but they also yield a number of public goods. Some of these are global such as carbon sequestration and genetic information, and some are local such as the regulation of the hydrological cycle, or micro-climatic regulation.”

6. Supply technology is the term used to describe the relationship between the benefits provided by particular public goods that are supplied by many countries and the contributions that each of those individual countries make – the significance of this is that technology supply influences the incentives around to encourage the supply of public goods as well as the costs associated with free-riding

behaviour (Perrings, 2014). Perrings (2014) identifies four types of supply technology: Additive (i.e., each community's contribution to the supply of a particular public good is unique and the benefits yielded from a specific public good are the sum of those individual contributions); Polar (this comprises two types – “best-shot”, the overall public good benefit received by communities is influenced by the community that is most effective in supplying that particular goods, and “weakest-link” - here the benefits received by communities from a public good are constrained by the least effective provider); stem-goods (i.e., a certain level of public goods provision is required to generate a flow of benefits but beyond that level of provision no extra benefit is received); and threshold goods (i.e., public goods are not supplied below a particular threshold of provision).

7. Free-riding can sometimes best be described in relation to national defence, as Zycher (2008) observes:

“National defence, like other public goods, has an important “free-rider” problem. That is, because people benefit whether or not they contribute toward defence, each person has an incentive to wait for others to provide the collective defence good, and thus hopes to get a “free ride.” Also, because a free rider’s consumption does not reduce the amount of defence available for others to consume, even those who pay have little incentive to prevent free riding by others.”

Chapter 11: Case Study 1 - Payments For Ecosystem Services: An Assessment Of Global Outcomes

“We need to develop and disseminate an entirely new paradigm and practice of collaboration that supersedes the traditional silos that have divided governments, philanthropies and private enterprises for decades and replace it with networks of partnerships working together to create a globally prosperous society.” (Simon Mainwaring, Huffington Post, 2011)

“Assessing global tendencies and impacts of conditional payments for environmental services (PES) programs is challenging because of their heterogeneity, and scarcity of comparative studies.” (Ezzine-de-Blas et al., 2016, pg.1)

There is a limited understanding of the conditions under which payments for ecosystem services (PES) programmes achieve improvements in ecosystem service flows, enhance natural resource sustainability or foster sustainable livelihoods. In this case study example, we systematically review global PES schemes – using a capital asset framework we evaluate these programmes in terms of their social, environmental, economic and institutional outcomes, and we place particular emphasis on efficiency, effectiveness and equity trade-offs.

11.1 Introduction

The MBI model, outlined in Chapter 10, has been applied in many developing countries in the form of payment for ecosystem services programmes (Shelley, 2011; van Noordwijk et al., 2012; Tacconi, 2012; Derissen and Latacz-Lohmann, 2013) as a policy tool intended to address a spectrum of land management challenges (Landen-Mills, 2002; Landell-Mills and Porras 2002; Wunder, 2006; Engel et al., 2008; Bond and Mayers, 2010). PES have been presented as an alternative to traditional command-and-control approaches, which through encouraging more decentralised management offer the potential to advance both conservation and rural livelihood development goals: thus they are often marketed as providing win-win opportunities – at once supporting conservation and the sustainable use of natural resources whilst concomitantly improving rural livelihoods (Ferraro and Kiss, 2002; van Noordwijk et al., 2007; Agrawal et al., 2008; Pokorny et al., 2012; Muradian and Rival, 2012). Yet, what constitutes PES, both in theory and practice, and PES success is open to debate (e.g. Wunder, 2005; Farley and Costanza, 2010; Muradian et al., 2010)¹. This is largely due to the plurality of financial arrangements underpinning PES schemes, which include government-financed, user-financed or hybrid co-financed arrangements, often involving external donors, such that the ways in which they function do not conform to a single operational standard (Schomers and

Matzdorf, 2013). Financially speaking, however, they can (generally) be thought of as a form of direct payment based on the beneficiary pays principle (Parker and Cranford, 2010). Within typical PES programs (e.g. Lin and Nakamura, 2012; Tacconi, 2012; Derissen and Latacz-Lohmann, 2013; Martin-Ortega et al., 2013), ES providers (e.g. landholders, farmers or communities) voluntarily participate in a program whereby they receive payments from ES buyers (e.g. a government, a utility or private organization). The set of transactions involved in these arrangements are generally facilitated by a single or multiple set of intermediary actors (e.g. a semi- autonomous body or non-governmental organization). In return for payments, providers adopt alternative land-use practices and management strategies that are considered to secure and deliver a set of important ES to a wider beneficiary population.

Institutionally, PES programmes are generally framed as decentralized instruments favoring bottom-up solutions to land management issues (Landell-Mills and Porras, 2002; Bond and Mayers, 2010). Despite the diversity of contexts in which PES schemes operate, they tend to adopt common modes of activity such as restricting agricultural development, proposing alternative cropping arrangements, reducing deforestation, and expanding forests (e.g. reforestation and afforestation), or protecting watershed and hydrological services (e.g. Asquith et al., 2008; Bennett, 2008; Muñoz-Piña et al., 2008; Wunder and Albán, 2008; Porras, 2010; World Bank, 2010; Kolinjivadi and Sunderland, 2012). Consequently, PES programmes generally involve multiple partners acting across sectors as well as spanning spatial and temporal scales (Schomers and Matzdorf, 2013).

However, the widespread adoption of PES masks important issues (Pirard et al., 2010). The validity and suitability of formulating PES theory on Coasean² grounds has been challenged because of the complexity, uncertainty, and asset specificity involved in managing ecosystem services (Farley and Costanza, 2010; Kosoy and Corbera, 2010; Muradian et al., 2010; Vatn, 2010; Muradian, 2013). Some argue that win-win conservation and development outcomes are likely if programmes are well designed (Pokorny et al., 2012; Kinzig et al., 2011), while others regard this as too optimistic given the influence of diverse contingent factors (Redford and Adams, 2009; Muradian et al., 2013). A number of practical obstacles may also hinder PES implementation, including: scheme design and payment structure (e.g. Engel et al., 2008; Kelsey Jack et al., 2008; Kemkes et al., 2010; Adhikari and Boag, 2012); modes of implementation (e.g. Engel and Palmer, 2008; Zhang and Pagiola, 2011); managing trade-offs arising from the need to balance efficiency, effectiveness and equity (e.g. Borner et al., 2010; Pascual et al., 2010, Narloch et al., 2011); institutional embeddedness and propensity to cooperate (e.g. Muradian et al., 2010; Vatn, 2010); spatial targeting, monitoring, participation, and compliance (e.g. Wünscher et al., 2008; Wendland et al., 2010); the adequacy of property

rights (Lockie, 2013); and social and well-being outcomes (e.g. Bulte et al., 2008; Pattanayak et al., 2010; Daw et al., 2011) [See Table 11.1 for a list of PES Review study summaries].

11.2 Study Aims

What, then, do these theoretical and practical debates mean for future PES prospects? Given that PES adoption will continue (Bond and Mayers, 2010), it is necessary to jointly assess both environmental and social effects to ensure long-term PES validation and effectiveness (Kelsey Jack et al., 2008; Farley and Costanza, 2010; Brouwer et al., 2011). To this end, we conducted a systematic review of the measured environmental and socio-economic outcomes of PES programmes. Systematic reviews are used widely in medical (Popay, 2006) and ecological sciences (Sutherland et al., 2004; Pullin et al., 2009) to gather evidence and generalise findings. We structured our review using a capital asset framework (CAF). The CAF originated as a rural livelihood assessment tool emphasising the interactions between individual- and community-level assets, and how collective action could be used to maintain various assets and resource flows to nurture local empowerment and foster development (Carney, 1998; Bebbington, 1999; Rudd, 2000; Green and Haines, 2008). The CAF connects socio-ecological context, institutional structure, the effects of changes in capital asset and their resource flows, and options for economic or political interventions based on actors' or societal values (Rudd, 2004). It has been used in diverse situations to analyse the transformative ability of assets to support rural livelihoods and reduce poverty in the Andes (Bebbington, 1999), assess poverty alleviation opportunities of a compensation-reward scheme for ecosystem services (van Noordwijk et al., 2007), identify barriers to the adoption of agricultural greenhouse gas mitigation measures in rural communities (Dulal et al., 2010), and appraise capacity-building requirements for tourism development in gateway communities bordering protected areas (Bennett et al., 2012).

We assessed the extent to which PES programmes represent effective environmental management tools based on their effects on social, environmental, financial and institutional capital assets. Our goal was to provide a means of appraising PES studies (and the programmes they describe) in a manner that enables improvements in scheme design, application and implementation. We systematically collated, consolidated and analysed PES literature describing specific programmes and the 'measured outcomes' of those programmes. We also collated observed barriers to PES uptake and the potential opportunities for enhancing PES programme success. Our approach builds on work by

Table 11.1 Summary of PES/PES-related review literature¹

Authors	Publication Date	Publication Type: Journal (J), Organisational Report (OR NGO, OR Govt, OR combined)	Publication Source	Review Type: Traditional (T), Narrative (S), Systematic, Hybrid (H)	Review Scope: National (N), Regional (R), Global (G)	Areas requiring further consideration to improve PES application
Landen-Mills, N. and Porras, I.T.	2002	OR NGO	IIED	H (A Traditional review with some systematic elements)	G (The document presents a global review of carbon (75), watershed (61), biodiversity (72), landscape beauty (51) and bundled services (28). Identifying a total of 287 market-oriented case-studies (proposed, active and inactive)	1. Formalise property rights and secure land tenure. 2. Link land management practices to service delivery, alongside clearly identifying and defining services and commodities. Integrate governance structures.
Mayrand, K. and Paquin, M.	2004	OR NGO	Unisféra	T (supplemented with expert opinion)	G (The purpose of the review document is to assess underlying differences and similarities as well as associated strengths and weaknesses of PES models, by evaluating schemes operating in the Western Hemisphere. The report focuses on 25 schemes operating in 15 countries)	3. Increase capacity building, legal and technical capacity.
Kumar, P. Bond, I. Dilaka <i>et al.</i>	2005 2007 2007	OR NGO OR NGO OR Govt	IISD IIED USAID	T/N T/N T	G G G	4. Improve institutional infrastructure, strengthen institutional resilience and overall sector cooperation.
Porras, I.T. <i>et al.</i>	2008	OR NGO	IIED	H (A Traditional review with some systematic elements)	G (Updated analysis of the Landen-Mills and Porras 2002 report)	5. Provide flexible contractual agreements and ensure compliance.
Wunder, S., Engel, S., & Pagiola, S. Southgate, D. and Wunder, S.	2008 2009	J J	Ecological Economics Journal of Sustainable Forestry	T T	G R (Latin America PWS)	6. Improve payment amounts and flows alongside better targeting.
Ferraro, P.J.	2009	J	Journal of Sustainable Forestry	T	R (Africa PWS)	7. Provide stable funds for projects.
Huang, M. <i>et al.</i>	2009	J	Journal of Sustainable Forestry	T	R (Asia PWS)	8. Reduce transaction costs and take proper account of opportunity costs.

Table 11.1 *Contd.*

Authors	Publication Date	Publication Type: Journal (J), Organisational Report (OR NGO, OR Govt, OR combined)	Publication Source	Review Type: Traditional (T), Narrative (S), Systematic, Hybrid (H)	Review Scope: National (N), Regional (R), Global (G)	Areas requiring further consideration to improve PES application
Prasetyo, F.A. <i>et al.</i>	2009	J	Journal of Sustainable Forestry	T	N (Indonesia)	9. Ensure greater participation and accessibility for poorer sectors.
Villamor, G.B. and Rodel, L.D.	2009	J	Journal of Sustainable Forestry	T	N (Philippines)	10. Need to situate PES practice so that it is seen as the interface between pico-, micro-, meso-, macro- and gigaeconomics where discounting, efficiency and brain systems interact.
Stanton, T. <i>et al.</i>	2010	OR NGO	Forest Trends/Ecosystem Market Place	H (A Traditional review with some systematic elements)	G	11. Generate a better understanding of how fairness and equity function in relation to rights and performance across scales
Morrison, A. and Aubrey, W.	2010	OR Combined	WWF/Federal Ministry for Economic Cooperation and Development (Germany)/BioClimate Research and Development	T	G	12. In relation to watershed schemes there is an increasing focus on trans-boundary programmes, climate mitigation and adaptation and bundling and stacking of ESs but there is a need for greater financial stability to ensure new or developing programmes once started can be maintained. Furthermore, greater private sector participation is required.
Yamaski, S. <i>et al.</i>	2010	J	CAB Reviews	T/N	General review of PES as a market mechanism not case study based	
Pattanayak <i>et al.</i>	2010	J	Review of Environmental Economics and Policy	T	G/R	

Table 11.1 *Contd.*

Authors	Publication Date	Publication Type: Journal (J), Organisational Report (OR NGO, OR Govt, OR combined)	Publication Source	Review Type: Traditional (T), Narrative (S), Systematic, Hybrid (H)	Review Scope: National (N), Regional (R), Global (G)	Areas requiring further consideration to improve PES application
Daniels, A.E. <i>et al.</i>	2010	J	Ecological Economics	H (A Traditional review with some systematic T)	N (Costa Rica)	
Nonga, F.N.	2011	J	Sustainable Development in Africa	T/S (meta-analysis) – primary and secondary data supplemented with a mail survey of PWS managers T	R (Congo Basin)	
Brouwer, R.; Tesfaye, A. and Pauw, P	2011	J	Environmental Conservation	T/S (meta-analysis) – primary and secondary data supplemented with a mail survey of PWS managers T	G (Investigates the connection between the institutional arrangement of PWS schemes and the effectiveness of environmental outcomes. Specifically, the institutional- economic factors that explain environmental performance.)	
Lin, H. and Nakamura, M	2012	J	Lakes & Reservoirs: Research and Management	T	G (This study collates information from 163 PWS schemes across 34 developing countries. Using these schemes, and in particular assessing their structural and institutional arrangements, the authors introduce the concept of an integrated ecosystem management approach to PWS, with particular reference to lake basin governance.)	
Noordwijk, M van <i>et al.</i>	2012	J	Annual Reviews in Environment and Resources	T	General review of PES as a market mechanism not case study based	
Bennett, G. <i>et al.</i>	2013	OR NGO	Forest Trends/Ecosystem Marketplace	T/S (Uses a combination of programme	G (Global review of the current state of watershed payment schemes – in the broadest sense)	

Table 11.1 *Contd.*

Schomers, S. and Matzdorf, B	2013	J	Ecosystem Services	information, interviews and published material) T/S	G (The paper reviews 457 articles obtained through a structured literature search, and addresses four areas: (1) the economic conceptualisations of PES, (2) Priority research foci for PES identified in the literature, (3) comparison of developed and developing nation applications of PES and (4) potential transference of best practice between developed and developing nations
Martin-Ortega, J.; Ojea, E. and Roux, C.	2013	J	Ecosystem Services	T/S	R (The authors collected 310 observations derived from 40 PWS schemes taken from a literature search spanning 1984 to 2011 concerning programmes operating in Latin America. The paper then sets out three objectives on this basis of this collected evidence: (i) evaluate and describe key PWS characteristics, (ii) Identify where the knowledge gaps lie and (iii) contrast their evidence with standard PES theory)

¹ See Table S11.1 in Suppl. Material B on CD for a fully annotated version of this table. The table lists 24 review studies; however, this is not necessarily an exhaustive list even though it does include some of the principal PES reviews undertaken over the last decade.

Wunder et al. (2008), Daniels et al. (2010) and Pattanayak et al. (2010) but, by adopting a CAF approach, introduces a new means by which PES programme management interventions can be systematically appraised.

11.3 Materials And Methods

Following various guidelines for systematic and related reviews (e.g. Petticrew and Egan, 2006; Cooper, 2010; Centre for Evidence- Based Conservation, 2013) our sequential four step process to the systematic review (Figure 11.1) proceeded from evidence gathering to critical analysis.

11.3.1 Step 1 – Search Strategy

Relevant studies were located via three sources: scientific databases; internet searches and websites; and journal special issues. Databases we searched included: ISI Web of Knowledge (all databases); Science Direct (SciVerse); Scirus; and OvidSP (Table S11.2 Suppl. B on CD). Internet searches were performed using Google (Table S11.3 Suppl. Material B). Searches used combinations of keywords and the first 50 hits retrieved were checked for relevance (Davis and Pullin, 2006; Bowler et al., 2010). We searched websites of specific organisations with known MBI expertise and involvement (e.g. FAO, World Bank, Global Environment Facility, WWF, Conservation International, Ecosystem marketplace, Watershed Markets, Katoomba group, World Agroforestry Centre and Centre for International Forestry Research). Journal special issues focusing on PES included three from Ecological Economics (65 (4), 69 (7), 69(11)), and one each from Journal of Sustainable Forestry (28 (3–5)) and Environmental Conservation (38 (4)). We restricted our source documents to those written in English but made efforts to locate English translations of non-English documents whenever possible. All document types were accepted (e.g. articles, conference papers, theses, chapters and reports as long as the provenance of the texts could be verified).

11.3.2 Steps 2 and 3 – Document Screening

The preliminary screening process focused on article title and abstract relevance, and used a standardised protocol applied to all documents to generate a first cut of “relevant” articles (Table S11.4 Suppl. Material B). A second, more detailed, screening was applied to those documents to obtain the final sample frame; we considered article type, theoretical content, and empirical evidence, and used a standardised protocol (Table S11.5 Suppl. Material B) in conjunction with additional study inclusion and exclusion criteria (Table 11.2).

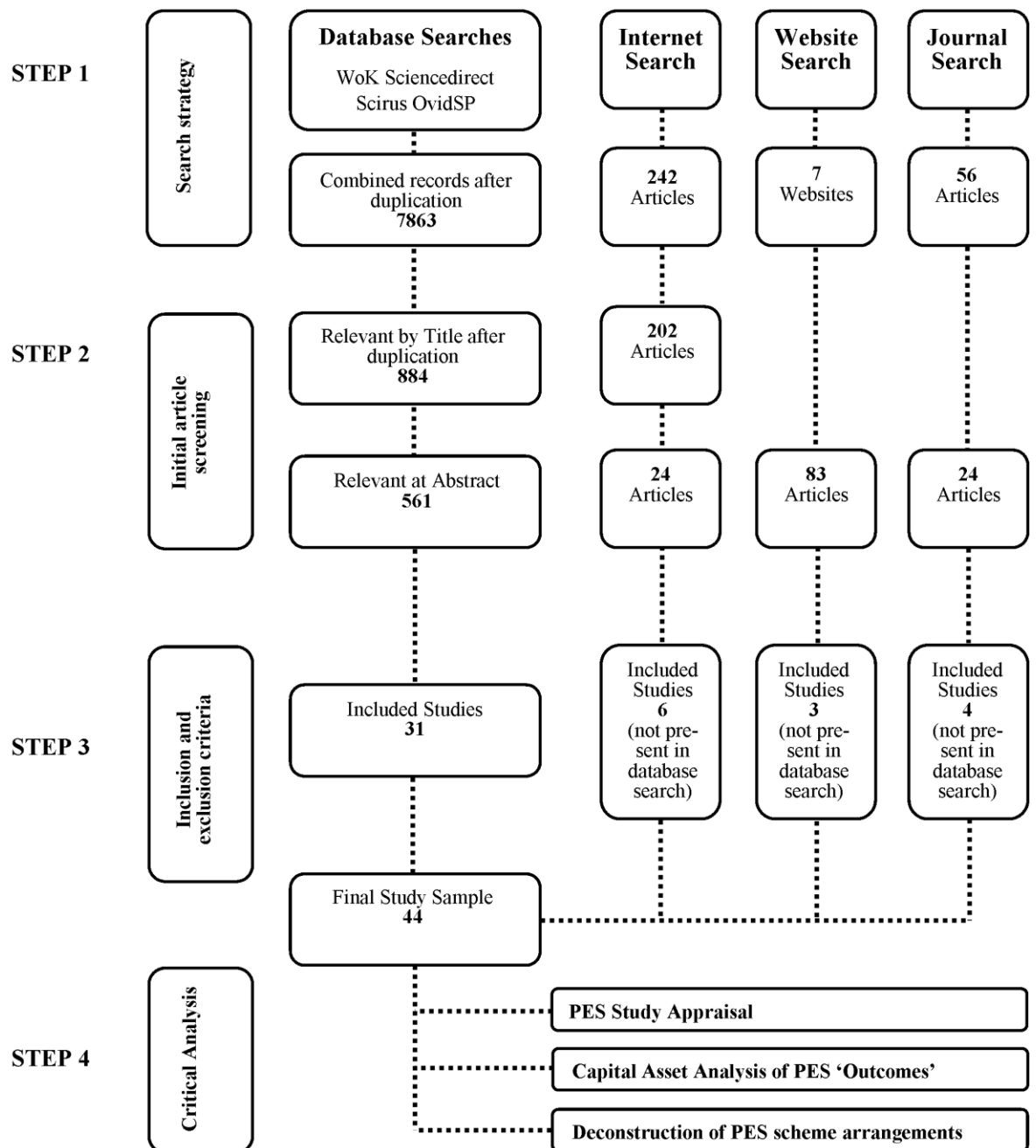


Figure 11.1 Flow diagram outlining the four steps of the systematic approach

11.3.3 Step 4 – Critical Analysis

Following Wunder et al. (2008), Pattanayak et al. (2010), and Daniels et al. (2010), we pursued three appraisal avenues to assemble our collection of studies: study appraisal (i.e. detailing the principal methodological characteristics of each study); PES programme evaluation (i.e. the application of the CAF to assess programme outcomes); and PES programme deconstruction (i.e. dissecting the operational, institutional, and financial arrangements of the specific projects identified within the collection of studies) (Figure 11.2). For each aspect, standardised coding protocols were employed to extract relevant information systematically and accurately across all studies (Tables S11.6–S11.12 Suppl. Material B [raw data tables are provided in a separate Suppl. Material D on the included CD]).

Table 11.2 Inclusion and exclusion criteria applied to select and determine the study sample

Inclusion Criteria	Exclusion Criteria
<p>1. The intervention being assessed by a study is wholly or primarily PES focused, where an intervention is defined as:</p> <p>An environmental externality addressed via a payment (which may or may not be performance related) received by a seller or provider of an environmental service from a private company, NGO, local or central government agency. The user is distinguishable from the seller, who is not a central government agency. The buyer does not have complete control over the production of the outcome, whereas the seller has partial or total control over the production of the outcome. Voluntary in principle on the supply side. (Based on adjustments to Wunder’s (2005) definition by Porras et al. (2008) and Ferraro (2009))</p> <p>2. The influence of PES interventions on specific environmental, socio-economic and/or institutional outcomes ought to be identifiable.</p>	<p>1. PES intervention is not the main aspect of the study assessed</p> <p>2. Articles focused on other market-based instruments, specifically:</p> <ul style="list-style-type: none"> · Cap and Trade schemes · Biodiversity/wetland banking · North American and EU agri-scheme payments · REDD/REDD+ <p>3. No detailed information regarding programme environmental, social, economic or institutional outcomes.</p> <p>4. General PES discussion/opinion papers concerning broad themes rather than specific PES programmes and their impacts</p>

Capital asset data were of two types. First, some data reflected the interpretation of theoretically relevant attributes for various assets. Second, in situ “measured outcomes” were detailed for individual studies. Those “measured outcomes” we considered to represent “effective” (i.e. beneficial or positive) programme impacts are detailed in Table 11.3.

A number of capital asset categorisations are recognised in the literature, from natural, human, social, cultural and produced (built, physical, or manufactured) capital (Bebbington, 1999) to financial and political capital (van Noordwijk et al., 2007; Bennett et al., 2012). Our framework consisted of human and social capital as an aggregated asset, natural capital, financial capital, and institutional capital that focused on conservation-relevant and development-relevant properties, characteristics or evaluation qualities. In other words, our ‘conservation-development perspective’ focused on those aspects relevant to those particular contexts (e.g. social mobility, access to social resources, land-use types, changes in ecosystem services, payment distribution and equity, and institutional accountability and transparency).

Natural capital refers to the structure, function and flows of ESs to humans as well as the land management practices and changes in those practices that PES programmes may cause (Costanza and Daly, 1992; Daily, 1997, van Noordwijk et al., 2007). Financial capital relates to the wealth of households and communities, the flow of funds available for

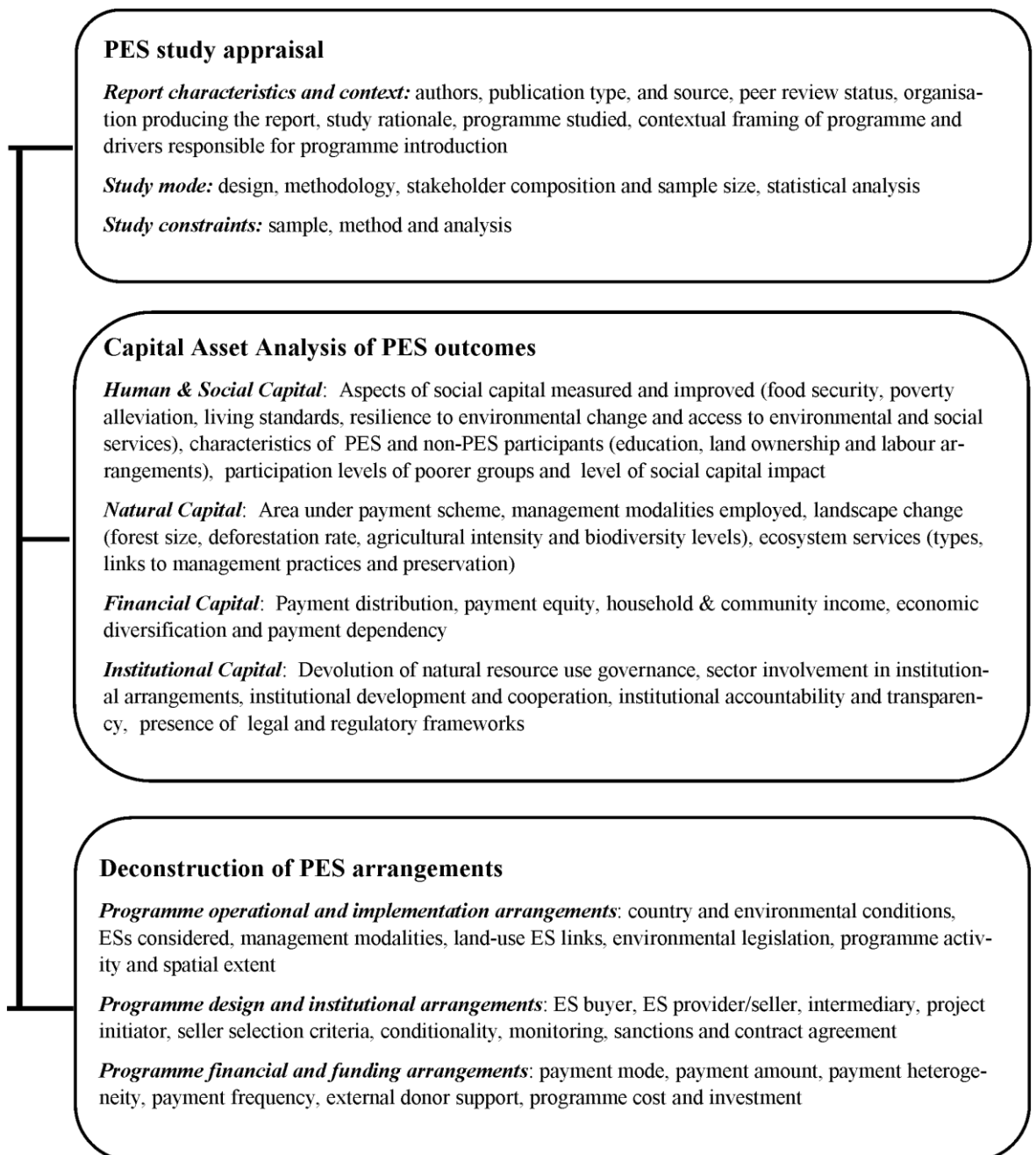


Figure 11.2 Critical analysis: a three part process comprising study appraisal, capital asset evaluation of PES 'outcomes' and deconstruction of programme arrangements

undertaking activities and payment distribution and equity (Rudd, 2004; Bennett et al., 2012). Human capital constitutes skills, knowledge, experience, and health at the individual level (Rudd, 2004; Brondizio et al., 2009; Behrman, 2011; Winters and Chiodi, 2011; Bennett et al., 2012; Moav and Neeman 2012), while social capital refers to social structure and relations that contribute to flows of norms and reputation-based trust (Bebbington, 1999; Rudd, 2000; Adler and Kwon, 2002; Brondizio et al., 2009). Finally, we use the term institutional capital to refer to aspects of resource governance and institutional transparency and accountability. Elsewhere, institutional capital has also been referred to as the structural attributes of organisations, institutional norms, and the capacity to build competencies (de los Hoyas and

Antunez Diaz, 2012; Valente, 2012). While what we denote as institutional capital reflects features of human and social capital, the scope of this asset does not fit neatly within common conceptions of human or social capital. In light of this we consider it justified to include a separate asset that considers the wider institutional, organisational and governance-related perspectives of a specifically environmental management intervention.

Table 11.3 CAF categorisation of “effective” PES programme “measured outcomes”

Capital Asset	‘Measured Outcomes’ of PES Programmes judged ‘Effective’*
Natural Capital	<ul style="list-style-type: none"> ● Increase in forest size, protected area extent and decrease in deforestation ● Reduction in agricultural intensity ● Alteration in agricultural practices (e.g. adoption of programme modalities) ● PES specifically acknowledged to be an effective and efficient mechanism to induce changes in land-use ● PES activities undertaken in areas of poor environmental condition ● Improvements in biodiversity (e.g. conservation of a specific species) ● Ecosystem service(s) identified ● Ecosystem service provision assessed ● Link between management practice and ecosystem service production ● Ecosystem service(s) preserved
Financial Capital	<ul style="list-style-type: none"> ● Small landholders receiving payments ● Medium landholders receiving payments ● Observed increase in household income ● Diversification of household economic activities ● Improved distribution of material wealth ● Payments favour poorer land owners ● PES participants more reliant on payments for household finances (i.e. better targeting of poorer sectors) ● PES participants have more diverse income streams than non-participants (i.e. more economically resilient) ● Payments are sufficient to meet household needs and/or provide a suitable alternative income stream
Institutional Capital	<ul style="list-style-type: none"> ● Community control over natural resource-use ● Decentralised administration control over fund disbursement and contract awards ● Greater involvement of local institutions ● Improved institutional relationships and cooperation ● Institutional accountability assessed ● Increased institutional accountability & transparency ● Increased transparency in funding chain ● Providers more accountable to beneficiaries ● Legal and regulatory measures in place to ensure proper resource-use
Social Capital	<ul style="list-style-type: none"> ● Improved food security ● Reduction in poverty ● Improved living standards ● Resilience to environmental change ● Better access to social and environmental services ● Increased poorer household participation

*We can say that there are certain ‘measured outcomes’ that are representative of an ‘effective’ PES programme. Here, our use of the word ‘effective’ implicitly acknowledges that a ‘measured outcome’ is positive or beneficial in some respect. There is no predefined exogenous objective method for determining what ‘measured outcome’ is deemed ‘effective’; determination is rather both normative and common sensical. These ‘measured outcomes’ derive from the coding applied to assess each individual study engaged in evaluating a PES programme.

11.4 Results And Discussion

11.4.1 Critical Analysis: Study Appraisal

We used a total of 44 studies in our analysis (Table 11.4). They were primarily from the peer-reviewed literature (71%) and in total considered 23 PES programmes operating at local and national scales in 13 countries (Table S11.13 Suppl. Material B). We found scholarly work concentrated largely on implementation and outcome evaluation, primarily in relation to natural and financial capital. The main geographic focus was Latin America, which has historically been the main testing ground for PES. However, PES initiatives were also identified in Asia (particularly China) and Africa, although these programmes were fewer in number.

Our cases employed multiple theoretical approaches to assess programme outcomes and highlight the discourses and drivers promoting the contextual development of PES. Eighty four per cent of studies were multi-modal, using one or more theoretical approaches and evaluation measures (Figure 11.3).

In general, studies assessed programme additionality (66%), livelihood sustainability (22%) and participation (20%). They situated programme-level developments within a predominantly historical (82%) and environmental conservation (95%) frame of reference that emphasised land-use change (98%), water protection (55%) and climate mitigation (50%) as the principal drivers of scheme introductions (Figure. 11.4). Similarly, in their regional analysis of payments for watershed services (PWS) in Latin America, Martin-Ortega et al. (2013) identified deforestation and loss of land cover to be a comparable driver (77%) of PWS scheme development. Poverty alleviation, surprisingly, was mentioned in only 27% of cases as a key driver of PES development despite the increasingly pro-poor rationale for PES and the recognition of the effects that poverty can have on natural capital (Bulte et al., 2008).

Various experimental designs were employed across the studies. Comparative matched-sample approaches commonly focused on qualitative assessments achieved through survey-related methodologies. However, relatively little attention was paid to assessing social and institutional factors and developing more explanatory social–ecological models (Figure 11.5). Studies exhibited an array of sampling (66%), methodological (75%) and analytical (27%) limitations (Figure 11.6).

Table 11.4 Summary of the final selected articles: geographical focus, PES schemes investigated, and scale of operation

Geographical location	No. of studies ¹	PES Programme and scale: Local (L), Regional (R), National (N)
Costa Rica	16	PSA ² (N)
Mexico	7	PSAH ³ (N), PSA-CABSA ⁴ (N), Fidecoagua (L)
Ecuador	4	Pimampiro (L), PROFAFOR ⁵ (R), SocioBosque (L)
Nicaragua	4	RISEMP ⁶ (L), PPSA-H ⁷ (L), San Pedro del Norte – PASOLAC ⁸ (L)
Bolivia	2	Los Negros (L), NKMCAP ⁹ (L)
Columbia	1	RISEMP (L)
Honduras	1	Jesus de Otoro – PASOLAC (L)
Brazil	1	Bolsa Floresta (L)
Madagascar	2	Durrel Conservation Trust PES Scheme (L)
Mozambique	1	Carbon Livelihoods Project
Kenya	1	WKEIMP ¹⁰ (R)
Cambodia	1	Payments for wildlife friendly products, community-based ecotourism, bird nest scheme (L)
China	5	SLCP ¹¹ (N), NFP ¹² (N)

¹ 44 case studies. The numbers do not sum to 44 as some studies focused on more than one PES programme.

² Pagos por servicios ambientales

³ Payments for Hydrological Environmental Services

⁴ PES programme for carbon sequestration and biodiversity conservation

⁵ Programma Face de Forestaciùn del Ecuador

⁶ Regional Integrated Silvopastoral Ecosystem Management Project – Operates transnationally but in each area at a local level

⁷ Proyecto de Pagos Por Servicios Ambientales Hidricos

⁸ Programa para la Agricultura Sostenible en Laderas da América Central – Operates transnationally but in each area at a local level

⁹ Noel Kempff Mercado Climate Action Project

¹⁰ Western Kenya Integrated Ecosystem Management Project

¹¹ Sloping Land Conversion Programme

¹² National Forest Programme

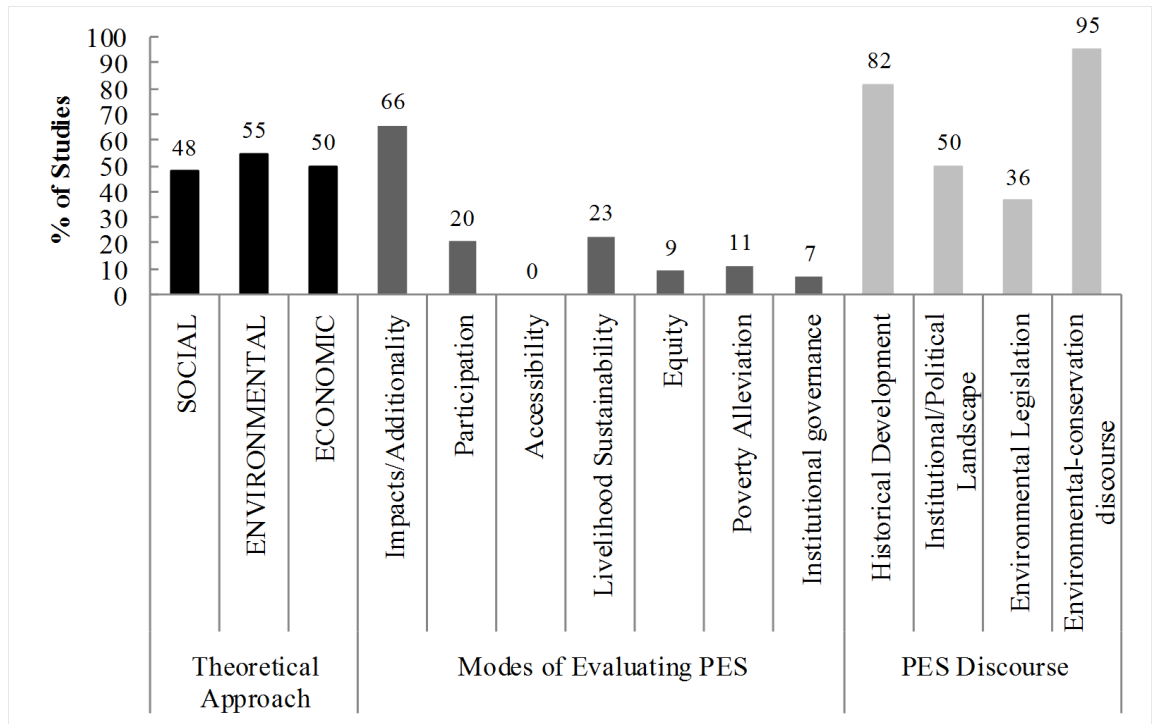


Figure 11.3 Theoretical approaches applied to PES studies: emphasising the discourse in which PES development is situated

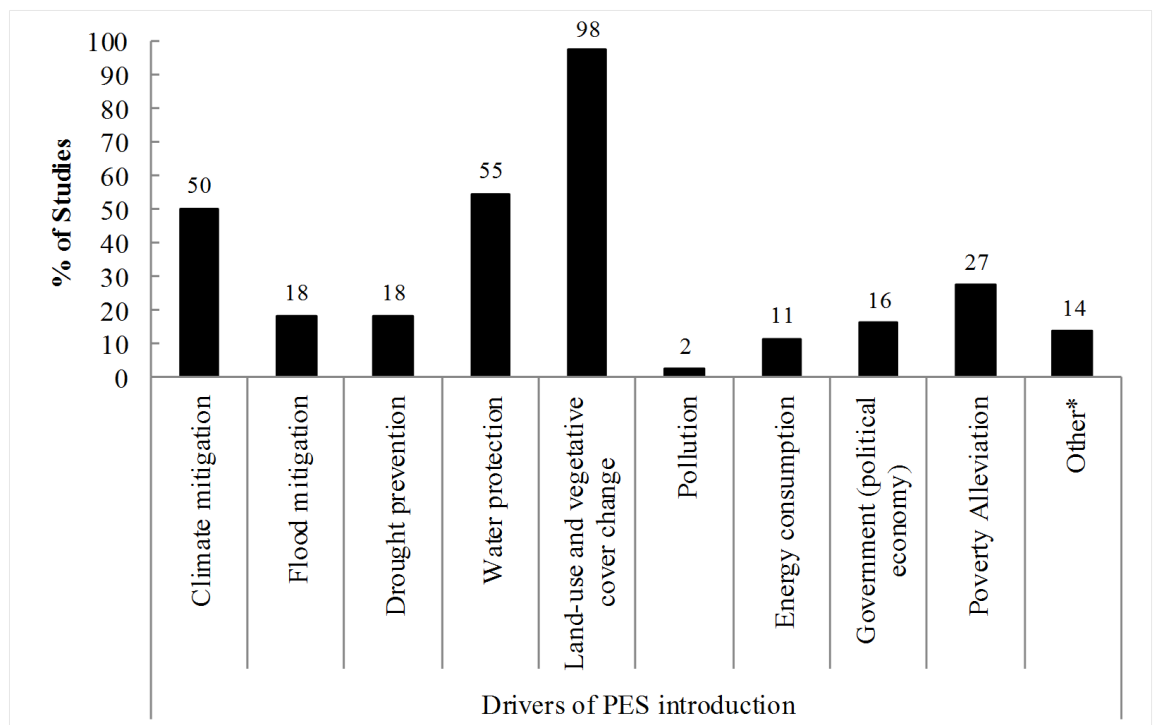


Figure 11.4 Drivers motivating PES scheme development. *Other refers to: urbanisation, population growth, food security and biodiversity threat

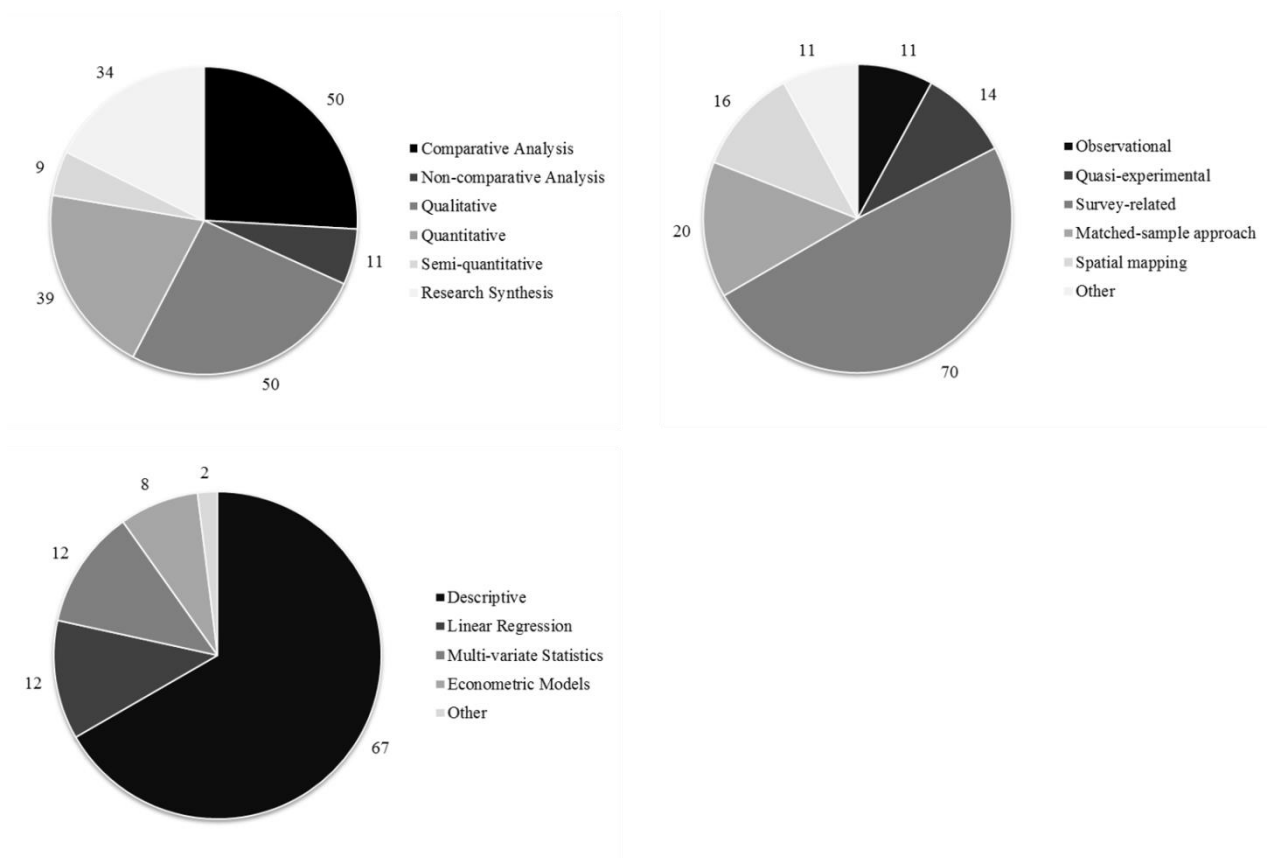


Figure 11.5 (a) Study design (b) Study mode (c) Data analysis. All numbers refer to percentages of studies

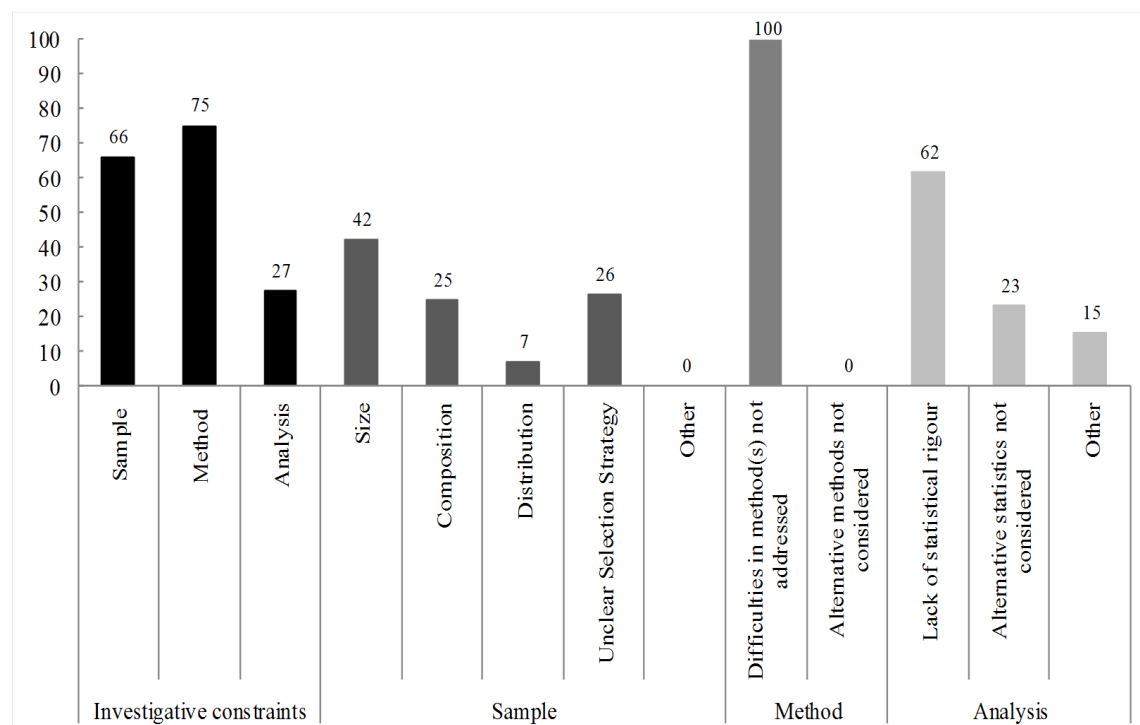


Figure 11.6 Investigative constraints of reviewed studies at the sample, method and analysis stages. All numbers refer to percentage of studies.

11.4.2 Critical Analysis: Evaluating Programme Arrangements And Outcomes

11.4.2.1 Human And Social Capital

Ecosystem services contribute to livelihood development at different spatial scales and through varying combinations (Willemen et al., 2013). Meeting development needs, alleviating poverty, and enhancing well-being are increasingly important roles for PES (Bulte et al., 2008; Lipper et al., 2009; Daw et al., 2011). However, only 52% of studies we assessed specifically evaluated human and social capital implications of PES programmes. The lack of focus on PES social dimensions may reflect the general division in the research community between those that view PES as a development tool (e.g. Muradian et al., 2010) and those arguing its development function is (and should be) secondary to its conservation function (e.g. Wunder, 2008). Milder et al. (2010) argue that the extent and influence of pro-poor PES have been inadequately quantified. While this is certainly the case, evidence from our study indicated that programmes can have a general, albeit conservative, positive social impact. Certainly a lack of evidence concerning the social impact of PES on non-participant households within targeted communities needs further investigation (Huang et al., 2009).

In part, the confusion regarding the social impacts of PES programmes, outlined in the preceding paragraph, arises due to the difficulties in comprehensively identifying potential ES beneficiaries and understanding how different programme strategies are likely to influence the distribution and magnitude of ES supply (Willemen et al., 2013). This suggests that PES may unrealistically promote win-win outcomes by simplistically claiming to have resolved the problems faced by earlier Integrated Conservation and Development Programmes (ICDPs) (Muradian et al., 2013). Research in Mexico, for example, has suggested that PES enhances a short-term utilitarian view of conservation (Rico García-Amado et al., 2013). In contrast, ICDPs are perceived as long-term conservation endeavours designed for specific community-level developments but may not be viable economically (Rico García-Amado et al., 2013). It is significant that the human and social capital measured outcomes in PES have quite broad human development implications relating to living standards (26% of studies examined), better access to environmental and social services (26%), poverty alleviation (17%), food security (13%), and resilience to environmental change (11%). These outcomes bolster recent commentaries advocating a realistic approach to PES design based on achieving attainable objectives (Muradian et al., 2013) and alignment of PES and ICDP practices that captures their respective benefits (Rico García-Amado et al., 2013).

The socially transformative capacity of PES is linked to scheme access, which is underpinned by eligibility and participation (Mahanty et al., 2013). A number of investigations have evaluated the extent to which ES sellers have benefitted from programme participation. The results have been mixed, although marginal benefits have been identified at the household

and community level (Milder et al., 2010). Twenty-nine per cent of studies in our review viewed the ability to access relevant scheme information as a major barrier to participation. In this respect social status and wealth may affect PES participation rates even when eligibility is not an issue (Mahanty et al., 2013). For example, where examined, those wishing to sign-up for entry into a PES programme were wealthier, better educated, owned larger tracts of land and were more socially mobile compared to non-participants. This supports the view (Mahanty et al., 2013) that skill level, education and negotiating ability are important determinants of scheme participation. Furthermore, poor economic development policies can constrain pro-poor livelihood strategies by failing to recognise the underlying characteristics of the poor (Smith, 2005; Fisher et al., 2008).

Stakeholder and community participation is vital for promoting individual and community empowerment, enabling access to resources and information, developing wider support networks and access to markets, and securing economic stability and land reform (Smith, 2005; Fisher et al., 2008). Tenure arrangements, community capacity, and coherent livelihood development strategies are critical issues for stimulating participation (Brewer et al., 2014). Despite the centrality of participation to programme success, its evaluation – particularly in relation to poorer households – is largely ignored, highlighting the limited role of social embeddedness in PES evaluation (Muradian et al., 2010). This sentiment is illustrated in our analysis, where only 27% of prior studies explicitly recognised the need to improve poorer household uptake rates.

The choice of ES providers is fundamental for PES to achieve significant poverty alleviation (Muradian et al., 2010). The chosen selection model must balance efficiency, effectiveness, and equity trade-offs (Unisfera International Centre et al., 2004) with fairness (Pascual et al., 2010) while maintaining cost-effectiveness (Chen et al., 2010). Ultimately, selection should reflect participant socio-economic circumstances and biophysical properties likely to maximise ES provision. We found most service sellers were farmers (51%), communal landholders (23%) and indigenous communities (14%), with 70% of programmes targeting one seller group. The selection of those sellers was often made primarily on ecologically important criteria (e.g. priority areas, biophysical conditions, strategic service site location, land and farm characteristics, herd size and livestock and the production of a management plan). The use of multiple criteria was generally low; 78% of programmes stipulated just one or two selection criteria. Still, this suggests natural capital optimisation over social capital maximisation, as most programmes considered few, if any, social criteria in their eligibility requirements. Bolivia's Los Negros programme is a case in point: the programme's criteria automatically excluded the poorest landless immigrants living within the PES implementation zone (Aquith et al., 2008).

Even where social criteria were considered (e.g. the Social Development Index devised for Costa Rica's PSA programme), they may be fundamentally at odds with the scale at which they need to operate (Porras, 2010; Matulis, 2013). The result may be the potential exclusion of large numbers of poorer households and de-emphasising social welfare concerns in programme design and implementation (WRI, 2005). However, in some cases, particularly in relation to China's Sloped Land Conservation Programme (SLCP), evidence suggests that poorer members of society were effectively captured by PES programmes. Liu et al. (2008) suggest that 30 million farming households have benefited directly from the SLCP. There was a strategy to target poorer marginalised households and communities to enhance SLCP impact (Yin et al., 2013).

11.4.2.2 Natural Capital

We found programmes extended across multiple landscape types operating mainly in agricultural (74%) and tropical rainforest and dry forest landscapes (65%). They ranged from lowland (69%) to highland (48%) geographies and across rural areas (52%). These broad landscape configurations mask high levels of heterogeneity, even over small ranges, with most being multi-functional landscapes dominated by smallholder farmers (Tschardt et al., 2012). Most programmes (91%) were implemented over spatial scales encompassing three or more distinct landscape types, and focused principally on delivering hydrological/watershed (52%), carbon/forest (61%), biodiversity (56.5%), and food and fibre (22%) services.

Despite landscape multi-functionality, only a few programmes targeted ES bundles (e.g. PSA in Costa Rica, Socio Bosque in Ecuador). Our findings contrast with Martin-Ortega et al. (2013), who found 73% of PWS transactions involved bundled services. Ingram et al. (2014) recently argued that bundling and stacking ESs can reduce the risks associated with unstable markets. However, 78% of programmes we examined focused quite narrowly on one or two ESs. Similarly, with respect to the land-use practices adopted by participants, 74% of programmes relied heavily on one or two management practices to achieve ES provision. The assumption that individual or coupled land-use practices are sufficient to generate ESs at adequate rates, spatial scales, and levels of availability currently informs most PES programme design (Schomers and Matzdorf, 2013). In total, 84% of studies that we examined measured aspects of natural capital primarily through documenting land-use changes, rather than focusing on the provision of jointly-occurring ESs. Land-use change is likely a poor proxy for ES provision because change occurs, generally, as a consequence of utilising just one or two land-use practices. This is insufficient to guarantee service supply especially in the case of multiple ESs (Bennett et al., 2009; Raudsepp-Hearne et al., 2010). Further, failing to acknowledge the connections between targeted ESs in PES programmes hinders the ability to assess ES provision, distribution and trade-offs, as well as to identify adequately factors

affecting service delivery such as the extent and type of land-use practices adopted by participants (Bürkhard et al., 2010; de Groot et al., 2010). Not accounting for this information could increase transaction costs, lead to contradictory regulations, degrade market compatibility, and forfeit potential win-win opportunities (Deal et al., 2012).

Similar to other research (Landen-Mills and Porras, 2002; Wunder et al., 2008), we found land-use practices were frequently geared towards forest protection (65%), reforestation and afforestation (52%), and reductions in extractive activities (30%). However, the extent to which these land-use practices deliver ESs is dependent upon high adoption rates and consistent employment by participants (Wunder et al., 2008). Adopted land management practices were generally regarded as effective in producing the stipulated land-use changes. In Colombia's Regional Integrated Silvopastoral Ecosystem Management Programme (RISEMP), for example, significant reductions in degraded pasture (78.3–7.1 ha) and natural pasture without trees (721–239 ha) alongside increases in improved pasture with high tree density (2.2–266 ha) were observed (Pagiola et al., 2010). At the global scale, increases in forest extent and decreases in deforestation rates were four fold more frequently identified than reductions in forest size and increases in deforestation rates. For example, China's SCLCP converted 408,000 ha yr⁻¹ of cropland to forest and grassland during the initial pilot phase (1998–2001), and a further 2.3 million ha yr⁻¹ from 2002 to 2003 (Bennett, 2008). Furthermore, one-third of studies demonstrated a notable reduction in the degree of agricultural intensity undertaken within project areas, with almost half of those suggesting shifts away from traditional cropping activities towards the development of timber plantations and forest management or protection.

The adoption of management practices can be hampered by technical, infrastructure, and payment constraints. For instance, we observed land management practice restrictions (27% of cases) and farm area or forest size requirements if taking land out of production was required (11%). This was particularly evident for programmes with specific criteria for management practices such as minimum farm size (e.g. PROFAFOR, Ecuador) or specific management plans (e.g. Los Negros, Bolivia). Reducing the number of management practices may increase the probability that they are jointly adopted and practiced (Engel et al., 2008; Wunder et al., 2008). However, although this action may increase the overall implementation of management practices the result may still be insufficient provision of ES because of the limited number of management practices employed (Bennett et al., 2009).

Even with 84% of studies targeting particular ecosystem services, 73% lacked evidence to demonstrate programmes were providing those services. This supports the view that land-use changes are not easily translated into ES provision (e.g. Bond, 2007; Wunder et al., 2008; Bennett et al., 2009). We found that 62% of studies described the links between land

management practices and ESs as assumed, with only 30% acknowledging a robust relationship between management practices and ES provision. The assessment of ES delivery is primarily associated with programmes focused on carbon management, for which clear protocols measuring carbon storage and sequestration rates exist (Wunder et al., 2008). For example, China's National Forest Conservation Programme (NFCP) is estimated to have sequestered 21 Tg of carbon between 1998 and 2004 and reduced carbon emissions by 23 Tg over the same period (Liu et al., 2008). To compensate for the short-fall in domestic timber supply, however, China rapidly expanded imports, thereby 'exporting' its 'timber footprint' abroad (Liu et al., 2008).

In 40% of studies, there were calls to improve the assessment and monitoring of ES production and land-use linkages. The lack of data and lag-effects makes it difficult to determine ES changes in most studies. The inability to adequately define and quantify ESs has in general reduced programme effectiveness and efficiency (Engel et al., 2008; Kroeger 2013). Overall understanding of dynamics is typically rudimentary and assumes sometimes tenuous causal linkages. This highlights the need for pre-planning, baseline studies, and effective demonstration projects (Yin et al., 2013). Bennett et al. (2009: 1395) argued that:

“without knowledge about the relationships among ecosystem services, we are at risk of incurring unwanted trade-offs, squandering opportunities to take advantage of synergies, and possibly experiencing dramatic and unexpected changes in provision of ecosystem services.”

This is a sentiment that we would reinforce.

11.4.2.3 Financial Capital

Not surprisingly, two thirds of studies focused on the financial capital implications of PES programmes. The potential of PES to strengthen livelihood development strategies rests upon the financial capacity of programmes to support household and community needs through the provision of payments (Wunder, 2008; Pascual et al., 2010; Narloch et al., 2011). Payments to ES providers were generally annual, ex post, on a per hectare basis, and for 'delivered' ES proxies (i.e. land-use changes) rather than ES supply. A number of schemes were sensitive to the variations in effort that different land-use practices require and paid accordingly (Wunder et al., 2008). For example, Costa Rica's PSA provided (in 2009) US\$64–80 ha⁻¹ yr⁻¹ for forest protection but US\$82–98 ha⁻¹ yr⁻¹ for reforestation. Ecuador's Socio Bosque programme employed a descending payment scale, reducing incremental per hectare payments as the area enrolled increased. The majority of programmes (60%) implemented a single payment system rather than adopting a multiple streams approach, with payments made predominantly in the form of cash (62%) or technical assistance (21%). Just 30% of programmes opted for two payment modes (primarily cash and

technical assistance), while only 9% employed three payment modes (i.e. the addition of in-kind payments) such as Ecuador's PROFAFOR and China's SLCP schemes. Consequently, the payment design of most programmes had an important effect on their capacity to aid individuals, households and communities because they fail to utilise the broadest array of available options (Wunder et al., 2008).

In some cases payments were made to families (e.g. Cambodia's payments for wildlife friendly products) or to communities (e.g. Madagascar's Durrell Conservation Trust scheme), but generally 58% of programmes allocated payments to small- (2–30 ha) and medium-sized (30–60 ha) landholders, versus 22% to large- size (60+ ha) landholders. This may reflect informed programme targeting of the poorest sectors to maximise the benefit from payments (Narloch et al., 2011). For example, due to the nature of its mandate China's SLCP preferentially targeted poorer land- owners (Bennett, 2008; Liu et al., 2008). At the other extreme, Costa Rica's PSA generally benefitted wealthier landowners because larger farms acquired proportionally more money (Miranda et al., 2003). Ecuador's Socio Bosque programme was hampered by the scheme's failure to distribute and apportion individual and collective contracts in a manner that sufficiently accounted for the number of beneficiaries per contract and their poverty status (Krause and Loft, 2013). However, Narloch et al. (2013) have demonstrated that conservation auctions, in relation to distributional outlay, can minimise the 'traditional' trade-offs between fairness and effectiveness. In addition, our analysis also suggests that payment distribution may be influenced by a range of other factors, including collective land ownership (e.g. indigenous communal lands), shared needs, technical assistance, land security and property rights, proximity to protected areas, and the type of scheme in operation.

We found that household level impacts of programmes were mixed. Fifty per cent of studies suggested programmes positively increased household income, particularly in China. Many cases failed, however, to provide evaluations of income streams alongside these observations. Where such information was detailed, 29% of studies demonstrated payments contributed between 0% and 50% of household income, with just 8% showing payments contributing to more than 50% of household income. This demonstrates the highly variable nature of payment contributions to household incomes.

Household wealth, particularly for comparatively poorer households, was identified by 11% of studies as an additional barrier to participation and, by extension, to programme effectiveness. Although 50% of studies established that programmes enabled a diversification of household economic activities, only 12% described payments as sufficient to meet household needs or provide an alternative income stream. Generally, payment contributions provided insufficient income to enhance household economic productivity and diversity. Yet

promoting household and community capabilities relies in part on generating adequate working capital (Smith, 2005) and stimulating wider rural economic growth (WRI, 2005). Expanding the number of revenue streams from the natural resource-base lessens the risks for families and communities relying on a single market (Ingram et al., 2014). Providing access to functioning markets and increasing household wealth is essential for generating diverse income streams, securing sustainability, and driving innovation (Smith, 2005; WRI, 2005; Wunder, 2008; Narloch et al., 2011).

Addressing the income stream shortfall as a constraint to improving living standards was mentioned by 59% of studies. Only 20% of studies demonstrated that PES schemes reduced wealth inequity. This has dual effects on poor landholders and on providers who bear opportunity costs (Wunder, 2008; Pascual et al., 2010; Narloch et al., 2011); 43% of studies highlighted opportunity costs as significant barriers to participation. A further 38% cited the low level of programme payments as responsible for reducing uptake and contract renewal rates. The studies we examined suggested that wealth distribution and equity are also influenced by factors related to payment design and broader institutional and socio-economic circumstances, in particular: sub-optimal targeting; land entitlement and formal property rights; centralisation of payment distribution; the utilisation of non-monetary payments; elite capture and under-representation of the highly marginalised; diversion of funds from the local community to project management budgets; asymmetric distribution of funds between communities and concessionaires; gender differences; community status and reductions in the extent of inequalities. Pirard et al., (2010) argued that in order to mitigate these complex and diverse issues PES (more broadly) should emulate the RISEMP agro-ecosystem business model as an example of a sustainable and self-sufficient wealth generating scheme.

Many of the issues concerning payment amounts, contributions to household incomes, wealth distribution, and equity are directly related to programme contracts. Negotiating these issues requires permanency, flexibility and compliance in contractual agreements (Ferraro, 2008). Programme permanency and contract flexibility were relatively heterogeneous, and determined by a range of factors including the adoption of specific management practices (e.g. Costa Rica's PSA scheme), service seller decision-making (e.g. Bolivia's Los Negros programme), and contractual extension (e.g. Ecuador's PROFAFOR and Pimampiro programmes). Eleven per cent of studies described the need to extend the time frame of projects and guarantee permanency. A further 13% acknowledged the need to improve contractual arrangements for the benefit of agreement holders (e.g. in terms of payment amounts), by improving programme permanency and renewal options as well as allowing more flexibility with regards to sanctioned management actions. Dealing with risk and uncertainty regarding payment cessation, adverse selection and moral hazard, and the extent to

which negotiated agreements spread unfairness by embedding asymmetric power relations, is vital (Ferraro, 2008; Wunder et al., 2008; Milne and Adams, 2012).

Solving the targeting and monitoring conundrum is also important (Sommerville et al., 2011; Wünscher and Engel, 2012). The priority for targeting has to be determining an effective basis for directing payments to locations that will enhance scheme additionality at least-cost while balancing potentially competing conservation and development objectives (Wünscher and Engel, 2012). Similarly, monitoring requires attention on multiple fronts: deciding what is to be measured and meaningfully quantified across the range of capital assets; identifying who is monitoring, how frequently, and at what cost; and linking land use and ES provision with payment heterogeneity (Sommerville et al., 2011). Effective monitoring requires stability over time (Lin and Nakamura, 2012). Ensuring agreement obligations are fulfilled is thus critical but only 48% of programmes we examined had a high degree of conditionality and 36% had medium to low levels of compliance. In theory most programmes subscribe to annual monitoring by local stakeholders and/or government-related officials, with many instituting sanctions for non-compliance. In Brazil's Bolsa Floresta programme, for example, a system of penalty cards is used to determine non-compliance and designate the appropriate sanction (Pereira, 2010). However, across all programmes, applications of sanctions are relatively rare. Clearly, substantial improvements are needed to make PES programme monitoring effective (Schomers and Matzdorf, 2013).

At the global scale:

“...there is an urgent need to mobilise substantial additional funds and develop effective mechanisms for global biodiversity conservation” (Hein et al., 2013: 91).

Fauzi and Anna (2013) identified 'fiscal constraints' as limits to long-term programme financial viability. We found that 48% of studies declared financial viability as a major barrier to PES effectiveness. Recently, a number of nascent watershed investment programmes have become inactive due to inadequate financing (Bennett et al., 2013). Programme investment levels vary widely, with national programme implementation requiring high levels of financing. China's SLCP was designed as a 10 year programme with a total budget of over US\$40 billion (Bennett, 2008) and Costa Rica's PSA programme received US\$175–206 million (1997–2008) (Porras, 2010). Mexico's Fidecoagua, a local programme, received only US\$0.5 million annually (2003–2009). Our analysis indicates that external donor investments, loans and grants are essential sources of financial capital, guaranteeing the financial viability of many PES programmes. Seventy four per cent of programmes received some form of external donor support, with 59% supported by a single donor and 41% by two or more donors. External donors range from international conservation agencies (e.g. Conservation International – see Niessen et al., 2010) and development agencies (e.g. Swiss Development Cooperation) to

major international corporations (e.g. British Petroleum Amoco, PacifiCorp). However, the most frequent external donor support organisations were the World Bank (WB) and Global Environment Facility (GEF), which in many cases provided full projecting costs or initial start-up capital. Indeed, WB and GEF have supplied 60% (approximately US\$11 Billion) of global biodiversity aid over the past three decades (Hein et al., 2013).

Transaction costs impose significant constraints on programme effectiveness (59% of studies). To ensure that locally-derived sources of finance can support programmes in the long-term, which 34% of studies highlighted as necessary to secure, requires that the full range and magnitude of transaction costs are accounted for (Fauzi and Anna, 2013; Marshall, 2013; McCann, 2013). The prohibitive nature of programme transaction costs (McCann et al., 2005; McCann, 2013) may dictate that bilateral donor funding is essential to implement PES programmes. Legrand et al., (2013) argued that the securing of national and international funds by programmes ought to be viewed as an institutional triumph.

11.4.2.4 Institutional Capital

Our analysis supports the view that institutional factors of PES programmes are ‘undervalued’ (Pascual et al., 2010) as only 58% of studies assessed institutional capital and context. There are typically knowledge gaps relating to direct and indirect, and short and long-term institutional performance (Legrand et al., 2013). Clearly,

“PES systems are never established in an institutional vacuum” (Vatn, 2010:1247).

Programme success relies on establishing institutions and maintaining functional institutional relationships (Ostrom, 2005), and strengthening institutional frameworks and ties (Legrand et al., 2013). Forty four per cent of studies noted that programmes improved institutional capacity, cooperation between sectors and across groups, and the level of engagement with local organisations (e.g. Costa Rica's PSA scheme) (Legrand et al., 2013). Yin et al. (2013) suggest that grass-roots inclusion, through direct stakeholder inputs, improves long-term PES programme stability and reduces inefficiencies. Extolling the virtues of cooperation, 50% of studies expressed the view that developing improved institutional coordination was especially important for facilitating and enhancing capacity- building and technical assistance. Thus, for institutions, achieving lasting outcomes requires understanding and assessing the relational interactions between agents, institutions, and sectors, and their collective cultural effects (Campbell et al., 2010; Legrand et al., 2013).

Legitimacy, transparency and accountability in particular are central for successfully building institutional capacity and increasing effectiveness (Lockwood et al., 2010; Ingram et al., 2014). Eleven per cent of studies that we examined advocated the need to optimise

governance, accountability, and transparency to improve programme effectiveness. Only 44% of studies addressed matters of institutional accountability. However, of these, 73% registered improvements in accountability and transparency. This refers mainly to instances where legal and regulatory mechanisms enabled appropriate resource-use (73%), as well as examples in which the funding chain was described as more transparent (27%) and the level of accountability between providers and beneficiaries was improved (36%). Notably, however, 36% of studies still indicated reduced transparency and accountability regarding institutional arrangements and operations.

The importance of property rights (i.e. their distribution, allocation, and social embeddedness) for PES effectiveness is widely acknowledged (Lin and Nakamura, 2012; Schomers and Matzdorf, 2013). Thirty two per cent of studies identified the lack of defined property rights and land tenure arrangements as a clear barrier to programme effectiveness. Clearly-defined tenure arrangements, which acknowledge local customary rights, may legitimise secure long-term resource access through the use of entitlements. Conversely, inappropriate tenure reforms could negatively affect livelihoods, and so need to be sensitive to contextual factors such as local power asymmetries, gender exclusion, or poor legal documentation of customary rights (WRI, 2005; Fisher et al., 2008). For example, consolidation of current inequalities has reinforced disparities in resource allocation and power structures with respect to water access in Ecuador's Pimampiro programme (Rodríguez de Francisco et al., 2013). Regulation of ownership and property rights needs to be open for transparent and simple fiscal mechanisms to operate in relation to payment arrangements (Fauzi and Anna, 2013). The legislative landscape is particularly important for land reforms, as government recognition provides legitimacy and legal instruments to formally institute land rights. Part of this involves access to legal remedies for PES programme damages (Kaul et al., 2003). Despite the importance of the legal landscape, the significance of the legislative framework in which PES programmes operate was mentioned in only 36% of studies.

Regarding the overall involvement of the State in PES, institutional governance programmes are considered predominantly state-centric (Schomers and Matzdorf, 2013). However, in cases where the State has the primary responsibility for being the originator and operator of programmes, only 28% of schemes we examined were of this type (e.g. China's SLCP and NFCP). In some circumstances these centralised tendencies may constrain participation options. For example, in a 2003 survey concerning SLCP operations fewer than 50% of participants thought that villages had been adequately consulted by State authorities regarding programme design and implementation, and 53% of households felt centralised control constrained their participation choice (Bennett, 2008). As Stanton et al. (2010) highlight, there can be a stark difference in the role played by the State compared to private

and voluntary sectors. All sectors procured environmental services but government and related bodies represented ES buyers in 50% of our cases, while private and voluntary sectors each accounted for 18% of buyers. This observation accords with those made previously by Brouwer et al., (2011).

Programmes operated in chiefly agrarian locations meaning that most service sellers were rural and community farmers; where government or private sector service sellers played a minimal role (3% and 6% of studies, respectively). Institutionally, those responsible for connecting ES providers and ES beneficiaries, and facilitating fund disbursement are the intermediaries (Huber- Stearns et al., 2013), who were active in 96% of programmes we examined (exceeding the 82% identified by Martin-Ortega et al., 2013). This demonstrates the crucial roles played by intermediaries in delivering effective PES programmes (see also Lin and Nakamura, 2012). Intermediaries may be individuals, groups, or organisations, and operate at different scales and in different economic sectors (Huber-Stearns et al., 2013). Local and national governments represented intermediaries in 40% of cases, usually in the form of semi-autonomous bodies acting as government subsidiaries (e.g. Comisión Nacional Forestal (CONAFOR) in Mexico and Fondo Nacional de Financiamiento Forestal (FONA- FIFO) in Costa Rica). NGOs acted as intermediaries in a third of programmes (e.g. Fundación para el Desarrollo de la Cordillera Volcánica (FUNDECOR) in Costa Rica, Nitalpan in Nicaragua, and Corporación para el Desarrollo de los Recursos Naturales (CEDER- ENA) in Ecuador). It could be argued that government is the most influential and powerful intermediary actor. Perhaps this is not surprising given the ‘functional diversity’ intermediaries display, for example, as mediators, information providers, arbitrators, administrators, and core network facilitators (Thuy et al., 2010; Huber-Stearns et al., 2013).

Clearly the influence of external intermediaries is substantial, particularly so in grass-roots community-driven situations, which emphasises the importance of including intermediary partners to represent the local context and stakeholder views (Thuy et al., 2010). Engagement of local intermediaries does have a decentralising effect, which we found in relation to local community oversight and fund disbursement (Thuy et al., 2010; Huber-Stearns et al., 2013). The involvement of fewer intermediary actors can reduce the negative impacts associated with organisational competition (Thuy et al., 2010; Brouwer et al., 2011). Consistent with this view, we found about 75% of programmes involved only a single intermediary partner. When well-run, intermediaries can help reduce transaction costs, and supply expertise to draw-up contracts and monitor PES-related activities. They do so via a complex combination of relationship-building, establishing reputation, and adapting to location conditions (Thuy et al., 2010). Intermediary functions and actions are not always, however, positive or allied to local sensitivities. They can legitimise and de-legitimise

processes; decentralisation does not always favour beneficial outcomes if it fails to take account of elite capture and accountability issues (Thuy et al., 2010; Huber-Stearns et al., 2013).

Actors in PES may play multiple roles. One quarter of projects we examined were initiated by service buyers, higher than the 16% observed by Martin-Ortega et al. (2013). Of those, 50% were initiated by national governments. To some extent this State-centric influence was counter-balanced by significant NGO involvement (43%) in project initiation (lower in our study than the 58% of NGO project promoter's identified by Martin-Ortega et al., 2013). However, NGOs may have considerable influence on national and sub-national governance and development issues, land-use policy, and advocacy linked to incentive-based mechanisms. This influence has grown alongside rapid sector expansion e.g. in Kenya, the number of NGOs (of all types) increased 15-fold between 1990 and 2008 and in Tanzania, the number of NGOs multiplied by approximately 250 times between 1990 and 2000. Growth in the number of NGOs has been witnessed worldwide (Banks and Hulme, 2012). Lane and Morrison (2006: 232), for example, referring mainly to the environmental NGO sector in Australia, noted that:

“...the extent of NGO involvement, both formal and informal, in environmental policy and management is so widespread [...] NGOs (and other forms of civil society) now assume a dominant, even-pre-eminent role in the ascendant model of governance”.

Due to their ubiquity and the niche NGOs have created for themselves between the State and Civil society, especially in developing countries, they have been cast and recast as both hero and villain: standing-up for the rights of the poor, dispossessed and marginalised; combating anti-democratic values embodied in poorly governed States and corporations and promoting environmental sustainability; yet failing to make headway in many of these areas through gradually de-politicising and re-focusing on service-delivery, up-scaling and technocratic professionalization aligned to donor priorities and funding, media image and political connections. This series of transformations has led to increased concerns regarding their underlying accountability, credibility and capacity to actively promote and reflect civil society values (Bebbington, 2004, 2005; Holmes, 2011; Banks and Hulme, 2012; Rusca and Schwartz, 2012; AbouAssi, 2013). Nevertheless, the development of so-called “horizontal” governance, in which PES has sometimes been contextualised, has been viewed as promoting decentralisation and benefitting grass-roots concerns (Agrawal, 2001; Barbosa, 2003; WRI, 2005). There are those that remain unconvinced and see such developments as sponsoring and implementing the priorities of an elite group of wealthy global institutions (McAfee and Shapiro, 2010; Holmes, 2011). In a number of cases, however, projects originated as multi-party programmes. Almost one-third of schemes had more than one initiator e.g. the State and

a NGO or utility, indicating a relatively high degree of cross-collaboration, balance of competences and influences in relation to programme design and implementation (Martin-Ortega et al., 2013).

Collectively then, reappraising the institutional relationships between the actors facilitating programme operations is crucial (Pascual et al., 2010; Muradian et al., 2010). Our analysis indicates that government influence extends across the entire PES system, whereas the involvement of the private and voluntary sectors is more restricted. Expanding private sector participation, particularly as ES sellers and project initiators, presents new opportunities. In our review, 16% of studies recommended encouraging the private sector to pay for ESs as a means of promoting PES effectiveness. Engaging with the private sector provides a mechanism to increase direct investment, supply needed know-how and facilities, and reduce state centrism through local and national firm participation (WRI, 2005; Blackman and Woodward, 2010). Business and industry account for 7% of global investment in watershed payment schemes, so there is clearly substantial room for PES programme growth (Bennett et al., 2013). Private sector demand for PES programmes is growing as the reasons motivating participation multiply (Waage et al., 2007). National firms appear more likely to invest in multiple ecosystem services (Koellner et al., 2010). Furthering private sector integration represents a means for expanding the portfolio diversity of PES operations across scales as well as enhancing corporate social responsibility and widening sustainability (Rio+20, 2012).

11.5 Final Remarks

PES programmes have recently been subjected to mounting scrutiny (Pirard et al., 2010; Muradian and Rival, 2012). Our CAF analytical approach in a systematic review has provided considerable insight into the workings and effectiveness of PES schemes and their 'measured outcomes' (Table S11.14 Suppl. Material B). We identified a number of important issues related to essential components (or absence thereof) needed for functional, effective PES schemes: proper protocols for assessing ES production and distribution; adequate accounting for social, human and institutional capital assets in PES design and programme outcomes; and viable long-term funding arrangements. Like Martin-Ortega et al. (2013), our analysis indicated that the theoretical underpinnings of PES, whether inclined towards Wunder's (2005) archetype or Muradian et al.'s (2010) model, are quite different to the real-world implementation of these schemes labelled with the same terminology. In this regard, there are opportunities for aligning theory and practice. We suggest three research themes that require further development if PES is to represent an effective natural resource management option in the future: connecting land-use practices and ES provision (e.g. Yin et al., 2013); ensuring

programmes provide adequate socio-economic contributions to livelihood development by focusing on the poorest sectors (e.g. Ingram et al., 2014); and developing appropriate property rights regimes and building institutional capacity and institutions that are robust, inclusive, transparent and accountable (e.g. Legrand et al., 2013).

In attempting to address whether PES programmes are effective, the answer is not straightforward. A diversity of PES programmes exist, each of which produces a different set of measured outcomes when assessed through a CAF lens. Any argument calling for PES to be employed as generic solutions to natural resource management challenges requires careful scrutiny. What constitutes impacts (good or bad) worthy of action depends on societal, political, and stakeholder values (Rudd, 2004; Ostrom, 2005). It is clear that both locally-administered and nationally-governed PES programmes can be effective and have positive measured outcomes across multiple capital assets. However, important issues remain regarding how PES schemes negotiate effectiveness, efficiency, and equity trade-offs. These depend on how programmes are constructed and administered, as well as monitored and evaluated, within an appropriate context.

In designing schemes and mitigating trade-offs, we advocate a function-oriented and outcome-led approach. That is, identifying and prioritising a set of scheme outcomes (the desired end-products of a programme) and reverse engineering the structural and institutional arrangements of a programme (the underlying functional properties of a scheme) to achieve those aims. Using a CAF approach in this regard may help achieve an optimal balance between conservation and development outcomes. The precise composition of conservation and development objectives needs to account for locally-generated concerns, and not result from a one-size-fits-all approach. There is potential for substantial PES expansion internationally, but these opportunities should be viewed alongside other natural resource management and poverty alleviation policy instruments. They should be subject to testing in with/without policy analyses in a way that accounts for causal linkages between intervention options, ES flows, and proxy measures for ES in the field, and programme outcomes. PES programmes should not necessarily be regarded as superior to other intervention options or a panacea to be implemented on blind faith.

Notes

1. The debate regarding what constitutes PES is on-going. Primarily, it marks the divergence between environmental economics (e.g., Wunder (2005) and Engel et al. (2008)) and ecological economics (e.g. Muradian et al. (2010) and Vatn (2010)) and exemplifies their differing approaches to the application of market methods as a means of suitably allocating scarce resources within a natural resource governance framework (Farley and Costanza, 2010; Beder, 2011). The environmental economic perspective maintains that market interventionism is an appropriate tool for addressing the provision of ecosystem services, even accounting for the complexities of ESs in terms of their intrinsic properties and potential to be 'commoditised'. The five point criteria outlined by Wunder (2005) that define the 'ideal'

PES scheme represents the archetypal environmental economics position. From this perspective a PES scheme is constituted by: (i) a voluntary transaction; (ii) with a well-defined ecosystem service (ES); (iii) bought by an ES user; (iv) from an ES provider (v) under conditions where the transaction stipulates the quantity and quality of ES. However, the main ecological economic critique argues against this 'rationalisation' of nature asserting that, according to this definition, most schemes would fail the PES test and in fact would be classified PES-like. Ecological economics also challenges the view that poverty alleviation is a secondary goal of PES (Muradian et al. 2010). In contrast, ecological economists argue for a far more broader and inclusive definition, one in which the emphasis is placed on the idea that PES involves the transfer of resources between social actors and collective land-use decision making in natural resource management. Equity and poverty alleviation are highlighted as critical aspects that can positively affect the end-points of PES programmes (Farley and Costanza, 2010). Ultimately, however, the particular economic perspective that is employed determines the design and implementation of the PES scheme, and therefore, its potential success or failure.

2. In a nutshell, the Coase Theorem stipulates that under a system where property rights are known, designated and enforceable through contractual arrangements, then suppliers and receivers of an externality can voluntarily negotiate an agreement that maximises social welfare under conditions of zero transaction costs. Importantly, the size of the generated externality is independent of the original property rights assemblage. In other words, if transaction costs are absent then markets would produce efficient outcomes and government intervention would be unnecessary, save for its usual role in defining and enforcing property rights and enforcing voluntary private contracts (Farley and Costanza, 2010; Dixit, 2014).

Chapter 12: Case Study 2 – Seagrasses And Incentives: Uniting Climate Mitigation, Conservation And Poverty Alleviation

“Inadequate protection and management of marine resources has profound consequences: the oceans house both essential species and critical ecological processes, and provide a vital source of food and livelihoods for large numbers of people, including the world’s poorest. Despite this importance, marine species and habitats are increasingly endangered and fisheries are collapsing around the world.” (Nielsen and Gjertsen, 2010, pg. 1)

“Beyond the sea, beyond the sea, my heart is gone, far, far from me; and ever on its track will flee my thoughts, my dreams, beyond the sea” (an extract from “Beyond the Sea” by Thomas Love Peacock)

Seagrass ecosystems provide numerous ecosystem services that support coastal communities around the world. They sustain abundant marine life as well as commercial and artisanal fisheries, and help protect shorelines from coastal erosion. Additionally, seagrass meadows are a globally significant sink for carbon and represent a key ecosystem for combating climate change. However, seagrass habitats are suffering rapid global decline. Despite recognition of the importance of “Blue Carbon”, no functioning seagrass restoration or conservation projects supported by carbon finance currently operate, and the policies and frameworks to achieve this have not been developed. The relative inattention that seagrass ecosystems have received represents both a serious oversight and a major missed opportunity. In this paper we review the prospects of further inclusion of seagrass ecosystems in climate policy frameworks, with a particular focus on carbon storage and sequestration, as well as the potential for developing payment for ecosystem service schemes that are complementary to carbon management.

12.1 Introduction

Seagrasses represent a diverse and globally distributed group of aquatic flowering plants (angiosperms) with up to 76 species occurring in boreal, temperate, and tropical waters (Green and Short, 2003). Seagrass meadows are commonly dominated by a single species, although in tropical regions meadows comprising 12 distinct species have been recorded. They are often significant or dominant primary producers, supporting local food-webs and driving local nutrient cycles (Hoard et al., 1989; Gullstrom et al., 2008; Hemming and Duarte, 2000). Seagrass meadows have evolved important physiological, morphological and ecological

adaptations to cope with the range of coastal marine environments they inhabit, with the spatial distribution of seagrass meadows heavily influenced by environmental factors such as light, temperature, salinity, nutrient availability and wave action (Hemming and Duarte, 2000; Orth et al., 2006). However, the shallow coastal habitat colonized by most seagrass meadows means they are especially prone to significant human-related disturbance (Waycott et al., 2009).

Human actions provide a triumvirate of environmental, biological and climatological stressors that act across spatial and temporal scales delivering locally-specific impacts (Orth et al., 2006). Drivers of seagrass ecosystem decline include: eutrophication and solid waste disposal (nutrient pollution); aquaculture; thermal pollution; physical alteration or habitat damage (via dredging, coastal infrastructural developments, land reclamation and mechanical destruction); disease spread and invasive species introductions; climate change; over-fishing; overexploitation; and land-runoff from deforestation, mining and agriculture (Duarte 2002; Erftemeijer and Lewis, 2006; Orth et al., 2006; Waycott et al., 2009; Short et al., 2011; Zuidema et al., 2011; Hicks and McClanahan, 2012; Cullen-Unsworth and Unsworth, 2013; Cullen-Unsworth et al., 2014; Baker et al., 2015).

Over several decades the global integrity of seagrass ecosystems has been seriously undermined by business-as-usual approaches to coastal development (Duarte, 2002). Occurrences fuelled by increasing population densities in coastal regions, which are about three times higher than the global average and increasing (Small and Nicholls, 2003). In some cases rapid population growth and urban expansion has shifted farming practices towards increased agricultural output leading to the persistent eutrophication of coastal lagoons and reduced seagrass biomass (Rivera-Guzmań et al., 2014). Similarly, nutrient loading and sedimentation have markedly reduced the extent of several seagrass meadow sites in the Western Pacific (Short et al., 2014).

Globally, 24% of seagrass species are now classified as threatened or near threatened on the IUCN's Red List (Short et al., 2011). Estimates of the rate of seagrass decline have increased over the last 70 years, from 0.9% yr⁻¹ prior to 1940 rising to 7% yr⁻¹ since 1980 (Waycott et al., 2009; Fourqurean et al., 2012; Duarte et al., 2013a). The global decline of seagrass ecosystems threatens to exacerbate climate change (Duarte et al., 2010; Kennedy et al., 2010; Fourqurean et al., 2012; Lavery et al., 2013), undermine the supply of a range of other ecosystem services (Bujang et al., 2006; Orth et al., 2006; Waycott et al., 2009; Short et al., 2011; Cullen-Unsworth and Unsworth 2013) and consequently detrimentally affect subsistence livelihoods (Cullen and Unsworth 2010; Nordlund et al., 2010).

This reality reflects the complexity of seagrass ecosystems, particularly the connections seagrass meadows have with marine and terrestrial systems, and therefore the difficulties and challenges associated with their management, which are embedded within broader coastal and ocean management issues (Rudd and Lawton 2013). For example, in a recent global ocean research priorities exercise (Rudd 2014) several top-ranked priorities had implications for seagrass ecosystems, including: “greenhouse gas flux” (7th); “climate change mitigation and manipulation” (8th); “ecosystem structure to service linkages” (16th); “upland hydrology effects on oceans” (24th); “coastal hazard management” (35th); “ecosystem management alternatives” (40th) and “integrated upland coastal management” (43rd). Our view is that research is needed on multiple fronts to create enabling conditions and the evidence base needed to craft innovative new policy tools for conservation and mitigating the potential adverse effects of climate change.

Our purpose here is to summarize the prospects for using new approaches to aid seagrass conservation. This will help address key coastal and ocean research questions, and provide substantive direction on future seagrass research needs. We address these issues in the context of incorporating seagrass habitats into climate change mitigation strategies jointly focused on ecosystem service provision, carbon management and livelihood support. In particular, we analyse prospective financing options in relation to carbon management, alongside other investment opportunities for including seagrass meadows into incentive-based mechanisms (e.g. PES) through a co-benefit and bundled ecosystem service approach. In so doing we consider science, policy and governance perspectives acknowledging the important barriers and challenges existing across those domains.

We examine five key issues. In Section 12.2, we summarize ecosystem services provided by seagrass ecosystems and the salient information needed concerning these ES to develop incentive schemes. In Section 12.3, we ask how ecosystem service valuation information could be applied to design and implement new policy innovations. In Section 12.4, we examine the prospects for seagrass carbon finance based on current climate policy frameworks. In Section 12.5, we broaden the scope to financing instruments that could be developed based on the multiple ES that seagrass ecosystems provide. Lastly, in Section 12.6, we summarize the key design, implementation and governance issues that must be addressed to bring functioning seagrass PES schemes to fruition. In addition, we highlight the relevant ocean priority research questions that relate to each stage (Rudd 2014), setting our seagrass-oriented research in the broader context of ocean research prioritization.

12.2 Seagrass Ecosystems And Ecosystem Services

Seagrass ecosystems provide supporting, regulating, provisioning, and cultural ecosystem services (Barbier et al., 2011; Raheem et al., 2012; Cullen-Unsworth and Unsworth 2013). It is important to emphasize that the different lineages and species of seagrass will differ to some extent in the number and magnitude of ES they provide, for example, the level of organic carbon present in living biomass and seagrass meadow sediments is affected by species differences (Fourqurean et al., 2012) as are the fish community assemblages seagrass meadows support (Rotherham and West, 2002). However, our purpose in this section is to provide a brief outline of the main ES that are generally attributed to seagrasses in the literature, and to highlight the information that is necessarily required about each of these ES for them to be included in an incentive-based payment mechanism (Table 12.1). We do not provide a species by species breakdown of seagrass ecosystem service provision, partly because much of this information remains to be obtained (this is what Table 12.1 to some extent addresses) but it would also be beyond the scope of the present review.

Table 12.1 Seagrass ecosystem services and the corresponding information needed to contribute towards incentive scheme development

Ecosystem Service	What we need to know^a
Climate regulation (carbon storage and sequestration)	<p>(a) The spatial distribution, density and species assemblage of seagrass meadows. Two relatively accurate and reliable means of achieving this are:</p> <ul style="list-style-type: none"> • Acoustic side scan sonar which is useful up to 25m depths and has been used to map seagrass communities in the Mediterranean (e.g., Sanchez-Carnero et al., 2012; Montefalcone et al., 2013). • Remote sensing, for example, Landsat 5 TM and 7 Enhanced Thematic Mapper, which is more appropriate for shallow waters of between 2 to 5m and has been used in Australia (e.g., Dekker et al., 2005; Phinn et al., 2008), Zanzibar (e.g., Gullström et al., 2006) and the Coral Triangle (Torres-Pulliza et al., 2013) <p>(b) Assessment of carbon stocks, rate of accumulation (e.g., Fourqurean et al., 2012; Duarte et al., 2013a; Macreadie et al., 2014), in particular:</p> <ul style="list-style-type: none"> • Belowground biomass and soil: soil depth (thickness of deposit), dry bulk density and organic carbon content (Fourqurean et al., 2012) • Aboveground biomass: represents only ~0.3% of total organic carbon stock (Duarte and Chiscano 1999) • Accumulation rate: direct measurement, radiocarbon, ²¹⁰Pb, soil elevation (Duarte et al., 2013a)
Erosion and natural hazard regulation (coastal protection)	<p>(a) Local vegetative characteristics such as canopy height, shoot density and below-ground biomass (e.g., Bouma and Amos, 2012; Christiansen et al., 2013; Ondiviela et al., 2014)</p> <p>(b) Bulk density, organic content of sediment and porosity (e.g., de Boer 2007)</p>

Table 12.1 *Contd.*

Ecosystem Service	What we need to know
Biodiversity	(a) Species inventory, richness, diversity and community structure (e.g., Bell and Pollard, 1989; Barnes 2013) (b) Habitat usage of fish species and correlations with life-cycle stages (e.g., Heck et al., 2003; Jaxion-Harm et al., 2012; Seitz et al., 2014) (c) Presence of charismatic and Red List species (e.g., Williams and Heck Jr, 2001)
Fisheries	(a) Fish species caught, landed and sold (e.g., average catch sizes, market value etc.) (b) Frequency, location(s) and time spent fishing, for example, by using participatory GIS (e.g., Baldwin et al., 2013; Baldwin and Oxenford 2014) (c) Degree of overlap between commercial and artisanal fish species (i.e. commercial fishing impacts on artisanal fishing activities) (d) Types of fishing methods and gear employed and their capacity to damage seagrass beds (e.g., Tudela, 2004) (e) Invertebrate gleaning activities (e.g., species gleaned, frequency of gleaning etc. Unsworth and Cullen, 2010)
Nutrient cycling and water quality Regulation	(a) Seagrass biomass and production (e.g., de Boer, 2007) (b) Levels of leaf litter (e.g., Chiu et al., 2013) (c) Water turbidity (e.g., Petus et al., 2014) (d) Dissolved nutrient concentration (e.g., Cabaco et al., 2013)
Cultural services (tourism and recreation)	(a) Hotels (coastal distribution and ownership of land) (b) Tourist numbers, demographics and usage of inshore areas (reasons for use) (c) Local employment of staff in tourism (community-based tourism (e.g., Salazar 2012 (Tanzania); Kibicho 2008; Steinicke and Neuburger, 2012 (Kenya)) (d) Local food supply to hotels (e.g., Pillay and Rogerson, 2013) (e) Associated infrastructure developments and impacts on seagrass meadow stability (e.g., Daby 2003 in Mauritias; Zuidema et al., 2011 Turks and Cacos Islands)
Cultural services (social-ecological)	(a) Composition of household income and reliance on seagrass-derived ecosystem services (b) Gender differences in use and benefits derived from seagrass meadows e.g., gleaning vs. fishing (e.g., Cullen-Unsworth et al., 2014) (c) Cultural significance of seagrass meadows to 'traditional way of life' (e.g., Unsworth and Cullen, 2010)
Ecosystem Service Threats	(a) Agricultural land run-off : nutrient loading (e.g., Waycott et al., 2009) (b) Coastal developments and population and urban impacts: infrastructure, conversion of seagrass meadow beds to alternative uses, sewage discharge (e.g., Short et al., 2011; 2014)

^a In relation to the information outlined three points need to be emphasised: First, it is not necessary to obtain detailed information on all ES provided by seagrasses to develop a payment scheme. Second, their needs to be agreement between the operating scale of the payment scheme and the scale at which ES information is acquired. Third, the information we list is not meant to be exhaustive.

12.2.1 *Regulating Services: Climate Regulation*

Historically, seagrass meadows had been virtually ignored in global carbon budgets (Duarte et al., 2005). More recently their role in combating climate change through carbon storage and sequestration has become more clearly recognised, while simultaneously the spatial extent of seagrass meadows has continued to decline (Duarte et al., 2010; Kennedy et al., 2010; Fourqurean et al., 2012; Duarte et al., 2013a; Lavery et al., 2013). Although a small fraction (18 to 60 x 10⁶ ha) of the world's ocean area, seagrass meadows are "responsible for 3 to 20% of the global carbon sequestration in marine sediments" (Duarte et al., 2013a, p.32)

and store 10% of annual buried organic carbon (C_{org}) (Fourqurean et al., 2012; Pendleton et al., 2012). Consequently, seagrass ecosystems play potentially central roles in how oceanic ecosystems can mitigate climate change, a question ranked 8th in global importance by marine scientists (Rudd 2014).

Seagrass meadows are highly productive systems, especially in Indo-Pacific regions, and provide habitat for diverse communities (Short et al., 2011). However, worldwide, seagrass standing biomass is small (76-151 Tg C) relative to the biomass of the vegetation in other coastal ecosystems (Fourqurean et al., 2012). Nonetheless, the high productivity of seagrass meadows, with potential net community production (NCP) of $6.7 \text{ t C ha}^{-1} \text{ yr}^{-1}$ (several times higher than NCP rates associated with Amazonian forests and North American wetlands), contributes significantly to their carbon sink capacity (Duarte et al., 2010). Approximately 20% to 60% of this aboveground productivity derives from the autotrophic epiphytes that seagrass meadows support (Duarte et al., 2013a). Moreover, seagrass meadows trap allochthonous material, including large amounts of particulate carbon, which combined with their ability to bury carbon enables seagrass meadows to store large amounts of carbon (Duarte et al., 2013a).

Carbon stored belowground, as dead roots and rhizomes and as C_{org} , may be stable for millennia (Duarte et al., 2010; 2013a). However, the amount of C_{org} locked beneath seagrass beds varies considerably according to the interplay of different abiotic and biotic drivers, with the result that in some cases deposits of organic-rich sediments beneath seagrass meadows can be up to 11m thick (Duarte et al., 2013a). In addition, most seagrass production (approximately 80%) is not consumed by herbivores and may therefore be buried, where a combination of low nutrient content and anoxic sediment conditions contributes to low rates of remineralization aiding long-term storage (Duarte et al., 2013a). Burial rates are therefore somewhat difficult to estimate; however, the most robust data suggests mean local C_{org} burial rates of $1.2\text{-}1.38 \text{ t C ha}^{-1} \text{ yr}^{-1}$: equivalent to 30-50% of NCP (Kennedy et al., 2010; Duarte et al., 2013b). Nevertheless, others (Siikamaki et al., 2013) have suggested a much lower burial rate, equivalent to $0.54 \text{ t C ha}^{-1} \text{ yr}^{-1}$.

Globally, the organic carbon that accumulates in the sediments below seagrass meadows is much greater (4.2 to 8.4 Pg C) than the biomass (Fourqurean et al., 2012). However, the areal extent of seagrass meadows is poorly mapped, meaning these estimates remain highly uncertain (Duarte et al., 2013b; Lavery et al., 2013). Further uncertainties arise from the fact that some 50% of below-ground carbon derives from autochthonous production while almost 50% is contributed from phytoplankton and terrestrial sources (Kennedy et al., 2010; Duarte et al., 2013a). Indeed, significant quantities of carbon are also

exported away from seagrass meadows to adjacent areas, although the fate of this carbon is poorly understood (Duarte et al., 2010; 2013a).

Despite the uncertainties, alongside the lack of attention given to the potential implications of extensive conversion of standing carbon pools beneath vegetative coastal ecosystems more generally, it is clear that seagrass meadows constitute an important global carbon sink whose continued loss threatens to exacerbate climate change (Duarte et al., 2010; Pendleton et al., 2012). Indeed, global carbon emissions may be enhanced by an additional 3% to 19% from the destruction of vegetative coastal ecosystems (Pendleton et al., 2012). Based on current assessments, seagrass biomass loss may release between 11-23 Tg C yr⁻¹ into the ocean-atmosphere system, and a further 63-297 Tg C yr⁻¹ into the ocean-atmosphere CO₂ reservoir through the oxidization of the underlying sediment (Fourqurean et al., 2012). Additionally, seagrass loss reduces the overall carbon accumulation rate (equivalent to between 6 and 24 Tg C yr⁻¹). Collectively, this represents considerable CO₂ emission potential (131-522 Mg CO₂ ha⁻¹), a figure comparable to roughly 10% of that emitted annually from land-use change, with associated economic costs approaching US\$1.9 to 13.7 billion yr⁻¹ (Fourqurean et al., 2012; Pendleton et al., 2012).

12.2.2 Regulating Services: Erosion And Natural Hazard Regulation

Coastal vegetated wetlands such as seagrass meadows can provide effective natural protection from the destructive powers of storms and wave action (Barbier et al., 2008; Bouma and Amos 2012; Duarte et al., 2013b). They are therefore important ecosystems to study in order to understand the spatial extent, frequency, and risk of marine hazards affecting coastal waters and how their effects can be minimized (ranked 35th in Rudd, 2014). Direct coastal protection is achieved through energy dissipation resulting from wave breaking, friction and energy reflection (Ondiviela et al., 2014); processes significantly influenced by seagrass shoot density and canopy structure (Hansen and Reidenbach 2013). Even low biomass and heavily grazed seagrass meadows can significantly reduce wave action by decreasing the hydrodynamic energy associated with current flow (Christiansen et al., 2013). For example, in temperate regions current velocities have been reduced by up to 60% in summer (high biomass) compared to 40% in winter (low biomass) in relation to adjacent non-vegetated sites (Hansen and Reidenbach 2013). By reducing wave action and current velocities seagrass habitats also protect the seafloor against hydrodynamic “shear stresses” (de Boer 2007).

Seagrass canopies act as efficient filters, stripping particles from the water column and adding to sediment accumulation (Hendriks et al., 2008). Soil accretion (~1.5mm yr⁻¹) is important in helping coastal wetlands, and seagrass meadows in particular, adapt to sea level

rise (Kirwan and Megonigal, 2013; Lavery et al., 2013), thus contributing to Rudd's (2014) 26th ranked question on sea level rise and vulnerable coasts. Below-ground seagrass biomass is particularly important for sediment accretion as well as stabilization against storm erosion (Bos et al., 2007; Christiansen et al., 2013). By helping to immobilise sediment, seagrass meadows also reduce re-suspension and increase water transparency (Duarte, 2002; Ondiviela et al., 2014). In the Arabian Gulf, for example, sediment stabilization and shoreline protection represent important ecosystem service functions of seagrass meadows (Erftemeijer and Shuail, 2012). Overall, the effectiveness and efficiency of the coastal protection services provided by seagrass ecosystems varies across spatial and temporal scales due to differences in species type (i.e. vegetative characteristics), coastal distribution, flow-vegetation interactions and water dynamic properties (Ondiviela et al., 2014). In monetary terms, the erosion control services provided by seagrass beds (inclusive of algal beds) have been estimated at US\$25,000 ha⁻¹yr⁻¹ (Costanza et al., 2014).

12.2.3 Provisioning Services: Biodiversity And Fish Nurseries

The physical and biological structure of seagrass meadows is central to their significance as a marine biotope (Gullström et al., 2008; Pogoreutz et al., 2012; Saenger et al., 2013). The high primary productivity of seagrass, their epiphytes and associated benthic algae provide an important energy source to support local, transient and distant food webs (Heck et al., 2008). In addition, the structural complexity of seagrass meadows offers sites for attachment and a place to avoid predation (Farina et al., 2009). These attributes mean seagrass meadows function as foraging areas, refuges and nursery habitats for diverse communities of marine life, many of which are commercially important or endangered (Bujang et al., 2006; Orth et al., 2006; Unsworth and Cullen 2010; Erftemeijer and Shuail 2012; Jaxion-Harm et al., 2012; Browne et al., 2013; Cullen-Unsworth and Unsworth 2013). Organic matter produced in seagrass meadows is also exported to adjacent ecosystems and supports a large range of marine and terrestrial consumers (Heck et al., 2008). Connectivity between mangrove and seagrass ecosystems has also been shown to be important for supporting inshore fisheries, the abundance and assemblage of fish and crustacean communities and fish life-cycle stages (Bosire et al., 2012; Honda et al., 2013; Saenger et al., 2013). Seagrass ecosystems are thus important for ocean priority research questions on biodiversity contributions to ecosystem function (ranked 6th) and biological connectivity (ranked 28th) (Rudd 2014).

12.2.4 Supporting Services: Nutrient Cycling

Seagrass meadows are directly involved in nutrient cycling through their uptake of water column nutrients, storage in biomass, detritus and sediment, and indirectly through the effect of seagrass metabolism on water column and sediment nutrient re-cycling (Saenger et al., 2013). The nutrient cycling capacity of seagrass meadows has been estimated to contribute

about US\$26,000 ha⁻¹yr⁻¹, or US\$1.9 trillion in aggregate, to the global economy (Waycott et al., 2009; Costanza et al., 2014).

12.2.5 Cultural Services: Social Relations

Wetland ecosystems play vital cultural, economic and ecological roles, supporting livelihoods and reducing poverty (Kumar et al., 2011; Senaratna Sellamuttu et al., 2011; Verma and Negandhi 2011). Frequently, the fish and marine invertebrate populations supported by intact seagrass ecosystems maintain stocks of commercial and artisanal importance, and their exploitation makes significant economic and food security contributions to many coastal communities (Jackson et al., 2001). In some cases seagrass supported fisheries may provide a harvest value of up to US\$3500 ha⁻¹ yr⁻¹ (Waycott et al., 2009). In Tarut Bay, (Arabian Gulf), seagrass ecosystems support a US\$22 million yr⁻¹ fishery (Erftemeijer and Shuaib 2012). Prawns are also the basis for extensive fisheries, particularly along warm-temperate and tropical coastlines, and previous estimates of the potential total annual yield from seagrass ecosystems in Northern Queensland, Australia, equated to a landed value of US\$0.41-1.35 million yr⁻¹ (Watson et al., 1993). In the Caribbean and Indo-Pacific region valuable species such as queen conch (*Euatombus gigas*), spiny lobster (Palinuridae), and smudgespot spinefoot (*Siganus canaliculatus*) also support local fisheries (Cullen-Unsworth and Unsworth 2013; Baker et al., 2015).

Shellfish gleaning frequently supports artisanal fishers' subsistence and generates income for rural households (Unsworth and Cullen 2010). Invertebrate harvesters in Zanzibar, East Africa, can earn between US\$8.51 to US\$17.01 per catch from gleaning activities, emphasizing the social-ecological connections between coastal community livelihoods and seagrass ecosystem functioning (Nordlund et al., 2010). In some locations, the scale of inshore fisheries supported by seagrass ecosystems have been shown to be more significant (in economic terms) than those supported by mangroves or coral reefs. Recent evidence from Chwaka Bay (Zanzibar) indicated that fishers spend 70% of their time fishing seagrass meadows and preferred fishing there compared to mangrove and coral reef habitats (De la Torre-Castro et al., 2014). As a consequence, over 50% of the fish sold in the central market derived from seagrass meadows. In Wakatobi National Park (Indonesia), 60% of invertebrate collectors and 40% of fishers and gleaners preferred harvesting from seagrass meadows compared to 20% of collectors, fishers and gleaners who preferred to harvest exclusively from coral reefs (Unsworth et al., 2010).

12.3 The Value Of Ecosystem Services Provided By Seagrass Ecosystems

As we saw in Part 3, valuing ecosystem services has become an increasingly important tool for demonstrating the significance of biodiversity and ES to society and informing policy decisions (Gomez-Baggethun et al., 2010; Brondizo et al., 2010; Dendoncker et al., 2014; Liekens and Dendoncker, 2014). Seagrass ecosystems provide a potentially tractable environment within which to conduct multi-faceted valuation research and address an important ocean research question (ranked 53rd, Rudd, 2014) on ecosystem service valuation implications.

Economic valuations of seagrass ecosystems remain few in number, with most focusing on the market value of commercial fisheries as the primary ecosystem service of importance (Table 12.2).

Table 12.2 Valuation studies of seagrass meadows

Study	Location	Description	Value
Watson et al., (1993)	Queensland (Australia)	Multi-species prawn fishery	A\$1.2 million yr ⁻¹
McArthur & Boland (2006)	South Australia (Australia)	Secondary fisheries production	A\$114 million yr ⁻¹
Unsworth et al., (2010)	Wakatobi National Park (Indonesia)	Ecological and socio-economic assessments of the importance of seagrass meadow fisheries	US\$230 million (value extrapolated to the national level)
Kamimura et al., (2011)	Seto Inland Sea (Japan)	Wild juvenile black rockfish (<i>Sebastes cheni</i>) production	US\$78600 yr ⁻¹
Rudd & Weigand (2011)	Newfoundland, Canada	Choice experiment to estimate citizens' willingness to pay (WTP) for improvements in three ecosystem services associated with a reduction in wastewater pollution in the Humber Arm, with eelgrass (<i>Zostera marina</i>) used as an indicator for estuarine biological diversity	\$2.63 sq km ⁻¹ household ⁻¹ yr ⁻¹
Lavery et al., (2013)	Australia	Estimation of the value of stored organic carbon beneath Australia's seagrass ecosystems (17 habitats, 10 seagrass species). Valuation based on the C _{org} content of the top 25cm of sediment	A\$3.9-5.4 billion
Vassallo et al., (2013)	Isle of Bergeggi (Italy)	Natural capital assessment of <i>Posidonia oceanica</i> seagrass meadows using emergy analysis. Focused on the collective value of four ecosystem services: nursery function, sedimentation and hydrodynamics, primary production and oxygen release	€172 m ⁻² a ⁻¹
Tuya et al., (2014)	Gran Canaria Island (Spain)	Primary and secondary fisheries associated with <i>Cymodocea nodosa</i> seagrass meadows	€673269 yr ⁻¹ (whole island value)
Blandon and zu Ermgassen (2014)	South Australia (Australia)	Meta-analysis of juvenile fish abundance to assess the juvenile fish enhancement capacity of seagrass ecosystems. Thirteen commercial fish established to be recruitment enhanced	Species were enhanced by approx. A\$230000 ha ⁻¹ yr ⁻¹

In several respects seagrass ecosystems have been marginalized in favour of other coastal and estuarine ecosystems, meaning valuation studies conducted for other wetland biotopes (i.e. mangroves, coral reefs and saltmarshes) are the only suitable avenue to identify comparative estimates for commonly shared ecosystem services that may offer insights into the expected range of values for seagrass meadows (Table 12. 3).

Table 12.3 Valuation studies of coastal and wetland ecosystem services

Study ^a	Description	Ecosystem Service Values
Barbier et al., (2011)	Global synthesis of estuarine and coastal ecosystem services	<i>Coastal protection</i> : (US\$174 ha ⁻¹ yr ⁻¹ coral reefs in the Indian Ocean, US\$236 ha ⁻¹ yr ⁻¹ saltmarshes in the US and US\$966-1082 ha ⁻¹ yr ⁻¹ mangroves in Thailand). <i>Maintenance of fisheries</i> : (US\$5-45000 Km ⁻² yr ⁻¹ coral reefs (local consumption and exports) in the Philippines, US\$647-981 acre ⁻¹ saltmarshes (recreational fishing) in Florida and US\$708-987 ha ⁻¹ mangroves in Thailand). <i>Carbon sequestration</i> : (US\$30.50 ha ⁻¹ yr ⁻¹ for saltmarshes and mangroves based on global sequestration rates). <i>Tourism, Recreation and Research</i> : (US\$88,000 coral reefs in the Seychelles, £31.60 person ⁻¹ saltmarsh (otter habitat creation) in the UK)
UNEP (2011)	Total economic value of the ecosystem services delivered by mangroves in Gazi Bay, Kenya	<i>Total Economic Valuation</i> : (US\$1092 ha ⁻¹ yr ⁻¹) e.g., <i>Fishery</i> : (US\$44 ha ⁻¹ yr ⁻¹) <i>Coastal protection</i> : (US\$91.7 ha ⁻¹ yr ⁻¹) <i>Carbon sequestration</i> : (US\$126 ha ⁻¹ yr ⁻¹) <i>Biodiversity</i> : (US\$5 ha ⁻¹ yr ⁻¹) <i>Existence value</i> : (US\$594.4 ha ⁻¹ yr ⁻¹)
Verma and Negandhi (2011)	Livelihood dependency and economic evaluation of the Bhopal wetland, India	<i>Fisheries production</i> : (US\$33 month ⁻¹ fisherman ⁻¹) <i>Boating activities</i> (US\$2264 yr ⁻¹ boatman ⁻¹) <i>waterchestnut cultivation</i> (US\$222 yr ⁻¹ family ⁻¹) <i>cloth washing activities</i> (US\$66 month ⁻¹ household ⁻¹) <i>secondary activities e.g., sugar cane juice sellers</i> (US\$6000 yr ⁻¹)
Brander et al., (2012)	Meta-analysis of the value of ES supplied by mangroves mainly in Southeast Asia. Valuations based predominantly on fisheries, fuel wood, coastal protection and water quality	Overall mean and (median) value: US\$4185(239) ha ⁻¹ yr ⁻¹
Salem and Mercer (2012)	Global estimates of mangrove ecosystem services	<i>Fisheries</i> : (US\$23,613 ha ⁻¹ yr ⁻¹). <i>Forestry</i> : (US\$38,115 ha ⁻¹ yr ⁻¹). <i>Recreation and Tourism</i> : (US\$37,927 ha ⁻¹ yr ⁻¹). <i>Non-Use</i> : (US\$17,373 ha ⁻¹ yr ⁻¹). <i>Water purification</i> : (US\$4,784 ha ⁻¹ yr ⁻¹)
Brander et al., (2013)	Global meta-analysis of ES delivered by wetland systems in agricultural landscapes, with a focus on three regulating services: flood control, water supply and nutrient cycling	Mean and (median) values presented <i>Flood control</i> : US\$6923(427) ha ⁻¹ yr ⁻¹ <i>Water supply</i> : US\$3398(57) ha ⁻¹ yr ⁻¹ <i>Nutrient cycling</i> : US\$5788(243) ha ⁻¹ yr ⁻¹
Camacho-Valdez et al., (2013)	Socio-economic benefit of saltmarshes in Northwest Mexico	US\$1 billion yr ⁻¹
James et al., (2013)	The social (non-monetary) values attached to mangroves across three villages in the Niger Delta region of Nigeria. Social values assessed were: therapeutic, amenity, heritage, spiritual and existence	Mean values for the village-level importance placed on these aspects of the social value of mangroves <i>Therapeutic</i> : (14%-71%) <i>Amenity</i> : (65%-73%) <i>Heritage</i> : (70%-92%) <i>Spiritual</i> : (44%-76%) <i>Existence</i> : (89%-91%)

Table 12.3 *Contd.*

Study	Description	Ecosystem Service Values
Kakuru et al., (2013)	Wetland ecosystem services in Uganda	<i>Flood control:</i> (US\$1.7 billion yr ⁻¹). <i>Water regulation:</i> (US\$7 million yr ⁻¹)

^aIt is important to note that the examples we cite in Table 12.3 are not meant to be exhaustive but rather illustrative of the different types of services and values attributed to a range of coastal wetland ecosystems, and are therefore to be seen as a guide for the range of possible valuations that may be attributed to seagrass ecosystems

Overall, the lack of in-depth local studies spanning different continents and regions valuing the breadth of ecosystem services provided by seagrass ecosystems needs to be remedied, with particular focus on qualitative value attributions associated with the social-ecological dynamics of seagrass systems (Cullen-Unsworth and Unsworth 2013). This view supports the wider sentiment articulated by Raheem et al., (2012:1169) that:

“...there is a dearth of spatially explicit non-market values for services provided by coastal and other ecosystems.”

And further supported by the Abu Dhabi Global Environmental Data Initiative (AGEDI 2014:10) who argue that the:

“...option of combining Blue Carbon with other ecosystem services valuation should be kept open to provide multiple potential values that can support conservation activities”.

Strengthening the evidence base regarding the global economic value of oceans (ranked 48th, Rudd 2014) requires site-specific seagrass ecosystem valuation efforts that can be used to derive transfer values from meta-analyses (e.g. Brander et al., 2012; Johnston et al., 2005).

12.4 Policy frameworks For Blue Carbon management

Recent thinking about Blue Carbon acknowledges the special importance of the carbon storage and sequestration capability of coastal and marine wetlands and organisms in global climate change scenarios and policies (Sifleet et al., 2011; Vaidyanathan 2011). Blue Carbon sinks capture and store amounts of carbon equivalent to up to half of global transport emissions (~ 400 Tg C yr⁻¹) yet their inclusion in current mitigation and adaptation programs has been very limited (UNEP 2009; Tommaso et al., 2014). Developments could occur in the regulated (compliance) or the unregulated (voluntary) carbon sectors. We take each in turn.

12.4.1 *The Regulated Sector*

12.4.1.1 Policies and Processes

Collectively, the United Nations Framework Convention on Climate Change (UNFCCC 1992, Article 4 (d)), Manado Ocean Declaration (2009), Cancún Agreement (2010) and Rio Ocean Declaration (2012) provide opportunities for development of Blue Carbon initiatives. In practice, however, current policy processes inadequately account for the restoration and protection of Blue Carbon systems (Grimsditch, 2011; Murray et al., 2011). This is due, in part, to the initial bias towards terrestrial climate change mitigation and adaptation activities within the UNFCCC, alongside the acknowledgment that practical expansion to coastal and marine systems (from principled intentions) would require further international agreement (Murray et al., 2012). However, as a recent report indicates (UNEP and CIFOR 2014: x):

“...climate change mitigation frameworks developed for terrestrial ecosystems can be extended to include coastal wetlands”.

There are clear points of entry for Blue Carbon funded activities under the parallel pathways of the UNFCCC, specifically: the Land Use and Land-Use Change and Forestry (LULUCF) and the clean development mechanism (CDM) of the Kyoto Protocol; and the Reduced Emissions from Deforestation and forest Degradation + (REDD+) and Nationally Appropriate Mitigation Actions (NAMAs) of the Durban Platform. In many cases these entry points require altering or reinterpreting definitions (Gordon et al., 2011; Grimsditch, 2011; Murray et al., 2011; 2012). Nevertheless, some argue that by the Paris Conference of the Parties (COP) 21 meeting in 2015 negotiations are likely to have reached a consensus for including an approach for Blue Carbon accounting under the UNFCCC (UNEP and CIFOR 2014).

12.4.1.2 Kyoto Protocol Opportunities

Limited possibilities exist within the Kyoto Protocol (Murray et al., 2012). However, some progress has been made through the recently updated Intergovernmental Panel for Climate Change (IPCC) guidelines. The so-called ‘Wetlands Supplement’ includes guidance for national governments to report carbon emissions and removals for specific management activities in coastal wetlands (e.g. mangroves, tidal marshes and seagrass meadows) (IPCC 2014). The activities that national governments will be able include in their national inventories for greenhouse gases covers forest management in mangroves, certain aspects of aquaculture, drainage and restoration or creation of coastal wetlands. However, this supplementary regulation is “encouraged but not mandatory in context of any other activities under Article 3, paragraphs 3 and 4, of the Kyoto Protocol” (UNFCCC 2014).

Moreover, extension of current LULUCF definitions to cover wetland ecosystems is lacking (Murray et al., 2012). However, with the publication of the IPCC Wetland Supplement the case for not including a broader set of definitions that specifically mention wetlands is harder to justify. Furthermore, activities under LULUCF could include avoided wetland degradation via alternative use or prohibiting disturbance (Herr et al., 2012). With regards to baseline credit mechanisms such as the CDM, in 2011 a mangrove project was approved as an afforestation and reforestation activity. However, the methodology applied is specifically for mangroves and not (so far at least) transferable to tidal marshes or seagrass meadows (Lovelock and McAllister, 2013). Moreover, the much more substantial avoided emissions resulting from protecting Blue Carbon pools remain outside this mechanism (Murray et al., 2011; 2012).

12.4.1.3 Durban Platform Opportunities

The Durban Platform provides more scope for Blue Carbon activities. Mangroves are now covered by REDD+ (Grimsditch, 2011). However, seagrass inclusion remains some way off: this would require a broader definition of “forests” as well as an extension of emission and reduction activities across all land-uses (i.e. Agriculture, Forestry and Other Land Uses, AFOLU) (Murray et al., 2011; 2012; Siikamaki et al., 2013). Nevertheless, AFOLU projects do include a variety of carbon accounting protocols relating to biomass, C_{org} and greenhouse gas emissions (UNEP and CIFOR, 2014). There have been calls to decouple carbon sequestration and emissions arising from habitat degradation (Grimsditch, 2011). This is particularly important for seagrass meadows where the ‘real’ carbon of interest is buried in the sediment. Under REDD+, deciding what aspects of the Blue Carbon pool (i.e. sediment/soil-carbon or above-ground biomass) count would be especially important (Murray et al., 2011). Extension of REDD+ to seagrass meadows could easily see them contributing to reduced emissions via the degradation pathway, through a focus on management strategies linked to tackling the negative impacts of nutrient loading for example (Seifert-Granzin, 2010). Developments to include tidal wetland restoration and conservation under REDD+ are currently on-going (UNEP and CIFOR, 2014).

NAMAs offer the most direct route for funding Blue Carbon enterprises because countries have autonomy over the activities that form part of their national strategies, and could reasonably protect and restore wetland and coastal ecosystems (Grimsditch 2011; Herr et al., 2012; Murray et al., 2012). Furthermore, the green climate fund provides finances for programs in accordance with NAMAs that could be directed towards Blue Carbon activities (Herr et al., 2012). However, the challenge remains that inclusion of these activities under a national framework would still require measurement, reporting and verification approval (Murray et al., 2012).

12.4.2 *The Voluntary Sector*

12.4.2.1 The Global Voluntary Carbon Market

The voluntary carbon market (VCM) accounts for 0.1% and 0.02% of the value and volume of the regulated global carbon market respectively (Benessaiah, 2012). Yet rapid sector expansion has led to increasing interest from governments, particularly in relation to carbon standards and registries (Peters-Stanley and Yin, 2013). The principal attraction of the VCM is its deregulated nature, which helps to reduce transaction costs and stimulate innovation. However, the trade-off to this regulatory flexibility is market uncertainty and depression of the carbon price, which can have serious implications for expected project returns (Benessaiah, 2012; Thompson et al., 2014). Project size is also a determinant of offset price, with smaller projects garnering higher carbon prices for carbon dioxide equivalent (CO₂e). The average carbon price for micro projects (i.e., those generating less than 5 Kt CO₂e yr⁻¹) was recently US\$10/tCO₂e, whereas the mean carbon price for mega projects (i.e. those generating more than 1 Mt CO₂e yr⁻¹) was US\$5.8/tCO₂e (Peters-Stanley and Yin, 2013).

Worldwide carbon standards have expanded from concentrating purely on carbon accounting to emphasising co-benefits (Peters-Stanley and Hamilton, 2012). This has been driven, particularly in the private sector, by an increasing interest in measuring and verifying non-carbon project outcomes (Peters-Stanley and Yin 2013). Programs are progressively focusing on climate change adaptation, public health, gender issues and biodiversity as additional attributes to non-carbon benefits (Peters-Stanley and Yin, 2013) (Table 12.4). For example, the verified carbon standard (VCS), which accounts for 55% of market share, considers climate, community and biodiversity (16%) and Social Carbon (2%) co-benefits (Peters-Stanley and Yin, 2013). This is important for ecosystems such as seagrass meadows that provide multiple benefits in addition to carbon storage as those benefits might be captured via broader standard attributes.

Another important development for coastal wetland systems such as seagrass meadows is that the VCM has highlighted the special connections between carbon and water. Both VCS and the American Carbon Registry (ACR) have coastal wetland accredited carbon accounting methodologies (Peters-Stanley and Hamilton 2012; Thomas 2014). For example, in the Mississippi Delta the ACR has developed a wetland restoration protocol (UNEP and CIFOR 2014). Furthermore, VCS has also developed a soil carbon sampling methodology that could be transferred to wetland and peatland ecosystems (Peters-Stanley and Yin, 2013).

Table 12.4 Carbon standards appropriate for joint environmental and development projects

Carbon Standard and Credits	Description
Gold Standard (<i>acquired Carbon Fix Standard</i>)	Carbon accounting + embedded co-benefits
Plan Vivo	Carbon accounting + embedded co-benefits
VCS	Carbon accounting + tagged co-benefits
VCS and CCB	This joint process is premised on the notion that forestry and land-use projects will be better able to meet emission reduction targets and achieve co-benefits if validation/verification costs are lowered
Social Carbon	Co-benefits (needs to be accompanied by a carbon accounting standard)
Global Conservation Standard	Developed for the purposes of ensuring conservation can deliver payments to local landholders, the accounting system is based on the 'stock' amount of identifiable and measurable ecosystem service benefits – credited through the use of Conservation Credit Units (CCUs). The first protocol established CCUs based on carbon stocks in vegetation.
Women's Carbon Standard	Certifying the role, engagement and leadership of women in carbon projects. Jointly administered by Women Organising for Change in Agriculture and Natural Resource Management – WOCAN
Vulnerability Reduction Credits	Acknowledges and qualifies reduction in community vulnerability arising from adaptation efforts. Administered by the Higher Ground Foundation
The Poverty Alleviation Criteria Tool	Measures the poverty alleviation outcomes resulting from forestry and other land-use projects implemented under the Panda Standard. Developed jointly by ACR (American Carbon Registry) and the China Beijing Environmental Exchange

Indeed, VCS methodologies cover the full array of Blue Carbon activities, from restoration and re-vegetation to conservation and management, and in late 2013, the 'Greenhouse Gas Accounting Methods for Tidal Wetlands and Seagrass Restoration' methodology was submitted to VCS and is currently awaiting approval (UNEP and CIFOR, 2014).

Although the increasing alignment between livelihood development and carbon management is welcomed, several challenges exist. Specifically, a lack of appropriate markets, negotiating trade-offs between maximizing economic efficiency and ensuring equity in benefit flows, and adequately socially embedding payment schemes. These challenges relate to broader issues of the transaction costs of ocean management (ranked 57th, Rudd 2014). Developing inclusive sustainable livelihood VCM projects depends on the provision of secure property rights and tenure arrangements regarding the ownership and use of resources. However, providing secure property rights alongside certification can be prohibitively expensive (e.g. CCB certification is estimated at US\$4000 – US\$8000) even though adequately accounting for costs and securing financial streams is essential (Benessaiah, 2012). Negotiating investment risk and return uncertainty are significant challenges in community-based carbon

projects where non-compliance and complex program arrangements are pressing issues. Likewise, the provision of “enabling institutions” for effective administrative, operational and implementation performance remains crucial. Nevertheless, the advantages of the voluntary carbon market outweigh the downsides and present a more immediately attractive option even if in some quarters the regulated carbon market is the preferred long-term option (Benessaiah, 2012; Ullman et al., 2012).

12.4.3 Multilateral Environmental Agreements

The sustainability of estuarine, coastal and marine habitats, with regards to their use, conservation, restoration and in climate change mitigation and adaptation have been alluded to under several regional and international multilateral agreements for example: the Convention on Biological Diversity (CBD); Ramsar Convention on Wetlands (Ramsar); UNEP Global Programme of Action for the Protection of the Marine Environment from Landbased Activities (GPA-Marine); Convention for the Protection of the Marine Environment and Coastal Areas of the South-East Pacific (Lima Convention) and the South Pacific Regional Environment Programme (SPREP). Although predominantly management and advocacy-related, some of these programs offer financial support for Blue Carbon activities (Laffoley, 2013).

12.4.4 National Level Policies

Research evaluating the ways in which vegetative coastal ecosystem services and carbon in particular can be included in national level statutes and policies is lacking, partly as a result of the highly individual nature of national legislation. However, Pendleton et al., (2013) have identified how such ‘coastal carbon’ could be incorporated under a subset of existing U.S. federal statutes and policies including the National Environmental Policy Act, the Comprehensive Environment Response, Compensation and Liability Act, the Oil Pollution Act, the Clean Water Act and the Coastal Zone Management Act amongst several others. The analysis indicates that although coastal carbon services are not currently accounted for under existing federal-level legislation, to do so would be relatively straightforward and consistent with the implementation of these regulations (Pendleton et al., 2013). Nevertheless, despite this relative ease, incorporating coastal carbon into existing federal legislation would require further improvements in the availability of expertise, guidance and procedures for assessing the value of coastal carbon, quantifying the impacts of projects on carbon storage and sequestration and mapping the spatial dynamics of coastal ecosystems. The lack of precedent (i.e. the formal assessment and analysis of the benefit-costs of coastal carbon economics values in these regulations) was also recognised as an important limitation that would need to be overcome for wider ‘coastal carbon functions’ to be frequently included in regulatory assessments (Pendleton et al., 2013). Importantly, these considerations are equally applicable

to State-level legislation as they are to other national legislative policies and statutes in other countries.

12.4.5 Blue Carbon Demonstration Sites And The Future

Recent research, policy and financing advancements in Blue Carbon relevant to seagrass meadows include global programs. The Blue Carbon Initiative (www.thebluecarboninitiative.org) focused on integrating Blue Carbon activities into the UNFCCC and other carbon financing mechanisms (Herr et al., 2012; Thomas, 2014). Charities such as The Ocean Foundation and partners (www.seagrassgrow.org) have developed a Blue Carbon calculator that determines CO₂ emission reduction offsets in terms of the protection and restoration of seagrass meadows (a method pending formal approval by the VCS). Collectively, developments such as the Blue Carbon portal (www.bluecarbonportal.org) and work by Bredbenner (2013) and Thomas (2014) have demonstrated the current global extent of Blue Carbon activities. In particular, significant work remains to establish a functioning global network of fully implemented Blue Carbon programmes involving the active transfer of carbon credits (Locatelli et al., 2014). In this regard, securing private financing of Blue Carbon activities will become increasingly important (Thomas, 2014). Presently, Blue Carbon programs are predominantly research-oriented, in the early stages of development and mangrove-focused, with few directed efforts towards seagrass ecosystems (Table 12.5) (Bredbenner 2013).

Table 12.5 Seagrass-related Blue Carbon initiatives (source: adapted from Bredbenner, 2013)

Blue Carbon Project	Description
Long-term ecological research in the Patos Lagoon Estuary (Brazil) – Institute of Oceanography and Federal University of Rio Grande	Spatial and temporal description of seagrass and macroalgae vegetation changes. Mapping, biomass and sedimentation sampling for carbon stock evaluation
National seagrass ecosystem mapping (Brazil) - Universidade Estadual de Rio de Janeiro, Universidade Federal do Rio Grande, Universidade Federal de Santa Catarina e Universidade Federal Rural de Pernambuco	Spatial mapping of Brazil's seagrass ecosystems, distribution and extent, and the determination of the associated carbon stock
Seagrass and Mangrove pilot assessments (Indonesia) - Agency for Research and Development of Marine and Fisheries, Ministry of Marine Affairs, Fisheries-Indonesia	Three pilot areas: Banten, East Kalimantan and North Sulawesi – field surveys, mapping and biophysical sampling of seagrass and mangrove systems to assess carbon storage and sequestration, alongside the socio-economic value of these systems for improving policy
Mangrove, saltmarsh and seagrass Blue Carbon potential (China) - Tsinghua University, Xiamen University, State Oceanic Administration	Assessment of the Blue Carbon potential of these ecosystems (i.e. carbon storage and sequestration) to provide evidence to support habitat restoration linked to carbon credit scheme

12.5 Seagrass Habitats: Prospects For PES

Here we explore opportunities for developing seagrass PES programmes. The options we describe should be seen as working in tandem with carbon-credit schemes not as mutually exclusive alternatives. We will not provide a general overview of PES and PES programmes here as these have already been described in Chapter 11; however, it is useful to recap the institutional context of PES which is generally framed as a decentralized set of instruments favouring bottom-up solutions to land management issues (Landen-Mills and Porras 2002; Bond and Mayers, 2010). Despite the diversity of contexts in which PES schemes operate, they tend to adopt common modes of activity such as restricting agricultural development, proposing alternative cropping arrangements, reducing deforestation and expanding forests (e.g., reforestation and afforestation), or protecting watershed and hydrological services (e.g. Aquith et al., 2008; Bennett 2008; Muñoz-Pina et al., 2008; Wunder and Alban 2008; Porras 2010; World Bank 2010; Kolinjivadi and Sunderland, 2013). Consequently, PES involves multiple partners across sectors as well as spanning spatial and temporal scales (Schomers and Matzdorf 2013). To function properly, schemes need to be acceptable to stakeholders, take the form of contractual obligations to which all participating parties agree, have specified objectives, be operationally transparent, and provide payments (in monetary or in-kind terms) to ES providers that account for (ideally) the full range of their opportunity costs (Wunder et al., 2008; Bosselmann and Lund, 2013; Hejnowicz et al., 2014).

12.5.1 PES Case Studies And Some Considerations

Examples relevant to guiding the development of seagrass payment schemes need to involve community approaches to natural resource management, as well as the provision of multiple ES with a focus on carbon management (e.g. Fisher et al., 2010 Table 12.6). Schemes seeking to deliver multiple ES via incentive mechanisms must also tackle the issue of stacking and bundling (Box 12.1). That is to say, determining what ES are to be provided, whether they will be paid for individually (i.e., stacked) or collectively (i.e., bundled), and what form payments will take (Bianco 2009; Ingram, 2012). Additionally, PES programs need to ensure that as part of their design and implementation they maximize biodiversity and social co-benefits by adopting a decoupled approach to benefit maximization (recognizing individual ES properties and spatial attributes), ensuring management decisions account for internal and external costs, and increasing social co-benefit provision by concentrating on economic and cultural context (Greiner and Stanley 2013; Phelps et al., 2013; Potts et al., 2013).

Table 12.6 Examples of PES schemes that jointly focus on carbon management and the provision of additional ecosystem services

PES Study	PES Description
Carbon Livelihoods Project: Mozambique <i>Source:</i> Hedge and Bull (2011) and Groom and Palmer (2012)	<p>The project operated across several villages and aimed to establish a viable alternative livelihood, agro-forestry and carbon credit scheme. Agroforestry was designed to generate carbon offsets alongside new ‘on-farm’ labour activities, whilst the alternative livelihood element promoted ‘off-farm’ micro-enterprises. Initially funded by the European Union, the programme became self-financing following sales of verified emissions reductions (VERs) in the voluntary carbon market. VER sales were used to establish an annual PES fund that dispensed payments to farmers over a seven year period. Remaining revenue was channelled into a community trust fund for development projects such as healthcare support. Adoption of agro-forestry practices meant households generated new ‘on-farm’ income by selling crops or harvesting non-timber forest products following cessation of carbon payments. Micro-enterprises such as bee-keeping, plant nurseries, carpentry and even a community sawmill provided viable and secure alternative revenue sources for farmers. In addition, some local people were hired by the project.</p> <p><i>Criticisms:</i> Carbon offset payments were less important (proportionally) than income from the project’s alternative revenue sources. Micro-enterprises potentially undermined the sustainability of ‘on-farm’ activities through changes in labour allocation. Gender discrimination contributed to uneven income distribution between male- and female-headed households, and project costs were significant; with two thirds of carbon offset sales revenue directed towards overheads.</p>
Western Kenya Integrated Ecosystem Management Programme (WKIEMP): Kenya <i>Source:</i> World Bank (2010)	<p>WKIEMP was initiated to provide a viable community-livelihood development model. Implemented across 15 micro-watersheds WKIEMP focused on land productivity and sustainable-use by supporting on-farm and off-farm conservation strategies and building institutional capacity; alongside promoting management interventions geared towards biodiversity and carbon sequestration and storage. Overall the project was moderately successful. Households did not receive payment; but derived income through improved land productivity, livelihood diversification and technical capacity. Estimated net present value to participating households is considered to be US\$1193 to US\$2844. Moreover, 60% of beneficiary households reported an increase in food production and consumption directly addressing poverty alleviation. Furthermore, the project established institutional networks to enhance the sustainability of community activities following project cessation such as basin technical committees that promoted cross-collaboration.</p> <p><i>Criticisms:</i> Two problems undermined WKIEMP’s notion of sustainability. First, project permanency: the project ran for only five years from 2005 to 2010. Second, the programme encountered fiscal constraints that hampered its implementation and operation leading to disjointed upstream and downstream management interventions. Overall, the failure to secure adequate co-financing of funds significantly impaired project performance.</p>
Socio Bosque: Ecuador <i>Source:</i> de Koning et al., (2011) and Krasue and Loft (2013)	<p>Socio Bosque is a nationwide government initiative designed to realise biodiversity conservation, climate mitigation and poverty alleviation benefits. Participants receive direct monetary transfers on a per hectare basis for protecting native forests and ecosystems through voluntary but monitored twenty year conservation agreements. Payments are made on a descending scale, with amounts reduced incrementally as the land enrolled increases providing a built-in equity mechanism. Participants are individual landowners or local indigenous communities, and so land is privately or communally owned. Only land that has a high probability of deforestation, sufficient carbon storage, water regulation and biodiversity capacity and is found in relatively socially-deprived areas is eligible for enrolment. Overall 260000 Ha yr-1 of forest have been protected. Remuneration is conditional, requiring compliance with a social investment plan (directing how incentives might best be used to improve social conditions) and conservation obligations. Social benefits are realised through monetary investments in health, education, household consumption, debt repayments, infrastructure and institutional capacity.</p> <p><i>Criticisms:</i> Payments allocated to participants are not equal: less than a fifth of households in community agreements receive more than US\$500 yr-1 compared to 92% of private landholders. The scheme has underperformed with regards to distributing individual and collective contracts in a way that accounts for the number of beneficiaries per contract and their poverty status.</p>

Box 12.1 Stacking and bundling ecosystem services

Stacking refers to the receipt of multiple payments for different ES provided from a single plot or parcel (Bianco, 2009; Cooley and Olander, 2012). Cooley and Olander (2012) recognise three forms of stacking, namely: *horizontal* (whereby individual management practices performed on spatially distinct areas each receive a payment); *vertical* (where a single management practice employed on spatially overlapping areas receives multiple payments) and *temporal* (essentially a *vertical* form of stacking where payments are disbursed over time according to the production of different ES).

Advantages of stacking: (i) delivers management that provides multiple services from programs concerned with specific services; (ii) potentially increases programme uptake rates and therefore ES provision, (iii) encourages large-scale projects that could not operate through single payments e.g., wetland restoration, (iv) may increase buyer diversification, and (v) incrementally stacking payments in an optimum way for a particular project can help raise necessary funds (Bianco, 2009; Cooley and Olander, 2012; Robert and Sterger, 2013).

Disadvantages of stacking: (i) stacking can make it difficult to demonstrate how ES delivered by mitigation projects have abated environmental impacts allowed through offset sales; (ii) stacking may undermine project ‘additionality’ e.g., if payments are more than that required to initiate a project, or are for an activity that would have occurred in the absence of the project, and (iii) stacking indirectly encourages ‘double counting’ – paying twice for (in essence) the same service where similar services overlap e.g., water quality credits and wetland mitigation credits (Bianco, 2009; Cooley and Olander, 2012).

In the case of bundling, single payments are received for the provision of multiple ES from an individual parcel – importantly payment amounts are not (generally speaking) based on the summation of the individual values of each ES (Cooley and Olander, 2012).

Advantages of bundling: (i) recognises the interconnectedness of ES processes and production; (ii) is beneficial for biodiversity and conservation (where broad conservation outcomes are sought); (iii) may increase the overall provision of individual ES from a parcel; (iv) can reduce administrative and transaction costs and raise price premiums, and (v) may reduce the degree of infrastructure needed to support a functioning market (Greenhalgh, 2008; Wendland et al., 2010; Deal et al., 2012; Robert and Sterger, 2013).

Disadvantages of bundling: (i) optimising multiple ES is difficult and given the uncertainty regarding quantification may lead to unintended trade-offs; (ii) limited knowledge concerning ES provision means accurately modelling ES spatial delivery and distribution is highly complex; (iii) regulatory requirements may mean that it is necessary to ‘unbundle’ specific services from the broader set; (iv) it can be difficult to demonstrate additionality and mitigate against double counting, and (v) performance related payments can be difficult to manage as ES bundle provision varies with time (Greenhalgh, 2008; Wendland et al., 2010; Deal et al., 2012; Robert and Sterger 2013).

Projects that employ either stacking or bundling need to ensure they have resolved the issues of additionality and double counting before proceeding (Bianco, 2009).

12.5.2 Seagrass PES Scheme Options

12.5.2.1 Regulating Fisheries And Developing Protected Areas

Many possible institutions are available to control and direct fishing activities along coasts and marine ecosystems (Rudd, 2004). They may involve fishing gear and net restrictions, limiting fishing permits to local residents and restricting the exploitation of connected habitats while providing alternative income generating projects and “legal” fishing equipment (e.g. Mnazi Bay Ruvuma Estuary Marine Park, Tanzania – Alberts et al., 2012; Mohammed 2012). Enforcing closed fishing seasons while providing wage supplements to

fishers to offset opportunity costs resulting from deferred fishing activities is another approach (e.g. the defeso system in Brazil – Bergossi et al., 2011, 2012). Seagrass PES schemes may often involve creating marine protected areas (MPAs), safeguarding the underlying resource base supporting coastal communities and compensating local fishers for lost income resulting from harvesting restrictions (Table 12.7).

Table 12.7 Examples of marine conservation agreements securing coastal conservation and livelihood development opportunities

Country	Project Summary
Ecuador, Galera-San Francisco Marine Area – operating since 2008	<ul style="list-style-type: none"> • Established to combat issues of overfishing, pollution, habitat destruction and coastal construction. • Local communities involved in the structuring of the conservation agreement and in the management of the conservation area. • Conservation agreement covers lobster fishing, no-take areas, fishing regulations and patrol zones. • Benefits to the community include employment in patrolling, management and user rights, access to markets for alternative income streams and capacity building. • Funded by the Nature Conservancy and Conservation International (via conservation stewardship programme) and Walton Foundation (via eastern tropical pacific seascape) – requires government investment to maintain the program in the long-term.
Fiji, Bio-prospecting and Live Rock Harvesting – earliest projects since 1997	<ul style="list-style-type: none"> • Example of locally managed marine areas (of which 200 currently exist involving 300 communities covering 30% of inshore fisheries). • <i>Bio-prospecting</i>: External private organisations make agreements with local communities facilitated by the University of South Pacific (USP) and regulated by the government; with benefits directed to local resource owners (fees paid by these companies are channelled to a district conservation and education trust fund). • <i>Live Rock Harvesting</i>: To substitute the removal of the natural reef base with artificially created reef-bases for aquarium traders. Local users are granted management and access rights over parts of the seabed. Walt Smith International signs agreements with local villages and trains individuals to artificially culture and harvest 'live rocks'. Villages pay US\$0.25/Kg of bare rock and receive US\$0.50/Kg of 'live rock'. USP also purchases 5000Kg of material for each village on the stipulation that almost two-thirds of the proceeds are put back into the live rock harvesting process.
Indonesia, Koon Island, Maluku Marine Conservation area – 2011 to 2014 (with option for yearly renewal)	<ul style="list-style-type: none"> • Comprises a marine protected area, a no-take-zone (to protect spawning grounds) and a rights-based sustainable fishery (also involving a local fishery cooperative partnering with a local fishing company). • Established to protect biodiversity, maintain a sustainable fishery and enhance community development. • A community foundation has been created (TUBIRNUIATA) to implement project activities such as patrols which employ paid community members. • Funding is mainly through philanthropic sources as well as WWF-Indonesia – also attempting to establish a number of ecotourism initiatives.

Table 12.7 *Contd.*

Country	Project Summary
Indonesia, Penemu and Bambu Islands, West Papua – Marine Conservation Area – from 2011 to 2036	<ul style="list-style-type: none"> • Comprises a no-take-zone and sustainable fishery, for the purposes of conservation, ecotourism and community development. • Project developed with a local non-profit organisation Taman Perlindungan Laut (TPL) and Sea Sanctuaries Trust (SST). • Marine conservation agreement is a contract between TPL/SST and the Pam Island Communities, with the purpose of developing ecotourism businesses to provide alternative livelihood revenue streams and sustain the program long-term. Benefiting local communities through employment opportunities, technical assistance and access to goods and services. • Aims to be self-funding after ten years.
Tanzania, Chumbe Island Coral Park, Zanzibar – established since 1992	<ul style="list-style-type: none"> • Private marine reserve, which includes 30 hectares designated as a marine reef sanctuary (coral reef and seagrass beds) plus an additional 20 hectares of coral rag forest, for the purposes of conservation, research, eco-tourism and local education. • Chumbe Island Coral Park Ltd established the park through management contracts and a lease from the Zanzibar government, and has since become an international ecotourism destination and conservation area. • The ecotourism component fully covers management costs. Several international conservation and development donors have been involved with specific local conservation and education programmes. • The Park trains and employs local people as rangers, guides and hospitality personnel. Guides and rangers also function as educators to communicate to local fisherman the importance of the reef bed and maintaining a no-take-zone. Local people have benefitted through increased incomes, access to markets for local goods, technical assistance and improved fish stocks.

Examples adapted from The Nature Conservancy's Marine Conservation Agreements: Practitioner's Toolkit (<http://www.mcatoolkit.org/>)

Designating “no-take-zones” to increase habitat cover and fish stocks, and compensating fishers for lost income is a strategy that some external non-governmental organisation (NGO) donors have used (e.g. Kuruwitu Conservation and Welfare Association in Kenya – Mohammed, 2012). Setting up seagrass PES schemes requires research in a number of areas identified as priorities (Rudd, 2014), including the role of MPAs on ecological resilience (ranked 30th) and their effect on human well-being (ranked 45th). Questions regarding compliance with rules (ranked 58th) and the capacity of communities to manage their coasts (ranked 56th) also demonstrate the potential value of seagrass PES development beyond the sector, as programs provide valuable opportunities to learn broad lessons about the interactions between social and ecological systems.

12.5.2.2 Ecotourism

MPA managers and coastal businesses may establish “green” levies or taxes for resort tourists and charge user-fees for diving access and licenses. Revenues generated by these charges can be re-invested to support continued management activities to enforce the operating rules and ensure compliance, conserve and restore seagrass beds, and create employment opportunities for local community members (Lutz, 2011). In this respect,

participation of the private sector can be transformative for scheme development by acting as a powerful ally in conservation outreach, providing new sources of financial support and creating employment and income opportunities alongside appropriate public sector institutions (e.g. the Indonesian Yayasan Karang Lestari coral restoration project and Marine tourism park on the island of Gili Trawangan – Bottema and Bush, 2012).

12.5.2.3 Linking Farming, Industry And Watershed And Coastal Management

Eutrophication and hypoxia resulting from nutrient loading and upland pollution are significant threats to the health of seagrass ecosystems (Waycott et al., 2009; Short et al., 2011, 2014). Because upstream land-use activities can negatively affect seagrass ecosystems (Freeman et al., 2008; Rivera-Guzmán et al., 2013) the conditions necessary for emulating watershed payment schemes are ripe (Porrás et al., 2013). This may involve cross-sector collaborative partnerships between local and international NGOs, who are often project initiators and intermediary facilitators, working together with public utilities, private firms and government organisations acting as ES buyers (Porrás et al., 2008; Schomers and Matzdorf 2013). Benefits to water quality and reduced water treatment costs save public utilities and private firms significant financial outlays, which may then be channelled into project start-up costs and payments for participants. Examples include the equitable PES schemes for watershed services in Tanzania and Honduras (Kosoy et al., 2007; CARE 2009; Branca et al., 2011). Collectively, these examples highlight the integrated nature of coastal and terrestrial systems and demonstrate that PES schemes which acknowledge these interactions begin to address Rudd's (2014) questions on “upland hydrology effects on oceans” and “integrated upland coastal management” ranked (24th) and (43rd) overall.

12.5.2.4 Biodiversity Conservation

Many turtle populations nest in coastal regions supported by seagrass ecosystems (Cullen-Unsworth and Unsworth, 2013). These iconic and charismatic species are increasingly threatened by poaching and so ensuring healthy nesting populations is vital (Koch et al., 2006). Protecting seagrass ecosystems may be a cost-effective and financially viable option for sea turtle conservation. Paying locals to monitor nesting sites and fishermen for releasing live turtles caught in fishing gear provides a direct and additional income stream for local communities (Ferraro, 2009; Mohammed, 2012). In nesting projects locals usually receive two payments: a flat fee for identifying nest locations; and a variable payment based on hatching success. Successful examples include Natamu Turtle Watch and Knunga Marine National Reserve Conservation and Development Project in Kenya and Sea Sense on Mafia Island in Tanzania (Ferraro 2009).

Due to positive willingness to pay (WTP) for sea turtle conservation among citizens of developed countries (e.g. Rudd, 2009), there are also opportunities for developing international PES schemes that transfer funds from developed countries, where WTP for iconic species conservation is high, to developing countries where turtle nesting grounds and critical life stages occur. For other seagrass-dependent iconic species that enjoy an international profile, there may be similar opportunities as for sea turtles. Seagrass ecosystem conservation and management may thus provide lessons in how triage decisions for species at risk (ranked 32nd, Rudd, 2014) are conceptualized and implemented (Hughes et al., 2009).

Importantly, the relationship between iconic species and seagrass meadows is highly dynamic. For example, in Australia it has been shown that dugong grazing intensity can impact the composition of seagrass meadow beds and their capacity to recover (Preen, 1995). It is therefore imperative for the conservation measures to adequately account for these potential negative impacts on the long-term condition of seagrass meadows, which may arise from population increases in iconic grazing species as a result of triage programmes.

12.5.2.5 Restoration

Seagrass ecosystems are declining yearly (Unsworth et al., 2014). To reverse this global trend seagrass restoration (in suitable areas) offers an effective means to rehabilitate carbon stores and sinks (Duarte et al., 2013c) whilst enhancing other equally important ecosystem services (Greiner et al., 2013). A recent seagrass restoration CO₂ accumulation model, examining long-term trends in carbon sequestration for several commonly planted seagrass species, demonstrated that at an optimal density carbon accumulation of 177-1337 t CO₂ ha⁻¹ after 50 years could be achieved (Duarte et al., 2013c). However, although seagrass restoration has a relatively long history, particularly in the USA, it still remains limited in scope and success (Fonseca, 2011). Nevertheless, the importance of restoration activities for coastal management has been highlighted by Rudd (2014), with the ocean priority research question addressing “restoration effectiveness” ranking (29th). Restoration programs also provide opportunities to generate significant socio-economic benefits.

However, seagrass restoration costs can be expensive. In the USA, projected costs were estimated at between US\$593,000 and US\$970,000 (1996 US\$) per hectare (author's conversion) once mapping and ground-truthing, planting, monitoring, contracting and government oversight were included (Fonseca, 2006). In addition, restoration programs suffer from a number of challenges associated with validation (i.e. monitoring), site selection, artificial colonization methods, management processes and lack of adequate scientific knowledge regarding seagrass ecology (Fonseca, 2011). Nonetheless, with respect to restoration program outlays, recent estimates in Australia have suggested somewhat more

feasible restoration costs of between AUS\$10,000 and AUS\$629,000 per hectare, with investments in restoration at the lower end implying pay-back times of 5 years or less (Blandon and zu Ermgassen, 2014). This is further supported by the work of Duarte et al., (2013c), which suggests that due to the value associated with the sequestered carbon restoration programs may be able to recover between US\$12,000 and US\$43,000 ha⁻¹ (constant dollars), enabling the recovery of full program costs where a carbon tax is in place. Furthermore, most restoration programs are likely to be undertaken in developing countries where capital and labour costs are much less prohibitive (Duarte et al., 2013c).

The Swahili Seas Mikoko Pamoja project (2010-2013) provides a successful example of a wetland restoration carbon finance program operating in a developing country context. Active in Gazi Bay, Kenya, the Mikoko Pamoja project has established a mangrove conservation and restoration program focused on the carbon storage value of mangroves to benefit poor coastal communities. The program operates an accredited Plan Vivo carbon credit scheme providing US\$13,000 annually, which is disbursed to conservation activities and community development projects. Moreover, since 2012 one of the project partner's, Earthwatch Institute, has employed local residents and volunteers to participate in mangrove management and restoration activities covering over 600 hectares. Finally, the project has also engaged in a number of capacity building initiatives through the provision of additional training and networking facilities (UNEP and CIFOR, 2014).

12.6 Possibilities for Implementing Seagrass Conservation Mechanisms

Deciding on the basic operational parameters for a PES program is only half the challenge; the other is to consider how broader institutional and governance elements weave together to influence scheme developments and outcomes: issues that need to be tackled at the design and implementation stage to ensure lasting results (Lin and Nakamura, 2012; Lin and Ueta, 2012). Collectively, these issues are intimately linked to three of the priority research questions identified by Rudd (2014), namely: “management capacity of human communities” (ranked 56th), “transaction costs of ocean management” (ranked 57th) and “property rights and conservation” (ranked 66th). Below we identify some of the most salient issues, incorporating insights from REDD+ and coastal resource management. As AGEDI (2012:10) note:

“Blue Carbon and PES project developers have the opportunity to learn from the challenges and successful outcomes from REDD+ projects that feature similar project elements”.

12.6.1 Institutions

Effective institutions are crucial to the successful implementation of incentive schemes and the resolution of coastal management problems (Rudd et al., 2003; Imperial, 2005; Schomers and Matzdorf, 2013; Somorin et al., 2014). In the process of establishing effective institutions the development of institutional flexibility is particularly important, as this enables programs to respond adaptively over time to changing circumstances and thus maintain their efficacy (Larson and Soto, 2008; Murdiyario et al., 2012). Securing institutional flexibility requires program arrangements that foster active connections and relations between actors, strong leadership and feedbacks in learning systems (Cox et al., 2011; Garbach et al., 2012; Legrand et al., 2013; Geist and Howlett, 2014).

In order to deliver these, programs need to be based on a platform of transparency, accountability and inclusivity (Lockwood et al., 2010; Larsen et al., 2011; Ingram et al., 2014). These aspects function as enabling properties, and the evidence clearly indicates that a lack of transparency and accountability can seriously impair institutional capacity and effectiveness (Somorin et al., 2014), whilst also undermining social capital (Rudd et al., 2003; Shiferaw et al., 2008). In addition, programmes that fail to consider the issue of inclusivity can ultimately disempower participant groups, and as a consequence, embed benefit sharing inequalities between households and communities (Krause et al., 2013).

12.6.2 Stakeholders and Participation

Devolving decision-making to stakeholder groups can be enormously beneficial (Larson and Soto, 2008), at once enhancing and strengthening intra-community ties as well as a sense of common identity (Rudd et al., 2003). Conversely, centralized administration can often stifle local-scale innovations and the development of shared visions (Pokorny et al., 2013). Programmes need to engage and connect with local stakeholders in order to maximise participation, which is central to providing effective management (Agrawal and Chhatre, 2006; Benjamin, 2008). Doing so legitimises decision-making and empowers individual and collective agency enabling the design process to align with, and support, local norms, values and beliefs (Kawowski et al., 2011; Brooks et al., 2012; Corbera, 2012; Bremer and Glavovic, 2013). This is essential for participant commitment (Murdiyario et al., 2012; Davenport and Seekamp, 2013) and acknowledges the relevance for effective governance of local users' knowledge (Andersson et al., 2014).

These processes can be supported by clarifying stakeholder roles and responsibilities and promoting leadership (Chhatre et al., 2012; Dent, 2012). Leadership, and especially local leadership, has been shown to be fundamental to delivering successful coastal management (Sutton and Rudd, 2014). Finally, it is important to acknowledge how participation is framed

in the context of power relations, as these can represent potent forces capable of distorting the meaningful involvement, agency and legitimacy of grassroots actors (Dewulf et al., 2011; Cook et al., 2013).

12.6.3 Tenure And Property Rights

Ownership in developing countries is often complicated by overlapping formal and informal (customary) tenure and rights-based arrangements (Awono et al., 2014; Resosudarmo et al., 2014; Rights and Resources Initiative, 2014; Sunderlin et al., 2014). Clearly defining, legitimising and enabling functioning property rights systems is essential for operationalizing incentive programs (Lockie, 2013). Such clarifications are critical for conditional payments where knowing who to pay (i.e. the right holder) and who is accountable for delivering project-level outcomes is necessary (Visseren-Hamakers et al., 2012; Duchelle et al., 2014; Sunderlin et al., 2014). Functioning tenure and rights-based systems also provide the framework to enforce property rights, securing contracts (Naughton-Treves and Wendland 2014) and combating weak governance (Resosudarmo et al., 2014).

This is particularly pertinent to coastal marine environments where complications concerning tenure, rights designations and authority are a direct challenge to introducing and enforcing incentive schemes (Mohammed, 2012), a state of affairs clearly linked to the ambiguities regarding property rights in coastal areas and the variety of users and user interests (Cicin-Sain, 1993). As part of the design process it is crucial to mitigate potential mismatches arising between the provision, delivery and bundle of property rights to reduce the likelihood of marine resource conflicts developing (Yandle, 2007), as well as to ensure that poorer sectors are not marginalised or power asymmetries and social inequalities reinforced (WRI 2005; Fisher et al., 2008).

12.6.4 Benefit Sharing

Distributing benefits and costs in a fair and equitable way is a fundamental aspect of delivering socially acceptable incentive schemes (McDermott et al., 2012; Rodriguez de Francisco et al., 2013). Traditionally, equity concerns have been side-lined in favour of a greater emphasis and focus on efficiency maximization (Pascual et al., 2010; Narloch et al., 2011, 2013). However, this trade-off can produce socially undesirable outcomes (Asquith et al., 2008). Incorporating social parameters in the targeting of schemes in order to widen access and participation whilst reducing the marginalization of poorer communities represents an important first step in reversing these potential trade-offs (Mahanty et al., 2013). These processes need to proceed in tandem with beneficiary identification and the evaluation of the potential socio-economic ramifications of ES provision and distribution (Willemsen et al., 2013). Additional considerations for effective benefit sharing include legitimising decision-

making processes via legal and procedural avenues (Murdiyario et al., 2012); adjusting compensation levels according to the capacity needs of individuals, households and communities (Mohammed 2012); and addressing the potential socio-economic impacts of programs on non-participants (Huang et al., 2009).

12.6.5 Delivering Ecosystem Services, Monitoring And Compliance

The central tenant of incentive schemes relates the provision of specified outputs to agreement obligations and payments (Ferraro, 2008; Wunder et al., 2008). Consequently, monitoring and compliance represent key contractual conditions for programs to deliver their principal objectives (Danielsen et al., 2013; Hejnowicz et al., 2014). These can be distilled into four broad areas:

First, measuring ES provision (Porras et al., 2013). This reduces the likelihood of producing a false picture of service provision, and provides a scientifically robust case for PES program design (Hejnowicz et al., 2014). It has been suggested that even though coastal systems may be data poor, there is sufficient knowledge of the management activities that improve resource protection and ES provision (Lau, 2013). Second, evaluating scheme additionality and demonstrating ‘added value’ by addressing the links between management interventions and program delivery (Ghazoul et al., 2010). Validating additionality requires baseline data, suitable metrics and performance indicators plus the targeting of PES to locations likely to maximize program benefits (Wünscher et al., 2008; Sommerville et al., 2011; Wünscher and Engel, 2012; Lau, 2013).

Third, assessing potential of spill-over effects (i.e. leakage) resulting from program implementation that may offset additionality gains (Engel et al., 2008; Porras et al., 2013). Fourth, monitoring contract conditionality and ensuring compliance (Ferraro, 2008). This requires establishing who is monitoring (i.e. users, communities or officials) and how frequently (Sommerville et al., 2011), providing sufficient payments to programme participants (Porras et al., 2013), and ensuring agreements are long-term arrangements with enforceable penalties for breaches of contract (Ferraro, 2008; Wunder et al., 2008). All have substantive effects on transaction costs of governance (ranked 57th, Rudd 2014) and will influence the long-term viability of PES structures.

12.6.6 Costs And Funding

The viability of PES programs relies upon consistent and sufficient financial flows, both in the short-term (i.e. covering costs needed to initiate and implement a project) and the long-term (i.e. securing the funds necessary to sustain an active project), without which lasting transformative change cannot be achieved (Hejnowicz et al., 2014; Kauffman, 2014). Programs need to be designed so that they sustain themselves through self-generated revenues

(Pirard et al., 2010). An added complication for seagrass PES schemes is that monitoring and enforcement in marine and coastal environments may require extra technical and specialist equipment not needed in the terrestrial sphere, adding significantly to program outlays (Lau, 2013). Securing long-term funding that reduces fiscal constraints but is not overly reliant on external donor funding is particularly important (Fauzi and Anna, 2013; Hein et al., 2013). Achieving both these objectives requires adequately accounting for the full range of transaction costs, which in some cases may be prohibitive for PES development (McCann et al., 2005; Marshall, 2013; McCann 2013).

12.7 Final Remarks

Seagrass ecosystems provide an array of globally and locally significant ecosystem services. From the perspective of climate change, it is their carbon sequestration and storage potential that is most attractive. Seagrass ecosystems are also home to diverse marine life that can directly or indirectly support the artisanal and commercial fisheries that help maintain resilience in human communities. In addition, they also play an important role in the conservation and maintenance of marine biological diversity and influence national or international non-market benefits deriving from endangered species such as sea turtles (Rudd, 2009). We have examined the prospects for financing seagrass conservation under a purely carbon approach and in conjunction with PES schemes that could help capture the benefits derived from multiple ecosystem services beyond carbon sequestration.

The prospects for developing a pure carbon credit scheme remain slim, especially if targeted at the regulatory carbon market. Opportunities exist, however, for voluntary carbon market schemes and these are far more promising. However, the instability of the voluntary carbon market and the impact this has on carbon prices makes a purely carbon-based approach questionable; fluctuating carbon prices mean projects cannot guarantee financial returns on investment or adequate payments to meet participants' needs. Nonetheless, voluntary carbon standards are channelling more effort into delivering co-benefits and, from this perspective, seagrass PES schemes may be highly complementary. Adopting a combined strategy would maximize conservation and livelihood outcomes so long as the design, implementation and institutional issues previously highlighted were adequately dealt with.

Providing the scientific evidence base for complex incentive schemes is challenging. This is particularly so with Blue Carbon systems where there remain many ecological, social and economic knowledge gaps that need to be negotiated in order to develop functional payment programs. However, we have mapped out what those potential knowledge gaps are in relation to seagrass ecosystems, in terms of basic ecosystem function-service information, ecosystem service valuation and research concerning the governance structures and apparatus

through which incentive schemes would need to operate. In so doing we have highlighted the importance and complexity of seagrass ecosystems and the value of conserving them. At the same time we have clearly identified how by conserving these systems, particularly through the use of innovative financial incentive mechanisms, we are also contributing to a broader set of significant global ocean priority research challenges.

Overall, a wide range of opportunities exist for including seagrass meadows in local PES schemes to combat climate change, secure seagrass conservation and enhance coastal community development. However, realizing the “true” potential of seagrass meadows requires international cooperation on two fronts: combating the threats that currently imperil the integrity of functioning seagrass ecosystems and including them in formal climate change policies such as REDD+. In this respect challenges and barriers remain but promising progress is being made; efforts to protect and rehabilitate seagrass ecosystems are crucial because of their widespread distribution, their central role in supporting functional coastal environments and the human communities that rely on those systems.

Chapter 13: Case Study 3 - Intermediaries And Agri-environment Schemes: Private Farm Advisor Perspectives On England's Environmental Stewardship Schemes

“The production of natural resources in agriculture, forestry and fisheries, stable natural hydrological cycles, fertile soils, a balanced climate and numerous other vital ecosystem services can only be permanently secured through the protection and sustainable use of biological diversity” (Sigmar Gabriel)

“The discovery of agriculture was the first big step toward a civilized life” (Arthur Kieth)

13.1 Introduction

Driven by a range of complex local and global drivers (e.g. globalisation, food security concerns) food production and domestic consumption patterns have undergone rapid transformations (e.g. FAO, 2003; OECD/FAO, 2011; Tscharrntke et al., 2012; Poppyet al., 2014). These changes have been accompanied by significant agricultural intensification and extensification (FAO, 2012, 2014; Godfray and Garnett, 2014). Striking a balance between intensification and extensification is a central challenge for modern food production systems (Pretty et al., 2010; Balmford et al., 2012; Grau et al., 2013). Without balance, environmental risks are high and may include deforestation and forest degradation, loss of bio-diversity, soil erosion, decreased water quality, water shortages, increases in greenhouse gas emissions and changes in biogeochemical cycles (e.g. Gibbs et al., 2010; Quinton et al., 2010; Lambin and Meyfroidt, 2011; Lenzen et al., 2012; Mills Busa, 2013; WWAP,2014).

In Europe aspects of the agricultural sector have also undergone a degree of intensification (OECD, 2008), with concomitant repercussions for ecosystems, biodiversity and water resources (e.g. Tscharrntke et al., 2005; Billeter et al., 2008; Henle et al., 2008;EEA, 2010; Pe'er et al., 2014; Zanten et al., 2014). The continuing problem European Union (EU) Member States face is trying to maintain thriving and competitive agricultural and forestry sectors whilst also ensuring a secure provision of environmental public goods (Allen and Hart, 2013). In response, to resolve this tension, incentive-based management strategies such as agri-environment schemes have been introduced throughout the EU (Deal et al., 2012; Lastra-Bravo et al., 2015; Lefebrve et al., 2015).

Initially optional, the 1992 MacSharry Reform of the Common Agricultural Policy (CAP) made AES a compulsory agricultural measure for all EU Member States; with further consolidation via the Agenda 2000 Reform leading to their provision under Pillar 2 of the CAP (European Commission, 2005; McCormack, 2012). Essentially, AES operate through voluntary contractual agreements and provide farmers with payments in return for the delivery of environmental public goods and services and/or the adoption of modern environmentally-friendly farming practices (Garrod, 2009; Lastra-Bravo et al., 2015; Lefebvre et al., 2015). Their implementation is based on the subsidiarity principle, meaning that AES are specially designed to negotiate the particular production-conservation circumstances faced by individual Member States, which they achieve by addressing three intertwined matters, namely: greening farming practices; reducing food production impacts on biodiversity and improving overall countryside management (European Commission, 2005; Smits et al., 2008; European Court of Auditors, 2011; McCormack, 2012; Allen and Hart, 2013; Burton and Schwarz, 2013).

Following their introduction in the UK in 1986 various versions of AES have affected more than 6 million Ha of agricultural land in England alone (Dobbs and Pretty, 2008; Gibbs, 2010; Tucker, 2010). The most significant recent variant, 'Environmental Stewardship', began in 2005 (Chaplin and Radley, 2010). Its purpose—to offer a fresher, more radical, two-tiered approach to land management characterised as “broad and shallow” and “narrow and deep” (Hart, 2010). The “broad and shallow” tier was designed as a non-competitive and open-access arrangement, while the “narrow and deep” component was configured as a targeted and competitive option for meeting priority environmental objectives (Boatman et al., 2010). In England, the Entry Level Stewardship (ELS) scheme represents the “broad and shallow” approach, which also includes organic (OELS) and upland (UELS) variants, while Higher Level Stewardship (HLS) represents the “narrow and deep” element (Boatman et al., 2010; Jones et al., 2010; Table S13.1 Suppl. Material C on CD).

So, how effective are AES schemes at meeting their stated environmental goals? At both the European (e.g. Kleijn and Sutherland, 2003; Kleijn et al., 2011) and UK (e.g. Whittingham, 2007; Boatman et al., 2008; Defra and Natural England, 2008; Whittingham, 2011) scale evidence suggests that their ability to provide environmental and conservation benefits have been relatively mixed. In respect of Environmental Stewardship the picture is similarly mixed, with both positive and negative impacts on the supply of environmental benefits identified. In particular, research has tended to focus on the biodiversity impacts of common in-field, margin and boundary options such as crop rotations, hedgerow management, riparian buffer strips and winter stubble regimes on farmland birds (e.g. Davey et al., 2010a,b; Field et al., 2010; Hinsley et al., 2010; Siriwardena, 2010; Baker et al., 2012; Goodwin et al., 2013; Gruaret et al., 2013), and to lesser extents on floristic diversity (e.g. Still and Byfield, 2010; Morris et al.,

2010), insect pollinators (e.g. Fuentes-Montemayor et al., 2011; Critchley et al., 2013; Dunn et al., 2013; Peyton et al., 2013), natural resource management (e.g. Ramwell and Boatman, 2010), and ecosystem services (e.g. Rollett et al., 2008; FERA, 2012).

Beyond biodiversity, other analyses have demonstrated that participation in Environmental Stewardship can deliver both human and social capital gains (Mills, 2012), whilst also enhancing local employment and boosting the rural economy (Courtney et al., 2013). Yet, it has also been established that the financial compensation mechanism operated by Environmental Stewardship may promote adverse selection as well as reduce the degree of environmental benefits secured (Fraser, 2009; Quillérou et al., 2011).

Concerning ourselves with the principal agents involved (e.g. farmers, land managers, independent farm advisors and Natural England) in the implementation of Environmental Stewardship, research has generally favoured addressing the farmer element: focusing primarily on understanding the views of farmers (e.g. FERA, 2013a) and their motivations for engagement in these schemes (e.g. Mills et al., 2013) with little attention paid to intermediaries (e.g. advisors)—particularly independent farm advisors. Yet, drawing on evidence from payment for ecosystem service programmes, a similar mechanism to AES, clearly demonstrates the importance of external advisors – especially as facilitators of agreement processes between participants and contracting authorities – due to their capacity to provide specialist knowledge and skills (e.g. Ferraro, 2008; Thuy et al., 2010; Lin and Nakamura, 2012; Huber-Stearns et al., 2013; Martin-Ortega et al., 2013; Schomers and Matzdorf, 2013; Hejnowicz et al., 2014).

13.2 Study Aims

In light of this, we posited that examining the farm advisor dimension would represent an important and justified avenue of exploration. By improving our understanding of the views and opinions of farm advisors regarding Environmental Stewardship, it may be possible to identify ways in which to improve the over-all implementation and effectiveness of AES: aspects important for achieving conservation objectives, public goods generation and farm business viability. In this research on the English experience, we report results from a survey designed to explore private farm advisors' views regarding their own particular role in the delivery of Environmental Stewardship agreements as well as their opinions concerning farmers, Natural England and other facets of Environmental Stewardship scheme implementation and operationalisation.

Our online survey adopted an exploratory approach, delving into the “world” of the farm advisor and concentrated on: (i) advisors' views regarding scheme constraints and client motivations and behaviours; (ii) advisors' modes of interaction with their clients and Natural

England; (iii) the determinants influencing the content of individual agreements; (iv) mechanisms for balancing client needs and the provision of sufficient levels of environmental public goods, and (v) recommendations for improving the delivery of AES.

It is important to point out that this investigation tells only part of a much larger story. As such, it should be viewed as the starting point, the first stepping stone, to further, more in depth examinations of the farm advisor role which by necessity would need to be triangulate with the views of farmers, land managers and those of Natural England.

13.3 Background: Evidence To Support Our Exploratory Approach

In concentrating on the areas (i–v) we were guided by evidence highlighting key determinants of voluntary incentive scheme operationalisation, implementation and effectiveness (e.g. Martin-Ortega et al., 2013; Hejnowicz et al., 2014); the general purpose and structure of AES (e.g. European Commission, 2005) and informed by the particular arrangements and specifications of individual Environmental Stewardship schemes (Natural England 2012a, b, c). We combined these different strands to develop an Environmental Stewardship framework (Figure. 13.1) to guide our survey. Reflecting this, the composition of our survey is underpinned by three major foci.

13.3.1 Actors And Their Interactions

13.3.1.1 Farmers: Motivation, Participation And Knowledge

What motivates participants is important; indeed, it is central to their participation (Lastra-Bravo et al., 2015). Motivations under-pinning farmer and land manager decision-making processes have been identified as a complex mosaic of extrinsic and intrinsic values (Siebert et al., 2006), as well as central to delivering effective AES programmes (Mills et al., 2013). The choices farmers make seem to be influenced by a range of external factors like environmental policies; internal drivers such as personal characteristics (e.g. age, education) and farm features, as well as interactive elements related to farm business arrangements and incentive design (Mills et al., 2013). In particular, economic factors related to household income, land tenure, family labour, and farm business structure appear to be particularly influential determinants of participation (Barreiro-Hurlé et al., 2010; Lastra-Bravo et al., 2015). These represent aspects frequently related to the need to maintain family and farm continuity (Farmar-Bowers and Lane, 2009; Ingram et al., 2013). The extent to which this mix of farm structural factors and personal farmer characteristics influences AES participation is affected by the likelihood of an agreement producing either major or minor changes to farm operations, and consequently, the potential impacts these changes may have on marginal profits, the size of transaction costs incurred and the level of utility derived from the delivery of environmental goods and services (Barreiro-Hurlé et al., 2010).

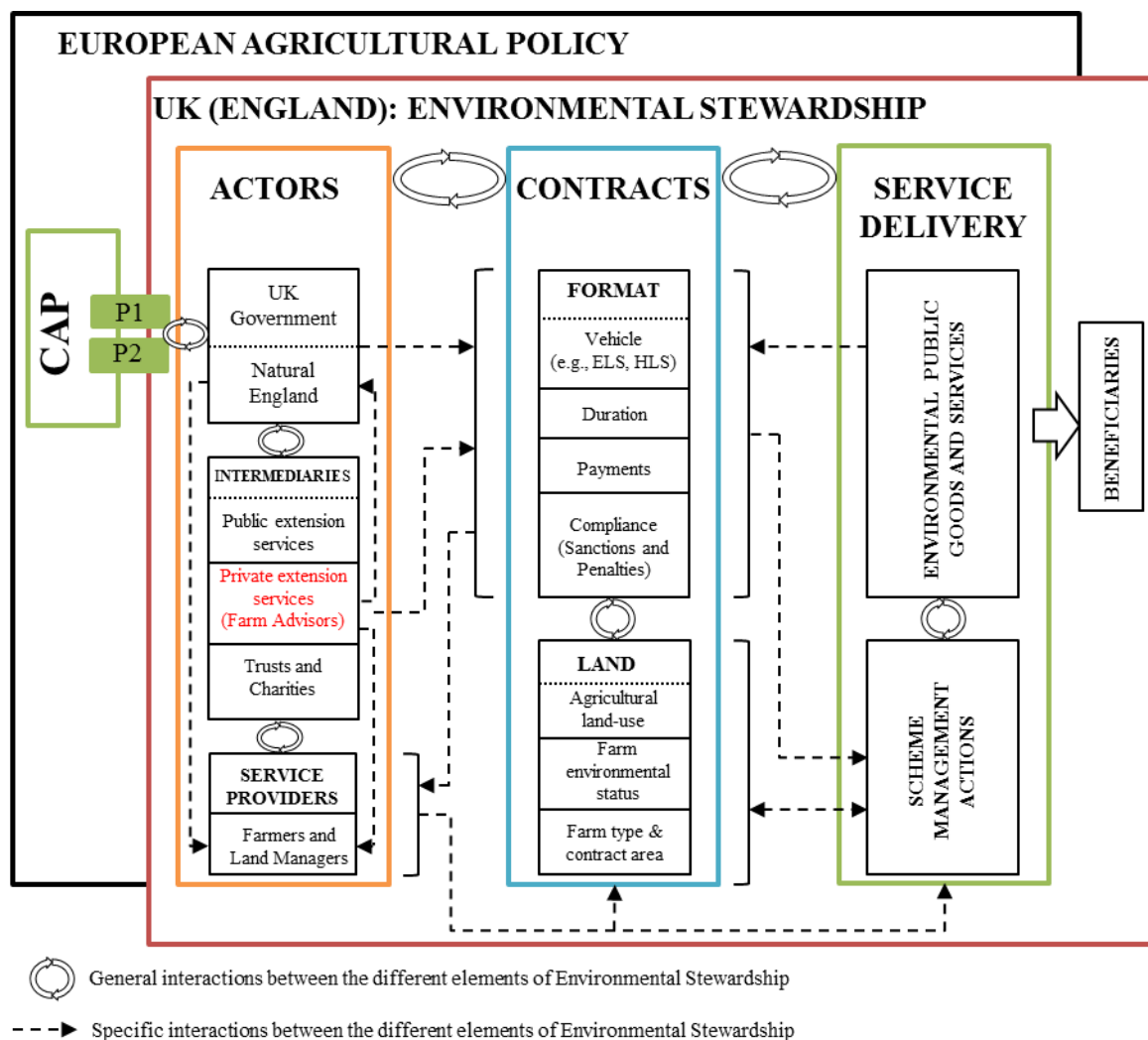


Figure 13.1 Environmental stewardship framework co-opted and adapted from a PES model by Martin-Ortega et al., (2013). The UK/England Stewardship Scheme is placed within the European agricultural policy context expressed through the linkage with the CAP (Pillar 1 and Pillar 2). The Environmental Stewardship programme is divided into three major component parts (actors, contracts and service delivery) with each of these subsequently subdivided into major constituent properties, characteristics and qualities. The framework also emphasises general interactions occurring between components parts as well as highlighting key interactions which are key foci of our survey.

Overall, as Siebert et al. (2006:319) note:

“There is an intricate interaction of contingencies affected by locality and specific context, such as agronomic, cultural, social and psychological factors, which [. . .] play interwoven roles in each [. . .] specific farm context”.

Similarly, alongside motivation, knowledge underpins successful agriculture but the types of knowledge required for engagement in particular land management activities can vary substantially (Winter, 1997). For example, as Ingram notes (2008a:224) in relation to farmers’ knowledge of soil management:

“...although farmers are largely knowledgeable many appear to lack the [...] knowledge necessary for carrying out more complex [...] management practices.”

Environmental stewardship more generally has been shown to enhance farmer skills and environmental knowledge, awareness and appreciation (Mills, 2012).

13.3.1.2 Advisors And Agencies

From this vantage the importance of both public and private advisory extension services and the advice they supply becomes apparent (Lastra-Bravo et al., 2015). The role of external advisors in bridging potential knowledge deficits, acting as a necessary pre-condition for realizing effective voluntary incentive schemes, and delivering successful public policy interventions are widely recognised (Juntti and Potter, 2002; Cooper et al., 2009; Vesterager and Lindegaard, 2012). Advisors are now required to explain regulatory processes and incentives as well as provide information and training (Vesterager and Lindegaard, 2012). Indeed, the evidence shows that informed farmers are far more likely to participate in AES (Lastra-Bravo et al., 2015). As Radley (2013) states, “good advisors” are central to the effectiveness of AES. Underscoring this point, private advisors have been shown to positively impact the promotion of both minor and major AES measures as well as influence the willingness of farmers to adopt such measures (Lastra-Bravo et al., 2015). Considered more generally, the evidence indicates that the interactions between the actors (and agencies) involved in voluntary incentive schemes, operating across different institutional levels, as well as the degree of decentralisation and devolution in decision-making, are central to the functionality and effectiveness of these interventions (e.g. Beckmann et al., 2009; Pascual et al., 2010; Legrand et al., 2013).

13.3.2 *Service Delivery*

13.3.2.1 Management Practices And The Provision Of Public Goods

The financial rules and decision-making framing AES design are central to how AES achieve environmental outcomes, economic efficiency and widespread uptake (Beckmann et al., 2009). At the heart of AES lies the provision of public goods. The capacity of voluntary management interventions to provide public goods rests on the premise that the measures specified by these initiatives are capable of generating the requisite environmental public goods (i.e. ecosystem services) at scale; and moreover, that participants (e.g. farmers) fully engage in implementing those management practices (Martin-Ortega et al., 2013; Hejnowicz et al., 2014). Essentially, it is about ensuring schemes are capable of demonstrating additionality, or added value, over and above the business-as-usual case in the absence of any intervention (Ghazoul et al., 2010).

Supplying public goods is an essential function of the CAP; but at the pan-European scale multiple environmental indicators suggest these are currently being undersupplied (Cooper et al., 2009; Pe'er et al., 2014). With respect to Environmental Stewardship there has been considerable discussion regarding ELS option management uptake and distribution, the inference being that this directly affects their capacity to facilitate the provision of sufficient environmental public goods (e.g., Hodge and Reader, 2010).

Optimal spatial targeting of management options is a key design challenge faced by incentive schemes (Wünscher et al., 2008). The reluctance of many EU/UK farmers to engage in significant environmental management has also been shown to inhibit uptake, although certain activities are guaranteed under cross-compliance (Rollett et al., 2008). Mechanisms to counteract these behavioural dispositions have been developed (Chaplin and Radley, 2010), such as: restricted option choices under a directed ELS regime (Boatman, 2013); focused initiatives like the “Making Environmental Stewardship More Effective” programme (Blainey, 2013) and the Campaign for the Farmed Environment (Gibbs, 2010). To varying degrees they have established that the delivery of environmental public goods can be enhanced (Boatman, 2013; Clothier, 2013; Defra and Natural England, 2013; Jones and Boatman, 2013). Taking the long view, however, there are those who argue that sustaining behavioural change that results in lasting environmental benefits remains a significant challenge (Burton and Paragahawewa, 2011)

13.3.3 Contracts

13.3.3.1 Agreement Arrangements And Conditions

Connected to issues of uptake and public goods provision are the other core elements of incentive scheme contracts, chiefly, payments, monitoring and compliance (Danielsen et al., 2013; Hejnowicz et al., 2014, 2015). The evidence clearly indicates that, for schemes to be effective, the payments entrants receive must be consistent with, and sufficient to cover, the opportunity costs they face through participation (Porrás et al., 2013; Hejnowicz et al., 2014). Consequently, in the case of AES, public agencies need to ensure that payments cover the operational and investment costs, production and profits foregone and private transaction costs incurred by farmers (Falconer 2000; Mettepenningen et al., 2009). This is important because, for example, regarding the issue of transaction costs, not only do private transaction costs represent a sizeable proportion of total AES-related costs but fixed transaction costs also act as a major contracting barrier (Falconer 2000; Ducoset al., 2009; Mettepenningen et al., 2009). Financial incentives are also key to farmers signing up to longer-term AES agreements, entering schemes that are more prescriptive and joining schemes with significant layers of bureaucracy (Ruto and Garrod, 2009). Hence the significance of payments should not be underestimated, as they directly impact overall income and the feasibility of participation

(Wunder et al., 2008) in addition to having fairness and equity ramifications (Borner et al., 2010; Pascual et al., 2010).

Providing a sufficient incentive also helps to ensure a higher degree of compliance; whilst this is important the other major factor that encourages greater compliance is the provision of con-tracts with enforceable sanctions and penalties (Ferraro, 2008; Wunder et al., 2008). Poor enforcement can often undermine scheme performance (Schomers and Matzdorf, 2013). Enforcement only works if there is an adequate monitoring regime in place, and in the case of Environmental Stewardship schemes this is recognised to require considerable improvement (Defra and Natural England, 2008; Boatman et al., 2010; Chaplin and Radley, 2010; Mountford et al., 2013; Radley, 2013). Collectively, these complex institutional arrangements represent fundamental aspects of the functioning and performance of any voluntary incentive programme (Hejnowicz et al., 2014, 2015).

13.4 Materials And Methods

13.4.1 Data Requirements

Our approach is exploratory. We reasoned the most expedient way to proceed to obtain a broad overview of farm advisors views was to use an online survey. In addition, due to the large number (>900) of Environmental Stewardship advisors this also seemed the most flexible and parsimonious choice. While interviews may provide a far more extensive and nuanced description of farm advisor views, we felt that it was important to gather information from as wide a variety of advisors as possible: given potentially important differences in opinions that might arise due to regional differences in priorities or in differences among advisors who specialized in ELS or HLS schemes. We also anticipated that a broad-based approach would identify specific issues and provide the necessary context to conduct more focused qualitative interviews with farm advisors as well as other agents in the process (e.g. farmers, Natural England advisors) in the future.

13.4.2 Sample

The sample was composed of Natural England registered stewardship advisors whose contact information, which was publically available, we obtained from Natural England's register.¹ After removing duplicate entries the register included information for 958 individual advisors from eight regions of England (North East, North West, East Midlands, West Midlands, East of England, South East, South West, and Yorkshire & Humberside).

13.4.3 Survey Instrument

The survey was constructed with Sawtooth Software's (www.sawtoothsoftware.com) SSI web-based interviewing platform. A pilot survey was tested on 24 stewardship advisors, three from each of eight regions; responses from the pilot informed the design of the final survey. The questions included in the survey explore those issues previously discussed in Section 2. The final survey [Final Survey, see Suppl. Material E on supplied CD] consisted of 39 questions divided into three parts. Most questions were closed, involving selection of radio buttons (single answers) or checkboxes (multiple answers) but several questions included space for providing extra comments. More specifically, Part 1 requested advisors' generic back-ground information, including expertise. Part 2 focused on the process of generating a stewardship agreement and comprised sub-sections covering advisors' views regarding: client motivations and knowledge; agreement preparation and constraints; inter-actions with clients and Natural England; how advisors balance farmer needs and Natural England objectives; and recommendations for improving environmental stewardship agreements in light of future changes to scheme delivery. Part 3 concentrated on the environmental content of agreements, respondents' views regarding Natural England advisor amendments to submitted HLS agreements, and advisors' perceptions of client understanding and acceptance of scheme payment levels and sanctions. Importantly, participants were fully aware of what professional opinions were being asked of them and could decide to opt-out at any stage of the survey.

13.4.4 Survey Implementation

Following standard social science survey protocol (Dilman et al., 2009), Environmental Stewardship advisors were contacted up to five times over a period of five weeks: (8th October to 7 November, 2013, with the survey open from 15 October to 15 November, 2013). Invitation emails contained a unique hypertext link enabling advisors to access the survey directly (Survey Protocol, see Suppl. Material C). Of 958 advisors from the register, we assumed that 840 had been contacted after adjusting for 118 non-delivery email notifications.

13.5 Results And Discussion

13.5.1 Advisor Characteristics

13.5.1.1 Demographics, Experience And Regional Distribution

A total of 354 respondents (42% of the sample) accessed the online survey platform and 251 (29.9%) completed the full survey (Table S13.2 Suppl. Material C). Given the number of completed responses, we can be fairly confident that the views expressed by respondents in our "survey sample" are broadly representative of those held by the "population sample".

Respondents were asked to indicate the regions in which they operated, in terms of commonality: 24.8% of indicated the South West; 20.3% the West Midlands; 17.5% the East of England and Yorkshire-Humberside; 16.3% the East Midlands and the South East; 14.6% the North West; and 8.5% the North East. The majority of respondents (75.3%) operated in one region, with only 14.0%, 5.6%, 2.8% and 2.0% working across two, three, four or five regions respectively.

Relatedly, regions appeared to differ according to their degree of advisor mobility (Table S13.3 Supplementary Material C). In some locations advisors demonstrated a more extensive “regional working network”, operating across a number of different regions, whereas in other cases these “networks” were more limited. In the East Midlands, for example, advisors exhibited the highest level of mobility with 60.0% working across additional regions. In the South West, on the other hand, just 24.6% of advisors operated outside their own region.

Advisor experience ranged considerably: from one to thirty years. However, the majority of respondents (56.5%) worked in an advisory capacity for 9.6 ± 5.6 yrs, although a large minority (39.04%) were considerably more experienced with 10–20 years of practice. Respondents operating across more regions also appeared to have longer experience as farm advisors, for example, 14.6 ± 2.0 yrs for those working across four and five regions compared to 9.4 ± 0.4 yrs for those working in a single region. Generally respondents indicated that they had knowledge and expertise in relation to two (31.5%) or three (42.2%) Environmental Stewardship schemes, with proficiency in ELS (93.4%) and HLS (82.8%) being the most common (Table S13.4 Suppl. Material C). There were distinctive patterns of regional expertise (Table S13.5 Suppl. Material C), probably reflecting the different geographies of these regions as well as the more limited application of some schemes compared to others (e.g. upland versions of ELS compared to standard ELS schemes).

13.5.2 Agreement Formation

13.5.2.1 Understanding Clients: Farmer Motivations

Identifying the most common motivating factors leading farmers and land managers to engage with Environmental Stewardship, we found that advisors felt both extrinsic and intrinsic values played a motivating role (Table 13.1). Evidence from previous research indicates that farmers’ and land managers’ participation in voluntary agri-environment schemes is influenced by a variety of attitudes and values (e.g. Siebert et al., 2006; Cross and Franks, 2007; Mills et al., 2013). Respondents ascribed a heterogeneity of motivations to the decision-making processes underlying farmer participation, this suggests connections to broader socio-cultural norms, worldviews and goals (Ingram et al., 2013; Mills et al., 2013)—advancing what Morris and Potter (1995) term a ‘participation spectrum’. Building on the

existing literature (e.g., Wilson and Hart 2000, 2001; Siebert et al., 2006; Cross and Franks, 2007; Defrancesco et al., 2008; Mills et al., 2013; Lastra-Bravo et al., 2015) our analysis of advisor opinions suggests that extrinsic values such as those related to financial gain, profit maximization, long-term security and capital investment represent the primary motivators encouraging farmer and land manager engagement with environmental stewardship. These observations accord with evidence from UELS agreement holders suggesting that scheme payments act as the principal motivating factors determining participation, with additional agronomic concerns such as the degree of alignment with existing farm practices also influencing engagement (CCRI, 2012). In this regard we identified so-called “calculating” (i.e. purely financial), “opportunistic” (i.e. income from existing practices), “optimizing” (i.e. production potential of marginal land) and, to a lesser extent, “compensatory” (i.e. regulatory obligation) motivating classifications as the primary drivers underlining farmer participation reinforcing previous analyses (e.g. Pike, 2008, 2013; Van Herzele et al., 2013).

Previous studies of AES participants have, for example, high-lighted support for environmentally-oriented concepts such as “wildlife and environment”, “improving the landscape” and “wildlife conservation benefits” (CCRI, 2012; FERA, 2013a). Indeed, a positive environmental attitude can be a component of farmers’ willingness to engage with AES (Lastra-Bravo et al., 2015). Similarly, respondents felt that intrinsic values were important secondary and tertiary motivators underlying Environmental Stewardship engagement, perhaps in this sense, complementing the more widespread financially-oriented motivations which they proposed that farmers hold: lending credence to the notion that extrinsic and intrinsic values need not be mutually exclusive (Mills et al., 2013). To an extent this may help temper concerns that without significant intrinsic motivations farmers lack the necessary incentive to deliver long-lasting environmental management improvements (Van Herzele et al., 2013).

13.5.2.1 Farmers’ Knowledge And Advisor Advice

Farmers’ capacity to undertake on-farm environmental management, in part, relies on their knowledge, skills and understanding of those management requirements, although competencies can vary significantly between individuals (Ingram, 2008a). Consequently, there is a growing recognition that the provision of advice, and the role of advisor, is central for helping farmers negotiate the progressively more complex demands of environmental management and agricultural production (Ingram, 2008b). The majority of respondents (60.0%) indicated that 50% or more of their clients had a clear notion of the stewardship scheme they wished to enter. However, only 27.6% of respondents agreed that a similar proportion of their clients also understood the intricacies of agreement arrangement (i.e. with particular reference to the application process)—in this case advisors may have been referring

Table 13.1 Stewardship advisor perceptions of client motivations

Motivation classification ^{a,b}	Motivating factors (<i>key themes</i>)	Primary reason (%, n=246)	Secondary reason (%, n=235)	Tertiary reason (%, n=153)
Extrinsic				
<i>Financial incentives (Cal)</i>	Economics (i.e. income, finance, money, cash, payment, compensation)	64.6	8.9	5.9
<i>Profit maximisation (Cal)</i>	Finance linked to farm viability and management	5.7	-	-
<i>Long-term security and farm viability (Opp)</i>	Continuation of current practices and extension of a prior environmental scheme	4.5	9.8	5.9
<i>Capital investment (Opt)</i>	Use of unproductive marginal land (linkages to finance and profitability)	4.1	12.3	5.9
	Finance linked to recouping monies from modulation	3.7	2.1	1.3
	Income diversification	1.22	2.1	-
	Capital works (e.g. finance, investment and enhancements)	-	6.4	6.5
	General improvement in farm management and operations	-	3.8	7.8
<i>Community image & recognition in wider society (Cat)</i>	Increase farm value	-	1.7	1.3
	Prestige and public perception	-	-	1.9
<i>Regulation (Com)</i>	Cross-compliance	-	0.4	5.9
<i>External non-regulatory obligation*</i>	Peer Pressure	-	0.8	1.9
	Encouragement from Natural England	-	0.4	0.6
Intrinsic				
<i>Personal sense of environmental responsibility and accountability (E)</i>	Environmental benefits (e.g. biodiversity, wildlife, conservation, farm environment)	13.8	39.2	35.3
	Environmental benefits linked to improvements for game and shooting	-	2.1	5.9
	Obligation and responsibility (e.g. moral and environmental aspect)	1.2	3.4	5.9
<i>Commitment and interest in the environment (E)</i>	Personal satisfaction and interest	-	-	5.2

^a Motivation classifications (i.e. extrinsic and intrinsic values) and sub-groupings are based on Mills et al. (2013)

^b (Cal, Calculating), (Cat, Catalysing), (Com, Compensating), (E, Engaged), (Opp, Opportunistic) and (Opt, Optimising) are categories referring to modes of agri-environment scheme participation identified by Van Herzele et al. (2013). See paper for category explanations.

*This category is not identified by Mills et al. (2013) but emerged from respondent comments.

to clients that were renewing agreements (FERA, 2013a). Interestingly, respondents' views to both of these questions also demonstrated a degree of regional variation, suggesting the importance of context, although the over-arching picture remained reasonably consistent (Table S13.6 Suppl. Material C). From an advisor perspective, the assertion that farmers express strong views regarding the schemes they wish to enter yet demonstrate more limited understanding of scheme-related processes and procedures should not, perhaps, be a surprise—after all one of the reasons farmers employ independent advisors is to help navigate

the complexity of scheme arrangements (Vesterager and Lindegaard, 2012). For example, as one respondent (id: GRFPA2) remarked:

“Most clients have a general idea of which scheme they would like to enter (i.e. ELS or ELS & HLS) and the majority know the basic options available under ELS & HLS. But I have not met one client yet who has read each handbook (160 pages + in each) from cover to cover before I meet them so they do not realise the complexities involved, particularly in HLS, nor do they always realise the restrictions/requirements involved in the management of some of the options. This always surprises me because I would always want to know the details of something I would be committing to for 10 years especially given the financial and management implications.”

In light of this, it is hardly unexpected to find that; over-all, respondents regarded their own advice as either “important” (37.2%) or “very important” (53.2%) for steering their clients towards the most suitable Environmental Stewardship schemes. Perhaps more surprising was the gender split between advisors, with 63.4% of female respondents regarding their advice as “very important” compared to 46.3% of their male counterparts. The reason for this apparent gender-based difference is unclear, though it does suggest that the potential role of gender in shaping professional advice needs further exploration. What we can say, however, is that although we expect respondents to validate their own importance, the evidence seems to suggest that the advice farmers receive can be beneficial. For instance, information from UELS agreement holders points to applicants requiring substantial amounts of advice (CCRI, 2012). Likewise for ELS participants, salient advice has been shown to be essential for enabling farmers to fulfil particular environmental management obligations (Lobley et al., 2010). Thematic analysis of additional open-ended comments not only supports this claim, but further extends it by revealing the wide variety of reasons farm advisors feel their advice is necessary for aiding farmer decision-making (Table S13.7 Suppl. Material C). These reasons include ensuring farmers select the most appropriate type of stewardship scheme and suite of environmental options; as respondent (id: 9V3PZG) clearly stated:

“Farmers are not aware of all the options on offer for each scheme (particularly HLS) and therefore which ones best suit their farm and farming practices. They are also not aware of all the management prescriptions in HLS as these are not in the handbook.”

They also reflect the opinion that farmers recognise the need for technical input and appreciate, trust and prefer independent advice; indeed, as one farm advisor (id: 5GHHXD) put it:

“They [that is the farmer] prefer private advice rather than Natural England sponsored/contracted advisors as they rather pay for impartial advice than possibly receive bias advice.”

There’s also a sense that respondents see their advice has having wider significance for their client’s farming system too; a view advanced by respondent (id: ZJGWWD):

“... other factors that ‘farmers’ need to consider and reflect on including tax implication of dual use and on their single payment, how their business structure is arranged [. . .], the working and timing of operations at the commencement of the agreement, implications particularly under the HLS regarding the reversion of arable to grass under several options that can affect the capital value of their land. . . .”

Many of these views are reinforced by farmers themselves (CCRI, 2012; FERA, 2013a).

13.5.3 Agreement Practicalities

13.5.3.1 Application Complexity

Respondents noted the variation in application completion times (to the point of submission) between ELS, OELS and UELS schemes, on the one hand, and HLS on the other (Table 13.2)—illustrating the different labour demands these schemes have.

Table 13.2 Average time taken by respondents to complete Environmental Stewardship agreements (applications)

ES Scheme	(% of Advisors, <i>n</i> =250)					
	1 to 3 months	4 to 6 months	7 to 9 months	10 to 12 months	1 year +	I don't know
ELS	88.4	4.4	0.0	0.4	0.0	6.8
OELS	57.2	4.8	1.2	0.0	0.0	36.8
U(O)ELS	39.6	4.0	1.6	1.2	0.4	53.2
HLS	20.4	30.4	21.2	8.8	5.6	13.6

The majority of respondents (88.4%) indicated that ELS applications take 1–3 months to complete, while 51.6% thought that completion times of between 4–9 months were more common for HLS. ELS agreements could sometimes be completed within a few days or weeks because of the efficiency of the ELS online application system. According to open-ended survey responses, major thematic factors contributing to the longer completion times of HLS agreement included: the requirement for detailed surveys (i.e. the Farm Environment Plan [FEP]); farm complexity and holding size; the need for meetings with Natural England (as well as Natural England’s capacity and decision-making); draft agreement checks; problems associated with rural land registry mapping, and obtaining historic environment record reports promptly. As one advisor (id: YMEVCN) commented:

“For ELS schemes; the application process is usually very quick— the application forms are available immediately online (or within a week on paper) then they usually take just a few hours to complete. Natural England then process the application to Agreement within 6 to 8 weeks. For HLS; often first contact is made up to or over a year in advance of the start of the Agreement. After this, the FEP must be carried out at the most appropriate time (e.g., mid-late June in species rich hay meadows or March/April on breeding wader ground). It’s the unnecessary to delay the submission of the application until after the FEP has been completed and approved (easily 8 to 12 weeks).It can then be another 8 to 12 weeks from the Agreement to be drawn-up by the HLS Adviser and studied in detail by the applicant before it goes live.”

The complexity of the application process represents a significant issue in the design and effectiveness of HLS, in particular, the task of undertaking and producing the FEP (Defra and Natural England, 2008). Yet 74.4% of respondents agreed that the FEP was “important” or “very important” to the subsequent design of HLS agreements. In addition, 57.6% of respondents indicated that the advice delivered by the FEP grant to farmers and land managers was “effective” or “highly effective”. The centrality of FEP to constructing agreements was confirmed in an HLS monitoring programme (FERA, 2013b; Mountford et al., 2013). Open-ended comments provided a mixture of views with some respondents arguing the value of the FEP, especially its usefulness for: mapping features and selected options; indicating the value of a holding; enabling advisors to advise on marginal areas of the holding and become familiar with the whole farm; and providing the ecological evidence base to support the correct choice of HLS options and indicators. As advocated by respondent (id: RG5J9W) the FEP represents a:

“Very useful exercise for the adviser to get a really good understanding of the farm, what is there and the farming system in place: it gives you the knowledge to sit down with the farmer and talk knowledgeably about their farm and make suggestions, also great to be able to tell them something they don’t know and helps to build a good relationship for developing a quality scheme.”

On the other hand, some took a more critical stance citing that the FEP was too time-consuming to produce and collected too much unnecessary information (FERA, 2013b). For example, as respondent (id: FJMNC3) notes succinctly:

“I completed a FEP for a farm. . .where over 75% of the FEP was irrelevant to the plan that was finally agreed. Natural England knows what it wants to focus on – there is no point in wasting time mapping areas that are irrelevant to a future scheme.”

Still, others claimed that the FEP was not always read in detail by Natural England project officers; did not properly consider the farm business, and was not used following approval – a view articulated by respondent (id: LDHK8D):

“We get the impression that the FEP is not read in detail by many NE project officers, and that the end stewardship agreement is worked up by verbal discussion rather than by close reference to the FEP – hence I rate the FEP as ‘important’, but not ‘very important’.”

To some extent these views leave a question mark over whether efforts to simplify FEP methodology and recording have been as successful as originally envisaged (Defra and Natural England, 2008).

Cost and time were identified as the most common principal constraints relating to the preparation and submission of Environmental Stewardship applications (Table 13.3). Administrative burdens magnify the transaction costs of HLSs, particularly due to labour-

intensive activities such as the FEP. Past surveys of both ELS and UELS agreement holders highlighted that the bureaucracy and complexity of schemes are perceived as daunting and can reduce the number of potential applicants (Cross and Franks, 2007; CCRI, 2012; FERA, 2013a). Reducing scheme complexity and procedures is the most effective means of curtailing transaction costs and increasing AES uptake by farmers (Mettepenningen et al., 2009). In this survey, 62.5% of respondents “somewhat agreed” or “highly agreed” that the HLS process needed to be simplified, while a further 80.2% agreed that farmers and land managers perceived the process of applying for environmental stewardship (in particular HLS) as too time consuming and complicated. This resonates with requests to make future Rural Development Programme funding applications far simpler and manageable through, for example, improving available guidance and streamlining application processes and requirements (Defra, 2013).

13.5.3.2 Advisor Roles

Our analysis indicated that respondent interaction (i.e. communication) differed according to agreement type and the corresponding actor. Respondents noted that all Environmental Stewardship schemes involved significant client interaction. However, respondents indicated that ELS, OELS and U(O)ELS schemes involved less client interaction (64.8%–77.6% “high” to “very high”) compared to HLS schemes (95.5% “high” to “very high”).

In the context of a “knowledge exchange encounter” (Ingram, 2008b) studies of advisor-farmer interactions have previously emphasised the power imbalance in that relationship, characterising the advisor as a prospective exploiter; however, following significant privatisation of advisory services farmer demand is potentially reframing that asymmetric power dynamic (Ingram, 2008b). Respondents suggested that interactions between themselves and their clients proceeded based on mutual decision-making (62.0%), or to a lesser extent according to their own requirements (27.0%). Advisors described their interactions with clients mainly in terms of client need (54.0%); with the frequency of interactions being mainly “sufficient” (40.8%) and/or “above average compared to other professionals” (28.8%).

The emerging narrative suggests that agreement preparation is not simply a series of box ticking exercises and procedures but also a social process. As Ingram (2008b:414) noted, with respect to the relational interaction between farmers and agronomists:

“...the practice the farmer implements is a negotiated or facilitated outcome between agronomists and farmer rather than a rigid prescriptive practice “adopted” by the farmer.”

Reinforcing this view, most respondents (80.0%) described the process of agreement preparation as a negotiation between client needs and their expert advice, with clients being

Table 13.3 Common constraints (C) identified by respondents in relation to Environmental Stewardship application preparation and submission processes

Constraints (Emergent Themes)	C1 (%, n=248)	C2 (%, n=211)	C3 (%, n=145)	C4 (%, n=76)	C5 (%, n=40)
Cost (e.g. for clients)	20.6	15.6	4.8	3.9	2.5
Time (e.g. application preparation and submission)	34.7	16.1	6.9	6.6	2.5
Time and cost	6.1	3.8	1.4	0.0	0.0
Costs versus potential rewards of scheme participation	3.2	1.9	2.1	3.9	0.0
Costs associated with implementing scheme options	0.0	4.3	3.5	6.6	2.5
Suitability and alignment with current farming practices (e.g. extent to which farming practices may need to be altered)	8.5	6.6	4.8	5.3	7.5
Farmer preferences, decision-making and expectations of scheme benefits	2.4	1.9	4.1	1.3	0.0
Farmer willingness to engage in the application process and implement scheme requirements	1.6	7.1	6.2	7.9	17.5
Farmer understanding, knowledge and awareness of schemes	0.4	2.4	4.	2.6	2.5
Scheme capacity to be effective and inherent limitations (e.g. environmental option choices and flexibility)	2.4	7.6	11.7	14.5	10.0
Scheme complexity (e.g. in preparation and implementation)	2.4	4.7	14.5	10.5	10.0
Mapping, visualisation tools, information procurement and surveying	5.6	3.3	8.3	2.6	5.0
Natural England	2.8	8.0	8.9	7.9	10.0
Farm limitations and suitability (e.g. size and features)	3.6	3.8	2.8	13.2	0.0
Bureaucracy (e.g. red tape)	0.8	0.0	0.0	1.3	0.0
Scheme funding availability (e.g. in relation to capital works)	0.4	0.5	2.8	2.6	0.0
Other	4.0	9.0	8.9	7.9	27.5

relatively flexible on the type of scheme (49.2%) and environmental objectives (56.1%) they would adopt. For example, as one advisor (id: D77SEQ) noted:

“Negotiation is the key to success – one needs to be able to understand the farm, the farmer and then the reasons for being invited and work out a scheme that will achieve success for both parties.”

Indeed, the overwhelming majority of advisors (92.0%) indicated that clients demonstrated a high degree of openness towards their advice. This suggests that farmers are a relatively pragmatic group – open to being persuaded on a range of possible recommendations concerning the type and composition of the agreements they enter – and that in this respect advisors can have an important role in guiding farmer-decision making processes. For example, evidence from UELS agreement holders suggests that a fifth of their option uptake is a result of external advice (CCRI, 2012). Likewise, ELS agreement holders have previously stated that advice is:

“. . .very useful for both option choice and option management”(FERA, 2013a:5).

In comparison with their clients, the levels interaction respondents had with Natural England advisors were substantially lower for ELS, OELS and UELS schemes (8.4% to 11.7% very low to low) but remained similar for HLS (82.0% high to very high). The vast majority (92.7%) of advisors indicated that they knew they could contact Natural England for information and advice if and when required. Their reasons for doing so were many and varied (Table 13.4); however, thematic analysis indicated that they were commonly related to issues concerning: clarification and guidance on scheme options, appropriateness and suitability; Natural England requirements/expectations for the farm area and application-related issues, details and administrative checks (e.g. help, advice, specific codes, vendor numbers etc.). Overall, additional comments verified that observed differences in the level of interaction respondents had with clients and Natural England reflected the underlying complexity of the Environmental Stewardship schemes.

13.5.4 Environmental Stewardship Performance

13.5.4.1 Public Goods: Promoting Environmental Objectives In Entry Level Stewardship

Delivering public goods implies a degree of spatial optimization to generate the requisite magnitude and distribution of environmental benefits (Garrod et al., 2012). There has been widespread discussion regarding the effectiveness of ELS option uptake and management activities in relation to realizing environmental benefits and value for money (Defra and Natural England, 2008; Hodge and Reader, 2010; Jones et al., 2010). In particular, arguments for increased integration of options across the landscape have been proposed (Chaplin and Radley, 2010), with research suggesting that farmers would buy into collaborative AES (Emery and Franks, 2012; McKenzie et al., 2013).

Table 13.4 Common reasons (R) respondents identified for contacting Natural England during agreement preparation

Emergent Themes	R1 (%)	R2 (%)	R3 (%)	R4 (%)	R5 (%)
Clarification and guidance on scheme options, appropriateness and suitability	23.2	16.4	13.2	8.5	20.0
Target options, priorities and features for the local area	6.3	6.0	2.8	1.7	-
Mapping-related issues	8.0	4.0	3.5	5.1	-
Natural England requirements/expectations for the farm area	12.7	6.5	2.8	6.8	5.0
Permission, approval and success of submitted application	3.8	2.0	2.1	-	-
Application-related issues, details and administrative checks (e.g. help, advice, specific codes, vendor numbers etc.)	10.5	7.0	4.9	10.2	10.0
Farm-related features, characteristics and aspects for inclusion in applications	9.3	4.5	5.6	1.7	5.0
Information related to previous applications and current on-farm/neighbouring farm schemes in operation (i.e. practices, compatibility etc.)	4.2	5.0	4.2	-	-
Budgetary availability, flexibility and constraints	2.1	2.5	4.2	1.7	-
Clarification of scheme rules and regulations	2.5	1.5	0.7	1.7	-
FEP/FER/HER related matters	2.5	4.0	8.3	3.4	5.0
Time scale of application submission/deadlines	1.7	3.5	5.6	6.8	5.0
Time scale for management options	0.4	-	-	-	-
Time to meet clients	0.4	0.5	-	1.7	5.0
Information and discussions regarding designated sites	1.3	3.0	1.4	5.1	-
Software/technical issues	0.4	0.5	-	1.7	-
Landowner-related matters (e.g. interest, commitment etc.)	0.4	2.0	4.9	5.1	5.0
Eligibility of land for Stewardship schemes and options	3.8	2.0	1.4	-	-
Field data, measurements and information	1.3	1.5	1.4	3.4	-
Agreement start dates	-	1.5	1.4	1.7	5.0
Capital works (i.e. availability, extent, investment and budgetary allocation)	-	9.0	4.9	5.1	15.0
Agreement amendments	-	1.5	1.4	3.4	5.0
Option/Scheme negotiations (e.g. composition of option bundles)	-	2.5	2.8	-	5.0
Payment rates of schemes	-	1.0	0.7	1.7	-
Discussions regarding the specificities of management options (e.g. implementation)	-	2.5	6.9	6.8	-
ELS/HLS procedural issues	-	6.0	5.6	3.4	-
Natural England processing of applications	-	0.5	1.4	1.7	5.0
Land transfer	-	0.5	-	-	-
Interactions with other agencies (e.g. English Heritage)	-	0.5	0.7	-	-
Natural England queries	-	0.5	0.7	-	-
Site visits	-	-	2.0	5.1	-
Implication of scheme implementation for the wider landscape	-	-	-	3.4	-
Other	5.1	2.0	4.9	3.4	5.0
Number of Respondents	237	201	144	59	20

ELS schemes are designed to achieve five broad environmental objectives: wildlife conservation (WC); landscape quality & character (LQ & C—particularly in relation to water); protection of the historic environment (PHE); natural resource conservation (NRC, particularly in relation to soil) and climate mitigation and adaptation (CM & A). We asked respondents to identify the environmental objectives most frequently met by the agreements they have been involved with (Table 13.5). Clearly, there are some that appear to be being met more than others. This suggests differences in the way agreements fulfil specific objectives and provide a comprehensive range of environmental benefits (Radley, 2013).

Table 13.5 Environmental objectives fulfilled by Environmental Stewardship agreements

Environmental objectives	Presence in stewardship application (% , n=250)	Commonly met environmental objectives	(%)
WC	94.0	WC, LQ & C, PHE	48.0
LQ & C	87.2	WC, LQ & C, NRC	27.6
PHE	67.2	WC, PHE, NRC	11.2
NRC	48.8	LQ & C, PHE, NRC	4.0
CM & A	4.8	-	-

For example, 94.0% of respondents indicated that most ELS schemes in which they have been involved fulfil WC objectives, whereas only 4.8% identified CM & A as being similarly fulfilled. This trend is also observed at the regional scale. It appears most applications (48.0%) focus on WC, LQ & C and PHE or WC, LQ & C and NRC (27.6%) environmental objectives. This supports previous agreement holder surveys indicating that “farmland wildlife”, “wildlife conservation benefits” and “resource protection” were important environmental issues affecting agricultural land (FERA, 2013a). To some extent variations in meeting particular environmental objectives may be a reflection of the outcomes of directed funding streams. For example, significant investments (approximately £200 million) have been channelled towards historic environment conservation activities since 2005 (Natural England, 2014).

They may also be a consequence of sectoral variation in agreement and option uptake (Defra and Natural England, 2008). It is certainly the case that most respondents (87.0%) regard their role as achieving the “best” agreement for their client. What “best” actually refers to needs further exploration; however, in this context there is the potential for farmer-advisor co-alignment: either to increase or decrease the environmental ambition of an agreement. One argument raised is that when advisors want to avoid losing the trust of farmers or negatively impacting farmer incomes they may focus on bolstering current on-farm agricultural practices (e.g. Sutherland et al., 2013). For instance, as respondent (id: 8RAV9H) stated:

“I see my role as trying to fit the scheme with the farming practices/available labour/capabilities & experience of the applicant. My aim is to enhance them all, and to ensure the success of them all.”

The majority of respondents (51.4%) stated that 50% or more of the applications they had been involved with included specific environmental objectives based on their priority status, implying that advisors strongly argued for the environmental component and/or that those farmers regarded this as important. As one ardent “green” thinking advisor (id: TGGE35) commented:

“I try to only work with those people that are truly engaged with the principles of Stewardship. If they are constantly trying to find ways out of doing what is necessary I explain that I can’t help them and leave.”

However, for a significant minority (48.59%) this was not the case, suggesting that advisors either failed to effectively promote and push the environmental argument or, that even if a forceful argument was presented; farmers considered other factors to be of greater bearing on their decision-making. A sentiment supported by respondent (id: 725VQ8) who acknowledge that:

“It depends if you are tuned into each other’s objectives. I do this with the aim of improving and promoting good environ-mentally sound farming practice – which has been in serious decline since the 80s. If my clients are similarly inclined, my advice is important and relevant. If they are not of the same opinion, my advice/persuasion falls on deaf ears and is of little importance.”

Regarding the latter, although we suggested in Section Advisor Roles that advisors were well placed to influence farmer decision-making – given the level of openness to advice they purportedly demonstrated – respondents identified a range of other significant farm-related factors that strongly affected farmers’ decision-making rationale to include/exclude specific management options. The most commonly identified reasons advisors noted were those connected to farm system compatibility. For example, environmental objectives are easy to implement (88.4%); is/are an extension of current farm practices (84.3%), do not significantly impact on the day to day farm routine (84.3%), provide a higher points value (67.1%) and requires few man hours (49.4%). Advisors seem to be indicating that farmers are predisposed towards selecting options that do not significantly influence their farm business (FERA, 2013a), bolstering the idea that the primary predictors of option uptake are agricultural related factors (Hodge and Reader,2010). Moreover, it supports the generally articulated view of AES that there is a:

“ . . .disjuncture between the policy’s supposedly overarching environmental rationale and its realisation in practice through the actions and behaviours of land managers” (Juntti and Potter,2002:216).

On the other hand, 80.0% of respondents also agreed that in preparing ELS and HLS agreements they needed to balance both client and Natural England needs. However, overall, opinion was split as to whether a stewardship agreement reflected the preferences of their clients or the priorities of Natural England. Nevertheless, when viewed through the lens of gender and expertise differences did emerge. For example, a higher percent-age of male respondents (46.3%) compared to female respondents (30.7%) agreed that agreements were more reflective of client preferences than Natural England. In fact; on this issue, the distribution of responses between male and female advisors was significantly different ($H(2) = 7.56, df = 1, p = 0.006$). However, when considering advisor expertise, those with wider experience of Environmental Stewardship schemes were more inclined to disagree with this position (44.7–54.3%) compared to those with less expertise (30.0–33.8%).

Importantly, 53.2% of respondents acknowledged that there was an inherent conflict between client needs and Natural England priorities, a pattern similarly observed if disaggregated by gender and expertise. This is significant, because it points toward an inherent tension in how agreements meet their statutory obligations whilst acknowledging that, to some degree, they must also co-align with farmer and land manager needs. Although it would require further investigation, it is conceivable that these various decision-making trade-offs may contribute to the skewed pattern of option uptake observed elsewhere (e.g. Boatman et al., 2007; Jones et al., 2010; Radley, 2013).

To an extent our results add weight to the contention that the voluntary nature of Environmental Stewardship schemes, and their option menus, predispose farmers to undertake only those environmental management activities that would have occurred in their absence (i.e. a lack of additionality); specifically, by subscribing to those practices that fit easily into existing farm activities – leading to the possibility of adverse selection in option choices where the envisaged level of environmental benefits cannot be guaranteed (Hodge and Reader, 2010). For example, research has found that:

“...farmers thought that 61% of features in ELS option would be managed the same if they had not gone into ELS” (FERA,2013a:9).

And furthermore, that between 21% and 52% of management work, in financial terms, would have occurred ‘in the absence of the scheme’ (Courtney et al., 2013).

13.5.4.2 Alteration Of Agreements: The Case Of HLS

A profusion of advisory services stretching across public and private sectors presents both challenges and opportunities: a pluralistic resource of diverse competencies enriching advice and extending the “agricultural knowledge system”; yet also providing an opportunity for fragmentation, duplication, and incoherence in policy and delivery; encouraging greater competition between service providers and leading to confusion amongst farmers (Juntti and Potter, 2002; Sutherland et al., 2013). How this plays out in practice can significantly influence the environmental performance of agreements (Sutherland et al., 2013). It is reasonable to posit that there are different points in the Environmental Stewardship process where the content of agreements can be revised. First, there are those opportunities that arise during agreement preparation, specifically, in relation to the dialogue between farmers, private advisors, and Natural England officers. Second, alterations can be made to applications post-submission when they are being reviewed by Natural England (here we are particularly referring to HLS applications). At this juncture, it is possible that agreements may be altered to favour Natural England’s environmental agenda and priorities—potentially shifting the pre-submission content away from more farmer-centric interests.

With this in mind, we asked respondents with experience of HLS agreements (n = 212) to comment on the extent to which Natural England advisors made alterations to the environmental content of HLS applications between the original and final approved application. Although some respondents (17.9%) specified that Natural England advisor decisions did not lead to any alterations in environmental content, the majority of respondents indicated that the environmental management composition of applications was either ‘somewhat different’ (40.1%) or ‘moderately different’ (38.7%). Notably, from the standpoint of transparency, a majority of respondents (56.3%) declared that Natural England “very often” or “always” informed them about changes that had been made:

“I have without exception found the Natural England advisors very helpful and communicative at all stages of the long process it takes to put an application into practice.” (Respondent id: WSGZAM)

Listing reasons as to why Natural England advisors altered the environmental content of submitted HLS applications, respondents expressed a range of views with several themes emerging (Table 13.6). The most frequently cited themes related to changes in Natural England’s environmental agenda (23.7%), financial and cost constraints (20.3%), and the calibre of Natural England advisors (11.6%). It is possible, of course, that Natural England modifications to HLS applications do improve their environmental content:

“I work very closely with Natural England and know advisors personally. Although they work to targets they seem to aim for the best ‘wildlife’ options overall.” (Respondent id: HT2RJQ)

Furthermore, a previous assessment of 174HLS agreements indicated that they were generally ‘well designed’ with 80%of agreements deemed likely to be effective in achieving most outcomes: in each of these cases it is likely that there was considerable Natural England oversight in the delivery of these schemes (Mountford et al., 2013). Conversely, extensive prescriptive revisions to AES have been shown to produce ‘excessive uniformity’ in habitat management which can undermine biodiversity as well as negatively affect farmer participation (Radley, 2013). The impact on farmers may be potentially quite severe, as respondent (id: 2NUE5U) strongly argued:

“Often the client has already committed to a huge expense to undertake the scheme that they feel they have to accommodate the changes in order to achieve a return. Many are bullied into the changes. This is ineffective as it doesn’t take into account if the changes can be managed effectively which can lead to failure in the long term.”

Table 13.6 Emergent themes describing the possible reasons (R) for HLS application modifications

Emergent themes	R1 (%, n=187)	R2 (%, n=116)	R3 (%, n=64)	R4 (%, n=13)	Importance Across Reasons (%)
Changes in Natural England targets and priorities for HLS	26.7	22.4	21.8	0.0	23.7
Budgetary/cost/financial constraints	22.9	18.9	12.5	30.8	20.3
Natural England advisor decision-making, viewpoint and knowledge	9.6	14.7	12.5	7.7	11.6
Other	8.7	12.9	14.1	15.4	11.1
Environmental option suitability	6.9	2.6	7.8	0.0	5.5
Farmers changed their minds	2.7	4.3	9.4	0.0	4.2
Conflict between HLS options and other scheme objectives	4.3	3.4	4.7	0.0	3.9
Farmer and NE negotiated changes	2.1	5.2	4.7	7.7	3.7
HLS prescriptions too burdensome	1.6	5.2	4.7	15.4	3.7
Too little evidence to justify & differences of opinion concerning option inclusion	1.6	4.3	4.7	7.7	3.2
Environmental option eligibility	4.2	0.9	0.0	0.0	2.4
Application mistakes	2.1	2.6	1.6	0.0	2.1
Rarely are changes/adjustments made prior to submission	4.3	0.0	0.0	0.0	2.1
Too many similar options included in HLS applications	1.6	0.9	0.0	7.7	1.3
Inclusion of too many options (overly ambitious application)	0.5	1.7	1.6	7.7	1.3

The majority of respondents (87.9%) expressed the opinion that they should be included in the discussions leading to changes in finalized HLS agreements. Similarly, 82.2% of respondents dis-agreed with the notion that it was appropriate for Natural England to make modifications to HLS applications without their input (Table S13.8 Suppl. Material C). For some advisors involvement is a question of process and procedure:

“If Natural England discuss with advisor then the advisor can understand the rationale and will be able to use advice on other applications. Sometimes the discussions with Natural England can lead to them reverting to original option.” (Respondent id: 3US4XH)

Yet, for other respondents, it is about mitigating inefficiencies and the potential straining of relations:

“I’ve had farmers very upset not understanding the changes and then creating a silly 3 way discussion where it is not easy to know exactly what has been said. It makes sense to have a proper open communication system. The farmer has employed me, why suddenly change to excluding the agent. This varies according to the NE officer.” (Respondent id: TFEPJJ)

Nevertheless, Natural England is the statutory authority in charge of implementing Environmental Stewardship and as such oversees the final decisions regarding individual

applications—they represent the legitimate institutional authority. It is interesting, therefore, that advisors should feel entitled to have a greater influence over Natural England decision-making. The reasons behind this require further investigation. Linked to these comments, 60.8% of respondents viewed Natural England’s modifications of HLS agreements as not made in the interests of their clients. In fact, 68.8% of respondents thought that revisions to HLS applications were made to favour Natural England interests (Table S13.8 Suppl. Material C). This point was forcefully conveyed by one advisor (id: MLZEYI) who commented that:

“In my experience they are ALWAYS made to save money and not in the best interests of the client or the environment and, if truth be known, not in the best interests of Natural England either in the long term.”

Equally, however, some advisors noted the difficult position Natural England advisors faced:

“Advisors are in the awkward position of trying to achieve Government objectives, habitat and species protection and encouraging land owners to enter the schemes. Advisors work hard to get as much for the money as possible and farmers want as much money as possible.” (Respondent id: QK59MK)

13.5.4.3 Payments, Costs And Income

Financial incentives are a central tool in steering farmers’ private interests to provide particular public goods, but to achieve their desired outcome payment levels must account for the opportunity and transaction costs that farmers might incur on entering voluntary schemes (Cooper et al., 2009). Stewardship schemes provide different standard payment amounts. Little consensus emerged in our survey regarding respondents’ perspectives on whether clients were satisfied with ELS payments (Table S13.9 Suppl. Material C). There was a sense from some open-ended responses that farmer expectations remain too high, with an attitude of ‘maximum gain and minimal cost/impact’. For example, as one advisor remarked:

“They [i.e. farmers] would always be happier with more!”(Respondent id: 7KMC97).

Other advisors observed that payments were sufficient so long as other farming business opportunities were not compromised:

“The £30/ha for ELS is ok provided there is not serious competition for more productive uses.” (Respondent id: 4DQMRP).

On the whole there was more general agreement among advisors that clients were satisfied with the standard payment amounts for OELS, UELS and HLS (Table S13.9 Suppl. Material). Having said that there were also clear differences of opinion, with one farm advisor suggesting that farmers might be overpaid:

“HLS payments are too high for the land based options in the uplands. Extremely large annual payments for delivering very little benefit.” (Respondent id: LGQECT)

Whilst in stark contrast, one respondent commented that the present level of HLS payments questions the scheme’s viability:

“HLS payments in recent years have been ‘trimmed’ down to the state that with some schemes the annual return has been so low as to make the scheme non-viable for the farmers.” (Respondent id: 7KDH6Y).

However, by and large, with regards to payment satisfaction our results seem to accord with the 66% of UELS agreement holders that considered scheme payments to be “generous” or “sufficient” (CCRI,2012). Ultimately, as respondent (id: P8XVWR) indicates:

“If they [i.e. farmers] were not happy they would not enter an agreement.”

Some studies claim farmers view Environmental Stewardship as an income top-up and stabilizer (a form of income security), particularly in instances where the farm business is vulnerable (Mills, 2012). In others, it has been argued that ELS entry may incur modest costs rather than adding to income (Udagawa et al., 2014). Our results suggest that respondents straddle both of these perspectives, as advisors were generally split over whether payments afforded their clients an adequate income stream or not (Table S13.9 Suppl. Material C), although one advisor did comment that:

“Some. . .farmers look on stewardship payments as income, but I do point out to them that they are required to work for their money – it’s not just a free hand-out.” (Respondent id: LDHK8D)

However, when specifically reflecting upon the connection between payments and costs (e.g. labour and materials), 56.6% of advisors suggested that clients considered payments insufficient to adequately cover changing input costs versus only 21.1% that thought the contrary. This view was echoed in open-ended comments, with some advisors stating payments did not reflect recent ‘escalations in commodity prices’ or ‘cost increases due to inflation’, for example:

“Payments should be index linked. A payment of £30/ha might have been acceptable 5 years ago, but costs have gone up a lot yet scheme incomes have remained the same.” (Respondent id: 4A29YS)

Others emphasised that ELS payments were “not cost effective for arable farmers” and that “crop values and greening measures will make it harder to encourage renewals”, as one advisor outlined:

“Payments are now seen as too low particularly in intensive arable areas where a typical comment is that ‘it’s hardly worth the hassle’.” (Respondent id: 6EATTB)

This is supported by evidence indicating that cereal incomes may be unduly affected by entering ELS schemes (Udagawa et al., 2014), as a respondent noted:

“The £30/ha on ELS was set at a time of low cereal prices and was to compensate for income forgone – in light of much higher prices for crops this aspect needs re-visiting.” (Respondent id: 4NGVTB).

While this may be the case other studies have suggested that the percentage of farmers who regard payments as sufficient to cover their costs has grown since 2005 (FERA, 2013a). Adopting a regional perspective revealed statistically significant heterogeneities in respondents’ views regarding scheme payment levels as well as income and input costs (Table S13.10, Figures S13.1 and S13.2 Suppl. Material C). These regional patterns likely reflect individual respondent experiences of specific farm-level socio-economic characteristics as well as the distinctive rural and wider economic circumstances encountered in these locations (see Farm Business Survey)². The implication seems to be that payment levels ought to account for these differences, and thus better reflect regional level conditions and farm business circumstances. Over-all, advisors’ comments suggest that payment levels are a real issue for farmers, in particular, whether the costs incurred actually make entry level schemes unsustainable in the long-term (Udagawa et al., 2014).

13.5.4.4 Compliance: Penalties And Sanctions

The successful provision of public goods relies on individuals complying with contractual arrangements: this requires agreements to have monitoring and conditionality elements (Hejnowicz et al., 2014). We queried respondents regarding elements of conditionality (i.e. sanctions and penalties) and found that, although a majority, only 51.4% agreed that their clients understood the extent of the penalties that may be applied should they fail to fully comply with agreements (Table S13.9 Suppl. Material C). Notably, a sizeable minority of advisors (33.9%) thought the opposite. Additional commentary raised a number of issues; for example, some advisors alleged that farmers and land managers, although aware penalties could be applied in cases of non-compliance, were often ignorant of the scope sanctions could take: partly as a consequence of the rules being over-complicated and poorly explained:

“Farmers are often oblivious to the penalties that would be applied for non-compliance. However, there are very few which would knowingly flout the rules and often farmers are found to be non-compliant only as a result of them not being aware of all of the requirements of the complicated scheme (particularly in the case of HLS).” (Respondent id: YMEVCN)

On closer inspection this observation does seem rather puzzling, surely the expectation is that part of the service advisors provide includes spelling out the likely repercussions of non-compliance? If an explanatory deficit exists then surely this rests, in part, on the shoulders of farm advisors themselves? Certainly some advisors made the effort to inform their clients

of non-compliance related issues but hinted at the fact that, in their opinion, “not . . . many other advisors do this” and also with regards to farmers “it’s amazing how quickly some of them forget!”.

Picking up on the latter point, ignorance and deliberate avoidance have been identified as issues for farmers failing to meet agreement prescriptions, a reality that would seem to support the contention that agreement holders ought to have better access to training in order for them to more closely adhere to agreement conditions (FERA, 2013a). The provision of training has been shown to positively influence farmer behaviour and management activities (Jones et al., 2013). Training, however, may not be sufficient because, as some respondents disclosed, a number of their clients took a fairly relaxed, even recalcitrant, stance towards compliance issues due to poor monitoring and enforcement:

“Past monitoring of schemes has been poor – I think some farmers believe they are unlikely to be caught breaching ELS/HLS options and may therefore continue existing practices (e.g., supplementary feeding, applying fertilizer) where the options actually forbid this.” (Respondent id: QKUYGC)

At the opposite end of the spectrum, however, a number of advisors commented that punitive sanctions actually dissuaded individuals from entering schemes as well as affecting the content of agreements:

“The issue of the current punitive level of sanctions has put off some farmers from going into schemes and has certainly reduced an agreements ‘ambition’.” (Respondent id: E9DAB2)

Indeed, some respondents suggested that farmers regarded sanctions and penalties as the thin-end-of-the-wedge:

“Penalties imposed as a result of inspection are often seen as pedantic and penny pinching for what appears to be minor infringements.” (Respondent id: 6EATTB)

Others went further, proposing that sanctions and penalties were inappropriate, poorly formulated and incorrectly realised:

“The main problems with the sanctions and penalties are that they come across as being draconian and in many cases incorrect and based on incorrect information either supplied by the inspector or interpreted by the administration staff.” (Respondent id: 97H33F)

and furthermore,

“Rules about penalties are over-complicated and poorly explained. No allowance for intent – issues with extenuating or mitigating circumstances are punished as severely as deliberate non-compliance.” (Respondent id: 36E4W9)

Reflecting this view, only 19.9% of respondents agreed that their clients regarded such sanctions as reasonable. Clearly, in the view of advisors, farmers regard the potential sanctions

imposed by Natural England as disproportionate, with some suggesting that the fault lies, in part, in the ‘uncertainty’ and ‘inconsistency’ with which Natural England tackle these issues. Perhaps one avenue to help address these compliance issues would be to adopt performance-based payment schemes: here payments are directly linked to the maximization of environmental benefits (Schomers and Matzdorf, 2013). Examples of agricultural results-based payment programmes throughout Europe indicate that they can be successful in generating beneficial ecological, economic and social outcomes (Burton and Schwarz, 2013), and research in England also suggests that payment by results is positively perceived by farmers (Schroeder et al., 2013).

13.5.5 .Moving Forwards

A new round of CAP reforms (2014–2020) has been introduced to provide a more streamlined, targeted and greener approach to agricultural production and the rural environment (European Commission, 2014). The degree of inter-pillar transfer from Pillar 1 to Pillar 2 in England, as a consequence, is set to increase from 9% to 15% over the next six years; and Defra has committed to allocate 87% of rural development funds to the environment (Defra, 2014; Natural England, 2014). These revisions will see Environmental Stewardship programmes eventually phased out and replaced by the New Environmental Land Management Scheme or NELMS for short (Natural England 2013). Although NELMS will be implemented in early 2016, current Environmental Stewardship agreement holders will still be delivering management under the ‘older’ system. The opportunity therefore exists for further suggestions for refinements to feed into the design and operationalization of NELMS based on lessons learned under the Environmental Stewardship programme.

Respondents were allowed to put forward four recommendations that could enhance Environmental Stewardship uptake and implementation (Table 13.7). Many recommendations were suggested and thematic analysis identified several broad themes, the most common of which centred on: reorganising Environmental Stewardship delivery (22.5%); simplifying scheme processes and procedures (19.5%); providing more information regarding environmental option management and implementation (14.3%), and improving the targeting of schemes (9.6%). Indeed, the views of respondents also echo those of Lastra-Bravo et al. (2015) who highlight the importance of “institutional design” and “stable policy” for aiding farmer engagement with, and adoption of, future agri-environmental schemes.

Overall, recommendations provided by respondents connected with the themes central to the NELMS programme (e.g. “delivering outcomes at a landscape scale”; “a participative and collaborative approach”; “outcome focused performance assessment”; “flexible and adaptable”; “locally tailored advice and training”; and “simplification”), as well as

reflecting research focusing on the design principles of future agri-environment schemes (FERA, 2013b). Respondent suggestions also aligned with those expressed in recent Defra consultations on CAP reform (Defra, 2014), which included the need to address ‘the risk of complexity’ and the importance of ‘targeting as a means to direct option choice’.

13.6 Final Remarks

Our survey has provided an important exploratory assessment of the English experience of Environmental Stewardship viewed through the lens of independent farm advisors: the views of whom are under-represented in the existing literature. In this regard, we have both strengthened and expanded upon the current literature concerning AES, and Environmental Stewardship in particular, by highlighting a broad range of farm advisor views that have not previously been, in this format at least, addressed or assessed.

Our findings indicate that farm advisors exhibit a wide range of expertise and experience, and that in some cases regionality, expertise and gender can play a part in influencing farm advisor perspectives and experience of Environmental Stewardship schemes. We have also shed some light on the facilitating role farm advisor advice plays in the preparation of Environmental Stewardship agreements, and shown that farm advisors regard what we term the “knowledge-exchange encounter” as a crucial aspect of this facilitative function.

Initial findings suggest that farm advisors face a difficult balancing act: preparing agreements based around the needs of their clients on the one hand, whilst on the other, ensuring submitted agreements are not at odds with Natural England requirements. In the view of most respondents, there is an inherent tension between farmer and Natural England objectives. This would seem to connect to the finding that, in the eyes of respondents, although farmers display a broad range of extrinsic and intrinsic motivations for engaging in Environmental Stewardship agreements, they are primarily motivated by financially-oriented reasons. And, in addition, they demonstrate a proclivity to make decisions about the environmental management content of agreements based on how closely this aligns with current on-farm practices and the farm business more generally.

Equally, given that farmers respect, and are open to, the advice they externally contract, farm advisors may have a decisive role in guiding farmer decision-making processes. Advisors, in this sense, potentially occupy an influential “soft power” position. This has the prospect of going in one of two directions, either: advisors can encourage farmers to undertake environmentally ambitious agreements that build on intrinsic ‘green’ motivations or, taking the opposite stance, draw on farmers’ extrinsic motivations and produce agreements requiring

minimal changes to on-farm practices primarily benefiting existing farm business arrangements.

Table 13.7 Recommendations (R) for improvements in Environmental Stewardship delivery and implementation

Emergent themes	R1 (%, n=168)	R2 (%, n=92)	R3 (%, n=56)	R4 (%, n=27)	Overall popularity (%)
Reorganization of Environmental Stewardship delivery (e.g. alternative ways to improve on-the-farm provision)	25.6	16.3	28.6	11.1	22.5
Simplification of Environmental Stewardship processes (e.g. streamline HLS application requirements)	26.2	17.4	7.1	11.1	19.5
Environmental Stewardship scheme options (e.g. degree of flexibility in option implementation)	10.1	21.7	17.9	7.4	14.3
Targeting Environmental Stewardship (e.g. tailoring to meet local environmental needs)	8.3	13.0	8.9	7.4	9.6
Natural England and Natural England Advisors (e.g. knowledge; interaction with farmers)	8.9	9.8	3.6	11.1	8.5
Environmental Stewardship payments (e.g. reassess payments to reflect environmental option requirements and changing labour and input costs)	9.5	6.5	3.6	0.0	7.0
Other	5.4	4.4	7.1	18.5	6.4
Consultation, dialogue and support (e.g. contact with industry; ES support provision for farmers)	4.2	2.2	5.4	11.1	4.4
Agent/advisor training (e.g. ensuring agents are suitability qualified and knowledgeable)	0.6	2.2	1.8	14.8	2.3
Mapping (e.g. improve online mapping tools)	0.6	3.3	1.8	3.7	1.8
Farmer focused (e.g. consideration of farmer viewpoints and operational constraints)	0.0	0.0	8.9	3.7	1.8
Environmental Stewardship scheme reinvention (e.g. establishing and dissolving schemes too frequently)	0.6	0.0	3.6	0.0	0.9
Farmer knowledge (e.g. improving knowledge of ES scheme management)	0.0	2.2	0.0	0.0	0.6
Scheme complementarity (e.g. reduce conflicts between different but co-implemented/managed environmental schemes)	0.0	1.1	1.8	0.0	0.6

Allied to these discussions are the observations that many respondents noted, namely, with particular reference to the HLS tier of Environmental Stewardship, that their clients found HLS application processes too burdensome and overly complex, a view they also concurred with, pointing to the need to simplify and streamline the system. Respondents suggested that the complexity of programme arrangements and processes may function as a barrier for farmers and land managers: potentially acting as a contributing factor to decrease the environmental ambition of agreements and increase the likelihood that the management practices incorporated into agreements mirror those of the 'farm system'. Perhaps it is the very labour intensive nature of producing an HLS agreement that provides one explanatory factor for the observation that respondents felt they ought (and were entitled) to be involved in the HLS revision process, a process in which they have no legitimate authority to intervene.

Overall, the narrative we have presented suggests that the knowledge-exchange encounter is not a simple straight forward interaction. Rather, advisers' opinions and comments suggest that there are often important tensions between the goals and agendas of the principal agents involved in preparing, implementing and delivering Environmental Stewardship.

In addition, our survey has also highlighted a feeling among advisors that, for farmers, scheme payments present a real issue, particularly because management interventions can have a considerable impact on overall farm income and; furthermore, may not adequately account for all the costs farmers incur and appropriately reflect regional socio-economic differences. Farm advisors also indicated, and to some extent this may be linked to the issues of scheme complexity we have previously discussed, that there are challenges associated with matters of scheme compliance and sanctions. Among respondents there was a general feeling that a significant minority of their clients were not fully aware of scheme-related penalties and sanctions, and in some cases adopted a fairly relaxed stance towards non-compliance. Such matters pose real issues for how enforcement works and the environmental management effectiveness of schemes, but also, raise important issues regarding proper informed consent (i.e. that agreement holders should be fully informed about, and understand, their contractual obligations at the outset of the process).

Looking ahead, to ensure the success of future AES programmes, teasing out the issues we have started to shed light on in this paper will be necessary, in particular: focusing in more depth on the relationships and tensions existing between farmers, farm advisors and Natural England. This would seem to be the most fertile ground for uncovering those factors determining the overall content, implementation and performance of AES agreements. Ultimately, if NELMS are to fruitfully replace and build on the successes of Environmental Stewardship, as well as avoid any of their pitfalls, then the issues raised by farm advisors in this

survey will be important food for thought in developing effective schemes that work in practice. Specifically, by acknowledging the importance of the different agendas and dialogues occurring between farmers, private advisors and Natural England; ensuring that participation and environmental ambition pays and contracts are properly enforced; and that the operation and implementation of schemes is simple, straightforward, easy to put into practice, accommodates farm production and does not alienate potential participants.

Notes

1. www.naturalengland.org.uk/farming/funding/es/agents/register.aspx
2. <http://www.farmbusinesssurvey.co.uk/regional/>

Part 7: A Proposal For A Landscape Approach To Future Garden Management

Improving the condition of our Post-Edenic world requires a recognition of, and also greater integration of, the multi-dimensionality of the social-ecological complexities of our Human Genesis project. If we are, in some sense, to reverse our Post-Lapsarian coupled environmental and social transgressions then we need ways of acknowledging, in an explicit manner, the criticality of the social, cultural, political, economic and institutional processes and dimensions that mould the way we operate in the world. In particular, we need to consider how these aspects are affected by environmental conditions but also, at the same time, feedback to shape the environment and influence human-nature relationships. We argue locating ecosystem services within a landscape framing and approach provides a route to achieving this.

Neil Armstrong's inspirational phrase "One small step for man, one giant leap for mankind", uttered as he stepped out onto the lunar surface captured completely the spirit and imagination of a revolutionary decade. Yet, mankind's "giant leap" was not the miraculous technological advancements that had led to such a supreme achievement – as significant as they were – rather, it was the recognition that, gazing back at our home planet across the emptiness of space, we understood for the first time its fragility and our shared heritage and connection with each other:

“Finally it shrank to the size of a marble, the most beautiful marble you can imagine. That beautiful, warm, living object looked so fragile, so delicate, that if you touched it with a finger it would crumble and fall apart.” (James Irwin, American Astronaut)

Following the birth of Fuller's 'spaceship earth' and our entrance into the Anthropocene, a world increasingly under pressure from human influence, we have begun to recognise the globally transformative effects our actions have on the Earth System (Adams, 2009; Steffen et al. 2011; Hughes et al. 2013; Vince, 2014).

Hubris has altered our perspective and we are aware, now more than ever, of the interdependence, across all scales, of environment and development issues and the connections between human wellbeing and the sustainability of planetary systems (Rockström et al., 2009; Wilkinson and Pickett, 2010; Griggs et al., 2013; Iverson et al., 2014). Many of

which have been outlined in earlier Chapters. Throughout, the mantra of sustainability has acted as a rallying call: shaping international development, conservation and environmental policies and conversations (Adams, 2004; 2009; Cornell, 2013).

Contemporary debates in sustainability have frequently been framed in terms of the mainstreaming of ecosystem services (Abson et al., 2014). Viewed in this manner sustainability is often characterised in relation to the maintenance, restoration, substitutability and depletion of ‘natural capital’ – increasingly regarded as the fundamental bedrock of ecosystem service provision, economic growth and social flourishing (Ang and Van Passel, 2012; Costanza et al., 2012; Hails and Ormerod, 2013). Here, securing an accessible, equitable and consistent supply of ecosystem services, which acknowledges the finite qualities of natural resources as well as the embeddedness of human society and economy within an ecological life-support system, is considered a prerequisite for human prosperity (de Groot et al., 2010; Costanza et al., 2012).

The human-centric presentation of the ecosystem services paradigm, particularly its economic appraisal of natural capital, one of its key features, is commonly regarded as responsible for advancing environmental concerns in political circles (Gómez-Baggethun et al., 2010; Villamagna et al., 2013). Yet, in some quarters, this particular formulation has met with widespread criticism (e.g. Vira and Adams, 2009; Norgaard, 2010; Spash and Aslaken, 2015). And even though many of these criticisms have been challenged with reasonably substantiated counterarguments (e.g. Schröter et al., 2014) important questions still remain to be answered, some of which have been framed in ways that further undermine the concept (Villamagna et al., 2013).

Consequently there have been calls for more holistic approaches to convey the ecosystem services paradigm – extending beyond the more ‘reductionist’ frameworks to emphasise the social-ecological complexities of human-nature relationships (e.g. Ang and Van Passel, 2012). This reformist agenda does not consider the ecosystem services concept as redundant, far from it; but rather; it expresses the need for it to become more fully integrated into everyday contexts and conversations (Villamagna et al., 2013). The message of these reformist arguments is that current ecosystem services frameworks are not fit for purpose, primarily; because of the disjuncture between how the concept translates into a chiefly landscape management context (Setten et al., 2012).

In this article we extend the arguments made by Setten et al., (2012), augmented by the work of Sayer et al., (2013) and Freeman et al., (2015), and build on their core message that landscape, at a both a function and conceptual level, acts as a cross-cutting theme to ground ES frameworks – anchoring their conceptual abstractness into a comprehensible form and

rendering them more susceptible to decision-making processes: improving both their practical and policy relevance and capacity to accommodate complex socio-cultural processes. In other words, landscape affords a much stronger and tangible social-ecological scaffold upon which credible ES frameworks can be developed.

Building on this foundation, in Chapter 14 we explore further, and in much greater depth, the notion and importance of landscape - discussing the multiple meanings attributed to landscape and in so doing touch upon issues of, for example, personal identity, psychology and culture, as well as politics, economics and power in relation to social justice and inequalities and natural resource exploitation. We show landscape to be highly complex in its production, development and perception and to explicitly recognise many issues that are not front-and-centre in current ecosystem service framings. In Chapter 15 we develop our landscape approach framework arguing that it progresses beyond the ecosystem-based approach outlined in the Malawi Principles and provides a far richer understanding of social-ecological systems and human-nature relations. Finally in Chapter 16 we fully develop our landscape approach detailing its implications for ecosystem services, arguing that this transformation in how ecosystem services is understood has important conceptual, practical and communication consequences for ES framings of human-nature relations and environmental management decision-making processes.

Chapter 14: Landscape: Meaning, Narrative And Unification

‘Once again do I behold these steep and lofty cliffs, which on a wild secluded scene impress thoughts of more deep seclusion; and connect the landscape with the quiet of the sky. The day is come when I again repose here, under this dark sycamore, and view these plots of cottage-ground, these orchard-tufts, which, at this season, with their unripe fruits, are clad in one green hue, and lose themselves among the woods and copses, nor disturb the wild green landscape.’ (Extract from ‘Lines Written A Few Miles Above Tintern Abbey’, *The Lyrical Ballads*, William Wordsworth, 1798)

‘It is said that goddesses (of rivers, forests, soils, and all things organic) are fecund, generous, never denying, ever demure and obedient, even when depleted, diverted, blocked, eviscerated, broken, burned and poisoned, for they are all Shakti, the female principle of divine energy.’ (Extract from ‘An Elegy for a Wounded Planet and a Plea for its Protection’, Suprabha Seshan, 2016)

14. 1 Landscape Framings

14.1.1 Interpretations Of Landscape

“Landscape” is so ubiquitous in everyday parlance; indeed, it has such an immediate cultural resonance, that its commonplace usage deceptively masks its underlying complexity. Like all descriptors ‘landscape’ has evolved through time, gaining new attributions and additional meanings down the generations; however, its formal origins lie in its Germanic Medieval inheritance (Tress and Tress, 2001). For example, the German word “Landschaft” where “schaft” or “schaffen” means “to make” brought forth the notion of something that is partially human created, while the early Dutch word “lantscap” conveyed the idea of “a land region or environment”, a similar meaning echoed by the Old English word “landscape” (Tress and Tress, 2001; Antrop, 2013). Clearly, from the outset, “landscape” was used to distil the social-physical expression of the connections between humans and nature: a set of relationships that have resulted in the contemporary notion of a complex multi-dimensional construct that draws-on “subjective observation and experience” and comprising “perceptive, aesthetic, artistic and existential meaning” (Antrop, 2013).

The criticality of landscape to the problems and solutions presented by current environmental management and conservation challenges is increasingly recognised (Lindenmayer et al., 2008). In Europe, for example, the formal recognition of the importance of landscape, not just for environmental management and conservation, but also for social and cultural prosperity was enshrined in the European Landscape Convention (ELC, 2000; Déjeant-Pons, 2006). For instance, as Déjeant-Pons (2006:365) states, conveying the underlying sentiment behind the ELC:

“A key factor in individual and social well-being and people’s quality of life, the landscape contributes to human development and serves to strengthen the European identity. It plays an important public interest role in the cultural, ecological, environmental and social fields and is a valuable resource conducive to economic activity, notably tourism [...] The Convention thus considers that landscape protection, management and planning entail “rights and responsibilities for everyone” and establishes the general legal principles which should serve as a basis for adopting national landscape policies and establishing international cooperation in such matters.”

Notably, the ELC clearly expresses the concept of landscape as being a holistic-perceptive phenomenon, specifically:

“...an area, as perceived by people, whose character is the result of the action and interaction of natural and/or human factors.”

There are two important features to note about this description of landscape. The first is the emphasis placed on the idea of landscape having bounded and organised area-based qualities: these conditions of restriction or containment draw on semiotic interpretations of landscape, where boundaries and borders are regarded as pathways of communication as well as facilitators of landscape diversity (Lindström et al., 2013). The second is the multivalent attributes landscapes have as a consequence of their perceived natures: aspects which evolve dynamically through the continuous interactions occurring between humans and the wider environment. In this sense changes in landscapes can emerge gradually or rapidly through so-called ‘dynamic non-equilibrium change processes’ (Lindström et al., 2013).

The implications are that landscape pattern, processes and meaning result from collective historical, social, economic and environmental structuring (Antrop, 2013). This is reflected in the ELC’s support for a “systems” appraisal of landscape that acknowledges ‘humans and nature are not two separate entities’ but coexist as “elements of a wider Earth system” (Cassar, 2013) where:

“...ecological processes originate from, and extend beyond, the boundaries of the ecosystem itself, and are embedded within a much wider landscape framework.” (Cassar, 2013:396)

Ultimately, the ELC projects a dynamic and fluidic notion of landscape as an interface subject to change (Antrop, 2013). This dynamic sentiment; that landscapes arise out of and are co-created and remade through human and environmental processes upon which socio-cultural meanings and identity are vested, is illustrated in Table 14.1.

Table 14.1 List of examples of the variety of meanings ascribed to 'landscape'

Source	Definition of landscape
Meinig (1979)	“Landscape is an attractive, important and ambiguous term [that] encompasses an assemblage of ordinary features which constitute an extraordinarily rich exhibit of the course and character of society.” (pg. 195)
Cosgrove (1984)	“Landscape is not merely the world we see, it is a construction, a composition of that world. Landscape is a way of seeing the world.” (pg. 13)
Coones (1985)	“The landscape [...] is in truth nothing less than the complex, interrelated and unified material product of the geographical environment, a seamless totality in which the immemorial processes of nature and the much recent activities of mankind interpenetrate” (pg. 5)
Mitchell (1994)	“Landscape is a medium in the fullest sense of the word. It is a material ‘means’ (to borrow Aristotle’s terminology) like language or paint, embedded in a tradition of cultural signification and communication, a body of symbolic forms capable of being invoked and reshaped to express meaning and values.” (pg. 14)
Muir (1999)	“It is clear that landscapes exist as historical texts. The historical aspects of landscape combine with aesthetic and place-related elements to constitute landscape as heritage. Landscape becomes, therefore, a significant component of the overall heritage which endows communities and nations with their identity.” (pg. 37)
Farina (2000)	“Human society and nature are the two main forces that shape landscape structure and drive landscape-level processes [...] Cultural landscapes are geographic areas in which the relationships between human activity and the environment have created ecological, socioeconomic, and cultural patterns and feedback mechanisms that govern the presence, distribution, and abundance of species assemblages [...] A cultural landscape is hierarchically organized in micro-units [...] which form the structural basis [...] of the cultural landscape. The successive aggregation of these microsystems [...] creates the complex landscape that is observed at scales of meters to kilometres” (pg. 313)
Naveh (2000)	“...landscapes and their multiscale dimensions as tangible, closely interwoven natural and cultural entities [...] Landscapes are therefore more than repeated ecosystems on kilometre-wide stretches [...] Landscapes range from the ecotope, as the smallest mappable landscape unit, to the ecosphere, as the largest global Total Human Ecosystem landscape.” (pg. 358)
Tress and Tress (2001)	“The transdisciplinary landscape concept is rooted in the different historical landscape concepts and is their logical culmination [...] and is built on five dimensions [...] the landscape concept, as introduced here, unites all five dimensions [...] These five dimensions are: (1) landscape as a spatial entity; (2) landscape as a mental entity; (3) landscape as a temporal dimension; (4) landscape as a nexus of culture and nature and (5) landscape as a complex system” (pg. 147)
Wylie (2005)	“Landscape is not just a way of seeing, a projection of cultural meaning. Nor, of course, is landscape simply something seen, a mute, external field. Nor, finally, can we speak altogether plausibly of the practice of self and landscape through notions of a phenomenological milieu of dwelling [...] Therein landscape might best be described in terms of entwined materialities and sensibilities with which we act and sense” (pg. 245)
Pedroli et al., (2006)	“Landscapes are clearly seen as the product of human’s interaction with nature, and as such have a biophysical dimension (human <i>causes</i> – landscape biophysical <i>effects</i>) as well as cultural history, and contemporary content” (pg. 426)
Rössler (2006)	“Cultural landscapes are at the interface between nature and culture, tangible and intangible heritage, biological and cultural diversity – they represent a closely woven net of relationships, the essence of culture and people’s identity [...] they are a symbol of the growing recognition of the fundamental links between local communities and their heritage, humankind and its natural environment.” (pg. 334)
Zaro et al., (2008)	“Landscapes represent a dynamic point of articulation between humans and the environment” [...] “landscapes should be treated as part of a historically contingent process that wholly integrates all biotic and abiotic elements” (pg. 261/264)

Table 14.1 *Contd.*

Source	Definition of Landscape
Swanwick (2009b)	“Landscapes embrace physical, natural, social and cultural dimensions of the environment and the interactions between them are arguably more comprehensive in their scope than ecosystems” (pg. S65)
Thermorshuizen and Opdam (2009)	“Landscapes are spatial human-ecological systems that deliver a wide range of functions that are or can be valued by humans because of economic, sociocultural and ecological reasons” (pg. 1041)
O’Farrell and Anderson (2010)	“...landscapes are complex entities; dynamic in and of themselves and further complicated in the human dimension of how they are perceived. Every landscape is a function of its abiotic and biotic template combined with its own unique history of human intervention.” (pg. 59)
Dramstad and Fjellstad (2011)	“Landscapes are continuously changing [...] are dynamic” (pg. 330)
Taylor and Lennon (2012)	“Landscapes can therefore be seen as a cultural construct in which our sense of place and memories in here and where we make places in a continuing process of inhabiting and changing the landscape.” (pg. 1)
Brook (2013)	“...the process of arriving at considered aesthetic judgements of landscapes begins with experiences of moving through places, including memory and attendant cognitive elements” [...] “...landscapes are lived in [...] landscapes are not just practical and cultural resources for humans; they also create ecological and morphological vale” (pg. 110/114)
Egoz (2013)	“Landscapes have been argued to be complex and dynamic entities, interpreted in different ways and thus the potent subject of conflicts and repositories of power relations and ideologies” (pg. 275)
Finch (2013)	“Landscape [...] encompasses material, cognitive and symbolic realizations of human-environmental relationships.” (pg. 143)
Gailing and Leibenath (2013)	“...the construction of landscapes can be viewed in a number of ways: as a physical shaping of nature by humans or as a social process in which meanings are attached to landscapes or in which landscapes themselves turn into symbols” (pg. 2)
Gulickx et al., (2013)	“Landscapes are spatial social-ecological systems that deliver a wide range of functions, which are valued by humans in terms of economic, sociocultural and ecological benefits.” (pg. 273)
Herring (2013)	“Landscape, then, is ever-changing, physically (in use and form), and in the ways individuals and communities perceive it.” (pg. 172)
Milani (2013)	“Landscape is a property of humans through their activity as free creators who modify, construct and transform the physical world by means of their talent, imagination and technology.” (pg. 74)
Pedroli et al., (2013)	“...an expression of societal development, landscape is dynamic by nature. Its history constitutes a fundamental basis for interpretation of changes, and appreciation of transformations” (pg. 692)
Roe (2013)	“...landscape is a reflection of human interactions with natural forces [...] concept supports the notion that landscape springs from interactions between culture and nature or humans and the land” (pg. 336)
Sayer et al., (2013)	“...define a landscape as an area delineated by an actor for a specific set of objectives. It constitutes an arena in which entities, including humans, interact according to rules (physical, biological, and social) that determine their relationships.” (pg. 8350)

Table 14.1 *Contd.*

Source	Definition of Landscape
Setten and Brown (2013)	“The material manifestation of landscapes and their role as a concretization of social relations means that struggle over its various forms, meanings and representations impinge on real people’s bodies and lives and the very structures and conditions of existence” (pg. 248)
Wattchow (2013)	“This reciprocity between people and locations on the Earth’s surface provides the reference point for all considerations of landscapes and places” [...] “Thus <i>landscape</i> has become a projection, a site of layered meanings, a receptacle for human values and experiences” (pg. 87/98)
Wu (2013)	“...landscapes are the scale at which people and nature mesh most acutely, and thus the composition and configuration of a landscape both profoundly affect, and are affected by, human activity” (pg. 1000)
Bastian et al., (2014)	“...part (at various scales) of the earth’s surface, which is shaped by natural conditions and formed by human influences to a different extent. It is perceived and felt by humans as characteristic, and it can be differentiated and classified according to defined rules” (pg. 1464)
Iverson et al., (2014)	“...historical perspectives of landscape when evaluated across space and time, increase our understanding of the dynamic nature of landscapes and provide a framework of reference for assessing patterns, processes, and functions as they pertain to provisioning, supporting, cultural and regulating services” (pg. 182-183)
Plieninger et al., (2014)	“Cultural landscapes are at the interface between nature and culture, tangible and intangible heritage, biological and cultural diversity – they represent a closely woven net of relationships, the essence of culture and people’s identity” (pg. 1)
Revill (2014)	“The term landscape can be used to describe a physical arrangement of topography, the shape and lie of the land, but it also refers to depiction in symbolic form as text, graphics or numeric data, as guidebooks, paintings, maps or tables, or equally environmental experience as worked earth, footfall, memory, solitude or belonging. Thus the term landscape seems to suggest object and interpretation, conscious arrangement and random scatter, ‘raw material’ and ‘artful construction’, ‘nature’ and ‘culture’. (pg. 335)
Freeman et al., (2015)	“...a landscape can be defined as a complex social-ecological system, usually made up of a mosaic of different land uses. The boundaries of the landscape can either be discrete [...] or fuzzy [...] In some cases there can also be multiple overlapping boundaries related to both social and ecological dimensions. The landscape itself will be largely context dependent [...] combining both social and ecological dimensions, spatially explicit patterns and processes and heterogeneity remain key defining characteristics of landscapes” (pg. 3)

14.1.2 *Landscape And Meaning*

Emerging from the conceptualisations of landscape presented in Table 1 is an underlying theme of complexity – that landscapes represent the tangible and visceral ways humans identify with, experience and understand the broader environment (Ward Thompson, 2013):

“...landscape is a relative and dynamic entity wherein, from antiquity up to the present day, nature and society, the gaze and the object of the gaze, constantly interact.” (Milani, 2013:75)

In other words, the meanings we attach to, as well as those that arise from, landscapes are determined by the complex interrelations between the visual, territorial and the spatial (Swanwick, 2009b). Beyond that, landscapes also present an additional meaning layer in the form of a sound-world: a symphony of rich, diverse and distinctive auditory sensations that

influence our understanding, experience and the meanings we ascribe to landscape – aspects that make strong connections to our memories and emotions. These sounds may be specific to particular landscapes as well as change over the course of a day or season affecting our perception and engagement with, and in, the landscape (Revill, 2014). As Revill (2014:334-335) comprehensively describes:

“...sound brings us into intimate contact with activities, actions and events that lie well outside the reach of other senses, behind us, round the corner or over the next hill [...] As Brandon LaBelle has shown, acoustic spaces are shared, conflictual, intimate and vital, combining points of focus with points of diffusion in ways that both contrast and complement other sensory experiences of landscape [...] Heard sounds give embodied sensation to feelings of depth, distance and clarity, delicacy and intimacy, transforming and animating the experience of landscape in ways that both complement and conflict with those of visual perspective [...] Yet there is certainly a sense in which sound seems to mediate landscape both intervening between world and perceiver shaping and transforming experience and acting as a point of departure for the creative cultural and emotional elaboration of landscape experience.”

Importantly, what Table 1 also demonstrates is an evolution of thought regarding the constitution of nature and culture and their dynamic relationship, whilst illustrating a gradual transition to the idea of ‘landscaping’ (Wylie, 2007:11):

“...we should think about practices, habits, actions and events, ongoing processes of reality and unreality that come before any separation of ‘nature’ and ‘culture’.”

Yet, at the same time, the various framings of landscape detailed in Table 1 also suggest an underlying theme of “tension”, a tension existing between “proximity and distance”, “body and mind” and “immersion and observation” (Wylie, 2007). These dualist interpretations open the door to phenomenological appraisals of landscape, in which human lived-experiences and ritualistic embodied practices combine to create a suite of dialectics between “landscape” and “life” (Wylie, 2013) or so-called “dialogical phenomena” (Lindström et al., 2013). Collectively, they reinforce the idea of an individual ‘cognitive landscape psychology’ through which our placed-based experiences are realised, as we and the landscape remain in a constant state of “becoming” (Waterton, 2013; Wattachow, 2013). Landscape does not therefore reach an endpoint but always remains “unfinished”, as Wylie notes referring to the work of Don Mitchell, the landscape remains in a continuous state of “production” and therefore remains “open to change”, “alteration” and “contestation” (Wylie, 2007).

These sentiments provide a platform, building on the work of Gilles Deleuze and Felix Guatari, supporting non-representational theories of the relationships between “body” and “landscape”, in which core non-representational concepts such as “time-space”, “practice”, “subject” and “agency” produce meaning-making through the performative acts our bodies facilitate, embody and co-produce with landscape in the form of experiences and

events (Wylie, 2007; Waterton, 2013). In this way landscapes are not simply “out there”, as in the disembodied and detached European aesthetic tradition, but are both externally and internally experienced (Brook, 2013; Milani, 2013) – a view echoing Ingold’s conceptualisation of “landscape-as-dwelling” or as “being-in-the-world” (Ingold, 2000).

14.1.3 Landscape: Psychology And Wellbeing

Landscapes connect to our emotions and link the imaginary with the material (Hawkins, 2013) presenting us with a set of phenomenological experiences (Swanwick, 2009b):

“...a milieu of engagement and involvement. Landscape as ‘life world’, as a world to live in, not a scene to view.” (Wylie, 2007:149 summarising the phenomenological appraisal of landscape pioneered by Merleau-Ponty)

Ultimately, our bodies internalise and reflect the engagement we have with the world around us in a “recursively intertwined” interaction that crystallises a “process of (re)formation” (Finch, 2013; Waterton, 2013). The notion of an integrated and perceived whole is succinctly expressed by Swanwick (2009a:351) in relation to people’s perception of UK upland landscapes, notably that:

“...people’s responses to an upland landscape like the North Pennines, and no doubt others, is that they do not necessarily compartmentalise values into separate categories but consider the landscape as a whole [...] the experience of landscape [...] cannot be easily separated out in the minds of the public, who see their surroundings in an integrated way.”

Unsurprisingly, yet nonetheless significant, as a consequence of these fundamental reciprocal connections the evidence suggests that when individuals and communities engage with each other through landscapes multiple benefits arise such as: improvements in social learning; the promotion of social capital and enhanced individual mental and physical health (Swanwick, 2009b). Many of these benefits are incorporated into the idea of “therapeutic landscapes” (Rose, 2012). Recent syntheses of environmental psychology demonstrate that human interactions with nature have a generally overall positive effect on cognitive capacities, emotional states and stress (Bratman et al., 2012). Advocates of the Attention Restoration Theory (ART) model in environmental psychology consider four aspects of the ‘natural’ environment or ‘landscape’ to be of particular importance in mediating these positive effects, namely; the extent of the experience; the sense of being away and escaping from daily life and routines; being intrigued and fascinated by these new surroundings and an alignment between individual needs and the capacity of the environment to meet those needs (Bratman et al., 2012). At the same time, it is also important to acknowledge that the landscapes we inhabit are not always beneficent and can frequently present considerable dangers to human populations, for example, in terms of natural disasters (e.g. volcanoes, earthquakes, flooding); outbreaks of

zoonotic diseases (e.g, avian flu, ebola) or landslides resulting from extensive soil erosion due to deforestation activities (Batram et al., 2012).

Despite some of the more severe negative impacts the wider landscape can bestow on individual, community and societal prosperity, observations like these highlight the deeply interwoven connections between landscape and personal identity expressed in concepts such as ‘rootedness’ and ‘uprootedness’ (Egoz, 2013). More profoundly, they evidence a “relational dynamic” between the individual “self” and the landscape, such that landscape as well as place may play a part in the stable formation of the individual and inform the connections between mind and body (Rose, 2012). This is central to the concept of “mentalising” and the notion that mentalising bridges the gap between imagination and reality, which in the context of individual identity and landscape posits that encounters between the individual and landscape can improve personal understanding, empathetic qualities and emotional regulation (Rose, 2012). Indeed, recent research suggests those that grow-up in a “green” environment, closer to and engaging with “nature” early on, demonstrate greater longer-term mental health through the development of a greener “self” (Windhorst and Williams, 2015). Collectively, these findings also lend credence to the idea that individuals have a ‘unique psycho-sociological perception’ of the cultural space they inhabit (Stobbelaar and Pedroli, 2011).

14.1.4 Landscape: Scale And Place

Hence identity, meaning and scale are intimately connected to the landscape setting:

“...identity processes are embedded within wider, dynamic cultural, political and economic forces [...] Our identities are shaped by the experiences we have with both social and non-social stimuli, the people and places that we encounter...”
(Devine-Wright and Clayton, 2010:267)

For example, as Bowring (2013) argues, attachments of “the global” to landscape are anathema given the “place-based” rootedness of the concept which tends to recreate feelings of “belonging” and “distinctiveness”: aspects at risk of homogenisation when conceived globally.

“From the panoptic viewpoint of the global, this physical particularity of place fades out of focus, becoming instead constituted by flows of people, information and material.” (Bowring, 2013:264)

Nevertheless, Bowring (2013) is quick to point out, quite rightly, that conceptions of the local are equally at risk from homogenisation due to overzealous “government directives” and “blanket approaches”. The language of scale is particularly potent and can be used to convey the need for prioritisation or the necessity of immediacy, not only by acknowledging that we perceive associations as “central” (i.e. local) or “peripheral” (i.e. global), but also by applying

these concepts we frame what is fundamental to understanding landscape dynamics and landscape construction in particular contexts (Bowring, 2013:269):

“In the context of landscape [...] the impact of the scalar relationship is vital, as it provides the frame for identity.”

It is equally important to note that although landscape and place are connected, the boundaries between them being porous, they are not the same:

“...landscape as a holistic term, larger than place – a term that gathers together body, place, perception and relationships between people and between people and things.” (Wiley, 2007:172 paraphrasing Tilley, 2004)

Landscape, then, presents a broader and richer tapestry of meaning and inclusivity in which considerations of place are subsumed. As Muir (1999:271) remarks:

“Both embody connotations of being defined in part by the existence and values of insiders, landscape being regarded as the environment perceived and place consisting in part of social networks [...] Landscape is likely to be an important factor in any discussion of place because it will normally be a crucial component of the sense-of-place.”

Ultimately, landscape harkens to a more fundamental, and in some ways timeless, sense of meaning and identity (Park and Coppack, 1994:63):

“Landscapes have a deeper significance, identity and character [...] This is because every landscape and artefact is associated with cherished attitudes, values and images, making them evocative, idiosyncratic and reflective of changing social values.”

Realised in this way landscape is not solely located in a purely terrestrial interpretation of the environment; but instead, refers to the space in which humans interact, affect and perceive, and is therefore equally applicable to the coastal and marine environments humans have made home (Herva and Ylimaunu, 2014). Furthermore, within the terrestrial sphere the idea of “landscape” does not exclusively refer to a rural or semi-rural construct, but also to peri-urban, urban and industrial environments. In this sense, landscape is equally as applicable to the study of the human built environment and the green spaces that populate these areas as it is to “wilder” wilderness regions.

14.1.5 Landscape: Control And Power

An interesting juxtaposition develops as the bounded and dynamic notions of landscape meet, in what Olwig (2013) refers to as the “German” concept of landscape and the “English” perception of landscape. These somewhat opposing conceptions have direct implications for how access and ownership of the landscape is regulated, and the way in which statutory and customary law interact (Olwig, 2013):

“Custom and culture defined a land, not physical geographic characteristics – it was a social entity that found physical expression in the area under its law [...] The land was [thus] initially defined by a given body of customary law that would have developed historically within and through the workings of the judicial bodies of a given legally defined community.” (Olwig, 2002:17)

Related to the “law of the landscape”, so-called, but also associated with the normative construction of landscape, is the idea that judgements lie at the heart of how landscapes are fashioned – hence issues of social justice are hardwired into the fabric of landscapes (Setten and Brown, 2013). As a result landscapes can be tools of domination, instruments that partake in, reproduce and exacerbate social struggles through their spatial configurations and the moral authority they legitimate (Setten and Brown, 2013). The ability to deploy power and ideologies to affect the nature of landscape are particularly well evidenced in post-colonial landscape studies, influenced by the poststructuralist work of Foucault and Derrida, where connections between landscape configurations, migration and human dignity have been explicitly highlighted (Egoz, 2013; Setten and Brown, 2013):

“...concern for many cultural geographers has been to demonstrate how certain ideal European landscape forms have been used to characterise, appropriate and judge non-European scenes.” (Wylie, 2007:124)

And, moreover:

“One important consequence of this naturalising and aestheticizing of the non-European ‘other’ is the [...] symbolic erasure of other possible histories of land occupation [...] parallels more literal processes of imperialist land appropriation and dispossession...” (Wylie, 2007:133)

Both of these examples illustrate the propensity of Occidental nations to ‘Orientalise’, in Edward Said’s sense, non-Western cultures. This form of social constructivism acknowledges the importance of dynamism, power and agency in the production of landscapes, and underscores their role as processes that mediate interactions within landscapes. Consequently, the arguments made for the role of formal and informal institutions in shaping individual and collective agency, as well as “power-laden political processes” exerting their influence on the constructed form of landscapes, becomes increasingly plain (Gailing and Leibenath, 2013).

Naturally, the assumption predominates that the particularities of the landscapes we inhabit, and within which we interact, can “affect how democracy develops in society” – with landscapes perceived as conduits for ‘more sustainable and democratic living’ (Roe, 2013). The problem with this assumption is that focusing too much on the advantages of participatory landscape decision-making processes can promote a naïve approach to participation, whereby social injustices and power asymmetries can become ingrained into practices seeking the opposite outcomes (Setten and Brown, 2013:248):

“The material manifestation of landscapes and their role as a concretization of social relations means that struggle over its various forms, meanings and representations impinge on real people’s bodies and lives and the very structures and conditions of existence. Consequently, landscapes are always about justice.”

14.1.6 *Landscape: Crucibles Of Settlement And Exploitation*

From the outset, the historical evolution and contemporary compositions of many landscapes is symbolic of directed and purposeful human exploitation and colonization: whether the result of land clearance for human settlements (e.g. Lucero et al., 2014; Marsh and Kealhofer, 2014; Cunfer and Krausmann, 2015; Kahn et al., 2015; Romanova, 2015), land conversion for agricultural production (e.g. Dull, 2007; Arvor et al., 2012; Van Vliet et al., 2013; FAO, 2014) or mining for natural resources (e.g. Sonter et al., 2014a; 2014b; Basommi et al., 2015; 2016). Since the seminal transition from hunter-gathers to farmers, natural resource-based exploitation and agricultural production have fuelled economic, urban, political, and societal development for millennia: transforming and shaping the environment around us to meet our needs and forging an extensive patchwork of human-dominated landscapes (Barbier, 2011).

The rise and, in many cases, subsequent demise of cities, nations and empires across the world – their achievements, their stories, their cultures – is inextricably linked to how those individual societies and the changing fortunes of their economies managed the pressing issues of land and natural resource scarcity, and for a time – for some transiently for others or more permanently – successfully bridged internal and external frontier-based exploitation and expansion with developments in agrarian technology, advances in new labour markets, and the coupling of primary domestic markets with international trade networks in goods and services and industrial developments (Barbier, 2011). As Barbier (2011:7) describes in his extensive and eloquently written tome *Scarcity and Frontiers*:

“Throughout much of history, a critical driving force behind global economic development has been the response of society to the scarcity of key natural resources. Increasing scarcity raises the cost of exploiting existing natural resources, and will induce incentives in all economies to innovate and conserve more of these resources. However, human society has also responded to natural resource scarcity not just through conserving scarce resources but also by obtaining and developing more of them. Since the Agricultural Transition over 12,000 years ago, exploiting new sources, or “frontiers,” of natural resources has often proved to be a pivotal human response to natural resource scarcity.”

Developments such as these continue apace producing lasting and transformative changes in landscapes – changes that demonstrate their effects in rural, urban and in inland and coastal settings as well as across landscape-types (e.g. forests, grasslands, agricultural rangelands, marine systems): The result of widespread and intensive resource exploitation promoted by global patterns of production and consumption – activities stimulated by the

rapid growth in international trade and commodity markets – driven by successive waves of economic globalisation and urbanisation: processes exemplified by the complex and distant teleconnections¹ between the sources of production – rooted primarily in the “Global South” – and the destinations of consumption – located primarily in the “Global North” – as well as the telecoupling² properties of these exchanges that emphasise the interactive and iterative qualities of these processes (e.g. Barbier, 2011; Lambin and Meyfroidt, 2011; Meyfroidt et al., 2013; Yu et al., 2013; Friss et al., 2016). For example, as Lambin and Meyfroidt (2011:3465) highlight:

“Globalization increases the worldwide interconnectedness of places and people through markets, information and capital flows, human migrations, and social and political institutions. Over the last 300y, the world economy has experienced an increasing separation between the location of production and consumption.”

Focusing particularly on the example of agricultural land-use impacts, the authors further explain how activities and processes such as these can result in widespread land-use displacement and leakage effects (*ibid.*):

“Agricultural intensification or land use zoning in a country may trigger compensating changes in trade flows and, thus, affect indirectly land use in other countries. Between 2000 and 2005, tropical deforestation was positively correlated with urban population growth and exports of agricultural products.”

As the work of Yu et al. (2013:1178) illustrates in relation to South-North commodity chains:

“Our analysis shows that 47% of Brazilian and 88% of Argentinean cropland is used for consumption purposes outside of their territories, mainly in EU countries and China.”

A point further underscored by Meyfroidt et al. (2013:3), again in connection to the implications of globalised production and consumption-based land-use activities on forests:

“Recent forest transitions in a handful of developing countries were associated with an outsourcing of land use and forest exploitation abroad via increased wood and sometimes agricultural products imports. Economic globalization thus facilitated a national-scale forest transition in some countries through a displacement of land use.”

Although displacement and leakage effects are substantial and increasingly known and understood other processes such as rebound³, cascade⁴ and remittance⁵ effects also figure in the impacts of globalised teleconnections on the transformation of land-uses and landscapes (Lambin and Meyfroidt, 2011). In an era of increasing demands on food, energy and water resources security and scarcity issues are also driving large-scale changes in land ownership through acquisition and land grabs by the State or powerful companies, this growing trend is producing significant social impacts, particularly on poorer marginalised communities, and

leading to dispossession, impoverishment, social inequities and negative gender and race relations (Amanor, 2013; Borrás Jr et al., 2013; White and White, 2013; White et al., 2013).

14.1. 7 Landscape: Cultural Perspectives

So far our framing of landscape has been primarily rooted in a European and North American socio-cultural and philosophical tradition. However, landscape is by no means purely the preserve of Western culture - notions of human-nature relationships mediated through “landscape” are deeply engrained in the cultural heritage and development of human societies around the world – for example, in Eastern traditions like Buddhism, Confucianism, Daoism and Hinduism these relations provide a framework and foundation for the elaboration of core belief systems (Wang et al., 2011; Grim and Tucker, 2014). In contrast to their Western counterparts, which have historically compartmentalised and juxtaposed conceptions of landscape, for example, as between “Romantic” and “Scientific” visions (Malini, 2013), these traditions weave together strands of the physical, ecological, geographical, economic, social, cultural, cognitive and meta-physical (Wang et al., 2011).

From time immemorial the sacred and the spiritual have expressed the tangible and intangible connections between human culture and landscape (Hind, 2007), with these primitive worldviews and beliefs shaping and moulding the diversities of human culture and our complex mythic creation stories and religious ideologies, as Jenkins and Chapple (2011:441) put it:

“Understanding the interaction of human and environmental systems requires understanding the religious dimensions to the integration of ecology and society.”

Ancient Maya, Inca and Aztec civilisations, for example, sprang out of the natural richness of the forests of Central and South America constructing fabulous temple cities (e.g. *Planeque* in Mexico and *Machu Picchu* in Peru) to the gods of nature: emblazoning monuments with beasts from the animal kingdom and linking cosmological patterns to the seasonal variations of growth and harvest. Across the vast terrain of North America the ancestral traditions of many Native Indian tribes are deeply connected to the worship and ritualization of the landscape. The Hopi of Arizona consider the protection of the land “a sacred duty”, whilst the Chugach people of Alaska view the “natural world and all that grows and moves upon it as irrevocably interconnected”. Similar belief systems can be identified in many of the tribal societies of Africa. The Dogon people of Mali, for example, have an animist tradition where *Lewe*, the earth god, can imprint his spirit on “animals”, “plants” and “stones”. Nowhere is this connection between the power and importance of “raw landscape” and ancestral spirits more evident than in the Aboriginal tribes of Australia and their concept of “dreamtime” (Hind, 2007).

The more complex civilisations of Asia have particularly rich and sophisticated conceptions of human-nature relationships. In Java, where the indigenous populations merged aspects of Buddhism and Shaivism, structures such as Borobudur “represent a three-dimensional mandala – a cosmographic model of a perfected world” (Hind, 2007). Indeed, the different features and composition of the landscape (e.g. water, mountains, sea) are encapsulated in broader cosmological and mystical beliefs regarding the origins and relations of nature and man, in which the landscape reflects the ‘functional’, ‘ideological’ and ‘aesthetic’ needs of the people (Amin, 2012:74):

“Water, mountain, sea and land i.e. the whole landscape, is the life giver.”

Similarly, the pantheistic tradition of Hinduism offers a deeply holistic interpretation of human-nature relations and its unfolding within a multidimensional universe (Thakur, 2012:161):

“The appreciation of nature and its elements encapsulated in the Vedas, Upanishads, Puranas [...] show the different ways in which man interacted with nature shaping a culture unique to the subcontinent and reflecting the underlying philosophy that is inclusive and based on a cyclic notion of time drawn from nature.”

Nowhere is this vision more profoundly expressed than in the temple complex of Angkor Wat, Cambodia, dedicated to the deity Vishnu the preserver of the universe (Hind 2007:194):

“The layout of Angkor Wat is a microcosm of the Hindu concept of the universe. Symbolising a mountain, three tiers support the temple: the first dedicated to the king, the second to the creation god Brahma and the moon; and the third to Vishnu. Five towers are set in a quincunx, representing Mount Meru, the Hindu home of the gods.”

Likewise, in China, alongside the more moral view of nature adopted by Confucianism, Daoism/Taoism evokes a similarly divinely inspired and shaped conception of human-nature connections, where physical elements of the landscape embody profound divine qualities (Han, 2012). For example, the Ancient Building Complex in Wudang Mountains was designed to capture the “powers of the mountain” (Hind, 2007:206):

“Taoism believes that nature itself is divine [...] Therefore the created world is to be revered. Mountains not only sustain a wealth of nature with their springs, caves, flora and fauna, they also support the sky and in so doing create the space in which life blossoms.”

The ancient Japanese religion of Shinto “the way of the gods” is similarly imbued with strong elements of nature and the ritualization of nature worship, in which not only physical aspects but events in the natural environment hold symbolic meaning and manifest spirit phenomena called kami (Britannica, 2016).

In many ways, these richly varied spiritual-cultural articulations of landscape connect directly to the field of historic cultural landscapes, in which the social, cultural (including the sacred), aesthetic, historical as well as scientific values of landscapes to society are considered of special significance (Herring, 2013). Emphasising the historical component of landscapes presents an embedded and experience-driven perspective of the human-environment relationship, in which the ephemeral existences of individuals are linked to longer-term trends of change that come to shape the visible landscape structure successive generations inherit (Finch, 2013). But, more profoundly perhaps, is landscape's capacity to inspire, to engender awe and wonderment in humanity, an aspect of landscape that has been central to its pervasive cultural influence (Collins, 2013). For example, from a European perspective, landscape has been an integral and rich source of creative inspiration to the various artistic, architectural, literary and musical developments, and the production of "Art" in its widest sense, over the last 500 years: from the Italian and Northern European Renaissance of the 15th and 16th Centuries through to the Baroque, neo-Classical, Romantic and Impressionist eras of the 17th, 18th and 19th Centuries, respectively, and onto the Post-Impressionist, Modernist and Post-Modernist movements of the 20th Century including much else in-between (Gombrich, 2007).

It is precisely this type of rich cultural legacy, at a global scale, which promulgated the development of UNESCO's Convention concerning the Protection of the World Cultural and Natural Heritage (1972), and in 1992, the subsequent inclusion and recognition of 'cultural landscapes' in the Convention (Lennon, 2012). Finally, alongside these global policy initiatives, landscape's capacity to transform human-nature relationships from one of opposition to one of inclusion has been integral to its, both implicit and explicit, presence in many contemporary political and philosophical environmental movements such as Deep Ecology (e.g. Naess, 2008[1973]; Devall, 2008[1980]), Social Ecology (e.g. Bookchin, 2008[1981]) and Spiritual Theology (e.g. Cobb Jr, 2008[1982]; Merchant, 2008[2003]).

14.1.8 Landscape in the 21st Century

Overall we argue that landscape is a dynamic, fluidic and functional construct arising as a combined product of socio-cultural, economic, political and bio-geophysical processes operating iteratively across multiple spatial and temporal scales (Figure 14.1 Wang et al., 2011). As a construct, we contend that landscape acknowledges human influences and connections in the creation of the external environment and construction of form: a creative process that feeds back into human culture and society contributing to both individual and collective meaning, identity and wellbeing. From this standpoint landscape ought to be viewed as an active fluidic space, contingent upon factors and pressures of change that may act both antagonistically and synergistically with a potential range of positive and negative outcomes

(Figure 14.2, Howard et al., 2013). In this regard we consider landscape as a quintessentially social-ecological phenomenon manifesting the qualities and properties of a complex system (Folke, 2006). However, we advance that landscape is not just an expression of the co-production and co-engineering of humans in nature, more than this, landscape is both internally and externally the embodiment of home, the reflection of our moral and ethical beliefs and values, and ultimately the co-imagining – a symbiosis if you will –and articulation of humanity’s inseparable connections to the richness of our own ancestry and the biosphere.

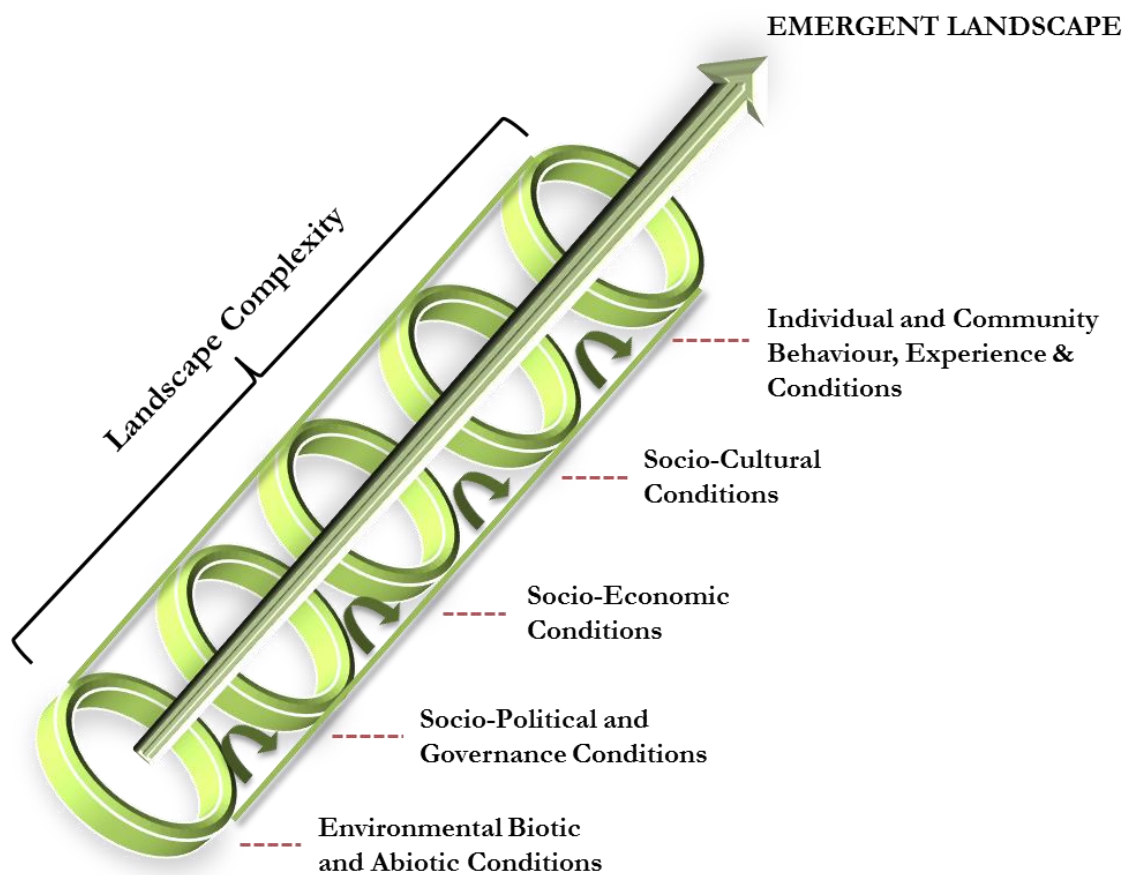


Figure 14.1 Representation describing landscape as an emergent property that arises from various interacting human-environmental dimensions (green rings), which are both dynamic, fluidic and operate in an iterative manner (landscape complexity) and through which landscape evolves.

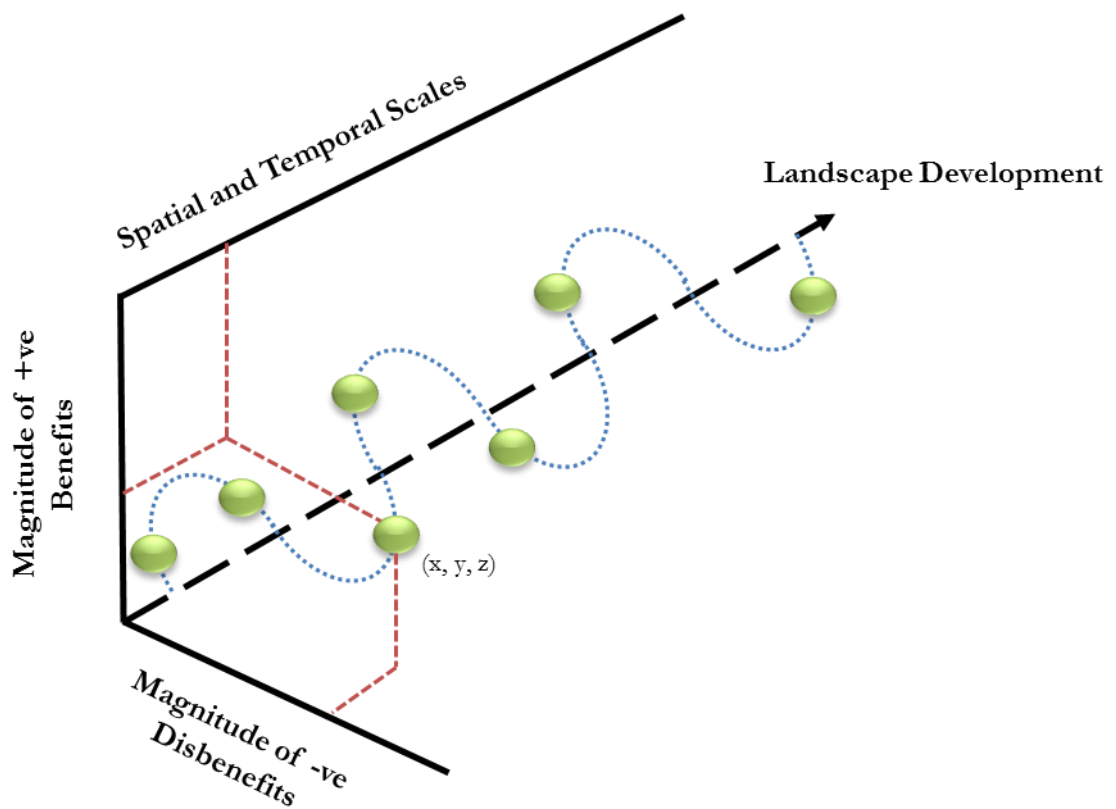


Figure 14.2 The landscape (green spheres - each one a separate variant) operates across different spatial and temporal scales and at each of these scales will provide both a number of benefits and dis-benefits (denoted by the x , y , z coordinates). Observed as a whole the landscape has a developmental trajectory, but not a pre-meditated one, one that arises through how the landscape has changed and evolved over time as a consequence of the interactions described in Figure 14.1 – we can say then that each landscape has its own landscape elasticity (dotted blue line) and the sum total of these elasticities described by each variant crystallizes the life history of a landscape.

Notes

1. As summarized by Friis et al. (2016:134) teleconnection refers to the:

“...spatial decoupling of land change drivers and outcomes [...] As captured in the prefix ‘tele’, the teleconnection concept invokes a sense of (large) spatial distance between the systems interacting to produce the connection [...] the concept has gained prominence in LSS [Land System Science] studies trying to come to grips with both environmental and socio-economic linkages between distant and seemingly unconnected land systems around the world. Many of these studies focus on international trade flows [...] The teleconnection concept has also been used to explore distal linkages between local land use change and livelihood transformations in relation to vulnerability and adaptation to global environmental change [...] Finally, the teleconnection concept has gained prominence in studies on urban dynamics and land use changes since urban expansion and the sustainability of cities are now highly dependent on the sustainability of their proximal and distant ‘hinterlands.’”

2. According to Friis et al. (2016:135) telecoupling extends the concept of teleconnection and:

“...is put forward in LSS to capture ‘not only the “action at a distance” but also the feedback between social processes and land outcomes in multiple interacting systems’.”

3. The term ‘rebound effect’ in this instance, according to Lambin and Meyfroidt (2011:3467) refers to:

“...a response of agents or of the economic system to new technologies or other measures introduced to reduce resource use.”

The idea here is that the introduction of a new technological development may lead to increased productive efficiency and as a result lower overall productivity costs including the cost of the good or service, the consequence of this is that consumption of the good or service naturally increases and thereby offsets the benefit of the new technology which was designed to reduce resource use (Lambin and Meyfroidt, 2011).

4. In a land-use context the ‘cascade effect’ according to Lambin and Meyfroidt (2011:3468) explains how a sequence of events moves through a system:

“Land-use change is driven by multiple, interacting factors that originate from the local to the global scales, involve feedback loops, and cascades through land use systems. A cascade effect is a chain of events due to a perturbation affecting a system [...] In land change, it occurs through indirect land use changes, a crucial issue when evaluating environmental impacts of biofuels, for example. The mechanism is similar to that of land use displacement, with an initial change in land use allocation causing multiple crop substitutions and land conversion in a place distant from the biofuel production site, thus leading to additional environmental effects that are not immediately measurable.”

5. The ‘remittance effect’ concerns the transfer of funds from populations that have migrated away from their home area, region or country and in their new existences elsewhere send part of their income back to their home communities, as Lambin and Meyfroidt (2011:3469) describe:

“Outmigration from rural regions affects land use through a decrease in labor force and in consumption needs, and an inflow of remittances. In 2009, 214 million international migrants in the world were sending back home an estimated 414 billion US\$ as remittances. This massive transfer of funds may facilitate the reconversion of family members at home to the rural nonfarm economy, thus decreasing pressure on land. An increase in wealth of rural households is generally associated with a decreased engagement in agriculture and diversification toward rural nonfarm activities. Alternatively, remittances can favor investments in mechanization and agricultural intensification. Migrants also directly purchase land in their home country, as a safety net and to maintain ties with their place of origin. Outmigration affects how land use decisions are made and may give rise to “remittance landscapes”

Chapter 15: Outlining a Landscape Approach

“The local needs often trump the larger issues like habitat or carbon storage, so there is a need to balance the two. In a humans-dominated landscape like in India, many functions of the landscape — food production for local subsistence, firewood collection, livestock grazing, tourism, wildlife habitat and agriculture — overlap and compete. In these cases, the landscape approach could provide viable conservation options.” (Professor Ruth DeFries, quoted in *The Hindu Newspaper*, December 2013)

“Landscape approaches are slowly starting to migrate into corporate thinking. Businesses are an essential actor; we need to accelerate this process.” (Lee Gross, Senior Manager, EcoAgriculture Partners, Global Landscape Forum 2014)

“Landscape approaches resonate well with indigenous people, It is how they conserve land, water and resources. So the landscape approach is very compatible.” (Victoria Tauli-Corpuz, UN Special Rapporteur on the Rights of Indigenous Peoples, Global Landscape Forum 2014)

15.1 Problems And Challenges: The Ecosystem Approach

Before we outline our thoughts of what a landscape approach comprises, it makes sense first to preface that discussion by considering the ecosystem approach, which we regard as a forerunner to landscape approach. The ecosystem approach (EA) has been increasingly implemented across a range of different sectors and international legal regimes, with the purpose of providing a policy narrative, and in some cases binding obligations, for a more modern and fully engaged type of environmental management (De Lucia, 2015). The conceptual foundations and implementation of the EA, in its most widespread form, is rooted in the so-called Malawi Principles (Box 15.1). Despite these principles some argue that the concept, particularly from a legal perspective, remains ambiguous, unstable and even elusive primarily because at its heart lies a genealogical tension between eco-centrism¹ and anthropocentrism² (De Lucia, 2015). The consequences of this genealogical tension it is argued is that the ecosystem approach suffers from a certain amount of “conceptual ambiguity”, “conflicting values”, “competing views on nature” and comprises a “confusing ensemble of labels and terminology” (De Lucia, 2015). A review by the CBD of the EA clearly identified the conceptual foundations as a problematically barrier in terms of the concept’s implementation (CBD/SBSTTA, 2007):

“The concept of the EA of the CBD is the centre of a critical debate concerning its theoretical foundation, its logical consistency and its value as a practical guide.”

The report carries on, saying that:

“...the wording of the principles and the guidelines is held so general that it cannot be used as a direct *modus operandi* to implementation.”

This reality, perhaps, partially explains why examples of projects employing the ecosystem approach do so with varying degrees of success and faithfulness to the original Malawi Principles. For example, in the marine fisheries sector where ecosystem-based management is a growing theme projects tend to only superficially relate to the sentiment of the Malawi Principles (e.g. Berghofer et al., 2008; Morishita, 2008; McFadden and Barnes, 2009; Osterblom et al., 2010; Paterson et al., 2010; Shannon et al., 2010). Whereas, in the terrestrial sphere, the ecosystem approach has been more rigorously applied with respect to the intentions of the Malawi Principles, for example, in Argentina (e.g. Ianni and Geneletti, 2010), Australia (e.g. Maynard et al., 2010), UK (e.g. Waters et al., 2012) and West Africa (e.g. UNEP, 2007). The unevenness with which the ecosystem approach is applied is supported by the observation of Waylen et al., (2014:1220) who remark that:

“...only 12% of its parties [i.e., those signed up to the Malawi Principles] stated that the principles were being applied despite widespread adoption in policy documents from intergovernmental and nongovernmental organizations.”

Box 15.1 The Malawi Principles

In a workshop on the Ecosystem Approach (Lilongwe, Malawi, 26-28 January 1998), whose report was presented at the Fourth Meeting of the Conference of the Parties to the Convention on Biological Diversity (Bratislava, Slovakia, 4-15 May 1998, UNEP/CBD/ COP/4/Inf.9), twelve principles/characteristics of the ecosystem approach to biodiversity management were identified:

1. Management objectives are a matter of societal choice.
2. Management should be decentralized to the lowest appropriate level.
3. Ecosystem managers should consider the effects of their activities on adjacent and other ecosystems.
4. Recognizing potential gains from management there is a need to understand the ecosystem in an economic context, considering e.g. mitigating market distortions, aligning incentives to promote sustainable use, and internalizing costs and benefits.
5. A key feature of the ecosystem approach includes conservation of ecosystem structure and functioning.
6. Ecosystems must be managed within the limits to their functioning.
7. The ecosystem approach should be undertaken at the appropriate scale.
8. Recognizing the varying temporal scales and lag effects which characterize ecosystem processes, objectives for ecosystem management should be set for the long term.
9. Management must recognize that change is inevitable.
10. The ecosystem approach should seek the appropriate balance between conservation and use of biodiversity.
11. The ecosystem approach should consider all forms of relevant information, including scientific and indigenous and local knowledge, innovations and practices.
12. The ecosystem approach should involve all relevant sectors of society and scientific disciplines.

Source: FAO

Indeed, it seems unclear, and is therefore open to debate, the extent to which the EA when applied merely helps to highlight the existing complexity of social-ecological systems or leads to the better management of these systems (Waylen et al., 2014). Historically, the Malawi Principles represent a significant contribution to the international development of the conservation and management of ecosystems and biodiversity. In particular, because they acknowledge the interdependency of human-environment relationships and recognising these interactions are central to the decision-making processes in environmental management (Waylen et al., 2014). Yet, it is also uncertain as to whether the principles are required (or intended) to be implemented together, as a whole, with some suggestions indicating that doing so make create additional problems between the more ecological-oriented principles and the more socially-focused principles (Waylen, 2014). Implementation is therefore represents a serious obstacle and challenge, as the CBD/SBSTTA (2007) report indicates several barriers exist with respect to the effective implementation of the EA:

“Ineffective stakeholder participation in planning and management; inconsistent use of terminology and definitions; the lack of capacity for decentralised and integrated management; insufficient institutional cooperation and capacity; lack of dedicated organisations able to support delivery of the EA; and overriding influence of perverse incentives and conflicting political priorities” (pg. 5)

This position is also supported by those who emphasize that the primary challenge to the effectiveness of the ecosystem approach resides in cross-disciplinary and cross-sector collaboration and coordination and the harmonization of different and diverse views and values, many of which may be oppositional (Richter et al., 2015). Part of the problem here is terminological, which by extension creates communication and comprehension challenges, as a recent assessment of the implementation of the EA in England has revealed (Natural England, 2015:3):

“The general public find it difficult to understand the terminology use to describe the ecosystem approach.”

Furthermore, although the Malawi Principles are a good starting point they still remain within the standard ecological paradigm, those aspects that do have a social perspective (principles 1, 2 , 11 and 12) are generally rather insubstantial and as references to the social dimensions of the environment they do not, in an overtly explicit manner, mention matters of governance, institutions, economics, and human capital etc.: in this sense they shy away from fully acknowledging the human-dominated nature of ecosystems and the transformation of those ecosystems into landscapes, which comprise the fundamental units of operation where planning, management, conservation, society, culture, human wellbeing, economics and governance intersect (Wu, 2013; Bastian et al., 2014).

If the ecosystem approach seeks to provide a holistic mechanism for the beneficial management of natural resources at the landscape level (Waltner-Toews et al., 2008), then ecosystem services ought to be understood as a mechanism for the production of services necessary to meet social and environmental objectives in a landscape context (Anton et al., 2010; de Groot et al., 2010). We argue that a landscape perspective provides a stronger, more rooted, social-ecological foundation and framework for ecosystem services to be situated in, and we believe it also has the added benefit of greater conceptual clarity, particularly from a planning and management viewpoint, and through a change of terminology is also far more easily comprehended and communicated (Bastian et al., 2014). For these reasons, we argue, adopting a “landscape approach” rather than an “ecosystem approach” may provide a more inclusive and effective decision-making arena for those involved in management and planning to coalesce around – the idea being that this would lead to more effective integrated management solutions capable of jointly maximising environmental and social benefits (Termoshuizen and Opdm, 2009). For example, as Swanwick (2009b:S65) notes:

“A landscape-centred approach, the interpretation of land as landscape may provide a more appropriate spatial framework than ecosystems for considering goods, services and benefits.”

15.2 A Landscape Approach

The use of landscape as a term for the combined management of social-ecological systems is gain-ground (Freeman et al., 2015; Reed et al., 2016). Evidence indicates that landscape-scale approaches have increased as sectoral modes of tackling environmental conservation issues have become increasingly inadequate, and their gradual evolution has converged on a more nuanced appraisal of the old conservation and development dichotomy, reconciling these often juxtaposed opposites through “interventions in different components of the landscape” (Sayer et al., 2013). For example, as Sayer et al., (2013) chart, the term ‘landscape approach’ refers to a multifaceted and diverse array of programmes that seek to:

“...address complex interactions across spatial scales and embrace the full complexity of human institutions and behaviour.” (pg. 8350)

This perspective is supported by Bastian et al., (2014:1464) who remark that:

“The attractiveness of the landscape approach results to a great extent from the fact that it contains natural, cultural and utility aspects alike, and sees the landscape as a hybrid system, with the interplay of nature, society and technology.”

Indeed, a recent analysis showed that there are over 80 terms referring to integrated styles of landscape management (Reed et al., 2016). However, whilst the terminology may

vary, as to some extent so does the application, there is a commonality of understanding that landscape approaches are concerned with:

“The landscape approach seeks to address global challenges of poverty alleviation, food security, climate change and biodiversity loss. Although it can be viewed as a refinement of prior approaches, it is distinct as it explicitly acknowledges that satisfying all stakeholders will often be unachievable. By bringing together the diverse range of stakeholders operating within the landscape and attempting to understand what each of their requirements and expectations are, trade-offs and synergies can be identified. Management plans should then aim to capitalize on the synergies while the trade-offs will enable planners to identify who is losing out and as such appropriate compensation or alternatives can be sought. Therefore, the landscape approach attempts, through participatory, inclusive negotiation and planning to minimize trade-offs and maximize synergies so that there are fewer losers and more winners.” (Reed et al., 2016:2544)

A similar view is arrived at by Freeman et al., (2015:3) who describe the landscape approach in the following way:

“...as addressing social-ecological systems at the landscape scale; related to resource management and/or environmental goals, and framed around the concept of multi-functionality, with the aim of achieving multiple objectives through the approach.”

Consequently landscape perspectives have become part of a number of global environmental initiatives, moving the focus from project-oriented action endpoints to the process by which actions are achieved (Sayer et al., 2013). This chimes with the idea of ‘action through landscapes’, what Swanwick (2009a:347) describes as:

“...landscape offers spatial units with a degree of unity in their character, and within which data can be assembled and policies can be created and delivered”.

Whilst we do not dismiss the variety of terminological expressions and applications of the landscape approach as unimportant, they are less problematic than the equivalent issues facing the ecosystem approach for two reasons: First, as we revealed in Chapter 14, landscape keys into an intuitive aspect of peoples’ understanding of the world – it may therefore as a result be many layered and multi-stranded, perhaps verge on the nebulous, but at the same time people have a deep sense of what it means, and what it means to them – the same cannot be said for the term ‘ecosystem’ and by extension the ‘ecosystem approach’ which is bound-up in ecological and scientific language. Secondly, “landscape” and the “landscape approach” are inherently cross-cutting framings naturally drawing together environmental, social, economic, political, and institutional dimensions (Freeman et al., 2015; Reed et al., 2016) and they thus also attach particular significance to the social and health benefits of landscape (Swanwick, 2009b). There have been other issues levelled at the landscape approach mainly relating to how various stakeholders communicate, engage and exchange information, knowledge and

ideas with each other, but by and large these are not systemic and entrenched problems instead they are issues faced by any developing discipline (Reed et al., 2016).

Where the landscape approach makes significant strides is in its worldview (Box 15.2). These principles argue that a person-centred approach is the preferred route to, for example, balance agricultural production and environmental priorities, with the landscape scale being the arena in which decisions regarding trade-offs or co-alignment between productive activities and conservation are most suitably undertaken (Sayer et al., 2013). These principles are not prescriptive guidelines but are considerations that an effective landscape approach ought to consider (Sayer et al., 2013; Reed et al., 2016). The principles outlined in Box 15.2 progress further than the Malawi Principles by more explicitly connecting social and ecological dimensions, directly expanding upon the social principles 1, 2, 11 and 12 of the Malawi Principles and highlighting the importance of communication and knowledge exchange. In this sense the landscape approach makes both conceptual and practical advancements. It is not simply a change of name, from ecosystem to landscape, but a change of ethos, of a worldview; it is more profound than banal, it is rich and complex as the notions of landscape described in Chapter 14 emphatically conveyed from which it draws.

Box 15.2 Principles of a Landscape Approach

1. Iterative management based on social learning, adaptation and trans-disciplinarity (cross-sector and cross-disciplinary collaborations)
2. Common entry points of concern (environmental, social, cultural (including spiritual), economic, institutional, political)
3. Address multiple-spatial and temporal scales
4. Approaches employed should target ecosystem service multi-functionality
5. Multiple stakeholders and constituencies (implementation requires a significant stakeholder participatory approach which should necessarily consider power-relations and agenda setting).
6. Strengthened stakeholder capacity (focused management decisions are required, which necessarily should include a human perspective and be easily conveyed and understood by multiple stakeholders).
7. Trust building – connecting key individuals and developing partnerships between actors in the development of a shared vision is essential.
8. Clarification of rights and responsibilities
9. Trade-offs between conflicting social objectives should be explicitly considered, including those that concern synergies and antagonisms between short and long-term ecological, social and political goals. Biophysical trade-offs between different ecosystem services should be addressed as well as the implications of these trade-offs for service providers and service users.
10. Implementation ought to be aware of legitimacy, social dynamics and costs (including a focus on social inequalities and social justice)
11. Participatory and user-friendly monitoring
12. Embed principles of sustainability, resilience and good governance, and focus on connections between environmental factors and human wellbeing.
13. Ensure institutional fit (management scales should match relevant ecological scales and should be facilitated by cross-sector cooperation)
14. Improve knowledge communication and exchange

Based on: Sayer et al., (2013), Freeman et al., (2015) and Reed et al., (2016)

This idea is perfectly captured by Sayer et al., (2013:8350) who acknowledge that:

“Landscapes provide the setting over which wicked problems unfold, and the landscape approach provides the social ecological systems’ framework by which we can grapple with them.”

Therefore, to reiterate:

“A landscape approach is a multi-faceted integrated strategy that aims to bring together multiple stakeholders from multiple sectors to provide solutions at multiple scales” (Reed et al., 2016:2551)

Based on these views we present our version of the landscape approach pictured in Figure 15.1. Effectively, this diagram crystallises the principles outlined in Box 15.2 but at the same time also explicitly presents ecosystem services and human-wellbeing as central tenets of the landscape approach.

15.3 Final Remarks

In this chapter, what we have sought to describe and layout our argument for a landscape approach to environmental management and to capture that both in terms of a set of principles but also diagrammatically. By doing this we have tried to emphasise the centrality of the connection between landscape – ecosystem services – and human-wellbeing. We have suggested that this development improves upon the difficulties associated with the ecosystem approach, and although we have acknowledged that the landscape approach has difficulties too, these are both lesser and more easily overcome than in comparison to the ecosystem approach. We have therefore tried to argue the strengths of the landscape approach but also, I hope, to have pointed out that the landscape approach draws upon the ecosystem approach (as instituted in the Malawi Principles). We see landscape, as represented in Chapter 14, as uniquely qualified for integrating ecosystem services, human-wellbeing and social-ecological thought into a vehicle (i.e. the landscape approach) that is comprehensible to all stakeholders, is conceptually robust and rich and has significant practical applications.

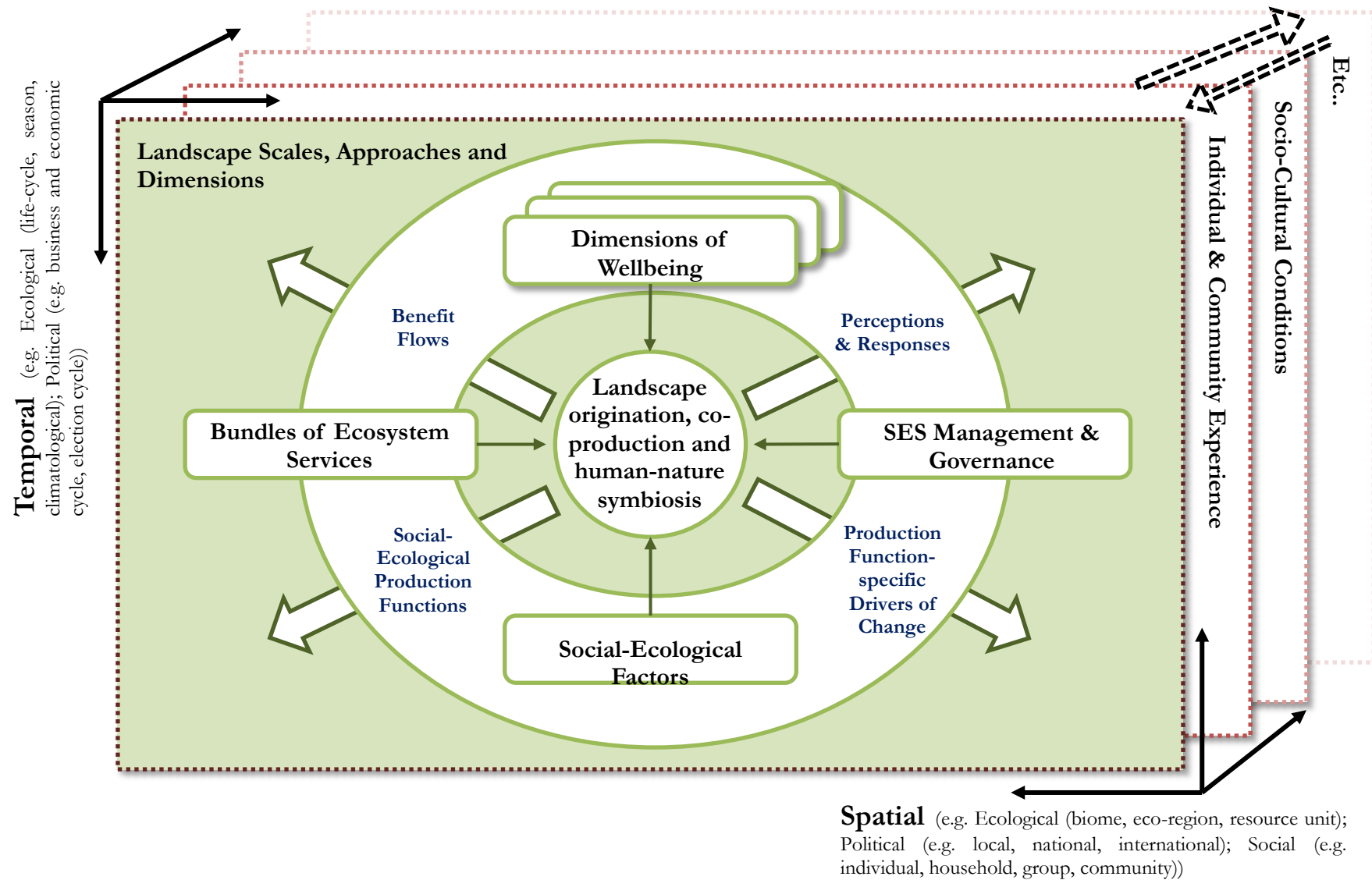


Figure 15.1 Landscape Approach capturing ecosystem services and human-wellbeing within a multi-scaled and multi-dimensional social-ecological framing. Landscape is situated in the centre of the framework where it interacts with and is co-produced by four main factors: human-wellbeing, governance, social-ecological factors and ecosystem services. Fuzzy edges around dimensions indicate that these are not bounded but in fact merge and interact with each other. Dotted arrows indicate cross-scale interactions and feedbacks (adapted from Reyers et al., 2013)

Notes

1. As defined in De Lucia (2015) eco-centrism is effectively a collection of counter-narratives against the dominant anthropocentric position or a 'counter-hegemonic ensemble of narratives' which seeks to expose the falsehood of the human-nature split.

2. In the context of De Lucia's (2015) legal examination of the ecosystem approach anthropocentrism refers to the dominant frameworks in legal and international policy fora, for example, regulations, protections etc. concerning the environment that privileges the human perspective. But, more than that, anthropocentrism goes one stage further and actively excludes certain aspects of humanity and natural ecosystems through the process of liberal modern law making. In other words, there is the potential for environmental laws to actually increase, as opposed to actively reducing, environmental harms.

Chapter 16: Landscaping Ecosystem Services

“Abandon the urge to simplify everything, to look for formulas and easy answers, and to begin to think multidimensionally, to glory in the mystery and paradoxes of life, not to be dismayed by the multitude of causes and consequences that are inherent in each experience – to appreciate the fact that life is complex.” (M. Scott Peck, American Psychiatrist)

“Today the network of relationships linking the human race to itself and to the rest of the biosphere is so complex that all aspects affect all others to an extraordinary degree. Someone should be studying the whole system, however crudely that has to be done, because no gluing together of partial studies of a complex nonlinear system can give a good idea of the behaviour of the whole.” (Murray Gell-Mann, American Physicist)

The two previous chapters dealt with the significant and multidimensional qualities and meanings attached to landscape (Chapter 14), and from there went on to argue, based upon those multidimensional properties, that a landscape approach provides a more effective framing for situating and illuminating the connections between ecosystem services and human-wellbeing in the context of environmental management (Chapter 15). In Chapter 16 we spell out in more detail what landscape and the landscape approach means for ecosystem services in both theory and practice. We start by emphasising that our new framework and approach is integrative and based on social-ecological complexity; from there we then drill down more deeply and explain the significance of the landscape approach framework we presented in Figure 15.1. What we describe in this chapter is the landscaping of ecosystem services.

16.1 Setting The Scene

16.1.1 *The Call for Holism*

Holism derives from the Greek word “Holos” meaning “all, whole, entire”, and according the Oxford English Dictionary (OED) was coined by Gen. J. C. Smuts (1870–1950) to “designate the tendency in nature to produce wholes (i.e. bodies or organisms) from the ordered grouping of unit structures”. In more modern parlance, reflecting a philosophical attribution, holism is taken to mean (according to Oxford Dictionaries):

“The theory that parts of a whole are in intimate interconnection, such that they cannot exist independently of the whole, or cannot be understood without reference to the whole, which is thus regarded as greater than the sum of its parts.”

Although we investigate many systems at a disaggregated sub-system level: a mechanism that reduces system-complexity and enables more straightforward methodologies and

hypothesis-driven approaches to answer increasingly focused questions – we walk a fine line by viewing everything through these disaggregated lenses:

“Human perversity, then, makes divisions of that which by nature is one and simple, and in attempting to obtain part of something which has no parts, succeeds in getting neither the part – which is nothing – nor the whole, which they are not interested in.” (Book III, *The Consolation of Philosophy*, Boethius A.D. 524)

The inclination is to presume that these discrete systems are representative of the “whole”; to the extent that if we were to understand every sub-system the “whole” would be rendered to us. This mode of analysis is adopted as a universal problem-solving panacea, but equally as profound, it becomes a system of thought that precludes other correspondingly reasonable means of endeavour. This failure can often result in the exacerbation of the problems we are seeking to resolve by creating spurious adaptations and short-termism in the solutions we develop (Norberg and Cumming, 2008).

In general, we rarely acknowledge that complex systems display characteristics that are not predictable from or reducible to a disaggregated level, and thus the need to employ alternative analytical strategies arises. At this point we must think about systems in more holistic terms and realise that this perspective of analysis is entirely compatible with a sub-system approach, and indeed, is essential for understanding complex macroscopic problems and the relationships between macro and micro scales and vice versa (Norberg and Cumming, 2008). Ultimately, both are required if we are to seek a more sustainable future where the complexities existing between social, economic, political and ecological processes and patterns are acknowledged and addressed concomitantly (Norberg and Cumming, 2008; Waltner-Toews et al., 2008).

16.1.2 Complexity And Scale: Necessary Understandings

The critical facets of interest with regards to the patterns and processes underlying social, economic and ecological phenomena and their connections tend to be located in discrete entities that exist within, at and across dimensions of time and space (Cumming and Norberg, 2008). These aspects are embodied in the definitions given to both ecosystem and society. According the Convention on Biological Diversity (Article 2) ecosystem refers to:

“...a dynamic complex of plant, animal and micro-organism communities and their non-living environment interacting as a functional unit.”

The fact that the identified “functional unit” is not further refined means that it can refer to any functioning unit at any scale. Society on the other hand variously refers to:

“...the state or condition of living in company with other people (i.e. in a community); and the system of customs and organization adopted by a group of people” (Oxford English Dictionary, 2014).

Both definitions emphasises scale, organisation and interaction as fundamental qualities: properties observed in complex systems (Mitchell, 2009).

Alongside the development of complexity theory pioneered by the Santa Fe Institute in California, the notion of complex adaptive systems (CAS) has gained prominence, as ‘systems’ that are defined by a number of shared common features (Levin, 1999; Wu and Marceau, 2002). These features include: (i) the capacity to demonstrate complex non-linear behaviours; (ii) a hierarchical organisation of system components (generally); (iii) multi-scale composition; (iv) feedback loops between components and scales, and (v) the capacity to self-organise and display emergent properties (Redman et al., 2004; Cumming and Norberg, 2008).

Scale is both an elemental feature of CAS and a fundamental tool with which to understand CAS, and underscores the proposition that complex systems are nested, comprising multiple sub-systems that function hierarchically (Berkes et al., 2003). Indeed, it is the resolution of scale that enables the differentiation between the “whole system” and “sub-system” elements to be recognised (Cumming and Norberg, 2008). Scale is quintessentially important because social and ecological systems occur across multiple scales, such that the way in which they operate (i.e. their magnitude) and the features they exhibit (i.e. their topology) may vary in a scale-dependent or scale-independent manner (Cumming and Norberg, 2008). For biophysical systems scale is generally understood within a relatively straightforward spatio-temporal frame of reference (e.g. species-area relationships and body size distributions) (Levin, 1998), whilst in social systems scale is also associated with notions of governance, institutions, organizations and power (Gibson et al., 2000). These scale related attributes and changes that occur across scales are fundamental to our understanding of how CAS function (Cumming and Norberg, 2008).

Non-linearity and uncertainty, meaning that CAS may organise around more than one equilibrium state, such that, as the conditions the system experiences change (for whatever suite of reasons) a critical threshold maybe reached whereby the system “flips” to an alternate “operating” state are fundamentally properties of scale (Berkes et al., 2003; Cumming and Norberg, 2008). Thus the mechanisms by which sub-system units interact and respond to interactions, in a networked way, to internal and external driver’s influences the behavioural response of the “whole system”, and in this respect cross-scale interactions are also important (Cumming and Norberg, 2008). For example, attempts to control “local uncertainty” or “variability” can result in fluctuations at higher spatial scales creating system sensitivities and

feedbacks that can result in unintended outcomes often termed “the conservation law of fragility” (Norberg and Cumming, 2008).

Hierarchical attributes to complex systems conveyed in concepts such as Koestler’s “Holons” (Koestler, 1967) and Holling’s “Panarchy” (Holling, 2001) represent a fundamental construct of CAS as multi-scaled systems exhibiting cross-scale interactions. The important insight here is that such interactions may occur in both bottom-up and top-down directions, affording the possibility that between scales systems may display “predictable” changes in behaviour and properties (Cumming and Norberg, 2008). The potential predictability of such scalar behavioural changes has important implications for recognising and understanding the influence of both positive and negative feedbacks within, and on, the system (Cumming and Norberg, 2008).

Linked to the scalar clustering of particular system attributes are the qualities of diversity and asymmetry and their associations with heterogeneity (Cumming and Norberg, 2008). Landscapes are often populated by smaller sub-regions of individual identity (i.e. locations) with the shared connection between them being simply their proximity to each other; such “distinct groups” are formed through the process of asymmetry (Cumming et al., 2008). Asymmetry is central to many ecological as well as socio-economic processes and can arise through internal and external drivers associated with biotic, abiotic and anthropogenic sources all of which can affect ecological community structure and environmental quality, as well as the creation of physical gradients (e.g. elevation) and boundaries (e.g. edge effects) in addition to influencing social structures, resource access and ownership (Cumming et al., 2008).

These qualities are particularly evident for landscapes in the context of heterogeneity where the temporal and spatial arrangement of features is especially important for the way landscapes function, particularly from a pattern-process standpoint (Cumming and Norberg, 2008). So, for example, due to the spatial and temporal arrangement of certain landscape properties particular processes may occur in a dispersed and multi-scalar manner (e.g., carbon storage and sequestration) or may be more highly localised (e.g. species dispersal). This can create ‘dependencies’ within parts of the system (e.g. sources and sinks), such that they can affect system diversity (e.g. ecological diversity) (Cumming and Norberg, 2008). Diversity, however, is a property that not only applies to the ecological sphere but also finds meaning in the social realm, for example, in relation to institutions or knowledge systems (Norberg and Cumming, 2008). In all cases tensions between diversification and selection within CAS, which are strongly related to the concepts of novelty, innovation and adaptation, shape the network dynamics of complex systems and their future development and sustainability trajectory (Norberg and Cumming, 2008).

Bridging these aspects is the idea of “self-organisation”, which is to say, that systems can form around points of instability in a manner that is dependent upon the system’s history, a kind of “path dependency” (Berkes et al., 2003). The capacity for landscape self-organisation is a process underpinned by asymmetry in pattern-process relationships (Cumming et al., 2008). Collectively, these features enable the ‘emergent’ capacity of the system to develop, in other words, to demonstrate qualities that cannot be predicted from an assessment of the system’s individual component parts i.e. the whole is greater than the sum of the parts (Berkes et al., 2003).

The extent of these so-called emergent properties is related to the size of the CAS, as size fundamentally dictates the magnitude of internal system processes alongside the capacity of the system to respond to external drivers. Hence the dynamical relationship between internal and external environmental interactions and variables is essential for appropriately managing the system (Cumming and Norberg, 2008). Properties such as resilience (i.e. the capacity of a system to absorb and adapt to change), robustness (i.e. the structural aspects of a system that allow the system to withstand disturbance in a manner that does not alter the system’s dynamics) and vulnerability (i.e. the absence of resilience and robustness) have been described as emergent properties of complex systems (Young et al., 2006).

16.2 Embedding Ecosystem Services in the Landscape

It is clear from Section 16.1 that holism and complexity, especially accounting for scale, ought to be central components of any approach (e.g. Burkhard et al., 2010) seeking to understand as well as implement sustainable and effective environmental management strategies that are capable of adequately delivering both environmental and social benefits (e.g. contributing to biodiversity conservation whilst also improving local livelihoods and sustaining basic levels of agricultural production). In addition, environmental strategies have to be inclusive and foster engagement as well as be clearly communicated by drawing-upon shared meanings, values and experiences between those directly engaged in management activities as well as those that benefit indirectly from the broader outcomes of management policies and programmes (Reed et al., 2010).

Combining and extending several recent frameworks (i.e. Wang et al., 2011; Bastian et al., 2012; Burkhard et al., 2012; Polishchuk and Rauschmayer, 2012; Bastian et al., 2013; Fisher et al., 2013; Villamanga et al., 2013; Reyers et al., 2013; Chapman, 2014; Duraiappah et al., 2014; McGinnis and Ostrom, 2014; Van Reeth, 2014; Spangenberg et al., 2014) this paper provides a more integrated and scale-based approach to embedding ecosystem services within a clearer, more policy and management oriented and conceptually robust framing, where

landscape and human action are pivotal elements (Figure 15.1). Below we elaborate further on the main elements of the framework.

16.2.1 Scale And Interactions

Following Fisher et al., (2013), Duraiappah et al., (2014) and McGinnis and Ostrom (2014) in particular scale is explicitly considered in three ways, namely: (i) the landscape approach framework is purposefully polycentric, acknowledging that each layer is nested in an ecological and socio-political governance dimension; (ii) the human experience of landscape and landscape management and policy is identified as the local scale at the individual, household and community level; and (iii) external drivers of the system at each scale, cross-scale interactions as well as feedbacks represent core facets of the framework design, and recognise these features as elements of the system that enable dynamism, change and adaptation.

16.2.1.1 Geographies Of Scale

From soil erosion, to farmer decision making to wildlife conservation and global sustainability issues, to the way actors and nations interact to solve these issues, scale and scale-related arguments are part and parcel of dealing with environmental challenges (Sayre, 2015). Yet although scale is a seemingly ubiquitous phenomenon in the human and social sciences scale as an abstract concept, as a property of social reality as well as a means of examination and its links to spaces for dialogue and interaction (i.e. politics) remains a highly contested and debated term (Moore, 2008; MacKinnon, 2011). There has been a growing discourse of commentary on the “social construction of scale” and “social-spatial relations” (e.g. Marston, 2000; Brenner, 2001; Jessop et al., 2008) as well as on the “politics of scale” (e.g. Delaney and Leitner, 1997; Cox, 1998; Brenner, 2001; Swyngedouw and Heynen, 2003). For example, as Delaney and Leitner (1997:96) explain:

“...the political construction of scale as a theoretical project necessarily involves attention to relationships between space and power, and to conceptions and ideologies of space and power that social actors bring to practical efforts to change the world and, of course, to resist change [...] Where scale emerges is in the fusion of ideologies and practices.”

These ideas are developed further in what Cox (1998) refers to as “spaces of dependence” and “spaces of engagement”, classifications that characterise social-relational events occurring between people and spaces and the interactions between people in those spaces (Cox, 1998)¹. Importantly, these social-relational events and interactions are not limited to a single scale, nor do agents for example, only have one type of “space dependence”: depending on the context and types of interaction multiple space dependencies may occur

simultaneously (Cox, 1998). What is significant is that these dependencies or engagements in some sense are argued to co-create spatial scales, such that an agent belongs or associates with a particular scale, through for example, the idea of “immobilization” as Cox (1998) explains:

“This immobilization in particular spaces of dependence-local economies, job markets, local government jurisdictions, etc.-is something that is shared. It is this sharing on the part of firms, workers, residents, other organizations like state agencies, that creates the possibility of local classes and specifically local interest groups organized along lines defined by the social division of labor.”

The scale component of our landscape approach emphasises that no one particular scale is entirely discrete and independent of the next, but instead is part of a larger continuum (i.e. scale boundaries are fuzzy and merge with each other), and by extension highlights the multidimensionality of human-wellbeing, social-ecological phenomena and environmental policy and governance (Duraiappah et al., 2014). Whilst it seems we are presenting a hierarchical scalar system, in fact we are not, by locating “landscape” at the centre of the framework we recognise that these classifications are socially constructed, not necessarily preformed physical realities nor absolute, and that they exist as a way of understanding and interpreting how the system operates: scale is both size, level and relation (Marston, 2000). In respect of the idea that landscapes and ecosystem services are used and consumed and to some extent socially constructed, then this view also aligns with the argument put forward by Marston (2000) that:

“...scale is constituted and reconstituted around relations of capitalist production, social reproduction and consumption.”

This perspective is supported by those authors identified by Brenner (2001) that are concerned with scale in terms of dynamism and process in the evolution of social practice. In his praiseworthy appreciation of Marston’s (2000) essay, Brenner’s (2001) critique is at pains to point out that scale is also over-extended and generalized: the problem this creates is that other important conceptual notions and categories such as locale, place, and territory cannot compete and are subsumed. In the same article Brenner makes the case for differentiating between a singular and plural form of the “politics of scale”. In the singular version scale effectively functions as a boundary concept; whereas in the plural description scale becomes about the differentiation of spatial units and their embeddedness, and so the pluralistic interpretation advances the idea of ‘scaling processes’ (Brenner, 2001)². By extension Brenner (2001:601) argues that:

“...geographical scale appears to ‘matter’ most to social outcomes – that is, to have the most obvious and far-reaching causal impacts – in those social processes or transformations which are described through a plural rather than through a singular notion of a politics of scale.”

By defining the “politics of space” in these terms Brenner aims to differentiate between these ‘structuration’ processes of social-space (i.e. the relational interactions of differentiated spatial units) and other socio-spatial forms of structuration such as “place-making” and “territorialisation”, and thereby reimagine the politics of scale as the “politics of scaling” (Brenner, 2001). The idea of a “politics of scaling” has been more recently supported by MacKinnon (2011:23) who makes the case:

“...that it is often not scale *per se* that is the prime object of contestation between social actors, but rather specific processes and institutionalized practices that are themselves differentially scaled.”

Importantly, landscape shares multiple structuration processes which help’s to move beyond a simple hierarchical assessment of ‘landscape space’. From this perspective we agree with some of the arguments raised by Marston et al., (2005) that vertical and hierarchical scale constructions can produce rigidity in thought and can act to determine the nature and state of objects and processes in a preformed manner.³ We are less concerned with arguments expressing the view that there is confusion between scale as size and scale as level because, as we previously remarked, by invoking landscape we emphasise social-ecological complexity and in that respect accord with the idea that it is possible to consider complex systems in a way that does not necessitate a hierarchical scale view, but instead can be examined in relation to flows and fluidity⁴ as well as material composition and decomposition⁵ (Marston et al., 2005). At the same time we do not reject scalar thinking altogether in favour of a purely “flat ontology” of scale using Marston et al (2005) phrase, instead we argue, after Jessop et al., (2008), that our landscape approach presents a multiplicity of socio-spatial relations which are “mutually constitutive” and “relationally intertwined”: this appraisal chimes with the notions of landscape sketched out in Chapter 14 and the processes and service flows that characterise ecosystem services.

By explicitly recognising scalar interactions and dependencies the our framework considers context to be an especially important factor in shaping social-ecological properties, and in particular, human engagement with the landscape as facilitated by human experience, values and beliefs manifested in decision-making processes and actions (Duraiappah et al., 2014; McGinnis and Ostrom, 2014). We also recognises that human actions are partially scale-specific and vary accordingly as individual values, beliefs and attitudes become subject to different synergistic and antagonistic social, economic and political factors (or pressures) related to particular challenges and barriers, cognitive dissonance and governance and policy drivers (Fisher et al., 2013; Duraiappah et al., 2014).

In the presentation of our landscape approach we would advance the view put forward by Sayre (2015), which at its heart states that we need to consider both the methodological and epistemological dimensions of scale: heeding the lessons of previous debates while at the same time recognising that we need to be pragmatic both in how we deploy scale in examining systems as well as in terms of the insights we hope to gain:

“Do not take the scales of one’s analysis for granted; identify the key processes that produce a phenomenon, and induce their scales empirically; be alert to how processes are rescaled, and to the possibility of non-linear, qualitative change across scales; be reflexive and critical about how observational scales may affect the patterns one sees in the resulting data. Overall, these guidelines suggest an open-ended approach to scale, with the potential for a virtually limitless array of particular applications.” (pg. 511)

16.2.1.2 Ecological Scales

Ecological scales are significant for the provision, distribution and values attributed to ecosystem services, with regards to supply and demand as well cross-scale interactions and potential service trade-offs (Cash et al., 2006; Burkhard et al., 2010; Scholes et al., 2013). Ecologists use scale in an non-fixed hierarchical sense, as Sayre (2015:509) explains:

“...ecologists frequently use hierarchy theory as a heuristic framework, with “levels” defined in a loosely functional sense (e.g., organism, population, community). But this is understood not as an exclusive hierarchy (in the state-bureaucratic, top-down sense discussed earlier) but as a constitutive hierarchy, in which phenomena at a “lower” or smaller scale may display different patterns when aggregated at a “higher” or larger scale—patterns that are irreducible to their smaller-scale components (so-called emergent properties, aka the whole is greater than the sum of its parts).

At their most fundamental ecological scales relate to biophysical processes and ecosystem functioning and connectivity, but in the context of a landscape approach, they also relate to how the connections between these biophysical scaling processes mediate the provision of ecosystem services at levels and magnitudes capable of meeting the needs of beneficiary populations (i.e. consumption demands), alongside other conservation and environmental policy objectives and competing land-uses (Bennett et al., 2009). From this standpoint, the “resource unit” represents the smallest ecological-scale through which human activities are mediated and in essence can be regarded as a subset of the landscape more broadly acknowledged (Duraiappah et al., 2014; McGinnis and Ostrom, 2014).

16.2.1.3 Governance And Management Scales

Governance scales acknowledge the plethora of simultaneously operating (and thus potentially competing and/or overlapping) policy drivers and management programmes, as well as customary and statutory laws underpinning the regulatory structures governing natural

resource access and use (McGinnis and Ostrom, 2014). Governance recognises the spatial qualities of political decision-making processes and shaping processes (Görg, 2007). In this respect scale provides a means of negotiating what Duraiappah et al., (2014) refer to as the problem of “institutional fit” (i.e. how to correctly align both “allocative”⁶ and “distributive”⁷ institutions), which are particularly prominent when considering the different characteristics exhibited by common pool resources and public goods (Wyborn and Bixler, 2013; Duraiappah et al., 2014). In many ways the obstacles encountered in the field of environmental governance arise from the spatial and temporal alignment or (more likely) mis-alignment of politics and environmental problems (i.e. in terms of their physical and social impacts), which have implications for the development of effective environmental policies and management strategies. For example, taking a spatial perspective, the mismatches between nation state boundaries and transboundary environmental problems, or from a temporal point of view, the disconnect between short-term political cycles and long term challenges like climate change (Meadowcroft, 2002). In this sense, our landscape approach can be seen to promote the idea of ‘landscape governance’ articulated by Görg (2007):

“...landscape governance deals with the interconnections between socially constructed spaces (the politics of scale) and “natural” conditions of places.” (pg. 954)

Thus, as Görg (2007:957) goes onto explain:

“The success of governance processes at one of the levels, for example the local level, is therefore dependent upon its relationship to other levels (referring to national laws, international agreements, regional and local actor constellation, etc.). Thus, the relationships between the levels and the possible hierarchies that exist between them are of major significance.”

However, at the same time, the significance of those socially constructed spaces, in terms of where effective environmental governance policies are formulated, have in one sense changed significantly:

“...in a post-Fordist constellation [...] The national state is no longer the central, pre-determined level of political processes; nor does it command the greatest power of shaping in relationship to other levels. Conversely, it is the international level (or the supranational level in the case of the EU), which, in many respects has attained primacy in the agenda setting process and in policy formulation, insofar as, for example, environmental measures are today often initiated and decided at the international (or EU) level. Nevertheless, the national level retains an essential function with regard to implementation, as evidenced by the way in which most international (environmental) agreements has generally only weak sanction mechanisms in order to enforce national compliance.” (Görg, 2007, pg. 958)

Yet, it is at the level where space is socially defined and produced and humans interact with the environment that environmental governance becomes concrete, has meaning and

resonance (Görg, 2007). Our proposed landscape approach acknowledges the intimate link between landscape and governance and in that way supports the contention of Görg (2007:961) that:

“What seems to make the landscape concept useful as a link between governance processes in multi-level-politics and natural–spatial conditions is precisely its hybrid character, that is, that societal and “natural” factors are intrinsically linked to one another. Cultural, aesthetic, economic and social dimensions are as much involved as ecological functioning or abiotic conditions.”

Adopting a more practical management stance, spatial and temporal scales are central to understanding landscape and pursuing management interventions and scientific evaluations (Pedroli et al., 2006). Scale is a core component of conducting ecosystem service assessments, a key property in identifying potential sources of uncertainty emanating from methodologies that attempt to negotiate scaling processes, and is an important factor to consider when accounting for the problems of aggregation (i.e. scaling) and disaggregation (i.e. downscaling) (Seppelt et al., 2012; Hou et al., 2013; Scholes et al., 2013). We therefore advocate, following other researchers, multi-scale and cross-scale assessments in order to: (i) evaluate ecological and social processes at the scales they operate; (ii) provide more spatial, temporal and causal information; (iii) provide independent validation of broad-scale processes by small-scale investigations, and (iv) foster a closer alignment between reporting scales and decision-making scales (Scholes et al., 2013).

16.2.1.4 Interactions

Evaluating cross-scale interactions is necessary to quantify off-site land management impacts. These can be particularly significant at larger regional scales where they can lead to sub-optimal outcomes and feedback negatively to lower spatial scale, such as the effects of the European Union’s bioenergy policy on land-use patterns in Indonesia (Seppelt et al., 2013). From this standpoint, the interaction and transmission of small-scale changes and perturbations to larger spatial scales and vice versa are important and these may be rapid, such that responses are amplified or slow manifesting a buffered response. Ultimately, rates depend on several factors (e.g. spatial configuration, intra- and inter-scale connectivity and intra- and inter-scale flows) and so are highly variable as they interact with and are affected by broader-scale forcing factors and feedbacks (Peters et al., 2004).

16.2.1.5 Scale Investigations

Finally, a landscape approach that does recognise scale affords an array of opportunities for analysis and investigation in relation to scale-specific and cross-scale effects with regards to ecological, social, economic, governance and management impacts and

outcomes (McGinnis and Ostrom, 2014). For example, assessing ecosystem service supply and demand budgets (e.g. Burkhard et al., 2010; 2012), developing land management and ecosystem services scenarios (e.g., Fleskens et al., 2014; UK NEA, 2011; 2014), evaluating the social and economic impacts of land management programmes on suppliers and beneficiaries of ecosystem services (e.g. Porras et al., 2008), and investigating the institutional regimes under which environmental policies are designed, framed and implemented (e.g. Young, 2010).

16.2.2 Social-Ecological Factors – Ecosystem Services Bundles – Human-Wellbeing

Ecosystem service occupy a position that recognises that they are subject to landscape change but also acknowledges that their flows underpin human-wellbeing, and so as a consequence they are regarded as being situated within a broader supply and demand chain (Burkhard et al., 2012). By extension, our landscape approach accepts that ecosystem service provision is the result of a combination of underlying biophysical processes, land management activities and individual and societal needs and preferences interacting across multiple scales (Paetzold et al., 2010). This is significant because the ability to distinguish between the different components of the productive base and the realised ecosystem services is central for effective land management (Burkhard et al., 2012; Spangenberg et al., 2014). Figure 16.1 presents a more in-depth and detailed description of the relationship that links ecosystem service bundles to human-wellbeing identified in Figure 15.1.

16.2.2.1 Ecosystem Services Supply: From Processes To Flows

Beginning with the productive base, we distinguish four components: ecosystem capital, ecosystem service potentials, ecosystem services benefits and values, and ecosystem service flows and capacity. We use the term “ecosystem capital” rather than “natural capital” to connote that ecosystems, in particular, are a form of renewable natural capital from which various outputs (e.g. flows of services) are ultimately derived (Van Reeth, 2014). Echoing several authors (e.g. Haines-Young et al., 2012; Fisher et al., 2013; Braat, 2014 and Spangenberg et al., 2014) we make the distinction between ecosystem structure and processes (Est & P) on the one hand and ecosystem service functions (ESF) on the other. Here ESF (e.g. wood production) arise from Est & P (e.g. net primary production), what Spangenberg et al., (2014) refer to as “elements” and collectively subsume under the category of “ecosystem properties” (Bastian et al., 2012). Thus, ecosystem properties are as Bastian et al., (2012:8) remark:

“...prerequisites for the human-wellbeing that is based on the natural capital of the earth from which goods and services are generated.”

Consequently, ESF are the first stage in a sequence leading to the production of ecosystem services and could best be described as representing “biophysical traits”, which are separate from but also related to ecosystem service potentials (ESP) (e.g. wood use for fuel), a term which recognises the early intervention of human agency in determining later stage ecosystem services (Bastian et al., 2012; Spangenberg, 2014; Spangenberg et al., 2014).

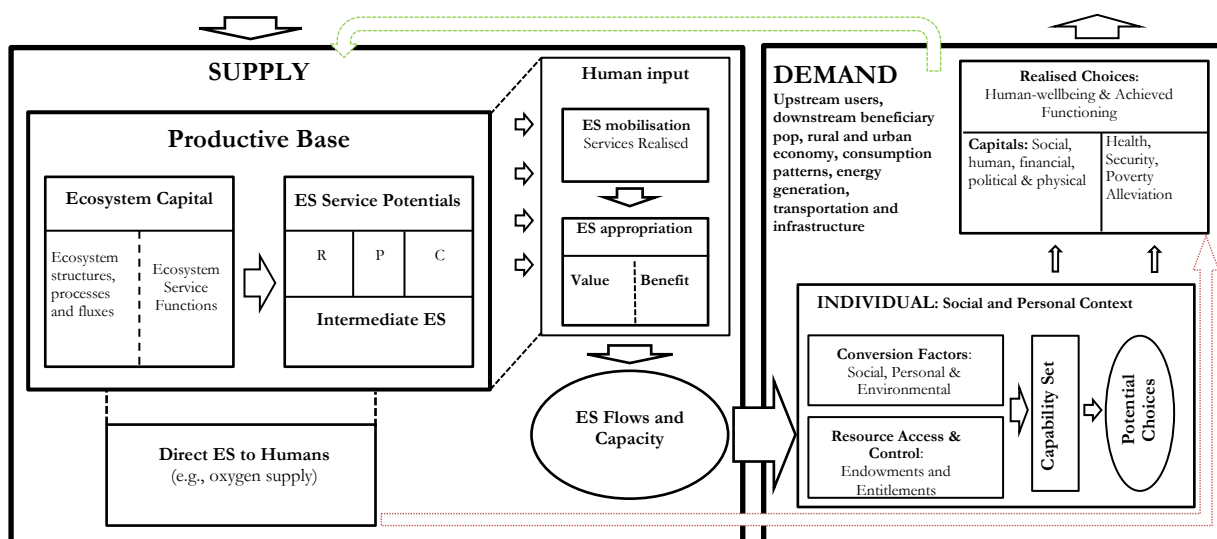


Figure 16.1 Presents a demand-supply framing of the production and provision of ecosystem services and links to human-wellbeing and welfare. Black-bordered arrows signify the direction of flow through the system. The red perforated arrow indicates that these ecosystem services directly underpin human-wellbeing. The green perforated arrow indicates a feedback link between demand and supply.

In this sense, as Fisher et al., (2013) describe, ecosystem service functions (or “ecological functions”) represent the phenotypic articulation of ecological processes and thus partly underpin the generation of provisioning, regulating and supporting services (Fisher et al., 2013). Why in part rather than in whole? Departing from Fisher et al., (2013) somewhat, and using Spangenberg et al., (2014) line of reasoning, ESP represent a separate step in the ‘classical’ cascade formulation of ecosystem services and effectively establish a proto-ecosystem services phase, which result from complex socio-cultural processes that determine the form of the finally realised set of ecosystem services (Spangenberg, 2014; Spangenberg et al., 2014). Essentially,

“...potentials describe the opportunity to use structures and processes of ecosystems and landscapes” (Bastian et al., 2014:9).

To some extent ESP are analogous to the notion of “intermediate ecosystem services”, but are also more significant in that they acknowledge the criticality of social processes as determining factors in the production of a specific bundle of ecosystem services through “use-value attribution” (Spangenberg et al., 2014). This is especially true for cultural ecosystem

services, which Fisher et al., (2013) describe as “emergent” properties arising from the unique intrinsic values human attach to ecosystems. The exception to this rule are those “direct” ecosystem services that support human-wellbeing without out any human input into their production or form (e.g. oxygen supply), which we clearly identify as being separate in the framework (Spangenberg et al., 2014).

Including the ESP step explicitly acknowledges the centrality of human agency by requiring human input and intervention (e.g. in the form of labour or capital) to be the transforming factor in converting the potentialities of these proto-services into a suite of finally realised services (Spangenberg, 2014; Spangenberg et al., 2014). Following Spangenberg et al., (2014) we identify two stages of human input described as mobilisation and appropriation. Mobilisation requires direct human investment (e.g. harvesting fuelwood), whilst appropriation converts a “service” into a benefit with associated values through a consumptive act (e.g. contribution of fuelwood to cooking or heating a home) or via commodification in a market setting (e.g. selling wood-based products) (Spangenberg, 2014; Spangenberg et al., 2014). The level of mobilisation and appropriation required is a consequence of the specific bundles of ecosystem services involved (Spangenberg et al., 2014).

Ecosystem service provision is a rate limiting step, the consequence of supply-demand dynamics, the crucial juncture between fundamental ecological processes, human input and human-wellbeing (Villamanga et al., 2013). Following Villiamanga et al., (2013) our framework recognises four components that are significant for ecosystem service provision and delivery, namely: capacity, flow, demand and pressure. Capacity refers to the underlying ability of the ecosystem to deliver a bundle of services (e.g. water supply, carbon sequestration) given a specific set of biophysical and social conditions, which directly relates to flow in terms of the amount of ecosystem service produced and used (e.g. quantity of crop harvested and consumed). Both capacity and flow are contingent and dynamic; however, whilst capacity is site-specific flow is not constrained to locations of ecosystem service production (Villamanga et al., 2013). Rather, a flow is related to the area where benefits can be experienced (i.e. a flow is a transmission of an ecosystem service from a source to a user), a situation largely governed by factors that influence beneficiary population distribution and resource access (Bagstad et al., 2012; Villamanga et al., 2013; Serna-Chavez et al., 2014).

Whereas demand denotes the societal requirements for specific ecosystem service bundles, pressures refer to the ecological impacts of both anthropogenic and environmental factors on “supply” which ultimately affect ecosystem service capacity and flows (Bastian et al., 2012). Demand and pressures modulate the system, and can be either internal or external to the particular scale of interest (Fisher et al., 2013; Villamanga et al., 2013). For example, the

impact of UK and European agricultural policies on England's upland systems (Condliffe, 2009) in terms of changes to moorland livestock management (e.g. Gardner et al., 2009) or game and sporting activities (e.g. Sotherton et al., 2009). In relation to external environmental and human drivers or pressures, these factors are not only direct stimulators of system change but they also interact with individuals, groups and communities to affect behaviours via "cultural exchange" for example (Fisher et al., 2013). From an analytical perspective making these components explicit elements of our framework enables a fuller integration with environmental planning and sustainability, facilitating assessments, evaluations and examinations of ecosystem service generation, provision and delivery in the context of changing ecological, social, economic and technological circumstances (Villamanga et al., 2013).

16.2.2.2 Ecosystem Services Demand: Capabilities And Human-Wellbeing

The demand-side element of the framework is the ultimate driver of dynamism within the system, acting as a pulling factor on ecosystem services flows and capacity and a modulator of service flows and capacity through direct feedbacks, and more circuitously, via encouraging the development of environmental management programmes that function to maintain the provision of ecosystem services through changes in land-use and land-cover (Villamanga et al., 2013). Demand also represents the point of conversion where ecosystem services become transformed into that multifaceted concept we call human-wellbeing. In this sense, echoing Bastian et al., (2012:9) ecosystem services can be perceived as the:

“...used or demanded contributions of ecosystems and landscapes to human benefits and the human-wellbeing”.

Importantly in our framework, although we recognise demand as an aggregated property of the beneficiary population and economy etc. (as evidenced in Fisher et al., 2013) we articulate that demand through the prism of the individual. Consequently, we follow Polishchuk and Rauschmayer's (2012) critical insight of conceiving human-wellbeing benefits from ecosystem services through the capability approach (CA).

Using the CA (e.g. Sen, 1999; Robeyns, 2005) the framework recognises the holistic nature of human-wellbeing, in particular, its multi-dimensionality and normative construction, and the importance of human agency and the exercise of choice. As a result, we reject a materialistic conception of human-wellbeing based on monetary and utilitarian values (Polishchuk and Rauschmayer 2012; Pelenc and Ballet, 2015). We emphasise, like CA, the notion of the "good life", in other words, a suite of actions and values both particular and common to individuals (termed "functionings") that when adopted enable individual flourishing (Polishchuk and Rauschmayer 2012). Importantly, functionings can only be

acquired or adopted if individuals have the freedom to achieve. Consequently, when characterised through the CA lens human-wellbeing comprises both “functionings” and “freedoms”, collectively referred to as “capabilities”, with an individual’s particular capabilities identified as their “capability set” (Polishchuk and Rauschmayer 2012; Pelenc and Ballet, 2015).

We agree with both Polishchuk and Rauschmayer (2012) and Pelenc and Ballet (2015) that capabilities underpin human-wellbeing, and that an individual’s capability set depends upon available resources (i.e. goods and services). Goods and services are deemed important to the extent that they affect individual capabilities, and so from this perspective provisioning, regulating and cultural services can be categorised as resources capable of affecting individual capabilities (Polishchuk and Rauschmayer, 2012). How this happens in reality is largely context dependent, relying on people’s ability to “make use” of goods and services (Polishchuk and Rauschmayer 2012; Pelenc and Ballet, 2015). In CA “conversion factors” are used to explain how individuals are able to “make use” of goods and services and enhance their capabilities (Polishchuk and Rauschmayer 2012; Pelenc and Ballet, 2015).

Conversion factors act as a processing lens through which goods and services are metabolised and assimilated, and are normally classified as comprising personal, social and environmental factors (Polishchuk and Rauschmayer 2012). Personal conversion factors refer to individual abilities (e.g. physical, cerebral, psychological etc.), whilst social conversion factors concern social practices, rituals and roles reflecting societal views regarding gender and religion for example. Finally, environmental conversion factors comprise a set of conditions or variables related to climate and geography for instance that may act to enhance or prohibit capabilities (Robeyns, 2005; Polishchuk and Rauschmayer 2012; Pelenc and Ballet, 2015). Consequently, ecosystem services can contribute directly to capabilities (e.g. through direct consumption of a provisioning services), or indirectly and in a more long-term manner through affecting personal and social conversion factors (e.g. ecosystem service impacts on health and education) that feedback into an individual’s capability set (Polishchuk and Rauschmayer 2012). Crucially, the concept of a general level of human-wellbeing derived from ecosystem services is false, ultimately, wellbeing impacts depend upon how ecosystem services are processed or converted at the individual level (Polishchuk and Rauschmayer 2012).

Alongside conversion factors CA also emphasizes the importance of freedoms, as previously mentioned, linking directly to another important component of “making use” of resources, namely “access” and “control” (Fisher et al., 2013; Pelenc and Ballet, 2015). Considered in this way the CA also encourages us to “see” resources in terms of endowments (i.e. rights over resources and the type of resource bundle) and entitlements (i.e. legitimacy to

effect control over the resource bundle), with governance regimes determining individual, group and community dynamics in relation to resource utilisation (Fisher et al., 2013; Pelenc and Ballet, 2015). Collectively, these represent a set of conditions with significant implications for human-wellbeing, livelihoods, wealth and poverty alleviation (Fisher et al., 2013). In many cases “access” and “control” may be more significant for human-wellbeing outcomes than the fundamental availability of resources (Fisher et al., 2013). Access and control also differ according to the type of resource, for example, controlling a physical provisioning service is much more straightforward compared to an intangible cultural service (Fisher et al., 2013).

In this respect choice is central to CA, and especially in relation to how people utilise ecosystem services according to personal preferences (Polishchuk and Rauschmayer 2012; Fisher et al., 2013). As Pelenc and Ballet (2015:38) describe:

“...the capability concept operates via a notion of freedom (i.e. positive freedom) that encompasses both potential choices (i.e. the set of achievable functionings) and realized choices (the set of chosen and achieved functionings).”

Accommodating this distinction we locate the finalised set of “achieved functionings” in a separate human-wellbeing element, distinct from the individual choice component, in part to highlight the significance and multifaceted nature of human-wellbeing. We assert that achieved functionings and human-wellbeing are mutualistic, underpinned by a suite of capital assets such as human capital (e.g. knowledge, skills and capacity), social capital (e.g. trust, shared knowledge, norms and rules associated with organisations and networks), financial capital (e.g. employment, income), physical capital (e.g. infrastructure, physical goods underpinning livelihoods, housing) and political capital (e.g. ability to use power in support of political or economic positions), situated alongside the broader backdrop of security, health and poverty alleviation (Sinha and Baumann, 2001; Fisher et al., 2013; Chapman, 2014). Collectively, these aspects relate ecosystem services to the individual, the individual to wider society, and underline the dynamic scalar nature of the framework (Fisher et al., 2013). To quote Pelenc and Ballet (2015:38) once again:

“CA makes it possible not only to drive the concept of well-being toward a more multidimensional conception, but also to distinguish between well-being achievement and well-being freedom”.

Reiterating the argument advanced by Polishchuk and Rauschmayer (2012), the coalescence of CA and ecosystem services provides a richer understanding of the relationships between particular bundles of ecosystem services and their transformation in to constituents of wellbeing, as well as the extent to which ecosystem services have the capacity to influence individual conversion factors and by extension individual capabilities (Polishchuk and Rauschmayer, 2012). Moreover, by adopting CA wellbeing is understood in a far more

nuanced manner than that conveyed by the standard utilitarian framing of human-wellbeing, which affords opportunities for further evaluating the relationships and influences occurring between less tangible ecosystem services and human-wellbeing (Polishchuk and Rauschmayer 2012).

16.2.3 Human-Wellbeing – SES Management And Governance – Social-Ecological Factors

Governance and management (the step that addresses the links between human-wellbeing, social-ecological factors influencing ecosystem service provision and the activities that shape the landscape in Figure 15.1) represent the instruments in our landscape approach where directed action towards achieving continued provision of ecosystem services, balancing conservation and development goals for biodiversity and human-wellbeing gains, ensuring sustainability and a stream of productive outputs are articulated (Figure 16.2, Primmer and Furman, 2012). We acknowledge two sets of drivers determining the development and implementation of environmental management programmes, specifically, intra-scale bottom-up processes deriving from the ecosystem service demand component and resultant impacts on human-wellbeing, and external inter-scale pressures such as top-down national environmental governance and policy drivers. Importantly, this provides the opportunity for ecosystem services and environmental and spatial planning to coalesce, areas which have traditionally been distinct both in their disciplinary outlook and discourses (Albert et al., 2014). Collectively, these factors signal the “need” for natural resource policy design and implementation, which follows the principle that there are ‘key resources’ related to ecosystem services provision, livelihoods, capabilities and development needs that programmes need to address to be effective (Chapman, 2014).

As such the landscape approach we advocate recognises the complex interactions and overlaps occurring between scale-specific environmental challenges and human-wellbeing, as well as the broader social and environmental context and the need to jointly negotiate these issues (Chapman, 2014; Keune et al., 2014a; Keune et al., 2014b). Like Keune et al., (2014a), we recognise the governance component of natural resource policy is a multidimensional construct concerned with the “structure” and “processes” of ecosystem service governance, and as such embodies the connections between the:

“...societal processes of participating in the decision-making, implementation, and evaluation of public policy.” (Keune et al., 2014a:65)

As further noted by Plummer et al., (2012:1):

“Efforts to guide nature-society interactions, sustain ecosystem services, and improve human-wellbeing require holistic, integrative and multilevel institutional arrangements.”

In this regard, in order for governance to be effective, we would argue strongly that our landscape approach recognises the need to align all aspects of institutional fit, or what Epstein et al., (2015) refer to as “ecological fit”⁸, “social fit”⁹ and “SES fit”¹⁰. These arguments partly refer back to the issue of Scale which we have previously discussed. At the same time we also acknowledge that governance is not homogenous but is multi-layered comprising a number of different approaches, for example, “hierarchical governance”¹¹, “scientific-technical governance”¹², “adaptive-collaborative governance”¹³ and “governing strategic behaviour”¹⁴ and all of these need to be recognised and accommodated (Primmer et al., 2015).

Increasing devolution of management activities to the lowest administrative scales means the knowledge regarding the biophysical structure and function of landscapes needs to connect to local scales of economic and socio-cultural decision-making (Termorshuizen and Opdam, 2009). For instance, as Termorshuizen and Opdam (2009) argue, local spatial scales are far more important to deliberative “reflexive” decision-making arrangements regarding land management planning. It is no surprise then that stakeholders should figure so prominently in the development and implementation of natural resource policies, specifically because of their pivotal roles in the actions, processes, interactions, dynamics and networks that underpin and facilitate ecosystem service governance (Reed et al., 2009; Termorshuizen and Opdam, 2009; Chapman, 2014; Keune et al., 2014a). This reflects the modern transition from top-down command-and-control arrangements to more organic, bottom-up and grass-roots focused management, where the emphasis is to shift the power-base from centralised structures to local decentralised sources of decision-making (Roe, 2013). This idea is neatly encapsulated by O’Farrell and Anderson (2010:62) who remark:

“The need for collaboration and the exchange of ideas with stakeholders is now recognised as critical to achieving this necessary transformation.”

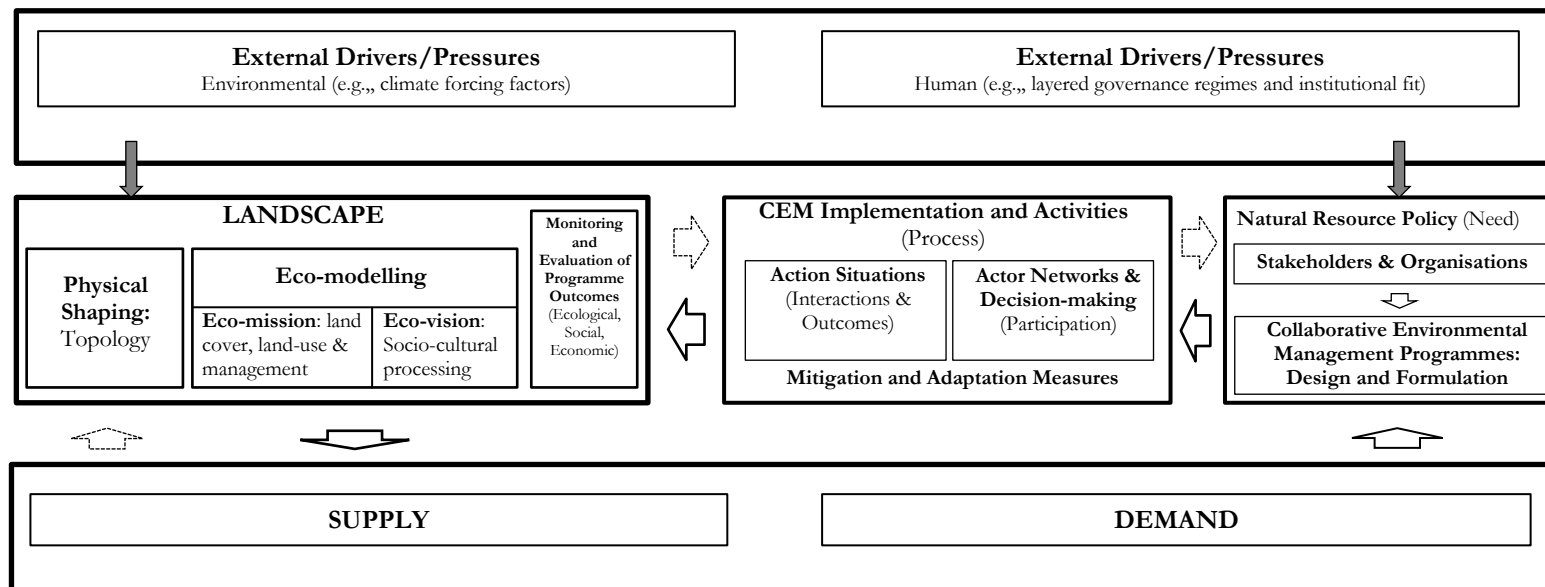


Figure 16.2 Presents the detailed links and processes connecting environmental management programmes and the construction, maintenance and conservation of the landscape. This is set within the broader context of multi-scaled environmental and governance and institutional drivers on the one hand, and supply-demand side drivers on the other. Filled arrows indicate the direction of those external environmental and governance drivers. Black-bordered arrows signify the direction of flow through the system. Perforated arrows indicate interactions between different components.

For example, in South Africa, Sitas et al., (2014) employed a stakeholder engagement process to highlight avenues for integrating ecosystem services into the decision-making processes underpinning landscape-scale planning arrangements. Similar participatory approaches for combining stakeholder ecosystem service demands with landscape development and planning processes have also been carried out in Spain (Palacios-Agundez et al., 2014). On the whole participation tends to be uncritically accepted as a universal good (Roe, 2013) primarily for the reason Sayer et al., (2013:8351) outline that:

“All stakeholders should be recognized, even though efficient pursuit of negotiated solutions may involve only a subset of stakeholders.”

Not championing participation could therefore be regarded as weaving together inequalities into the decision-making process, which in turn could prove detrimental to programme interventions (Sayer et al., 2013), as Setten and Brown (2013:246) comment:

“...a justification for landscape participation is hence a notion of social justice...”

However, as Roe (2013:340) conveys there are circumstances in which:

“...participation under existing institutional and political conditions may simply reinforce existing boundaries and barriers.”

The general supposition framing these comments is the notion that communicating the “stakes” stakeholders have largely improves the decision-making process; however, such thinking also runs the risk of failing to acknowledge that people do not all share ‘equal’ stakes (Roe, 2013). The potential for power asymmetries to have a decisive role in the formulation and implementation of management programmes consequently looms large. From this perspective landscape can be seen as central to the exercise of these power dynamics, as Roe (2013:341) astutely remarks:

“The increasing recognition of considerations of power and control in the landscape has considerable importance for the study of landscape and participation [...] The way landscape becomes an enabler or focus for expression of power and protest is now apparent.”

Improving the management and sustainability of SES therefore requires greater attention to be paid to understanding how power relations, equity and justice affect ‘ecosystem stewardship’ (Fischer et al., 2015).

The composition and selection of stakeholder groups, as a consequence, can therefore be a crucial element in the successful development and implementation of institutional governance arrangements and policy measures (Reed et al., 2009). As Sayer et al., (2013:8354) advance:

“The quality of stakeholder engagement, the degree to which various stakeholders concerns are acknowledged, and the investment in building trust and developing a shared vision will ultimately dictate the success or failure of the process.”

To negotiate this complexity we advocate a collaborative adaptive management approach in order to fully account for the multi-tiered and multi-faceted nature of institutional governance issues (Plummer et al., 2012; Chapman, 2014; Fischer et al., 2015). Adaptive management and its variants have become increasingly popular over recent decades (Plummer et al., 2012); even though the successful implementation of adaptive management programmes is limited and they suffer from a number of significant challenges including goal formulation and outcome evaluation (Plummer et al., 2012; Westgate et al., 2013). Nevertheless, the significance of an adaptive management approach resides in the emphasis it places on deliberative processes of engagement and decision-making, social learning, problem-solving, reflexivity, knowledge plurality, collaboration and consultation, scale and greater alignment between biophysical and social processes (Cundill et al., 2012; Plummer et al., 2012; Chapman, 2014; Keune et al., 2014b; McAllister and Taylor, 2015). These aspects are fundamental to the design and formulation of effective environmental management interventions for the continued delivery of environmental and social benefits (Chapman, 2014).

Operation and implementation phases and other additional “process” activities (or elements) are especially important (Plummer et al., 2012; Chapman, 2014), particularly in terms of delivering programmes capable of providing effective “adaptation” and “mitigation” measures (Fisher et al., 2013). This is because they enable individuals, households and communities to cope with environmental change and act concertedly to address these drivers (e.g. mitigation) or adjust behaviour (e.g. adaptation) through the development of a diverse strategy portfolio (Fisher et al., 2013).

At the heart of these “process” elements is the institutional analysis and development framework constituents of actor participation and decision-making located within an ‘action situation’ setting (Ostrom, 2011; McGinnis and Ostrom, 2014). This emphasises the importance of local conditions, interactions between individuals (whether acting on their own behalf or as agents of particular groups or organizations) and the choice context in realising desired outcomes (McGinnis and Ostrom, 2014). In this context we can distinguish actor-relevant norms and broader institutional norms determined by particular socio-cultural settings, which are subject to the influence of governance systems: rules, property systems and network structures (McGinnis and Ostrom, 2014). In this respect property-rights are particularly important in that they specify the relationships between people and resources with regards to both duties and obligations – factors that can significantly affect ecosystem service provision (Schlager and Ostrom, 1992; Cash et al., 2006; van Laerhoven and Ostrom, 2007;

Armitage, 2008; Lockwood et al., 2010, McGinnis and Ostrom, 2014). As Vatn (2005:253) describes:

“Property rights define who has access to which resources or benefit streams and under what conditions. They distribute access to resources between the members of a society and regulate conflicting uses.”

Two important points are worthy of note. First, that “landscape” is the object of property rights not ecosystem services, even though people may have the right to use and access particular ecosystem services (e.g. timber) provided by the landscape. Second, in general terms, but also in cases where complicated spatial relationships exist between property right arrangements and the provision and distribution of ecosystem services, multiple input streams are crucial to the decision-making processes that underpin effective land management. Consequently, we support social learning¹⁵, soft systems thinking¹⁶ and knowledge exchange¹⁷ as key elements to develop the required capacity to achieve environmental management programme outcomes (Reed et al., 2010; Cundill and Rodela, 2012; Cundill et al., 2012; Chapman, 2014; Reed et al., 2014).

16.2.4 Landscape

Our conception of landscape closely aligns with the ideas presented by Wang et al., (2011): namely, that landscape arises through the process of “ecoscape modelling”. As a concept, although ecoscape modelling does not have widespread literature coverage its philosophical disposition aligns with our thinking concerning landscape production which we outlined in Chapter 14 (see Figures 14.1 and 14.2). Ecoscape modelling can be broken down into two components: First eco-vision, which refers to the human social construction of landscape as mediated by individual perception and imagination, in other words, the consolidation of the physical aspects of landscape transformed and internalised into our minds. Not only does this represent a self-enrichment process, capable of feeding back into natural resource use and management behaviours, but it also represents the profound connection between humans and nature occurring on an individual and community level (Wang et al., 2011). Second eco-mission, this consists of three elements, namely, ‘reform’, ‘restore’ and ‘regulate’ and refers to the sustainable shaping of the environment through human action and intervention (e.g. the implementation of particular management programmes such as restoration initiatives). The main thrust of the argument is that ecological planning, management and capacity building combines with an effort to better regulate the eco-dynamics of the constructed space that is landscape (Wang et al., 2011).

Crucially, landscape is the lynch-pin between nature and society, the interface between environmental management programmes and underlying ecological structures and processes that contribute to the downstream production of ecosystem services (Thermorshuizen and

Opdam, 2009). Landscape-scale approaches now represent a new form of management, better suited to the conservation of meta-populations or trans-frontier landscapes for example (Hodder et al., 2014). The spatial and temporal dynamics of landscape, in terms of land-use change, land cover and particular land-uses are central to understanding ecosystem services (Crossman et al., 2013). Ultimately, landscape represents the arena in which environmental management programmes assess and evaluate their outcomes, successes and failures (Chapman, 2014). Primarily, this is because the majority of environmental management programmes seek to transform the landscape via the implementation of alternative management regimes and land-uses arising from shifts in agricultural and forestry production modes for example (Crossman et al., 2013).

The resultant changes in land-use, which can influence underlying biophysical and hydrological processes, can both positively and negatively affect ecosystem services provision through changes in the spatial configuration of land-use patterns (e.g. Martinez et al., 2009; Ostle, 2009; Fürst et al., 2011; Nagendra et al., 2013; Hodder et al., 2014). For example, as Nagendra et al., (2013:503) state:

“...changes and modifications in land cover/land use constitute the most dominant drivers of biodiversity loss globally [...] and [...] ultimately affect the structure and function of ecosystems, and alter their capacity to provide sustained ES.”

Sentiments echoed by Crossman et al., (2013:509):

“Land use and management decisions have a strong bearing on the condition and integrity of ecosystems and their components which in turn have a substantial impact on the supply of ecosystem services.”

In light of this reality monitoring programme outcomes is especially important (Chapman, 2014). However, in many instances, monitoring can be hampered by a range of ecological and socio-economic constraints and uncertainties, such as: adequately linking proxies of change to underlying ecological production functions that connect to specific land cover and vegetation types (Nagendra et al., 2013), or understanding the social consequences of change that may result from alterations in biophysical conditions or from political drivers that affect water and land security (Crossman et al., 2013). Furthermore, difficulties can arise through the conflation of what is considered a programme process and what constitutes a programme outcome (Chapman, 2014). In being aware of these difficulties we support the idea of distinguishing “mediator” variables (i.e. those associated with pathway mechanisms) from ‘moderator’ variables (i.e. those associated with the relationship between programme and outcome) in programme evaluations, with outcome assessments feeding back into design and implementation phases to improve and inform future programme developments (Chapman, 2014; Hejnowicz et al., 2014).

16.3 Landscaping: Implications For Ecosystem Services

16.3.1 Multi-functionality And Connectivity

Landscaping ecosystem services implicitly recognises the importance of landscapes in providing a wealth of social, cultural and environmental benefits, in other words, delivering multi-functionality (Termorshuizen and Opdam, 2009). Yet, for a landscape structuring of ecosystem services, it is not simply the implied connections between landscape and multi-functionality that are important, but also the explicit associations multi-functionality has with the protection and creation of landscape spaces – aspects that relate to landscape planning and development (Selman, 2009). Second, the connections between multi-functionality and ‘multi-scale approaches’ to the construction of diverse land-use patterns – patterns which enable the conservation and provision of underlying biophysical processes and biodiversity (Lovell and Johnston, 2009). And finally, the capacity of multi-functionality to probe important facets of landscape, for example, “composition”, “configuration” and “patch potential” (O’Farrell and Anderson, 2010).

These contributions of landscape in shaping a holistic vision of multi-functionality are especially significant in a world in which the direction and magnitude of global drivers of landscape and land-use change have led to the simplification of ecosystems, in many respects, developments commensurate with the promotion of provisioning ecosystem services for food and energy security (Pedroli et al., 2006; Mastrangelo et al., 2014; Perrings, 2014). These seemingly relentless changes also pose great difficulties for sustainable land management activities, especially for example, those focused on the development of agro-ecosystems (Lescourret et al., 2015).

Probing the conceptual underpinnings of multi-functionality a stage further, we need to disaggregate the terms “functional” and “multiple” rather than perceiving multi-functionality as simply a “layering” concept (Selman, 2009). Often “function” refers to the dynamic and interactive qualities related to the larger system, emphasising a particularly eco-centric perspective, where functions underpin the “services” conveyed by the ecosystem services concept (Selman, 2009). On the other hand, “multiple” relates to a number of different pursuits, where land is regarded as capable of delivering several distinct outputs to meet social, economic and ecological demands concurrently across space and time (Selman, 2009). Emphasising “landscape multi-functionality” promotes a focus on the human-designed and constructed elements of multi-functionality, and the necessity to understand the dynamical relationships that shape landscapes (O’Farrell and Anderson, 2010). In particular, the “power relations, political agendas and politicised issues” that frame the decision-space governing those who decide upon, use and benefit from the final landscape product (O’Farrell and Anderson, 2010). Moreover, it is important to bear in mind that functions can be both

beneficial and adverse due to their spatial and temporal availability but, despite their Jekyll and Hyde character, they can usefully inform public policy by highlighting potential trade-offs stemming from planned landscape developments (Bolliger et al., 2011).

Overall, multi-functionality is closely aligned with ideas of “richness”, “diversity”, “heterogeneity”, “multiple use”, “connectedness of production”, “joint ecosystem service supply”, “complementarity” and “synergy”, but also; perhaps somewhat more profoundly, core social-ecological system notions of “place”, “resilience” and “culture” (Lovell and Johnston, 2009; Selman, 2009; O’Farrell and Anderson, 2010; Mastrangelo et al., 2014). Indeed, multi-functionality is regarded as an integrative concept, connecting ideas of “complexity”, “connectivity” and “social-ecological systems” with “human-value systems”, as well as being an “emergent property” central to achieving a “sustainable landscape system” (Selman, 2009). Connectivity is a particularly important concept in relation to ecosystem services provision and regulation (Mitchell et al., 2013). However, as Mitchell et al., (2013: 894) point out:

“We do not fully understand how changes in landscape connectivity affect ecosystem service provision, despite theory suggesting connectivity is important.”

Their review indicates that landscape connectivity can have both positive and negative consequences for ecosystem service supply and provision, depending upon the service in question, through the influence connectivity can have on biotic and abiotic patterns and dynamics. Indeed, in this vein, Mitchell et al., (2013: 905) further argue that:

“Understanding the relative importance of connectivity within and between ecosystems will also be important for predicting how land-use will affect ecosystem services, and for designing landscape management techniques to maximise multiple ecosystem services.”

According to Selman (2009) six distinct qualities exemplify the core concept of landscape multi-functionality, namely: interactivity, scale, social construction, social learning and collective action, social-ecological dynamism and stakeholder partnerships and joined-up governance. Many of these aspects including scale, stakeholder participation and policy integration are important and popular areas of landscape multi-functionality research (Mastrangelo et al., 2014).

Ultimately for Selman (2009:49) multi-functionality can be defined from a landscape planning standpoint in the following way:

“Multi-functionality provides us with a way of understanding change and delivering joined-up policy and the landscape scale, where its core property of interactivity can be harnessed in ways that produce qualities valued by people.”

The most notably aspect of Selman's (2009) definition is its ultimately person-centred foundation, a viewpoint that resonates with the eco-semiotic subject-centred approach of Almo Farina (Lindström et al., 2013), and a principle also shared by Lovell and Johnston (2009) and Reyers et al., (2012) who both emphasise the broader ecosystem benefits emanating from landscapes:

“Multi-functional landscapes are landscapes which provide multiple environmental, social and economic functions and are able to achieve multiple societal needs including energy and food production, management of waste, conservation of biodiversity and the management of water quantity and quality across the landscape; the improvement of landscape heterogeneity and therefore resilience; and the provision of recreational opportunities.” (Reyers et al., 2012:1122 adapting Lovell and Johnston's (2009) definition)

Evident from the definition presented by Reyers et al., (2012) is the embedded notion and connection between multi-functionality and landscape sustainability, a sentiment advanced by O'Farrell and Anderson (2010:59):

“Sustainable multi-functional landscapes are landscapes created and managed to integrate human production and landscape use into the ecological fabric of a landscape maintaining critical ecosystem function, service flows and biodiversity retention.”

Bringing these conceptions of multi-functionality together in a more systematised and comprehensive manner, Mastrangelo et al., (2014:353) provide a three-fold description of multi-functionality framed in a pattern- and process-based understanding of ecosystem service generation:

“...we distinguish a pattern-based multi-functionality when it is conceived as the joint supply of multiple ES in space but without regard for the ecological processes underlying this pattern [...] a process-based multi-functionality as the joint supply of ES in space caused by well-understood relationships of synergy or complementarity among them [...] a socially-relevant, process-based multi-functionality when it is conceived as the joint supply of ES of relevance for local stakeholders, which result from complementary or synergistic relationships among ES.”

16.3.2. Communication, Understanding And Enrichment

Several years ago Termorshuizen and Opdam (2009) introduced the concept of “landscape services”. This concept is predicated on the idea that “landscape” provides a more suitable vehicle for conveying value, discussing sustainability and framing conservation and development issues because of its inherent immediacy and capacity to access and key into tangible and deep-rooted sense of meanings. In other words, landscape and landscape services as a “value-delivering system” provides a far more tractable notion than the standard eco-physical concept of ecosystem services¹⁸; and moreover, has the added advantage of enabling a greater alignment between ecological knowledge and landscape planning (Termorshuizen and

Opdam, 2009). The argued benefits of employing landscape are its ability to provide a clearer connection between landscape configuration, composition and functioning (i.e. pattern-process relationships), engage stakeholders and better unify scientific disciplines from landscape architecture through to conservation (Termorshuizen and Opdam, 2009). Landscape expresses a more spatially constructed, richly contextualised, planning-oriented and human-centred approach (Bastian et al., 2014). The fundamental supposition is that landscape services, and in particular “landscape”, functions as a galvanising and unifying concept that is more immediate and less remote than the esoteric concepts of “ecosystem” and “ecosystem services”, and as such acts as a “bridging concept” linking science, policy and people (Dramstad and Fjellstad, 2011).

Clearly, in this sense, landscaping the ecosystem services concept is argued to improve its communicative capacity, knowledge sharing and overall implementation and wider acceptance. We know from the literature concerning effective communication, knowledge exchange and transfer as well as semantics that commonality of language is critical for communicating concepts, ideas, thoughts and meaning and thereby engaging stakeholders in participatory processes of decision-making (e.g. Eppler, 2006; Boroditsky, 2011; Phillipson et al., 2012; Reimer, 2013; Thomas and McDonagh, 2013; Fazey et al., 2014; Gärdenfors, 2014; Malt et al., 2015; Imai et al., 2016; Tamariz and Kirby, 2016). For example, as Eppler (2006:1) explains:

“The efficient and effective transfer of experiences, insights and know-how among different experts and decision-makers is a pre-requisite for high-quality decision-making and coordinated organizational action.”

Providing avenues for knowledge communication¹⁹ and opening up space for knowledge dialogues²⁰ is central to the objective of “effective communication” and overcoming knowledge communication barriers (Eppler, 2006), which can impact on stakeholder engagement processes and the exchange of knowledge between stakeholders (Phillipdon et al., 2012). Effective communication is also fundamentally about the concepts we are trying to convey, and so linguistic and semantic comprehension are central elements underpinning true cognitive understanding as well as representing the platform upon which the power of a concept to communicate meaning rests (Reimer, 2013:4):

“The cognitive resources which an agent draws on to think about an aspect of the environment must permit successful interaction with that aspect [...] Our concepts of the external environment must, in other words, be ‘appropriately connected to the world’.”

As Reimer (2013:5) goes on to explain:

“If there is not some level on which speakers of the same language share the same concepts, it is impossible to see how communication and mutual understanding

occur: concepts must, by definition, be shared if there is to be communication and understanding in the normal sense of these terms.”

This line of thought is further supported by Thomas and McDonagh (2013) who emphasise the idea of a “shared language” in the determination of comprehensible communication, understanding and knowledge sharing:

“Shared language refers to people developing understanding amongst themselves based on language (e.g., spoken, text) to help them communicate more effectively [...] Shared language is critical to collaboration [...] The language we use and our ability to share language with others impact [...] perceptions and perspectives...” (pg. 46)

Or to put it another way, as Tamariz and Kirby (2016:38) state:

“Linguistic signals fulfil their communicative function because they have conventionally associated meanings which are shared by a community of speakers.”

It is essential then that conceptual language, which conveys complex thoughts, ideas and meaning, has an intelligible commonality about it (Reimer, 2013), for as Wittgenstein famously remarked in his *Philosophical Investigations* (1955, para. 203):

“Language is a labyrinth of paths. You approach from one side and know your way about; you approach the same place from another side and no longer know your way about.”

In essence this is because, as Boroditsky (2011:64) points out:

“...language shapes even the most fundamental dimensions of human experience: space, time, causality and relationships to others.”

This is why, as Boroditsky (2011:65) goes on to remark:

“...language influences what we remember”

Ultimately then, commonality of language allows people to develop and coordinate meanings, and in essence, negotiate “communicative elements” which enable the intelligibility of conveyed concepts and foster a meeting of minds: a level of compatibility that allows goals, objectives and a shared vision to form (Gärdenfors, 2014:3-4):

“...meanings of expression do not reside in the world or (solely) in the image schemas of individual users but they emerge from the communicative interactions of language users. Consequently, meanings are in the coordinated heads of users [...] The fundamental role of human communication is, indeed, to affect the state of mind of others – bringing about cognitive change [...] A meeting of minds means that representations in the minds of the communicators become sufficiently compatible to satisfy the goals that promoted the communication.”

In this way language reflects “cultural specific value systems and epistemologies” which affect the way knowledge is represented and acquired (Imai et al., 2016). Yet, at the same time

the “mental representation” of concepts and the dimensions of knowledge communicated are not static or rigid, rather language can be “flexibly utilized” and modified ‘from situation to situation’ (Malt et al., 2015).

16.3.3 Reconnecting Ecosystem Services

Against this backdrop there has been a growing coalescence between land science, ecosystem services, and the “landscape” articulation of ecosystem services, particularly in relation to cultural landscapes and services (Plieninger et al., 2014; 2015). Much of this has been driven by the connections and contributions landscapes make to human-wellbeing. The subsequent re-imagining of the standard ecosystem service model through landscape has afforded a greater plurality of exploration and deeper integration of notions of place, connectedness, physical and mental health, social relations and social practice, continuity and identity (Vallés-Planells et al., 2014; Villamagna and Giesecke, 2014, Martínez-Juarez et al., 2015; Pröbstl-Haider, 2015; Wilhite, 2015). Holistic socio-cultural valuation approaches, in particular, those seeking to uncover the connectedness between particular landscape types, ecosystem services and cultural value and wellbeing are becoming increasingly widespread, for instance, there are recent examples in relation to transhumance (López-Santiago et al., 2014) and flood alleviation (Barbedo et al., 2014). Assessments of drivers of cultural landscape change have also become prominent, for example, in relation to evaluating the influence of broad-scale social, institutional and political factors on use-rights (i.e. land, property), entitlements and ownership on rural livelihoods and traditional relationships with the land (Gu and Subramanian, 2014).

However, it has been pointed out (Plieninger et al., 2015) that the conceptual foundations of “cultural ecosystem services”, and its inherent usage in relation to “landscape”, is not neutral but in fact highly normative and contested, particularly when viewed from the vantage of different sectors (e.g. planning) and stakeholders (e.g. landowners, communities). There have also been claims that cultural ecosystem services’ framing of human-nature relations is both “separationist” and “reductionist” because it caricatures these relationships (i.e. culture) as merely a provided service, ignoring the fundamental social-ecological context in which these relationships and their benefits co-evolve (Setten et al., 2012; Plieninger et al., 2015). Nevertheless, recent evidence seems to suggest that cultural ecosystem services can support landscape multi-functionality, provide the necessary motivation for owning and managing land, and draw together urban and rural communities in a way that reengages and reacquaints them with the landscape and heritage values and engenders a new stewardship ethos (Plieninger et al., 2015).

A landscape perspective is also relevant for understanding provisioning and regulating services (Iverson et al., 2014). For example, floodplain management programmes in Europe have rolled-out integrated restoration and rehabilitation strategies resulting in an increase in overall landscape multi-functionality including provisioning ecosystem services (Schindler et al., 2014). Similarly, in Chile, the sustainability of watershed resources has been improved through adaptive management programmes designed to balance competing urban and rural demands, resulting in more effective delivery of several ecosystem services including water quality and quantity (Donoso et al., 2014). Investigations in the Amazon modelling large-scale drivers of land-use change and composition have established how the joint impacts of deforestation and climate change can affect underlying regulating services like hydrological cycles and processes, creating feedbacks at both local and regional scales which exacerbate the probability of drought and flood-like conditions (Santos de Lima et al., 2014). Other models have established how landscape context and geography is critical to the erosion control responses of different mountainous environments experiencing human-induced changes in vegetation patterns and landscape composition (Vanacker et al., 2014).

It is quite apparent then that the most important aspect of “landscape” is its holistic recognition of human-nature interactions and the criticality of bridging divides between ecological and social dimensions (Bastian et al., 2014). Though we are in agreement with Bastian et al., (2014) that terminology such as “landscape services” is unlikely to replace or compete with ecosystem services, especially within specialized fora, in reality the most important issue concerns changing perceptions and attitudes and presenting a new way of thinking. It is about recognising that conveying the ecosystem services concept in a landscape vehicle is a far more effective means of communicating and managing landscapes for multiple functions, goods and services than treating them as separate and independent entities (Termorshuizen and Opdam, 2009; Bastian et al., 2014). Thus, as Sybre and Walz (2012:80) express:

“The landscape services approach in a wider sense allows us to take social/cultural services better into account because they depend strongly on heritage assets, structural characteristics, historical conditions and even cultural specifics; which can hardly be subsumed to ecosystems.”

16.3.4 Environmental And Social Justice

A concern for justice²¹ is important. Such a concern has implications for how we view the world; how we use (or misuse) natural resources and how we treat each other (particularly in terms of human rights); and furthermore, how the consequences that result from our actions and interventions influence whether others’ experience of those outcomes is positive or negative, as Sikor (2013a:2) explains:

“Justice matters [...] because interventions almost always affect the distribution of benefits and responsibilities, different people’s participation in decision-making or the recognition of their identities and histories. These justice-relevant outcomes in turn influence people’s reaction to governance interventions, making justice an integral component of environmental management.”

Thus, by logical extension, outcomes can be perceived and experienced as being simultaneously just and unjust, as Martin (2013:99) points out:

“...any decision or process might be considered just by some and unjust by others, either because they prioritise different justice dimensions or they attach different principles to them. Furthermore, each actor might themselves consider something simultaneously just and unjust, for example where an exclusionary decision-making procedure leads to distributional outcomes that are considered fair.”

Like environmental challenges and problems that occur at multiple scales justice also has local and global “framings”: notions of justice that are formed and implemented at the national and sub-national level, and global concepts of justice that are instituted and executed at the international scale via collective agreement (Martin, 2013)²².

The ecosystem services framework implicitly evokes justice through its central claim that ecosystem services support and underpin human-wellbeing, and that concern for human welfare should be the guiding principle for engaging in environmental management activities (Sikor, 2013a). However, as a consequence of its utilitarian framing, its preponderance to emphasise instrumental values, “atomise” services and present decision-making frequently from the standpoint of trade-offs, the ecosystem service framework primarily favours distributive concerns over-and-above “procedural” and “recognition” justice elements (Sikor, 2013a).

If we consider the three dimensions of justice from an ecosystem services perspective, then distributive concerns might consider who receives ecosystem services and who is responsible for their provision. On a procedural front, we might want to know and be concerned about which populations, service beneficiaries for example, are involved in the decision-making processes surrounding the maintenance and delivery of ecosystem services, and with regards to recognition we might be interested in examining whether there are any social-cultural dimensions or differences in ecosystem service provision and access. And of course, from a global framing perspective, transboundary environmental issues concerning, for example, food, energy and water security related to climate change or diffuse pollution and their impacts on human welfare may figure more heavily (Martin 2013; Sikor, 2013a).

The environmental justice implications of ecosystem services are clear, for example, when considering the influence social and political processes have on their distribution, where social groups and power relations can strongly affect the scalar delivery and accrual of

ecosystem services (Ernstson, 2013). For instance, as described by Spangenberg et al., (2014:52):

‘Societal process, with their typical mix of traditional norms, changing preferences and [...] power relations decide on the [...] services delivered and their beneficiaries.’

However, because landscapes are partly socially engineered and conceived so too are the individual and societal wide benefits we accrue from landscapes’ underlying biophysical processes (Thermorshuizen and Opdam, 2009). In other words, by recognising ecosystem services are co-produced, not simply the result of discrete objective physical processes, we automatically acknowledge the wider social and political events and value articulating discourses from which they originate (Ernstson, 2013). Paraphrasing Spangenberg et al., (2014), “human agency” determines the provision of ecosystem services. In particular, the flow of ecosystem services is related to the historical input of human capital (i.e. in the form of labour), and so their materiality is not simply a natural resource but also a “discursive resource” (Ernstson, 2013). In fact, it is these socio-economic inputs that Spangenberg et al. (2014) argue enable the distinctions between ecosystem functions; ecosystem service potentials and ecosystem services highlighted in Figure 16.2, as Lele (2013:132) comments:

“Obtaining benefits from ecosystem processes usually requires the investment of human labour and human-made capital for harnessing the ‘service’ [...] The entire activity of production and consumption is embedded in social structures and institutions that determine who gets access to which resource, capital or labour, and also influences strongly the pattern of demand and consumption of goods and services.”

For Lele (2013) the major problems with the current framing of ecosystem services are two-fold: First, by focusing on aggregate human-wellbeing it fails to consider “fairness”, “equity” and “justice” in an adequate manner, and second, its consideration of environment-society relations is an oversimplification – this may explain why these issues are so prescient in management programmes like PES (e.g. Fisher, 2013; Martin et al., 2014; Hejnowicz et al., 2014; 2015), REDD (Sikor, 2013b) and river basin management (Zeitoun and McLaughlin, 2013).

In Chapter 11, for example, we highlighted the lack of focus on human and social capital in the design and implementation of PES programmes, and how their institutional arrangements posed difficulties for achieving favourable justice, equity and benefit sharing outcomes in terms of sustainable livelihoods and poverty alleviation, in particular because many schemes failed to target the poorer and more marginalised groups. Similarly, in Chapter 12, we also pointed out that improving PES effectiveness would require a better understanding and acknowledgement of the social and institutional factors governing how PES affects communities. In other examples of how justice figures in the implementation of

PES programmes, Fisher (2013) demonstrated that where conditionality becomes the prime focus of PES other important concerns arising from the implementation of the intervention are excluded: a situation that precipitates significant justice issues:

“...the central emphasis upon monitoring environmental performance within conditional interventions comes to dominate such interventions, to the exclusion of other concerns [...] a focus on technical procedures [...] makes projects upwards accountable, rather than accountable to participants, with consequences for procedural (participant) justice [...] important implications for distributional equity, as conditional interventions automatically seek those best able to secure an ecosystem service.” (pg. 32)

We also need to grapple with the idea that there are different worlds of justice, for example, Martin et al., (2014) identify – again in relation to PES – the disconnect between local community views and conceptions of justice and the imported notions of justice that are inherent in Western constructions of PES. Similarly, Sikor (2013b) notes that REDD programmes incur distributive, participation and recognition justice and injustice challenges in relation to their scale of implementation, benefit distribution and carbon measurement, such that the way these governance interventions are instituted means that they can change, weaken or strengthen existing governance arrangements through the implicit social values they carry. Many biodiversity conservation efforts and programmes also focus on “benefit sharing” – effectively sharing the beneficial proceeds of an intervention to improve the social justice elements of these programmes. However, as Martin et al., (2013:70) acknowledge:

“...we open up the possibility that even ‘successful’ benefit sharing might not necessarily service social justice where it successfully provides for material needs whilst reducing opportunities to meet other social needs [...] We find that benefit-sharing approaches can serve distributional justice but can also reinforce dominant ways of thinking about conservation problems that can themselves produce injustices.”

As a framework ecosystem services does not explicitly accommodate justice and injustice concerns, rather it implicitly installs a utilitarian and libertarian quality of justice in its theoretical framing (Lele, 2013; Sikor et al., 2013). However, “landscape” and a “landscape approach” explicitly recognise multiple justices and injustices. For example, a landscape approach recognises that within landscapes are embedded political, social, economic and cultural processes and practices that fundamentally affect human welfare and human rights, as we have seen from Chapter 14 (e.g. colonialist landscapes, natural resource frontiers). Hence, the litany of arguments critiquing ecosystem services on the basis that it favours particular modes of justice over others; that it’s too technically oriented; focused too narrowly on the positive aspects of service provision at the expense of examining ecosystem disservices, and that it has a tendency to ignore existing inequalities in health and power relations whilst at the same time reinforcing them through a sole focus on aggregate human-wellbeing. In many

respects these critiques can be assuaged by situating the ES concept within a landscape framing, as Sikor et al., (2013:198) remark:

“The justices and injustices of ecosystem services [...] are also about underlying issues of knowledge, value and governance [...] in short matters of social recognition.”

It is these matters, we would argue, that landscaping ecosystem services brings to the table.

16.3.5 Planning And Management

The comprehensive alignment of ecosystems services and landscape planning remains some way off, even though the last few years have witnessed an increasing emphasis on achieving this integration (e.g. von Haaren and Albert, 2011; Albert et al., 2014; Galler et al., 2016). Planners have had a mixed reaction to the widespread policy mainstreaming of ecosystem services, while some have been attracted to its capacity, especially within multi-sectoral processes, to crystallise the links between biodiversity and human-wellbeing; others have been turned off by its conceptual ambiguity, the dominance of monetary valuations and the difficulty in communicating its relevance to stakeholders (Albert et al., 2014). Nonetheless opportunities exist for combining ecosystem services and landscape planning, and many reasons why this represents a fruitful avenue to pursue. For example, landscape planning has the potential to embed a broad range of ecosystem services as well as provide directions for implementation strategies (von Haaren and Albert, 2011; Albert et al., 2014), whilst strongly emphasising the human influence on ecosystems (von Haaren and Albert, 2011). Furthermore, fully integrating a planning ethos into the standard ecosystem services model may provide a more suitable avenue to connect individual and societal demands in a way that takes account of legitimised social values and decision-making processes (von Haaren and Albert, 2011). From this perspective, it is argued that the transparency offered by landscape planning provides an avenue to make the normative aspects of ecosystem services more evident (von Haaren and Albert, 2011).

It is often stated that ecosystem service governance needs to be innovative, community-focused, and collaborative and involve mutual learning. Unsurprisingly, landscape planning provides a participatory approach to achieve those governance aims (Fürst et al., 2014). The substance and effectiveness of these participatory approaches relies on a shared knowledge and social network development. As Fürst et al., (2014) argue; integrating ecosystem services within a landscape framing furthers the development of an integrated and shared knowledge base, as well as promoting a shared vision and encouraging the development of social and human capital. However, as Albert et al., (2014) rightly point out; correctly integrating ecosystem services necessitates “careful scoping of context, objectives

and capacities”. While this is sound advice it should not be seen as an excuse to avoid incorporating ecosystem services into landscape planning because, as the evidence from regional planning suggests, integration helps to address the problem of “fit” between local knowledge about land use and management and wide societal needs (Fürst et al., 2013).

It has been consistently argued that a landscape platform for ecosystem services actively supports collaborative decision-making amongst diverse actors (Albert et al., 2014; Frank et al., 2014). These types of collaborative actions create the possibility of more “place-specific planning and implementation strategies” based on acknowledging social and ecological objectives (Albert et al., 2014). For example, Fürst et al., (2013) clearly make the case that by embedding ecosystem services in landscape planning the capacity to accommodate multiple knowledges is improved. The authors also highlight that planning across spatial scales is enhanced by the incorporation of “different data sets, sources and methodologies”, providing information to actors that is practically useful as well as meaningful and understandable (Fürst et al., 2013).

Finally, the statutory dimension of landscape planning focuses primarily on public goods provision which largely excludes “marketable” goods from accounting procedures. This contrasts starkly with ecosystem services where “market” goods are frequently involved in valuation exercises (e.g. in relation to provisioning services). As von Haaren and Albert (2011) note, including these “commercial goods” as part of the planning process could generate new options for planning arrangements and implementation. The authors also argue that from an ecosystem service perspective, the differentiation between public and private goods is beneficial (von Haaren and Albert, 2011). Overall, it is clear that when embedded within a landscape perspective the capacity for ecosystem services to aid landscape planning processes, decision-making and implementation bares most fruit (Galler et al., 2016).

16.4 Final Remarks

Over the course of the last three chapters we have attempted to present our view of landscape, and in particular, how our landscape approach to ecosystem services improves the social-ecological foundations of the ecosystem services framework, and how it more strongly connects to human-wellbeing and welfare. In many respects the reasons for this reside with the fact that landscape explicitly recognises a whole host of processes such as political, economic, social, cultural and environmental as well as more personal aspects of the individual (e.g. psychology, meaning and identity) alongside historical, legal and rights-based perspectives that are either absent, excluded or only implicitly considered by the current framing of ecosystem services.

We have argued that “landscaping” ecosystem services through the construct of landscape provides a stronger, more effective and robust, integrated concept better equipped to tackle the considerable challenges associated with environmental conservation, management and planning. Our underlying position has been that the connectedness and meanings humans imbue and attach to landscape affords a common language, dialogue and space capable of bridging and enriching science, policy and stakeholder narratives which together provide a more sustainable and coherent framework for articulating the ecosystem services paradigm. In this regard acknowledging the impact humans have on ecosystems is to recognize that these radical transformations result in new articulations of the environment which we identify as landscapes (Tarolli et al., 2014).

Landscapes are not just “other” but are also “everyday”, places we inhabit, experience and rely on to meet our basic needs, not simply locations to escape to (literally and metaphorically) but areas full of meaning and identity that hold memories and connect us to our ancestors (Howard et al., 2013). This description of landscape expresses the duty of care (or stewardship) we owe to these “hybrid” environments, and highlights the political and governance implications for responsibly managing our activities in ways that enhance and reduce respectively the environmental and social benefits and costs of our actions (Biermann, 2014). Embedding ecosystem services within the landscape construct imbues the concept with greater immediacy, tangibility and far richer layers of meaning and understanding for practitioners, stakeholders and the general public, whilst also emphasising the interdependence of humans on the environment (Termorshuizen and Opdam, 2009; Wu, 2013).

Ultimately, managing landscapes for multiple benefits, balancing competing human demands and meeting different policy and governance objectives is a serious challenge (Defries et al., 2004). Indeed, an emphasis on “landscape” aligns more strongly with sustainability science and concepts such as “landscape sustainability” and “sustainable landscapes” (Wu, 2013). Achieving these multiple objectives requires: (i) the design, production and maintenance of multi-functional landscapes; (ii) a closer connection between the social and the ecological to enable improved landscape planning and management; (iii) a greater alignment between natural resource governance and sustainable development emphasising stakeholder participatory processes; (iv) a more nuanced understanding of local and global scale interactions between social, economic, ecological and political drivers and, (v) a systems approach to consolidate this complexity in a holistic way (Anton et al., 2010; de Groot et al., 2010). Many of these issues were spelled out in relation to PES in Chapters 11 and 12.

In the final analysis human beings are meaning makers; we share a cultural and social history that in large part has been shaped by the environments we inhabit: they are part of the

our identity, common heritage and the story of our species is woven through their fundamental and ever changing fabric. In the modern age, the environments in which we inhabit are now largely human-dominated, directly or indirectly, and our activities have profound impacts on the way these environments function, for good or ill. Ecosystem services has become a powerful new paradigm to convey human-nature relations, but the capacity for ecosystem services to aid our ability to manage our activities sustainably relies on it having meaning for each of us. Without meaning and identity we lose attachment and our values ebb away, landscape provides the missing meaning for ecosystem services.

Notes

1. As defined by Cox (1998:2):

“Spaces of dependence are defined by those more-or-less localized social relations upon which we depend for the realization of essential interests and for which there are no substitutes elsewhere; they define place-specific conditions for our material well-being and our sense of significance. These spaces are inserted in broader sets of relationships of a more global character and these constantly threaten to undermine or dissolve them. People, firms, state agencies, etc., organize in order to secure the conditions for the continued existence of their spaces of dependence but in so doing they have to engage with other centers of social power: local government, the national press, perhaps the international press, for example. In so doing they construct a different form of space which I call here a space of engagement: the space in which the politics of securing a space of dependence unfolds.”

2. According to Brenner (2001:599) the singular notion of the ‘politics of scale’ refers to:

“...the production, reconfiguration or contestation of some aspect of sociospatial organization within a relatively bounded geographical arena – usually labeled the local, the urban, the regional, the national and so forth. In this singular aspect of the ‘politics of scale’, the word ‘of’ connotes a relatively differentiated and self-enclosed geographical unit.”

On the other hand the plural conceptualisation of the ‘politics of scale’ refers to:

“...the production, reconfiguration or contestation of particular differentiations, orderings and hierarchies among geographical scales. In this plural aspect, the word ‘of’ connotes not only the production of differentiated spatial units as such, but also, more generally, their embeddedness and positionalities in relation to a multitude of smaller or larger spatial units within a multitiered, hierarchically configured geographical scaffolding [...] Here, then, geographical scale is understood primarily as a modality of hierarchization and rehierarchization through which processes of sociospatial differentiation unfold both materially and discursively.” (pg. 600)

3. As Marston et al., (2005) conclude at the end of their article critiquing hierarchical scalar constructions:

“...we are convinced that the local-to global conceptual architecture intrinsic to hierarchical scale carries with it presuppositions that can delimit entry points into politics – and the openness of the political – by pre-assigning to it a cordoned register for resistance.” (pg. 427)

4. According to Marston et al., (2005) who argue that one can do away with hierarchical scalar representations altogether they assert that one way of achieving this outcome is to think in terms of flows and fluidity:

“According to this approach, the material world is subsumed under the concepts of movement and mobility, replacing old notions of fixity and categorization with absolute deterritorialization and openness” (pg. 423)

However, they do not align themselves with this view based on its potential liberalist extremes.

5. From this perspective, one that Marston et al., (2005) find particular agreement with, the concern is with materiality, and in particular the way material materializes and dematerializes according to the operations of the complex system, as they describe this approach:

“...focuses on both material composition and decomposition, maintaining that complex systems generate both systematic orderings and open, creative events.”

They go on to argue that is approach is preferable because:

“Leaving room for systemic orders avoids the problems attendant to imagining a world of utter openness and fluidity that inevitably dissolves into problematic idealism. Further, this approach allows us to avoid falling into the trap of naïve voluntarism by embedding individuals within milieu of force relations unfolding within the context of orders that constrict and practices that normativize.” (pg. 424)

6. According to Duraiappah et al., (2014) this term refers to institutions that:

“...mediate interactions between the natural system, natural capital and ecosystem services”

And specifically they determine:

“...how capital assets are distributed to be used efficiently and effectively” (pg. 98)

7. According to Duraiappah et al., (2014) this term refers to institutions that:

“...oversee the access and distribution of the various capital assets across the various social entities at the respective levels and across multiple scales” (pg. 98)

8. Ecological fit takes a technical perspective and is concerned with the relationship between ecological and biophysical problems and the institutions designed to address these issues (i.e., is there a good alignment, are they well-matched). Ecological attributes of the system are identified and then compared to the attributes of the governing institution(s). The more complex the system the more attributes that have to be compared (Epstein et al., 2015).

9. Social fit addresses the alignment between the values, needs and preferences of human agents (e.g., their beliefs or even psychological make-up) and governing institutions. Social fit has three aspects to it: (i) overlap between operational rules and the social context – match between rules and resource use; (ii) the suitability of rule-making taking into account the needs and expectations of human actors, and (iii) the scale of fit between institutions and levels of social organization – do institutions act to enable and enhance group capabilities (Epstein et al., 2015).

10. SES fit concerns uncovering how context-specific institutional arrangements and factors contribute to the overall sustainability of the system, in particular, by focusing on interactions between institutions and different elements of the system. In other words, it asks the question

whether institutions are fit for purpose in the sense that they are capable of governing complex systems (Epstein et al., 2015).

11. Hierarchical governance refers to supra-level policies that are adopted at an international level of political process and filter down to affect national and sub-national policies (i.e., the direction of flow is from higher level to lower level governance) (Primmer et al., 2015)

12. Scientific-technical governance particularly focuses on the impacts governance has on biodiversity and the development of governance to better manage biodiversity through scientific knowledge, support systems and platforms, based on accumulated evidence, to formulate effective policy (Primmer et al., 2015)

13. Adaptive collaborative governance effectively relates to the idea of sustainable environmental management – it integrates a range of different norms, stakeholders and communities in order to engage in ‘collective governance’. Particular emphasis is placed on knowledge accumulation, knowledge exchange and social learning (Primmer et al., 2015)

14. Governing strategic behaviour refers to the reality that in some cases there will be individuals or groups of actors that do not participate in governance activities and the implementation of policies because they see these are the dominant imposition of a particular discourse. Instead they perceive the dominant governance paradigm as a barrier and contestable (Primmer et al., 2015)

15. We follow Reed et al. (2010) in defining social learning as a:

“...process of social change in which people learn from each other in ways that benefit wider social-ecological systems”.

16. Soft systems thinking emphasises an open-ended and pluralistic discourse, moving away from a problem-solution orientation, to a form of decision-making between actors that generates co-created frames of reference (Cundill et al., 2012).

17. Knowledge exchange refers to a situation in which knowledge is ‘exchanged’ between a chain of knowledge producers, intermediaries and knowledge users (consumers), and primarily relates to the use of knowledge in a policy and practice arena, where concerns relate not only its relevancy, legitimacy and accessibility but also its production, translation and dissemination (Reed et al., 2014).

18. In a recent Strengths-Weaknesses-Opportunities-Threats (SWOT) assessment of the ecosystem services framework conducted amongst young ecosystem service specialists, whilst the ecosystem services concept was acknowledged to have a number of strengths significant weaknesses and threats included its ‘ambiguous language’, ‘inaccessibility to non-specialists’ and ‘lack of awareness across the general public’ (Bull et al., 2016).

19. In Eppler’s (2006) terms ‘knowledge communication’ is the:

“...deliberate activity of interactively conveying and co-constructing insights, assessments, experiences, or skills through verbal and non-verbal means.” (pg. 2)

In this sense, Eppler argues that knowledge communication is:

“...the elicitation of one’s rationale and reasoning [...] of one’s perspective, rankings and priorities, and of one’s hunches and intuition.” (pg. 3)

20. According to Eppler (2006) knowledge dialogues, of which he categories into four distinct types, are ‘synchronous real-time interactions’ which emphasize collaboration and interaction specifically for the mediation of knowledge exchange and communication. The four types of knowledge dialogues are as follows: Crealogues (creation of new insights); sharealogues

(facilitation of knowledge transfer); assessalogues (new insight evaluation) and doalogues (understanding into action).

21. Justice is a multidimensional concept normally divided into distributive, participatory (procedural) and recognition aspects. Distributive justice concerns focus on how benefits and costs (or goods and bads) are partitioned between individuals. On the other hand, participation or procedural justice concerns address decision processes – how they are made, the roles of particular individuals within those decision processes and the rules that govern decision-making. Finally, recognition justice is about people’s identities and histories, about respect and power, and therefore whose culture dominates. At the same time environmental justice addresses a plurality of issues along these lines, for example, intra- and inter-generational notions of justice, justice as occurring between particular social groups and inter-species justice (Martin, 2013; Sikor, 2013a).

22. For example, as Martin (2013:100) states:

“...the addition of a global framing is made necessary by the scales of contemporary environmental problems such as radioactive fallout, the political-economic forces that pattern the uneven access to resources and exposure to harms, the vocabularies and narratives that are used to make claims about these perceived inequities, and the associated moral communities.”

17. Discussion

In the Twenty First Century we face a complex and challenged world, with multiple intertwining global environmental challenges from climate change and marine pollution to food, water and energy security; to biodiversity loss (Vince, 2014; Sachs, 2015). At the same time these global environmental challenges interact with, and at some scales and in particular contexts are driven by, widespread social challenges such as human poverty, health and disease, poor governance, and social inequalities in wealth and access to basic services (Wilkinson and Pickett, 2012). Finding sustainable ways of meeting these challenges is perhaps the biggest challenge of all.

At the outset of the thesis we began by describing, through the Garden of Eden device, a situation in which human actions have led to a widespread deterioration in global environmental systems: transforming our idealised “Edenic” state into a series of post-lapsarian degraded gardens. We suggested that in more recent environmental conservation history, in theory, practice and in policy, there has been a turn towards viewing, what are now regarded as linked environment-development issues, from a human-wellbeing (welfare) perspective – a perspective which has come to be known as the ecosystem services framework. Here nature provides “services”, partially through human transformation of material inputs, which are essential to our wellbeing. In a sense we now have a ‘service-driven’ view of human-environment relations, one that to some extent is replacing an older ‘protectionist’ conservation that split humans and nature into two different camps. It was argued then that the purpose of this thesis was to explore ecosystem service theories and applications as well as to provide both practical and policy-relevant evaluations, using the “Garden of Eden” device as a narrative arc. So what have we found out; what important issues (challenges, barriers and opportunities) have emerged, and what does this tell us about the road ahead?

17.1 Summary Of Main Outcomes

In Part 1 *The State of the Planetary Garden* Chapter 1 we discovered a world which has been transformed, severely impacted by human activities, activities whose far reaching effects have been noted on a myriad of different biosphere systems from the land, to the sea to the air. We might say that what Chapter 1 identified was a growing sense, to use Ed Barbier’s phrase, of “Ecological Scarcity”: that our natural resource base (what is more frequently referred to as natural capital) is being degraded and eroded (Barbier, 2011). Yet, in a perverse way, this erosion also acknowledges – and this was also clear from Chapter 1 – the sheer

number of individuals, communities, groups, and populations (you only need to look at the forestry sector) whose livelihoods depend on these resources. So, whilst Chapter 1 painted a rather bleak picture of the current state of the Garden, for instance, in terms of the scale of our impacts as documented by the example of planetary boundaries, on the other hand, it also evidenced that our stake or (self-interest) in the functioning of that Garden, in terms of livelihood dependence for example, means that we have the necessary impetus required to take action to prevent (and halt) any further escalation in the planetary-wide degradation we have witnessed since the 1950s (Vince, 2014).

In Chapter 2 we explored the connections between life in the Garden and ecosystem services. During our exploration we established that the set of relationships existing between biodiversity, ecosystem functioning and service generation are complex but, at the same time, are also starting to become clearer as the research investigating these linkages becomes more robust. At its core biodiversity is seen as central to the provision of ecosystem services: we must therefore regard the continued escalation in biodiversity loss (e.g. Butchart et al., 2010; WWF Living Planet Report, 2014) as severely compromising the future provision of ecosystem services and human-wellbeing (Cardinale et al., 2012). These issues, as we explain, are fuelling many national, international and global research and policy programmes (e.g. Future Earth, IPBES) dedicated to increasing our understanding of biodiversity-ecosystem service connections, assessing the impacts we are having on those systems and recommending ways of mitigating, managing and adapting to a changing environment.

Moving on to Part 2 *The Human Garden* we explored the idea that in our post-lapsarian existence we have inherited a human-dominated Garden, but while this may be the case – that the handprint and footprint of humanity touches nearly every aspect of the biosphere – it is nevertheless, and this is the critical point, a rich and complex social-ecological system. In other words, we described the importance of understanding human-nature relations as a complex set of social-ecological interactions and connections, and that it is in essence this “connectivity” that is central to how our Garden functions. In Chapter 3 we established that social-ecological thought has been a growing area of research within the ecosystem services narrative and has provided us with a rich body of discourse and insights – not simply in terms of investigations into underlying human-nature relations in terrestrial and marine environments but also in its application to the governance and management of those systems. We also catalogued in Chapter 3 the growing significance of urban environments in the everyday lives of humanity as we took a glimpse at the life of *Homo urbanus* (Vince, 2014). Specifically, we acknowledged how our urban creations are functioning as a new source of ecosystem services but, at the same time, due to their proliferation as growing and developing entities represent hugely significant drivers of global environmental change and centres of

immense natural resource consumption: creations that are increasingly imperilling the capacity of the Earth System to continue to provide a sustainable flow of ecosystem services. We emphasized that how we manage and govern our urban future, in terms of the development of green infrastructure and smart city developments, will be central to their sustainability – in terms of their own functioning and the wider natural resource base on which they rely. Having provided a broad overview of social-ecological thought, in Chapter 4 we highlighted and evidenced key concepts in social-ecological systems analysis which have become dominant in global conversations about ecosystem services, in particular, the concepts of resilience and regime shifts. We set out how these conceptual ideas and research avenues, despite certain flaws, have shaped our thinking regarding the way social-ecological systems operate and are affected by human activities, emphasising the important theoretical and policy insights they have provided.

Parts 1 and 2 formed the background to our discussions in Part 3 *The Garden in the Age of Sustainability*. In Chapter 5 we concentrated our attention on the narrative of sustainable development and how, increasingly, the ecosystem services paradigm is framed within a sustainability agenda, where a growing emphasis is placed on the interdependence of environment-development problems (and solutions), highlighted in international policy developments such as the Aichi 2020 biodiversity targets and the new SDGs (Sachs, 2015). It was quite apparent from our discussions that the sustainability agenda is multi-dimensional and its implications for ecosystem services comprises a number of separate strands and discourses of investigation and analysis: some of which focus on environmental sustainability, others on social and economic sustainability, and still others on conflict and sustainability alongside poverty and human-wellbeing linkages. Major themes underpinning these discourses are cast in terms of the Anthropocene, notions of stewardship, and chronic societal inequalities in wealth, prosperity, health, race, gender, education and governance – a whole gamut of issues that act as fissures to undermine how social-ecological systems function (Wilkinson and Pickett, 2012; Sachs, 2015). The evidence we examined pointed to the fact that poverty and development challenges can seriously undermine ecosystem service provision and also generate significant ecosystem disservices (e.g. zoonotic diseases which impact directly on human health). What is also clear is that poor environmental conditions can worsen current social, economic and political inequalities, entrench poverty traps and undermine human health; creating a viscous form of path-dependency, which can have significant ramifications for food, water and energy security. Yet, although the sustainability of the Earth System is questionable based on our current business-as-usual trajectory, what Chapter 5 also demonstrated is the significant progress that the last three decades have witnessed with regards to a wide number of environmental and social sustainability problems, and while much

progress still needs to be achieved, efforts such as the 2030 Agenda for Sustainable Development hold out the prospect of moving us in the right direction (UN, 2016).

In Part 4 *Assessing the Garden*, Chapter 6 briefly explored the range of practical and methodological approaches that are currently applied to examine ecosystem service generation, provision and distribution, in both an ecological and policy context, and we focused on three areas: assessing trade-offs, mapping and modelling, and indicator developments. Collectively, this triumvirate approach highlighted the broad application of these tools and instruments and demonstrated how, in a short space of time, they have enhanced our understanding of the factors that affect the spatial provision and distribution of ecosystem services and how these relate to and impact different social and economic correlates of human-wellbeing – such tools have become an important component of environmental management decision-making and policy formulation (Burkhard et al., 2012). Overall, we showed there is a continuing research and policy agenda concerned with uncovering how the social-ecological interactions within the Garden determine and influence ecosystem service provision and distribution and the interrelations between ecosystem services and human-wellbeing.

Quantification is also a central step in the process of ecosystem service valuation, particularly in relation to monetary valuation assessments, and increasingly for non-monetary assessments too: it is difficult to attribute “value” to something, or the changes in something, if that property cannot be captured (Perrings, 2014). This observation provides a neat route into discussions covered by Part 5 *Valuing the Garden*. A sizeable dimension to the ecosystem service framework relates to the processes and problems of valuation. Ecosystem services is at its core inherently anthropocentric, it is concerned with how changes in the “natural world” (i.e. natural capital and ecosystem services) affect human welfare and wellbeing – it is these welfare impacts that generate the concern and interest in valuation. A substantial focus of ecosystem services research to date and a considerable component of the mainstreaming of ecosystem services into policy making and general environmental decision-making has focused on the assessment and development of valuation processes.

In Chapter 7 we provided an overview of the main theoretical and practical debates surrounding the underlying rationale for valuation as well as describing recent developments in valuation methodologies. From our discussions it is clear that the major fault lines running through valuation arguments centre on those who, on the one hand, regard monetary valuation as overly dominant in the language and assessment of ecosystem services and view it as reductionist (i.e. it favours instrumental value over intrinsic value) and generally inappropriate (i.e. particularly in terms of the assumptions it makes regarding how people make value judgements and the significance they assign to monetary values). From this

perspective the argument runs that there has been a widespread failure to capture the “full” range and value of ecosystem services, which has impoverished policy and side-lined other more apt non-monetary valuation approaches. On the other hand, the counter-argument if you will, there are those that regard monetary valuation, where price signals are considered a reasonable indicator of scarcity and peoples preferences, as an essential tool for articulating the value we place on ecosystem services. Why? Because, ultimately, these values are viewed as absolutely necessary for directing the choices we need to make between different decision-making pathways, each of which will have pluses and minuses for ecosystem service provision and human-wellbeing.

Having aired the various underlying theoretical as well as practical debates, in Chapter 8 we sought to provide a general flavour of the current application of valuation assessments to a range of ecosystem services across a broad array of scales – from global assessments of the planetary-wide significance of ecosystem services to the human economy, to the value of local services derived from a forest or wetland. What we established in this brief tour was the widespread variations in values across different services, between different geographies and within services at the same scale. At the same time, we also demonstrated that current valuation studies are more engaged with assessing bundles of ecosystem services across diverse ecosystems, and they are doing so with more refinement, moving beyond earlier more simplistic valuation studies with a blinkered single service perspective. Having said that, provisioning and regulating services are still the most frequently ‘valued’ types of ecosystem services and studies that focus on assigning values to the connections between services and human-wellbeing remain in the minority. Finally, in Chapter 9 we examined some of the principal culprits behind the variations we see in valuations and the difficulties inherent in ascertaining accurate values, which we identified as emanating from uncertainty, benefit transfer and discounting. Whilst we established that steps have been taken to reduce the errors associated with each of these aspects, they will nevertheless continue to present as sources of underlying variance in the process of valuation.

Ecosystem services is not simply about the re-orientation of human values at the centre of environmental issues, and the diagnosis of those problems and challenges from an anthropocentric perspective, it’s also about the practical engagement and management of those environmental issues. Crucially, it’s about how we provide a sustainable flow of ecosystem services, how we meet the challenges posed by competing resource needs and environmental uses, and how we balance environment and development pathways in the Global North and the Global South. Much of what we have discussed thus far (from Part 1 to Part 5) finds its inculcation in Part 6 *Managing the Garden*. In this section, we highlighted two types of environmental management incentive instruments, namely payments for ecosystem

services and agri-environment schemes, as policy tools increasingly used to navigate the difficulties posed by these complex issues. Prefacing the examination of these policy tools Chapter 10 provided a short discussion of ecosystem services and the provision of public goods, outlining the theoretical underpinnings upon which the incentive programmes we investigated in Chapters 11, 12 and 13 are based.

Chapter 11 presented a global evaluation of PES schemes; our purpose was to assess their effectiveness through a capital asset examination of their outcomes. Specifically, we used a capital asset framework to evaluate PES programmes in terms of their social, environmental, economic and institutional outcomes, focusing on efficiency, effectiveness and equity trade-offs. Most PES schemes operate in low and middle income countries. We found that PES schemes can provide positive conservation and development outcomes with respect to livelihoods, land-use change, household and community incomes, and governance. We also established that programmes differ with regards to contract agreements, payment modes, and compliance, and have diverse cross-sector institutional arrangements that remain primarily state-structured and external donor-financed. At the same time there appears to be a consistent lack of focus on evaluating and fostering human, social and institutional capital. This reflects general inattention to how programmes consider the causal links between ES and outcomes. To enhance ES production and PES scheme accessibility and participation, we argued that: the linkages between ES production and land-use practices needed strengthening; private and voluntary sector involvement needed boosting; property rights and tenure reform needed to be encouraged, financial flows and viability required securing, and the distribution of programme costs and benefits among participants needed to be adequately accounting for. In attempting to achieve these progressive outcomes we argued that a function-oriented outcome-led approach, combined with a capital asset perspective, should guide the design and implementation of PES schemes.

In Chapter 12 we carried on the PES theme, but this time looked at the prospects of developing PES programmes for a globally important marine ecosystem like seagrasses, alongside other 'Blue carbon' management schemes, as a mechanism to enhance their protection and reduce their loss for climatological, conservation and development reasons. We detailed the array of ecosystem services seagrasses provide (from carbon storage and sequestration to cultural services), outlined the prospects for including seagrasses under current global climate policy frameworks and carbon management schemes, and then suggested how PES schemes might be developed alongside these based on various avenues such as fisheries, ecotourism, restoration etc. We suggested that the likelihood of developing seagrass Blue carbon programmes based on the regulated carbon market, as it currently stands, is relatively slim but that the voluntary carbon market presents more realistic opportunities.

An additional benefit of the voluntary carbon market we argued was their increasing focus on delivering projects with co-benefits (i.e. livelihood benefits), and we suggested that this could represent an important avenue to explore in relation to the parallel development of PES schemes. Overall, we argued that a combined and complementary strategy would yield the greatest set of conservation and livelihood benefits. But, at the same time, we pointed out, as we had previously argued in Chapter 11, that achieving these outcomes would require a necessary focus on PES design and implementation, in particular a focus on: institutional arrangements, stakeholders, tenure and property rights, benefit sharing, provision and monitoring of ES, and financial viability.

The two previous chapters both examined PES schemes, one from a global-scale perspective ranging across multiple ecosystems and the other concentrated on a single ecosystem; albeit seagrasses they do have a global distribution. At the same time these chapters implicitly and explicitly focused on conservation, development and resource management issues largely predominating in low and middle income countries - although many of the lessons they gleaned would be applicable to other geographic contexts. Nevertheless, they highlighted a set of incentive-based schemes that are currently enjoying a flourishing in a largely non-Western context. Moving on from PES schemes, Chapter 13 turns to an examination of agri-environment schemes. In many respects AES can be considered a cousin to PES, but one that operates in the formal agricultural sector of the 'Global North'. Agri-environment schemes have been around for decades as a means of balancing production activities against other rural land management issues. In Chapter 13 we focused our attention on the agri-environment context of Europe and specifically the UK and England as we examined Environmental Stewardship schemes. In particular, we were concerned to highlight the role and views of farm advisors, a constituency that is highly involved in the implementation of these schemes but also at the same time a constituency whose views are rarely assessed in peer reviewed discourse.

Our examination of farm advisors indicated that the majority had knowledge and expertise in relation to two (31.5%) or three (42.2%) Environmental Stewardship schemes, with proficiency in ELS (93.4%) and HLS (82.8%) being the most common. On average, advisors had 9.6 ± 5.6 yrs of experience and operated (75.3%) in a single region of England. Although our results concentrated upon a relatively simple set of initial topics of inquiry, the survey revealed a number of interesting findings. Firstly; for example, that in the opinion of farm advisors the 'knowledge-exchange encounter' occurring between themselves, their clients and Natural England is fundamental to the environmental effectiveness of these schemes as well as their farm business compatibility. Secondly, advisors suggested that beneath this 'encounter' lie tensions arising from the competing agendas and objectives of the different

actors involved which can affect the content of agreements; for instance, farmer selection of management options versus Natural England's target environmental objectives. Farm advisors suggested that they had to negotiate this balance whilst also serving the needs of their clients. Thirdly, respondents raised issues concerning the complicated nature of scheme arrangements, both from their own and farmers' perspectives, as well as the adequacy of payments to cover input costs and matters regarding contractual compliance, all of which they proposed affected farmer participation. Overall, we argued that the future design of agri-environment schemes must acknowledge the different agendas and dialogues occurring between farmers, advisors, and Natural England; participation by farmers must cover their opportunity costs, but not come at the expense of environmental ambition, and therefore contracts must be properly enforced; and just as importantly the operation and implementation of these schemes must be simple and straightforward and accommodate farm business requirements.

Part 6 has clearly shown how the ecosystem services concept can be applied to the use and development of financial incentive mechanisms for sustainable environmental management, highlighting its importance as a policy instrument in the agricultural sector but also in contexts where less top-down regulatory arrangements might be preferred as a way of achieving collaborative approaches to common-pool resource management issues (Keune et al., 2014).

Whilst Part 6 demonstrated that taking an ecosystem services perspective to the development of environmental management incentive programmes can yield beneficial outcomes (in ecological, social and economic terms), it also flagged up a number of issues particularly around social, institutional and governance related areas that are not necessarily well served by how these programmes have so far been implemented. Why? Primarily because the social-ecological foundations of ecosystem services have not been fully explored and instituted in a way that draws out the complex set of interrelated environmental, social, cultural, political, economic and institutional dimensions that are at play. In the final section of the thesis Part 7 *A Proposal for a Landscape Approach to Future Garden Management* we put forward the idea that situating ecosystem services within a landscape-context will help address some of these shortcomings.

In Chapter 14 we addressed the concept of landscape, showing it to be a multi-dimensional and highly complex and dynamic social-ecological construction that carries significant sets of meanings, with important implications for the way people perceive, interact with and view human-nature relations. We highlighted the fact that landscape has features that explicitly acknowledge ideas such as identity, psychology, wellbeing, scale and place, control, power, and justice, settlement and exploitation and cultural and religious practices – aspects that are either entirely absent or largely ignored in the “mainstream” interpretation of

ecosystem services. In Chapter 15 we turned our attention to transforming the complex multidimensionality of landscape introduced in Chapter 14 into a “landscape approach”, with a general explanatory synopsis. In this chapter we suggested that our more integrative approach improved and advanced both the conceptual, practical and communicative foundations of the ecosystem approach instituted by the Malawi Principles. The specifics of the landscape approach were more fully explored in Chapter 16, where we fleshed out in greater detail the different components of the landscape approach framework we presented: addressing issues such as scale, ecosystem service provision, distribution and translation into human-wellbeing, governance, and management. We also noted the approach’s implications for ecosystem services in five broad areas: landscape multi-functionality and connectivity; communication; ecosystems and cultural services; environmental and social justice; and planning and management.

Overall, we argued that articulating ecosystem services within a landscape framing roots the concept in ecological, social, cultural, economic, institutional, governance and political dimensions; provides the most suitable vehicle for communicating the relevancy of the ecosystem service concept; and affords a more inclusive, immediate and participatory approach to landscape management by providing stakeholders and decision-makers with an accessible and common language that is practically and policy-relevant. In essence, emphasising the complex social-ecological dynamics underpinning ecosystem services offers a more realistic chance of addressing fundamental sustainability and governance challenges, because notions of scale and interconnectedness are explicitly considered at the outset. Moreover, it provides common points of articulation for practitioners and policy-makers to engage in meaningful decision-making processes, ultimately bridging divides amongst sectors and enabling win-win situations (Keune et al., 2014).

17.2 What Has This Thesis Demonstrated?

First, it has confirmed the conceptual expansiveness of the ecosystem services concept and its increasingly accepted characterisation of environmental management and biodiversity conservation issues in science-policy circles, as evidenced by the developments such as the 2020 Aichi targets, IPBES and Future Earth. Clearly, in this respect, the ecosystem services paradigm has been highly influential in a very short space of time. Conceptually, ecosystem services lends itself to combining with the development agenda and other service and target oriented sectors, for example, there is a high degree of overlap between the Aichi targets and the SDGs, which represents an important acknowledgement for achieving the 2030 Agenda for Sustainability.

Secondly, the thesis has sought to evaluate some of the ways that ecosystem services is practically applied to deal with complex natural resource management issues, namely, in the form of environmental incentive-based measures. From this perspective the research presented has furthered the theoretical development of PES by providing an approach to rigorously assess programme outcomes and design robust and functional schemes. Moreover, it has highlighted the range of barriers and challenges that exist and need to be overcome in order to successfully implement a functioning and effective incentive-based intervention measure. In addition, I have argued, in the case of seagrass ecosystems, that PES schemes can be developed and designed to be complementary to other carbon management programmes based on additional ecosystem services so as to maximise environmental and livelihood benefits. This is a key development with important global implications for poverty alleviation programmes in coastal communities and climate change mitigation and adaptation strategies. In relation to agri-environment schemes, by viewing these schemes from a farm advisors perspective I have offered an appraisal of how various actor, agreement and institutional arrangements affect their implementation and performance. Recognition of this fact is an important advance that could be factored in to the future design and implementation of agri-environment schemes and lead to an overall boost in programme efficiency and effectiveness.

Thirdly; however, the thesis has illustrated the many pitfalls that continue to exist within the ecosystem services field, particularly in terms of the application of valuation methodologies and the translation of ecosystem services into practical decision-making arenas (e.g. incentive-based policy measures like PES and AES). Consequently, the thesis has strongly argued that ecosystem services needs to be grounded in a social-ecological framing, and we have offered the landscape approach as the 'best' expression of that social-ecological framing because it provides a means of applying the ecosystem services concept in a consistent, coherent and legitimate manner. Why? Because this integrated approach emphasises the inextricable linkages between nature and society; it provides a common language for all stakeholders to engage with; it highlights all the relevant factors, drivers, and pressures affecting the system; it offers points of articulation where the system can be investigated and it ensures that social and ecological concerns (and not just economic and financial but also cultural and religious, power and justice) matters are addressed.

17.3 Moving Forwards: Research Developments

17.3.1 Connecting With Cultural Services

The ecosystem services concept still has much to give as well as much to prove. Going forwards, there is considerable scope for concentrating on the social and cultural values that are embedded in a landscape approach to land management, values that are on the whole

overlooked by monetary valuations of ecosystem services in a concept that has the potential to be quite removed from everyday experiences. The power of ecosystem services lies in its integrative qualities and its capacity to connect meaningfully with stakeholders and policy-makers, at present that power is still limited. Framing the concept in a way that connects to everyday experiences, which draws on a multitude of values and belief systems about the natural environment, and at a scale that is policy-relevant is far more likely to generate effective and appropriate policies and management decisions. In this regard, focusing on the developing our knowledge of cultural ecosystem services and cultural landscapes using participatory and deliberative techniques is crucial. The use of the landscape construct in this situation will be highly tractable (Bohnet and Konold, 2015; Tilliger et al., 2015).

17.3.2 Globalization And Ecosystem Services

As our landscape approach advanced, viewing ecosystem services in a supply-demand perspective emphasises the importance of assessing the drivers that provide momentum and dynamism to the system. A major driver that works across scales to influence SES dynamics is globalisation and trade. The relationships between national resource trade (exports and imports) and domestic consumption patterns, so-called teleconnections, is still a burgeoning area but is critical to questions of global sustainability. Connecting the discourse of global trade and ecosystem services is essential because the impacts of national resource use patterns have considerable local and global scale social, ecological, political and governance ramifications, which lie at the heart of energy, food and water security challenges. Trade and globalization also provides another common point of articulation between environment and development issues, and offers avenues to explore the links between conservation and poverty in relation to income inequalities, social justice, global power asymmetries, human rights and supply-chain management (Koellner, 2011; Challies et al., 2014; Peters, 2014; Lenschow et al., 2016).

17.3.3 PES Developments

Global and regional analyses of PES continue to support their potential to negotiate environment and development issues, whilst at the same time highlight persistent flaws in their design and institutional operation and implementation arrangements that undermine their effectiveness (Ezzine-de-Blas et al., 2016; Grima et al., 2016; Raes et al., 2016). Embedding PES within a social-ecological framing is regarded as a necessary step to improve their social, ecological and economic impacts (Bennett and Gosnell, 2015), as well as consolidating their theoretical and conceptual foundations (Van Hecken et al., 2015a). Applying the landscape approach to the development and application of PES schemes, for example, means paying greater attention to issues of programme effectiveness and equity in relation to institutional and social settings (Calvet-Mir et al., 2015; Chen et al., 2015) and

fairness with regards to the impact and distribution of programme outcomes (Leimona et al., 2015a).

Achieving this will require an improved set of social criteria for determining how PES schemes affect participants, for example, based on the idea of human capabilities (Kolinjivadi et al., 2015a), as well as enhanced social targeting of PES schemes that acknowledges a pluralistic value system – not simply a one-size-fits-all approach based on liberal “efficiency” values (Kolinjivadi et al., 2015b). Increasing participation rates, strengthening stakeholder engagement, and improving stakeholder collaboration in knowledge production and development regarding the design and implementation of PES (Leimona et al., 2015b; Page and Bellotti, 2015; Perevochtchikova et al., 2015) represents three complementary opportunities to meet these goals. Improved monitoring and evaluation of programme outcomes will be central in determining whether schemes are meeting their stated goals (Pham, 2015). Assessing the institutional, economic and political factors and settings in which PES programmes operate and are implemented will be necessary to judge how these influences shape scheme targeting, monitoring, evaluation and outcomes (Birmont and Karsenty, 2015; Hayes et al., 2015) as well as embed particular value narratives in their design and implementation (Van Hecken et al., 2015b). Finally, further growth in PES applications will need to address financial viability, and consider more innovative sources of funding, rather than relying solely on government-funding and international donors such as the World Bank and GEF. Much more consideration needs to be focused on private investment of which there are many options such as green bonds, crowd funding, and trust funds to name but a few, as recently highlighted by the Global Canopy Fund (Oakes et al., 2012).

17.3.4 Investment, Wealth Creation and Sustainability

Private investment in global conservation financing is growing, and needs to continue to grow if the conservation funding gap is to be bridged (Perings, 2014). I think it is important not to see the increasing involvement of the private sector as selling-out or leading to the privatisation of nature (Arsel and Büscher, 2012). Adopting this position is a simplification and misrepresents reality. That is not to say that private investment should become the main source of future conservation financing or that it should replace current funding sources, or indeed, that in any one project the private sector will (or should) be the only sector (and partner) represented. The likelihood is that multi-partner projects that share expertise and the knowledge of different sectors will be the most successful. However, it is to suggest, rather pragmatically, to the conservation community that PES-type programmes cannot function without secure and viable sources of funding, and at present many programmes cease after only a few years or don't even make it off the page due to inadequate funds. In addition, the private sector also offers additional benefits in terms of enterprise development. There is a

chance that greater private sector involvement in PES projects would deliver programmes with a focus on being financially self-sufficient through a suite of income generating activities (Credit Suisse, WWF, McKinsey & Company. 2014; NatureVest and EKO. 2014). Wealth creation offers the potential to boost livelihoods, enhance local economies and reduce poverty (Sachs, 2015), and depending on the sustainability trajectory of those wealth creation pathways poverty reduction effects may also decrease peoples' direct dependence on biodiversity exploitation (Perrings, 2014). In particular, investments in rural agriculture (FAO, 2013), for example, offer up the opportunity to build capacity (FAO, 2014), increase employment (FAO, 2016), improve rural institutions (FAO, 2012) and strengthen weak governance regimes (Sachs, 2015). Such developments should not only be thought of as restricted to developing world environments either, innovative sources of funding could also be used by the farming community in developed counties to provide more flexible sources of funding for on-farm management activities that are complementary to existing statutory agri-environment schemes for example, like the Agri-Tech Catalyst Fund in the UK and the European Innovation Partnership for Agricultural Productivity and Sustainability (UK Government, 2016).

Collectively, these research developments represent opportunities to address the three ecological scarcity challenges identified by Barbier (2011), namely: the sustainability challenge, the equity challenge, and the funding challenge.

Returning to the Introduction's Garden of Eden device, I hope to have demonstrated that the ecosystem services concept, its practical application in the form of incentive schemes such as PES and AES, and its re-contextualisation within a broader landscape approach does, in fact, offer a more promising avenue to secure future Gardens of Eden and regain paradise compared to Twentieth Century conservation endeavours. If we advocate for the strengths of the ecosystem services paradigm whilst also bearing in mind its potential shortcomings then it offers a promising vehicle to reconcile many current unsustainable practices and put us on a path to greater future prosperity. In so doing we may yet overcome the main challenge that Perrings (2014:484) identifies:

“...provision of the many benefits offered by the world's ecosystems depends on the establishment of multiple governance mechanisms operating at many different spatial and temporal scales, all of which share one fundamental characteristic. No matter what the scale, no matter who the constituents, all seek to regulate the activities of people exploiting the common pool dimensions of ecosystems to meet the many objectives of those constituents.”

As Barbier (2011:300-301) observes:

“Overcoming these institutional rigidities will be necessary if progress towards more sustainable forms of economic development is to be realized [...] Capitalizing on nature does not mean selling off the natural environment and

ecosystems. It means recognizing the value of ecosystems as natural assets so that we no longer view natural capital as free.”

LIST OF COMMON ABBREVIATIONS

AES	Agri-Environment Scheme
CAS	Complex Adaptive System
CBA	Cost Benefit Analysis
ELS	Entry Level Stewardship
ES	Ecosystem Services
ESP	Ecosystem Service Paradigm
ESV	Ecosystem/Environmental Services Valuation
EU	European Union
HLS	Higher Level Stewardship
MA	Millennium Ecosystem Assessment
MBI	Market Based Instrument
MDGs	Millennium Development Goals
NGO	Non Governmental Organisation
OELS	Organic Entry Level Stewardship
PB(F)	Planetary Boundaries (Framework)
PES	Payments for Ecosystem Services
SDGs	Sustainable Development Goals
SES	Social-Ecological System
TEEB	The Economics of Ecosystems and Biodiversity
U(O)ELS	Upland Organic Entry Level Stewardship
UN	United Nations
WTA	Willingness to Accept
WTP	Willingness to Pay

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