Jane Carter Ingram · Fabrice DeClerck
Cristina Rumbaitis del Rio *Editors*

Integrating Ecology and Poverty Reduction

The Application of Ecology in Development Solutions

Foreword by Professor Jeffrey D. Sachs,
Director of the Earth Institute

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Editors
Jane Carter Ingram
Wildlife Conservation Society
Bronx, NY, USA
cingram@wcs.org

Cristina Rumbaitis del Rio
The Rockefeller Foundation
New York, NY, USA
crumbaitisdelrio@rockfound.org

Fabrice DeClerck
Centro Agronómico Tropical de
Investigación y Enseñanza (CATIE),
Turrialba, Costa Rica
fdeclerck@catie.ac.cr

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Foreword

Humanity has entered the Anthropocene. If ever there was a time when we could take nature's beneficence for granted, it has passed. With seven billion people on the planet, and the eight-billionth arrival expected by 2025, human pressures on every ecosystem have multiplied, in some cases to the breaking point. The famine in the Horn of Africa reminds us that productive and resilient ecosystems are important not only for human well-being but also for human survival, especially in the dire circumstances of impoverished populations.

The urgent need to sustain ecosystems in the face of climate change, growing human populations, and rising demands for a multitude of primary commodities and agricultural outputs is giving rise to a burgeoning new discipline of sustainable development. More than ever, we need to understand how society depends on a range of complex and subtle ecosystem functions, and conversely, how ecosystem functions are impacted by human activities. The intellectual challenge is enormous. Both ecosystems and human systems are immensely complex. Their interactions add further dimensions of complexity. And understanding natural and human systems requires a range of analytical tools that surpass traditional academics' disciplinary boundaries.

The present volumes, *Integrating Ecology and Poverty Reduction*, are a powerful and innovative addition to this vital field of research. These volumes are also a personal thrill for me, since their genesis is the multidisciplinary setting of the Earth Institute at Columbia University. I am most grateful to our former Earth Institute postdocs who conceived and carried out these studies. They and the contributors to these volumes have earned our admiration and gratitude.

Every chapter in these volumes shows that the emerging scientific discipline of sustainable development is both vital and difficult. This is especially the case when it is viewed as an applied science that aims to find practical solutions in specific human-ecological contexts. It is one thing to recognize that ecosystem functions are vital to a society's health and economic productivity (as explored in the first volume, *Integrating Ecology and Poverty Reduction: Ecological Dimensions*), and quite another to devise institutions and policies that protect ecosystems in the face of climate change, growing populations, and rising economic pressures (as explored in the second volume, *Integrating Ecology and Poverty Reduction: The Application of Ecology in Development*). The case studies in these volumes describe as many failures as successes in the policy sphere and illuminate the subtle and multidimensional approaches to both science and policy that are necessary for success in managing complex and interacting systems.

Despite the range of geographies, ecologies, and development challenges covered in these volumes, there is a unified and highly successful intellectual approach. This is development seen through the ecologist's eyes and with the ecologist's tools. The overriding theme is how the science of ecology – with its focus on complex systems, interacting components and networks, threshold effects, and strong nonlinearities – can and should inform development thinking and design.

As one would expect, the detailed ecological context of development looms large. The details of ecological stress, resource ownership, community organization, gender relations, migration patterns, biodiversity, land use patterns, transport conditions, and vulnerability to environmental hazards and climate change, all condition the interactions of society and ecosystems, and all shape the ways to find sustainable approaches to economic development. It is a vast challenge to understand these complex relations. It is an even greater challenge to ensure that the impacted communities themselves can appreciate the ecological and social context in which they operate, so that they can devise effective means to solve pressing problems.

The chapters put a great deal of emphasis on how ecological knowledge is shared and diffused within a community. There is need for formal training and scientific knowledge, of species, climate, and ecological changes. There is need for a deep understanding of the key actors in the communities. There is an especially vital need for gender awareness and women's empowerment. Women are often disempowered in local communities, and yet play the vital role in managing croplands, water resources, fuelwood, and other ecosystem services. Without women's empowerment, sustainable solutions are impossible to identify, much less to achieve.

Population dynamics, including the challenges of the demographic transition to low fertility rates and the management of migration, loom large in the challenges. Both the issues of natural population increase caused by continued high fertility rates in low-income settings and the challenges of massive migration, from rural to urban areas and across national boundaries, are among the most vexing problems of sustainable development. Population growth is highest in the poorest and most fragile ecosystems, such as the drylands of the Horn of Africa. Migration from such regions can also trigger social conflicts and violence. Migration is leading to a dramatic surge of urbanization, beyond the planning and management capacity of many sprawling urban areas. The second volume has excellent discussions of these dimensions of demographic-ecological interactions.

Many of the chapters in the second half of the second volume deal with various strategies for monetizing the social value of ecosystem services. The basic idea is straightforward: since ecosystem services provide great value to society, there ought to be a way to create economic incentives to sustain those services, and more generally to benefit poor communities that manage the services. Yet the wonderful case studies and analyses make clear that this strategy is much easier said than done. There is no off-the-shelf strategy for creating appropriate incentives. Each situation, type of ecosystem service, and pattern of local culture and politics calls for a tailored design.

The cases are fascinating. We gain insight into community-based management of forests, fisheries, non-forest products, biodiversity conservation, ecotourism, and much more. We learn about a fascinating project to "pay for ecosystem services" (PES) in a wildlife reserve in Tanzania. Even though the community receives very modest compensation for its conservation activities, and for forgoing other economic activities around the site, the project has proved to be very popular with the community and has successfully combined conservation with development initiatives; in short, PES proved to be "a highly cost-effective model for community-based conservation" (p. 167). In other cases, however, with different ecological and social dynamics, PES proved to be less robust and less effective.

What is most exciting about these volumes is the consistently high quality of ecological analysis combined with an equally high quality of keen social observation. This collection of chapters is, in short, sustainable development analysis at its best, drawing strength by acknowledging the complexity of biological and social systems, avoiding oversimplification, and always giving due attention to the interactions of nature, culture, and economy. Readers will savor these chapters as bold and cutting-edge approaches to a budding scientific discipline of enormous practical importance. The field of sustainable development is enormously enriched by this pioneering effort.

Jeffrey D. Sachs
Professor, Director of the Earth Institute

Preface

The two volumes comprising the series *Integrating Ecology and Poverty Reduction* address the ecological dimensions of some of the major challenges of reducing poverty in developing countries (Vol. 1) and present potential solutions and opportunities for more effectively leveraging ecological science and tools to address some of those challenges (Vol. 2). Collectively, we hope these volumes serve to foster a deeper, more nuanced understanding of the ecological dimensions of various aspects of poverty, particularly in rural areas of developing countries where some of the world's poorest people live, and a heightened appreciation for the role that ecological science and tools can play in poverty reduction efforts. We acknowledge that no development challenge is uniquely ecological in its provenance or its resolution, but posit that ecological science and tools are critical components of effective solutions to some of the world's most vexing international problems.

The second volume of this series, *Integrating Ecology and Poverty Reduction: The Application of Ecology in Development*, builds upon the first volume, *Integrating Ecology and Poverty Reduction: Ecological Dimensions*, by exploring the way in which ecological science and tools can be applied to address major development challenges associated with rural poverty. In Vol. 2, we explore how ecological principles and practices can be integrated, conceptually and practically, into social, economic, and political norms and processes to reduce poverty and positively influence the environment upon which humans depend. Specifically, these chapters explore how ecological approaches and considerations can be useful for enhancing the positive impacts of education, gender relations, demographic shifts and dynamics, markets, and governance for poverty reduction. As one of the final chapters on the future and evolving role of ecological science points out, sustainable development must be built upon an ecological foundation if it is to be realized. The chapters in this volume illustrate how traditional paradigms and forces guiding development can be steered along more sustainable trajectories by utilizing ecology to inform project planning, policy development, market development, and decision-making.

Bronx, NY — Jane Carter Ingram
Turrialba, Costa Rica — Fabrice DeClerck
New York, NY — Cristina Rumbaitis del Rio

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Contents

Contributors

Susana B. Adamo Center for International Earth Science Information Network (CIESIN), The Earth Institute at Columbia University, New York, New York, USA

Sandra Baptista Center for International Earth Science Information Network (CIESIN), The Earth Institute at Columbia University, New York, New York, USA

Jason Bremner Population Reference Bureau, Washington, DC, USA

Natalia Estrada Carmona Livestock and Environmental Management Group, Division of Research and Development, Centro Agronómico Tropical de Investigación y Enseñanza (CATIE), Cartago, Turrialba, Costa Rica

David Carr Department of Geography, University of California at Santa Barbara, Santa Barbara, CA, USA

Esteve Corbera School of International Development, University of East Anglia, Norwich, England

Sara R. Curran Center for Studies in Demography and Ecology, University of Washington, Seattle, Washington, USA

Jason Davis Department of Geography, University of California at Santa Barbara, Santa Barbara, CA, USA

Fabrice DeClerck Centro Agronómico Tropical de Investigación y Enseñanza (CATIE), Cartago, Turrialba, Costa Rica

Cristina Rumbaitis del Rio The Rockefeller Foundation, New York, NY, USA

Alex de Sherbinin Center for International Earth Science Information Network (CIESIN), The Earth Institute at Columbia University, New York, New York, USA

Mary Emery Bioeconomy Institute, Iowa State University, Ames, Iowa, USA

Manuel Estrada Independent Climate Change Consultant, Mexico

Edith Fernandez-Baca Consortium for the Sustainable Development of the Andean Ecoregion (CONDESAN), Lima, Peru

Brendan Fisher Woodrow Wilson School of Public and International Affairs, Princeton University, Princeton, New Jersey, USA

Isabel Gutierrez-Montes Centro Agronómico Tropical de Investigación y Enseñanza (CATIE), Cartago, Turrialba, Costa Rica

Margaret Buck Holland Geography & Environmental Systems, University of Maryland, Baltimore, Maryland, USA

Jane Carter Ingram Wildlife Conservation Society, Bronx, New York, USA

Michael Jenkins Forest Trends, Washington, District of Columbia, USA

Peter Marcotullio Department of Geography, Hunter College, City University of New York, New York, USA

Caleb McClennen Wildlife Conservation Society, Bronx, New York, USA

Shahid Naeem Department of Ecology, Evolution, and Environmental Biology & Center for Environmental Research and Conservation, Columbia University, New York, New York, USA

Lisa Naughton-Treves Department of Geography, Land Tenure Center, Nelson Institute for Environmental Studies, University of Wisconsin-Madison, USA

Fred Nelson Maliasili Initiatives, Underhill, Vermont, USA

Hassan Sachedina Wildlife Works, Nairobi, Kenya

Robin R. Sears The School for Field Studies, Salem, Massachusetts, USA

Angela M. Steward The School for Field Studies, Salem, Massachusetts, USA

Chapter 1
Introduction to Integrating Ecology and Poverty Reduction

Fabrice DeClerck, Jane Carter Ingram, and Cristina Rumbaitis del Rio

Background

At the writing of this book, the world is at a critical crossroads. The year 2010 was the United Nations (U.N.) year of biodiversity and the year when the targets of the Convention of Biological Diversity (CBD), which was signed in 2002, were supposed to have been met. The CBD aimed to achieve by 2010 a "significant reduction of the current rate of biodiversity loss at the global, regional and national levels as a contribution to poverty reduction and to the benefit of all life on Earth." However, progress remains elusive – species extinction rates continue to be 1,000 times greater than background rates in the geological record (Secretariat of the CBD 2006; Walpole et al. 2009, 2010; Butchart et al. 2010).

We are also at a critical stock-taking point on progress towards meeting the Millennium Development Goals (MDGs), a set of time-bound goals for achieving measurable improvements in the lives of the world's poorest people by the year 2015 (www.un.org/millenniumgoals/). The MDGs were agreed upon by every member nation of the United Nations in 2000 as a global commitment to reducing extreme poverty. Progress towards the goals was recently reviewed in an MDG summit convened during the 2010 annual General Assembly meeting. The eight goals can be summarized as follows: (1) eradicate extreme economic poverty and hunger; (2) achieve universal primary education; (3) promote gender equality and empower women; (4) reduce child mortality; (5) improve maternal health; (6) combat HIV/AIDS, malaria, and other diseases; (7) ensure environmental sustainability; and (8) develop a global partnership for development.

Despite the historical separation between biodiversity conservation and poverty reduction efforts (Adams et al. 2004; Sanderson and Redford 2003, 2004; Redford

F. DeClerck (✉)
Centro Agronómico Tropical de Investigación y Enseñanza (CATIE),
Cartago, Turrialba, Costa Rica
e-mail: fdeclerck@catie.ac.cr

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et al. 2008), there is increasing consensus that the maintenance of biodiversity is an integral part of reducing extreme poverty reduction. Biodiversity conservation is a core focus of the MDGs, in particular, MDG 7 that focuses on environmental sustainability and includes the CBD goal of achieving a significant reduction in the rate of biodiversity loss. Progress towards MDG 7 is measured in terms of the proportion of land area covered by forest, a reduction of carbon dioxide (CO_2) emissions and of ozone-depleting substances, the proportion of fish stocks within safe biological limits, a reduction in the proportion of the total water resources used, an increase in the proportion of terrestrial and marine areas protected, and a reduction in the proportion of species threatened with extinction (Secretariat of the CBD 2006).

Despite widespread international commitment to all of these goals, including MDG 7, integrating environmental sustainability, and biodiversity conservation specifically, into development projects and national development strategies remains a challenge. In 2004, Adams et al. wrote that biodiversity conservation scientists face a dilemma as a result of the increasing global concern that international conservation efforts are in conflict with efforts to reduce poverty and that lasting positive outcomes of conservation-with-development projects are elusive. Indeed, many perceive biodiversity conservation and poverty reduction to be two completely disparate goals. Adams et al. (2004) addressed these perceived conflicts and proposed a typology for clarifying the different relationships between conservation and poverty reduction: (1) poverty and conservation are separate policy realms, (2) poverty is a critical constraint on conservation, (3) conservation should not compromise poverty reduction, and (4) poverty reduction depends on living resource conservation. Much of this discussion, however, has focused on the impact that protected areas and reserves have on poverty reduction – which in many cases will be minimal. For example, Redford et al. (2008) demonstrate that only about 0.25% of the world's poorest people are found in areas that are somewhat or extremely wild.

These two volumes focus predominately on the fourth typology proposed by Adams et al. (2004), that poverty reduction depends on living resource conservation. However, there are several important clarifications to be made. First, the chapters included in this volume push beyond the notion that poverty reduction is disproportionately dependent on living species simply for production services obtained from nature, but that integrating ecological concepts into development strategies can be a useful approach for achieving multiple MDGs and improving livelihoods (Rumbaitis del Rio et al. 2005; DeClerck et al. 2006). Second, we distinguish between integrating ecological tools into development practice, and the conservation of critically endangered biodiversity. That is, many of the interventions and tools highlighted in this volume address conserving ecological integrity in human-dominated landscapes with the specific aim of sustaining and restoring ecosystem services that contribute to human well-being. Multiple studies have demonstrated that practices that target biodiversity conservation in human-dominated landscapes can make significant contributions to biodiversity conservation (see Gardner et al. 2010), and to ecosystem services (Naeem et al. 2009), but that these interventions often fail to protect sensitive species (Milder et al. 2010). Thus, ecological science will continue to be important for informing conservation planning aimed at protecting threatened biodiversity, but will also be critical for successfully achieving the MDGs in human-dominated landscapes that may not be high priorities for biodiversity conservation, but where poverty is high and persistent (Kareiva and Mavier 2007). Finally, we acknowledge that poverty is a multi-dimensional condition resulting from a lack of access to material and non-material needs (Anand and Sen 2000; Alkire and Santos 2010). However, in these volumes, we have focused and expanded upon the multiple components of poverty represented by the MDG framework (Sachs 2005; UN 2010), while recognizing that some aspects of poverty may not be addressed explicitly in these volumes. Rather, these volumes can be viewed as a starting point for illustrating how ecology underpins certain components of poverty (Volume 1) and considering how several types of mediating social forces can be leveraged to increase the benefits that ecosystems provide to the poor (Volume 2).

Certainly, conservation and poverty reduction "win-win" situations are by no means commonplace nor easy to achieve, as they may require compromise with respect to one or both goals. For example, in a global meta-analysis using 11 case studies from Latin America, Africa, and Asia, Tekelenburg et al. (2009) investigated how biodiversity and poverty are related to each other by exploring the ways in which indicators of conservation and development changed over a 10-year period. In all but one example, gains in biodiversity were uncorrelated with poverty reduction. The single example of gains in both was found within the Chorotega Biological Corridor in the Guanacaste peninsula of Costa Rica. The Chorotega Biological Corridor is part of the greater Mesoamerican Biological Corridor (MBC), which aims to facilitate the movement of biodiversity from southern Mexico to northern Colombia. Although at its conception, the MBC consisted entirely of conservation goals (biological connectivity), recent analysis of the most functional corridors indicate that these goals have been supplemented with more development-focused goals such as ensuring water quantity and quality (Estrada and DeClerck 2010; see also Chap. 14 in this volume on PES). Although many factors have led to positive results for conservation and livelihoods in the Chorotega Biological Corridor, part of the success can be attributed to the integration of local needs (water) with conservation goals.

Achieving conservation and poverty reduction goals, as exhibited by the Chorotega example, will require cross-disciplinary approaches, which have been growing (NAS 2005; Ostrom et al. 2007; Ostrom 2009). Thus, it is now timely to ask what is and should be the role of ecology in efforts to alleviate poverty? Why should ecological understanding of the way in which biological communities work be relevant to solving complex development problems? How can ecological knowledge be integrated into cross-disciplinary approaches to support development planning? These questions are the central starting points for these volumes. While the importance of ecosystem services for human well-being is now widely accepted, the challenge remains as to how we can practically maintain biodiversity and ecosystem function alongside poverty reduction initiatives? This is the key challenge this volume seeks to explore across a range of development goals and through the lens of several potential solutions that may provide a way to achieve both conservation and poverty reduction. Specifically, this volume explores what the role of ecologists and the science of ecology is in addressing these challenges and contributing to potential solutions.

The Science of Ecology

Ecology is the science of studying the interactions of organisms and their environment. During the relatively short history of ecology as a field of study, this has focused on understanding how populations of species are shaped and influenced by the environment (e.g., temperature, humidity, latitude, and elevation) and by interactions with other species (e.g., predation, competition for resources, and cooperation). Much of the early work of ecologists has specifically and intentionally focused on areas characterized by low human impact – relatively intact wilderness or protected areas, or laboratory microcosms – with the explicit goal of understanding how ecological communities are formed and operate in the absence of human influences. Much of this early ecological research documented the effects of human perturbations on ecosystems as an external forcing, but has not looked at humans as an important component in the system. In large part, traditional ecology has sought to minimize human influence and even to exclude the human footprint in our understanding of how the biosphere works, rather than disentangling the complex relationships between humans, other species, and the physical environment. For example, Real and Brown's (1991) edited volume "Foundations of Ecology" includes 40 classic ecological papers that form the theoretical foundation for most students of ecology. However, not a single one of these papers includes humans as a critical ecological player. In fact, much of the research about the interactions between humans and the ecosystems in which they live, also referred to as social-ecological systems, has occurred within disciplines such as geography and the burgeoning field of sustainability science (Kauffman 2009) and has been promoted within programs such as the International Human Dimensions Program (IHDP, www.ihdp.org). Only recently have ecologists shifted their focus to consider not only how humans impact the environment, but also how functional ecosystems contribute to human well-being (Daily 1997; Rumbaitis et al. 2005; DeClerck et al. 2006; Kareiva and Marvier 2007; Naeem et al. 2009).

An important first question is what are the contributions of ecology and its subdisciplines, beyond conservation implications? As previously stated, ecology is the science of studying organisms in their environment and of understanding the relationships between communities of organisms. This includes a multitude of branches such as population ecology that specializes in how organisms of the *same* species interact with one another to acquire resources and reproduce. In contrast, community ecology studies the interactions *among* species, which includes multiple classes of interactions such as predation and competition, but also facilitation and cooperation. Landscape ecology, one of the youngest branches of ecology, considers how spatial context or position in a landscape affects ecological interactions.

Early ecologists focused primarily on the impacts of the environment on the distribution of organisms and ecological communities through observations of how these communities changed from the poles to the tropics, or at smaller scales, from valley bottoms to mountaintops, rather than how organisms and communities influence the functioning of the environment in which they exist. Today, ecologists increasingly recognize that species are not just passive recipients of the environment, but that they play a very active role in shaping and driving ecological processes (Naeem 2002). This functional view of biodiversity is a significant paradigm shift that pushes the work of ecologists into the cross-disciplinary realm where biodiversity and ecosystems are understood to be essential contributors to human well-being through the provisioning of essential goods and services (Naeem et al. 2009). The chapters comprising these volumes reflect on that role with a particular focus on how ecological knowledge, tools, and understanding can contribute to improving the living conditions of the world's poorest people.

A Functional Role for Ecology in Poverty Reduction

The important distinction between the MDGs and other development initiatives is the renewed focus on cross-disciplinary (Fig. 1.1; Eigenbrode et al. 2007), and multi-scalar approaches. Past development interventions have been criticized for their shotgun approach. In many cases, there has been little to no interaction among different disciplines, or when there was, there have been negative impacts, where the advances made by one discipline negated the efforts made by another. Multiple development projects have resulted in unintended consequences, where well-meaning interventions have not considered the indirect, systemic effects of actions (Ranganathan et al. 2008).

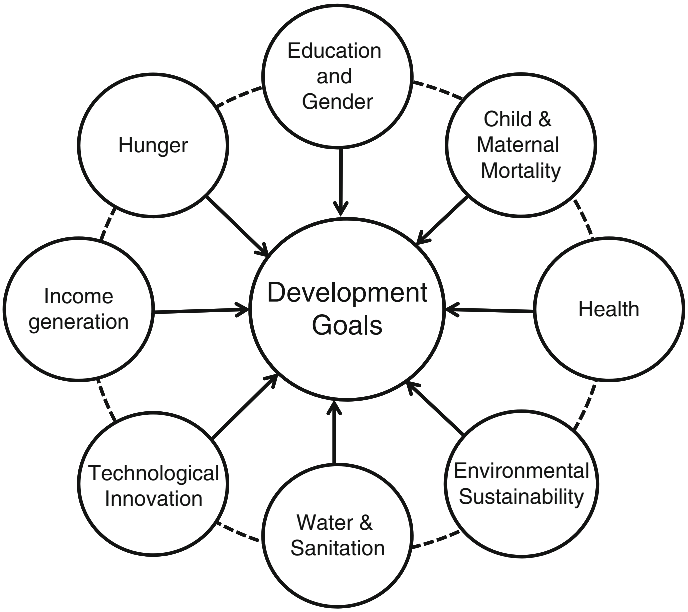

Fig. 1.1 The Millennium Development Goals were agreed upon by all member countries of the UN in 2000 and aim to significantly reduce poverty by 2015. This schematic illustrates development challenges the Millennium Development Goals aim to address

One example is agriculture's Green Revolution, which undoubtedly saved millions of lives by increasing the agricultural productivity of the world's most important grain crops, but with a large associated environmental cost (Tilman 1998). The question remains, whether the negative environmental impacts of the Green Revolution might have been reduced had ecosystem science been more developed as a discipline at the time, and had there been greater dialogue between ecologists and agronomists on the imperative to sustainably meet global food production needs without compromising the ecosystem services important for meeting other basic needs?

This is much more difficult than it might appear and although development goals, including the MDGs, may be multidisciplinary and combine several usually separate branches of learning; they are far from being truly interdisciplinary by fostering increased interaction and integration of contributing disciplines. The primary difference between the two, according to Eigenbrode et al. (2005), is that multidisciplinary research is conducted by scientists from different disciplines, but is designed to address a question pertaining to a single system. In contrast, interdisciplinary research requires a greater degree of coordination among disciplines from the start with research questions that often span several temporal and spatial scales and fields of study. When considering the MDGs as presented in Fig. 1.1, it is easy for a professional in a specific discipline to focus on the goal most relevant to his or her work. This approach, however, limits the opportunity for finding novel solutions and avoiding conflicts (NAS 2005).

We propose, however, that rather than identifying with individual goals, professionals consider each goal through the lens of their respective discipline (Fig. 1.2). For example, ecologists could consider their role not only in ensuring environmental sustainability, but also in reducing hunger, improving maternal health, or achieving universal primary education. Certainly, ecological expertise, knowledge, and methods, which we term the "ecological toolbox" (Rumbaitis et al. 2005), will have limited application in achieving some development goals, and greater application in others, but we may be surprised by the solutions that arise simply by looking at a problem in a new light. Such an exercise serves not only to identify how ecologists can contribute to areas outside of their typical remit, and to highlight the interaction between the fields, but also serves to highlight areas of potential conflict between fields where cross-disciplinary discussion and considerable negotiations will be needed to identify tradeoffs and/or negative impacts before they occur.

Of course, we do not suggest that ecology or any single approach is a panacea capable of solving all of the world's most pressing problems, or even a single problem alone (Ostrom et al. 2007; Ostrom 2009). However, we do strongly believe that ecology can make significant contributions to most of the MDGs, and that the integration of the ecological perspective with that of other disciplines will present solutions that are novel, sustainable, and may result in fewer trade-offs in the long-term than quick-fix solutions that deliver immediate returns on a single development goal. Examples of this integrative thinking are becoming more popular. For instance, the increasing collaboration of ecologists with agronomists in the field of agroecology focuses largely on how ecological interactions can be used to reduce the need for

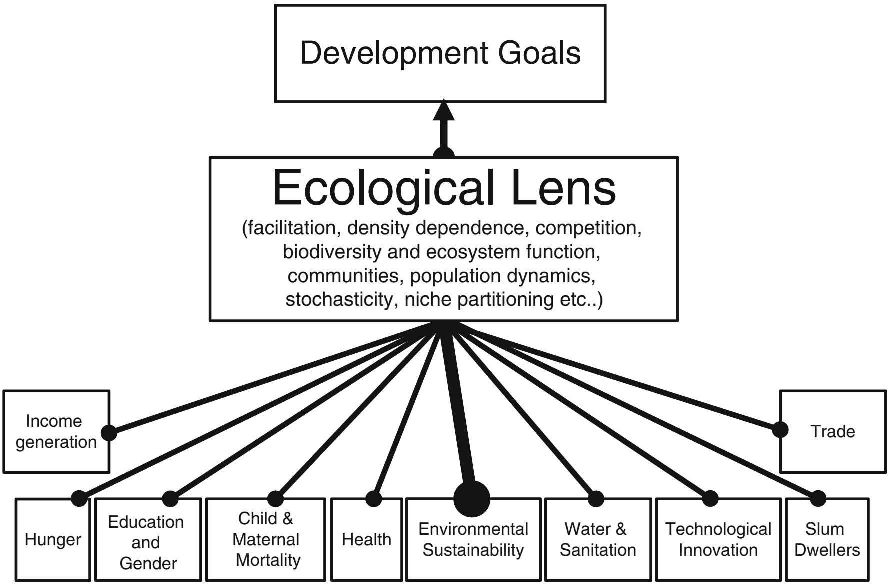

Fig. 1.2 The role of ecology in achieving poverty reduction should not be restricted to development goals that are explicitly environmental. Rather, ecology offers useful concepts and tools for achieving progress towards other development goals, as discussed throughout these volumes and illustrated in this figure. For some development goals, the role of ecology will be more direct and significant than for others. Nevertheless, considering a problem through the lens of multiple disciplines, as encouraged throughout these volumes and as demonstrated herein with the field of ecology, may lead to new, innovative solutions for addressing poverty

agrochemicals while maintaining competitive yields (Smukler et al. Chap. 3, Vol. 1). Many ecologists also work directly alongside engineers and farmers to design riparian (riverside) forests whose functional role is to improve water quality before it enters rivers and streams, reducing the cost of water treatment for downstream communities. Interaction of ecologists with nutritionists and medical professionals has shed new light on how species composition, interactions, and distributions can be manipulated to decrease malnutrition (Chap. 4, Vol. 1) and risk of infectious diseases (Chaps. 13 and 14, Vol. 1). The purpose of these volumes is to focus specifically on these issues in relation to major development challenges and how knowledge of interactions and trade-offs can be integrated into solutions.

Organization of These Volumes

To prepare the two volumes comprising the series *Integrating Ecology and Poverty Reduction*, we have asked authors to address a major development challenge or solution and to assess if/how an ecological approach is relevant within that context and the advantages and/or limitations of using the ecological toolbox. This task was more straightforward for some development goals and solutions than others. Nevertheless, all of the chapters have highlighted the utility of ecological science for addressing development problems and solutions through the direct application of ecological theory and tools, as well as the more indirect application of ecological thinking, which emphasizes the importance of spatial and temporal scales, feedbacks, and trade-offs. We recognize that entire books can be written on each of the topics presented herein and, thus, we do not attempt to cover all possible applications of ecology with respect to development challenges, a task that is beyond the scope of this project. Rather, these two volumes seek to highlight how major development challenges can be viewed through an ecological lens and addressed through the use and applications of the ecological toolbox. We do not propose that ecology alone will be able to answer many of these critical questions; rather, we suggest that ecological science combined with the tools of other disciplines can make a greater contribution to developing a sustainable future and reducing the tremendous poverty that persists in our world.

The series is divided into two volumes. The first volume, *Integrating Ecology and Poverty Reduction: Ecological Dimensions*, focuses on the ecological dimensions to global development challenges. The chapters in this volume deal with the biophysical aspects of ecology and demonstrate two primary points. First, that understanding the ecological foundations of human-dominated landscapes can provide a better understanding of how we are impacted by ecological processes. The American conservationist Aldo Leopold once famously stated that "to keep every cog and wheel is the first precaution of intelligent tinkering." We would add to that by stating that applying the right tool for the job should be the second rule of intelligent tinkering. In the chapters included in this section, we explore the direct application of ecological tools to achieving distinct development goals of reducing hunger, improving human health and nutrition, decreasing vulnerability to extreme events, and increasing access to clean water and energy. These chapters present specific examples of the application of ecological principles in poverty reduction – or examples of how ecological tools fit and function in a development toolbox.

The second volume, *Integrating Ecology and Poverty Reduction: The Application of Ecology in Development Solutions*, focuses on mediating forces and solutions for poverty reduction and addresses the relevance and role of ecology in relation to these. We recognize that the mediating forces and the solutions that we have addressed – Education, Gender, Demography, Innovative Financing, and Ecosystem Governance – represent far from an exhaustive list of topics that we could have covered in this volume. Nevertheless, these chapters collectively address many ways in which humans interact with each other and the ecosystems in which they live and how these interactions inform how ecological science and tools can be applied to positively influence forces shaping human societies and the creation of solutions that conserve biodiversity and ecological processes alongside poverty reduction. For example, demographic trends in population growth, urbanization, and migration influence the nature of human interactions with the environment (see Chapters on Population). Similarly, gender dynamics influence how the sexes perceive and interact with the environment and how natural resource access and management decisions are made (Gutierrez et al. Chap. 4, this volume). An understanding of the dynamics underlying these forces must be factored into developing successful poverty reduction measures in communities that rely directly on natural resources for their livelihoods. In the section on Innovative Financing, the authors of various chapters demonstrate that implementation of mechanisms, such as Payments for Ecosystem Services, requires an application of sound ecological science and tools, in addition to an understanding of the social, economic, and governance constraints and opportunities where such programs may be developed, if they are to be effective. In the section on Ecosystem Governance, the authors emphasize the importance of strong ecological science, tools, and targets for governing and managing a land or seascape for multiple, often conflicting purposes. These chapters demonstrate that reducing poverty will require understanding the interplay of ecological, social, economic, and political systems and illustrate that ecologically sound solutions will require major shifts in conventional thinking, which society may or may not be willing to make. In a final concluding piece, Naeem critically addresses the overarching role of ecology in sustainable development, and states what this role is currently and proposes what it should be, if we are to have truly, sustainable development.

Conclusions

Traditionally, the science of ecology has not been an integral component of many aspects of international development for a variety of reasons. Increasingly, however, there has been a renewed interest in finding more sustainable means of development, grounded in ecological knowledge. Yet, a range of concepts and approaches that are becoming more widely used across a range of sectors, such as ecosystem services, resilience, and social-ecological systems thinking, are signs of a paradigm shift. Our goal with these volumes is to build upon this recent momentum at this important moment in time to increase the dialogue between ecologists and development practitioners. We have produced these volumes for both audiences in the hope that ecologists who read them will see the contribution that the field can make to poverty reduction and that development practitioners will gain an understanding of the contribution that ecology as a discipline can make to sustainable development. Our ultimate intention is that these volumes will facilitate increased dialogue among multiple disciplines, including ecology, and that this dialogue will result in a more effective use of ecological science and tools to improve the livelihoods of the world's poorest people alongside the conservation of functioning ecosystems.

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Chapter 2
Introduction: Gender, Education and Ecology

Fabrice DeClerck and Jane Carter Ingram

Throughout this volume, as we seek to think about new or enhanced ways in which ecology can be applied to address poverty, it is critical to consider the social, cultural, and economic traditions that may support or challenge the adoption of an ecologically based approach to development. Two key, interconnected areas in which societal norms are critical to furthering poverty reduction and sustainable natural resource management in developing countries include education and gender. Education is widely recognized as an important component in reducing poverty and a key to wealth creation (UNESCO 2003). While the rural poor in general lack access to formal education, women and girls have significantly fewer opportunities to access education than men and boys (UNESCO 2003). Education is not the only sphere in which gender inequalities exist. Women perform 66% of the world's work, produce 50% of the food, but earn 10% of the income and own 1% of the property (UNICEF 2007). Yet, improving the lives of women and girls can be an effective way to prevent disease, reduce hunger, and raise Gross Domestic Product (Kristof and WuDunn 2009). Increasing appreciation of the importance of education and gender equality in development initiatives is reflected in Millennium Development Goal (MDG) 2, which focuses on education, and MDG 3, which focuses on gender equality.

In this section, authors have considered how ecological science and tools might be related to the challenges of education and gender equality and how ecology can be better integrated into ongoing initiatives to address gender and education challenges. Both chapters in this section begin with the premise that the rural poor are heavily and directly dependent on functioning ecosystems for their well-being. A majority of these people lack formal education, yet, there is a wealth of local ecological information held by rural communities, and much of this information is gender specific. Understanding how rural communities perceive, understand, and

F. DeClerck (✉)
Centro Agronómico Tropical de Investigación y Enseñanza (CATIE),
Cartago, Turrialba, Costa Rica
e-mail: fdeclerk@catie.ac.cr

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interact with each other and the ecosystems in which they reside is critical to developing strategies for sustainable development. Towards this end, both chapters emphasize the importance of local context and "systems thinking" for developing sustainable development approaches.

For example, agro-ecologists have long been interested in understanding farmer perspectives regarding increasing tree densities and coffee- and pasture-based agroforestry systems of Mesoamerica. Interviews with rural farmers have demonstrated that, even without formal education or training, they may have a sophisticated understanding of the ecological traits for different species, understand the spatial dynamics of pests and diseases (particularly when their neighbors follow unsanitary crop management practices), and also integrate concepts of resource limitation into their management practices. During interviews with coffee farmers of Costa Rica, one farmer favored using laurel (*Cordia alliodora*) as a shade tree both because of its timber value and because of its synchronous flowering with coffee, thereby attracting pollinators. This farmer operates under the assumption that pollinators are scarce in coffee farms due to pollen limitation. Another farmer of the same region, however, stated that he does not include laurel in his coffee farms for exactly the opposite reason stating that laurel flowers at the same time as coffee, thereby reducing the number of flower visits because of pollinator scarcity. Thus, it is clear that both farmers managed pollination as an ecosystem service and both implemented farming practices based on the notion that managing flower resources could influence productivity.

Much ecological information is also gender based and complementary. In a project studying local knowledge of trees in silvopastoral systems of Nicaragua, interviews that include both the male and female heads of households yield more information than interviews with only one head of household. For example, women tended to possess in-depth information on the medicinal values of tree species, including veterinary uses, whereas, men tend to focus more on the production aspects of different species.

In the chapter on Education, Ecology, and Poverty Reduction, Sears and Steward point out that scientific ecological knowledge and local ecological knowledge share similar traits. Alone, neither is complete. Local ecological knowledge is essential from several points of view. First, as highlighted above, it is derived from the local environment, and therefore is highly context specific. Second, because it is embedded in community practices, it responds to the local needs of the population. Countless examples exist of development interventions that have failed after not taking into consideration this perspective. Finally, because it is multi-generational, local knowledge includes critical information on how societies have dealt with disturbances and other challenges in the past—information that may be critical for developing adaptation strategies for future perturbations. Local knowledge may not be sufficient alone to deal with exogenous, new challenges such as climate change and opportunities such as payments for ecosystem services—these may require additional capacity building. Training programs that are grounded in the local social and ecological context are more effective at equipping communities with the skills needed to adapt to changing environmental conditions and to fully engage in emerging environmental markets than programs that do not incorporate aspects of local knowledge. However, as Sears and Steward discuss, retaining smart, highly skilled professionals in rural areas to work on these issues remains a challenge.

In the chapter on gender, Gutierrez-Montes et al. outline the importance of women in natural resource management and the different ways that men and women perceive and use natural resources, and propose ways in which women can be more fully integrated into natural resource management. The authors state that poverty reduction will require understanding the linkages among the social construction of gender, the context of local access and decision making over different natural assets, and the impacts of environmental change on those assets. Such information can be used to ensure that women more fully participate in decision making and project planning.

Sears and Steward, and Gutierrez et al. provide an illustration of several social barriers to effective ecological management as related to education and power. These chapters show that developing an understanding of the local social and ecological context and empowering people who hold valuable local knowledge to have a voice in natural resource management are important parts of implementing poverty reduction strategies that are grounded in the sustainable use of ecosystem services.

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Chapter 3
Education, Ecology and Poverty Reduction

Robin R. Sears and Angela M. Steward

Introduction

Many of the world's poorest people live in rural areas. They depend on environmental resources for their health and well-being. The rural poor depend disproportionately on trees, freshwater streams, pollinators, mangroves, and rainfall (UN Millennium Project 2005). Many rural people have grown up in those rural areas; they come from generations of rural producers—herders, farmers, hunters, and gathers—who collectively possess an unwritten library of local knowledge, intelligence, skills, and technologies about how to survive and thrive on what local ecosystems can provide.

Rural residents have adapted their production and extraction systems over time to new information, new technologies, and changing conditions. Those changes have been drastic in the past 100 years and have included changes to ecosystems, hydrological systems, and climate conditions, so much so that many rural people today live in environments so different from their grandparents' and parents' time that the local knowledge that was passed and had evolved over generations is no longer adequate. Most rural people today live with far less biodiversity and many with greater population densities, both leading to far more competition for resources and potential social conflict over resources. Many rural people live with less water than their grandparents, a changing and more unpredictable climate, and greater incidence of pests and disease outbreaks (see the section on Health Vol. 1 for chapters on this). Similarly, many of the rural poor have been displaced and find themselves far from the lands they know.

Changes in the social, political, and economic conditions also influence the rural landscape and rural people's livelihoods and well-being. Changes in land use and natural resource policies may restrict access to certain species, such as fisheries species,

R.R. Sears (✉)
The School for Field Studies, 10 Federal St, Salem, MA 01970, USA
e-mail: rsears@fieldstudies.org

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or prohibit certain practices, such as hunting. Alternatively, they may encourage specific production techniques or production of commodities, such as soybean or oil palm monocultures, bolstered by rural subsidy or credit programs. For example, the Green Revolution in Southeast Asia changed the landscape from small, diversified production systems to plantation monocultures (Ali 2004). The current energy crisis is changing landscapes around the world as new acreage is dedicated to biofuel production—African oil palm, sugar cane, and corn, which are replacing natural forest, grasslands, or local production systems.

Adaptation to *gradual* changes in environmental, economic, political, or social systems often can be generated from existing resources and institutions. However, adapting to *sudden* changes, new opportunities, or changes in ecosystems that pass a local knowledge threshold may require new educational and training opportunities from external actors. A village of hunters will need guidance and training when their livelihood is ended by new conservation laws prohibiting bush-meat trade. Farmers in sub-Saharan Africa accustomed to rain-fed agriculture in regions subject to decreasing rainfall and more frequent, longer droughts will need assistance adapting their production systems to new climate conditions, or getting out of farming altogether. Andean migrants who grew up farming at 1,500-m elevation may need training in lowland farming techniques when they flee from violent conflict in the coca-growing regions to the Amazon basin in hopes for a more secure life.

In this chapter, we explore how education based on an improved understanding of the ecosystems in which people live and enhanced communication among stakeholders can assist the poor in negotiating environmental changes and the emergence of new economic opportunities while contributing to poverty reduction goals. Specifically, there are three reasons for improving ecological knowledge in development sectors today. First, natural environments are changing rapidly due to degradation from either overuse or pollution. Changes to local and regional climates are affecting land use options; population pressure on existing limited resources such as freshwater is driving disease and conflict; and hazardous waste in air, water, and foods, coupled with poor sanitation also drives widespread and serious health problems. Adaptation to changing environmental conditions and mitigation of pollution requires an understanding of the ecological dynamics of the systems. Second, the emergence of environmental markets is creating new economic incentives for ecosystem management and conservation and new potential opportunities for improving rural livelihoods. Understanding the ecological basis for both the management and measurement of those marketable ecosystem services is critical to ensure the participation of rural people who are the stewards of many of these resources. Third, a shift from single sector development and extension—such as focusing just on agriculture or energy—to integrated sustainable development, including economic and ecological sustainability and social justice, requires systems thinking and understanding of both environmental and social complexity. Ecological literacy, we believe, can help not only the rural poor but also development extension agents to navigate these changes.

This discussion focuses on three populations who are involved in issues related to environment and rural development. These are the development and extension agents, who deliver educational and capacity building programs; rural people, who often possess a great deal of local ecological knowledge, and to whom the agents are reaching out; and scientists and conservationists who generate scientific ecological knowledge used by both extension agents and rural people. Our thesis is that building ecological literacy among all three actors can help to reduce the semantic barriers between these populations and empower rural people to participate more equitably in economic opportunities for rural development and conservation. Furthermore, an examination of the purposes and approaches to ecological education can yield recommendations for empowering rural populations to fully and effectively participate in conservation and development opportunities in the face of environmental, social, economic, and political changes.

For the purposes of this chapter, we limit the scope of education to the transfer of knowledge about ecology, resource use and environmental management, as opposed to the broader field of environmental education that also examines the socio-economic problems and solutions underlying environmental conservation and management. In this case, we focus on educational activities aimed specifically at improving ecological literacy. We consider the role of ecological knowledge as a tool for empowerment, self-governance, and access for rural resource users.

In this chapter, we draw the links between education and development, environment and rural livelihoods, and environmental education for rural development; discuss the types of environmental knowledge and actors who use it; identify the opportunities for improving rural livelihoods through payments for environmental services and the opportunities for ecological education in this context; and present a case study of community-based education focused on rural livelihoods and production to illustrate these concepts and approaches.

Development, Environment and Education

Development and Education

Because many rural people depend directly on environmental goods and services for subsistence and economic well-being, a critical element of sustainable development in rural areas is environmental sustainability. The status of the environment, and the health and well-being of rural people in developing countries is influenced by many factors, among them the norms, policies and institutions related to trade and governance. The integration of the rural poor into systems of trade and governance depends on the poor's access to, and use of information, which in turn depends on their access to infrastructure and services, such as communication systems and educational opportunities. Those factors, in turn, influence the manner in which individuals engage in productive and extractive activities that may affect the environment.

In the development arena, education has long been heralded as a pre-requisite for macro-economic development and the common discourse about the role of education in achieving development goals is about giving poor people the tools to engage more significantly in the market economy. Literacy, especially in reading and mathematics, is purported to help people participate more meaningfully in social and economic institutions. A passage from a document entitled *Reshaping Education for Sustainable Development* illustrates the expectation and emphasis of the link between education and economic development:

> The function of education in sustainable development is mainly to develop human capital and encourage technical progress, as well as fostering the cultural conditions favoring social and economic change. This is the key to creative and effective utilization of human potential and all forms of capital, ensuring rapid and more equitable economic growth while diminishing environmental impacts. Empirical evidence demonstrates that general education is positively correlated with productivity and technical progress, because it enables companies to obtain and evaluate information on new technologies and economic opportunities.
>
> (Albala-Bertrand 1992, p. 3 cited in Sauvé 2005)

Related to this discourse on education and development is the idea that literacy and subsequent household economic development through better jobs can also alleviate the poor's—at least the rural poor—dependence on the natural environment. This can lead to both development and conservation gains: it could free those who depend on ecosystem services from vulnerability to climate and biotic fluctuations, and it could alleviate pressure on natural ecosystems, both of which are dependent on decoupling rural livelihood and well-being from environmental resources.

At the same time, development paradigms increasingly embrace the notion that rural poverty can be alleviated somewhat not by decoupling rural livelihood from environmental resources, but by engaging rural dwellers in emerging markets and payment schemes for ecosystem services and new environmental products they can steward (Wunder 2005). These new markets for ecosystem services include watershed protection, carbon sequestration for climate change mitigation, and biodiversity conservation (See Chapters on Payments for Ecosystem Services, this volume). Payment schemes for these important ecosystem services are lauded by economists, development agents, and natural scientists alike as mechanisms for providing income to rural residents while conserving biodiversity (although, see chapters by Estrada and Corbera, and Fisher, this volume).

Similarly, markets for environmentally and socially "friendly" agricultural and forest products are stimulating a return to production systems that are ecologically complex and that have greater conservation value than monocultures. Demand by wealthy consumers for shade-grown coffee, organic bananas, sustainably managed timber, and fair-trade spices has spurred a return to bio-dynamic ways of farming, diversified agro-ecosystems, and sustainable forest management.

Sound ecological management is essential for the provision of multiple ecosystem services and environmentally friendly products. Management may be based on scientific and technical knowledge and applied through extension services and development projects; it may be driven by local knowledge and practice; or it may be a combination of both. While rural producers may have managed diversified systems using ecological techniques in the past, one generation of monoculture farming may significantly deplete the collective library of local knowledge on traditional agricultural practices.

Equally critical for environmental markets to function are sound and transparent techniques for measuring and accounting of the sustainability and output of services from productive ecosystems. The quantification, validation, and measurement of ecosystem services, such as pollinator species diversity near agricultural fields, water quality in a managed watershed, tons of carbon stored in a forest, or output of timber from a sustainably managed forest require scientific understanding of ecosystems, how they function, and the components that comprise them.

We propose that while the measurement of ecosystem services largely falls within the domain of ecosystem science, much of the management of ecosystem services may fall within the domain of rural producers' local knowledge. We suggest that to develop sound policies, markets and payment schemes for environmental services, and the equitable engagement of rural landholders, we must first break down semantic barriers between ecosystem science and ecosystem management, and between scientific ecological knowledge and local ecological knowledge. By understanding the capacity for rural ecosystems to provide ecosystem services, scientists can develop practical ways to measure those services. By understanding the scientific basis for measuring ecosystem services, rural landholders can make informed management decisions about engaging in payment for ecosystem service schemes and markets.

Ecological education is essential for reducing the semantic barriers between scientists and rural producers, and reducing such semantic barriers can empower the rural poor to participate more equitably in ecosystem service market opportunities for development and conservation. However, a first step in doing this is understanding the nexus between ecological education and rural development and distinguishing between environmental education and ecological education where ecological education is more specific and focuses on species-environment and species-species interactions, including how changing the species composition of an ecological community impacts these interactions and the provisioning of ecosystem services. Environmental education in contrast broadly examines how natural environments function, the relationships between humans and nature, and the interactions between social and natural systems (Sauvé 2005). Ecological education is often a primary component of environmental education. For the sake of simplification, we refer to environmental education throughout the remainder of this document as encompassing ecological education.

Environmental Education and Development

In developing countries, environmental education emphasizes identification of both environmental problems, socio-economic concerns that are linked to the environment, and practical solutions that are relevant to sustainable development in rural areas (Bekalo and Bangay 2002). Of relevance here are the two related goals of environmental education as a way of promoting biodiversity conservation and natural resource management for rural poverty reduction.

Education and training of rural people in developing countries about management of environmental resources, such as water, soil and forests, has been a focus in rural development initiatives for decades. More recently, environmental education has become a component of conservation initiatives, development programs, and integrated conservation and development programs (ICDP), the latter signaling the links between environment and development discourses and goals. The objective of environmental education is to influence the behavior of individuals, through increasing their knowledge of how their actions, and those of others, affect biodiversity, which, in turn, affects those who depend on that biodiversity for their livelihood and well-being. The goal is that participants in the educational activity will adopt the values and practices espoused in the curriculum and change their behavior. Much research in environmental education is about the uptake of the values and practices taught in the educational activity. The same is true for rural extension services, a form of education focused primarily on practical aspects of environmental management and agricultural production.

From a pedagogical perspective, environmental education can be characterized as positivist, or rational and instrumental because it is used to solve problems; interpretive, because it provides learners with a venue for understanding aspects of the environment; and socially critical because it promotes “the analysis of the social dynamics underpinning environmental realities and problems” (Sauvé 2005). In developing countries, a comprehensive educational program presenting all three of these perspectives would be useful, since rural poverty and environmental degradation are situated in broader social, political, and economic contexts. In the past several decades, however, environmental education has taken on, as Sauvé and Berryman (2005) suggest, purely instrumental roles, such as to implement “a globalized mix of (highly questionable) developmental and environmental agendas” (p. 230), particularly with resource-dependent populations. Their critique stems partly from the fact that the stimulus for and curriculum of environmental education programs often comes from places and institutions situated far from the site of delivery or object of study and the realities and conditions of those sites. This could be viewed as a neo-colonialist approach, imposing outside conservation agendas on developing countries and disempowered people.

In the 1980s and 1990s, conservation efforts shifted towards strategies that couple conservation with economic development, or at least with livelihood security. Today, just about all conservation initiatives take into consideration rural livelihood improvement. Projects now strive for conservation gains in one area, such as designating no-use zones or prohibiting the use of certain species, while improving livelihoods in another area. Livelihood gains may be manifested through sustainably increasing production of a marketable crop or through profiting from off-farm employment, such as work in an ecotourism enterprise. Those initiatives are not always successful, because of underlying institutional, geographical, or cultural challenges, but the underlying principle rooted in sustainable development is important.

Ecological knowledge is important for achieving both objectives, biodiversity conservation and improving rural livelihoods. The prevailing notion of technical environmental education is that a scientific understanding of ecosystems and natural resources, and technical training about natural resource and environmental management, will allow managers to control environmental factors using inputs and engineering, such as fertilizers and irrigation channels. The desired result is to increase production and reduce risk due to factors related to environmental uncertainty. However, local ecological knowledge is also critical to the success of conservation and development initiatives. Ecological education using both local and scientific ecological knowledge can help rural people manage natural resources more effectively and to enhance benefits from ecosystem functions and services.

Ecological Knowledge and Those Who Use It

We discuss three types of actors who deal with ecological knowledge in this section. Rural people who depend on natural resources, "rural producers," are both generators and users of ecological knowledge. Extension agents, including agricultural and natural resource extensionists as well as development agents, are technical workers whose main role is to translate and diffuse information from researchers and policy makers to farmers. The third population of important actors in rural development and environmental management includes researchers, and these come in two types: technical research scientists who engage in experimental research and field trials directly related to natural resource management and agriculture; and academic researchers in both the natural and social sciences, who focus on topics such as ecosystem structure and function, local ecological knowledge, and rural production and extraction systems, among others.

Each of these actors possesses knowledge about the ecological systems in which they work. We consider very broadly two kinds of knowledge, scientific ecological knowledge (SEK) and local ecological knowledge (LEK), though much hybridization occurs between the two. An individual or community may obtain or generate information and knowledge about ecosystems from many disparate sources, including both formal and informal knowledge networks. In this section, we describe these types of knowledge and the roles of extension agents, scientists, and local people themselves in using and disseminating this information to achieve rural poverty reduction.

Types of Knowledge

Local Ecological Knowledge

Local knowledge is a broad term referring to the skills and information held by local populations, often referring to natural resources, but this may also include social and cultural aspects of society. It is situated knowledge, based in the conditions and experiences of the holder. As with culture, local knowledge is by no means static, rather it adapts to changing conditions and opportunities. Development and agricultural agencies, scientists, and conservation organizations have recognized the vast wealth of local knowledge in agriculture (DuPré 1991; Scoones and Thompson 1994), medicine (Balick et al. 1996; Schultes and von Reis 1995), and, more recently, natural resource management and conservation (Posey and Balée 1989) and development (Bicker et al. 2004).

Resource management seems to work best when it is driven and monitored by the resource owners and users themselves, such as community management of fisheries in the Brazilian Amazon lakes (Castello 2004; McGrath 1999). Residents' knowledge about the life history of fishes, their behavior, including reproduction, and their population size helps communities work with natural resource authorities (who do not have that information) to develop and implement sound management plans. Knowledge about the environment and resource management is also sometimes embedded in local customs, rules, and norms that govern natural resource use (e.g., Begossi 2001), though as the social fabric breaks down in a community, rules may no longer be followed, and knowledge may be lost or unused. The disintegration of social networks and local institutions that support community-based resource management is a problem for both the well-being of residents, as well as, for conservation of natural resources and biodiversity (Alcorn 1994; Barrett et al. 2005). On the other hand, new opportunities and experiences, such as off-farm labor, are sources of new knowledge that can be integrated with existing knowledge (Sears et al. 2007).

Scientific Ecological Knowledge

Scientists conduct research in the field and laboratory to answer very specific questions, with both practical and theoretical applications. Scientists and technical researchers also turn to the field for generation of knowledge, but their observations and experiments follow a formal scientific approach and research design, whereas local resource users largely rely on trial-and-error. Most ecological research is based in reductionist science, examining single components and simplified dynamics in ecosystems, rather than on systems research (Ison et al. 1997). The stringent rules of reductionist scientific inquiry, isolating causal factors and mechanisms, render much of the research information too narrow to be effectively applied in complex ecological systems. In the case of ecosystem-based management for conservation and poverty reduction, the utility of the scientific and technical information is only as good as its relevance to local people and conditions (Scoones and Thompson 1994).

Experimental assessment of environmental phenomena and productivity can be especially useful to verify and enhance LEK (Walker et al. 1999). Relevance of the results and recommendations emerging from scientific and technical experiments will be improved by using local terms and concepts (Steiner 1998). Ecological studies whose objectives are to assist with rural production systems should focus on systems that will yield results that are specific and relevant to that location, technically appropriate, and where implementation will be compatible with existing social structure and dynamics (Hanyani-Mlambo and Hebinck 1996) and market opportunities.

Scientists turn to libraries and to their professional colleagues for information, as well as field studies, while rural producers turn to their grandparents and neighbors to share information. Increasingly, and taking a cue from ethnologists, ecologists are also turning to the local residents for information about ecosystem components and functions (for example, see Arce-Nazario 2007).

Actors

Extensionists

Rural development and extension technicians work for government and non-governmental agencies to deliver educational and technical training to rural people in an effort to improve rural livelihoods and well-being. Extension agents are knowledge brokers (Bentley et al. 2004) who can play an important role in the rural communities by providing information, ideas, and new technologies and practices to producers and resource managers. They can inform rural people about new market opportunities or government and non-governmental programs, such as seed distribution, land titling, and agricultural credit programs. Their goal is technical and human capacity building to help rural people increase production and add value to existing resources.

There are many types of extension training programs, ranging from classical agriculture and forestry technical schools to integrated sustainable development programs. Classical extension education programs tend to train students to specialize in one cultivar or system, such as soybeans or orchards. The curriculum and practicum are wholly based in technical and scientific knowledge. While some technical schools have embraced the value of local ecological knowledge, some urban technical schools are still biased against local empirical knowledge. In those schools, students from rural areas may be taught to undervalue and replace what they may know from the farm with technical information generated in research centers, and, in turn, fail to appreciate LEK in the field. The danger of unlearning rural epistemology, especially in regions where landowners rely on diversified production systems, is that extension agents end up providing information and technologies that are at best inadequate and at worst damaging to both the ecosystem and to local livelihoods (Hanyani-Mlambo and Hebinck 1996). Thus, it is critical that technical schools today help students understand and negotiate different types of ecological knowledge and production practices.

It is also important to avoid the classical diffusionist model of development, disseminating knowledge uni-directionally from the scientist to the farmer. Extension agents should be encouraged to engage in a two-way knowledge exchange between the field and the research agency, and between the farmer and the policy decision makers. Because of the nature of some state extension and international development programs, extension agents often strive for rapid results in the field. They want to reduce the number of factors in a system that they cannot control, including biotic and abiotic. For this goal, technical information generated from simplified experimental situations is most useful for ease of explanation, demonstration, and application.

Successful extension agents are those who are able to articulate both local and scientific knowledge, such as learning the vernacular and scientific names of species, recognizing scientific ecological concepts in the farmers' descriptions of their production systems, and understanding the connectivity among the multiple components of a system. They are then better able to translate the SEK to LEK, and vice versa, making both knowledge sets available to all actors.

Rural Producers

Rural producers have a different ecological epistemology, relying on an empirical understanding of the systems in which they work. The terms and concepts they use to describe ecological systems and processes may be quite different than the technical language of extension agents. These epistemological and semantic differences can cause misunderstanding between the two populations. Sensitivity on the part of the extension agents, and, in many cases, research on the claims of farmers, is helpful to overcome these barriers.

Some rural farmers and natural resource users develop a rich knowledge of the environment through engaging in processes of inquiry and deduction in their daily lives and practices, generating empirical evidence through on-farm trial-and-error experimentation and innovation (Padoch and de Jong 1992). Years and generations of trial and error through calculated risks have yielded a capacity in some rural producers to build multifunctional, productive landscapes that provide livelihood and food security as well as environmental protection (Pinedo-Vasquez and Sears 2011). With the application of system-level management, they are prepared to respond to changes in environmental conditions, as well as, new market opportunities (Pinedo-Vasquez et al. 2002). They also gain and share knowledge about the environment, its productivity, management, and utility, through informal knowledge networks, such as talking and visiting with neighbors and visitors. Integration of information they gain from external sources such as off-farm labor and extension services results in hybrid knowledge and innovative practices (Hanyani-Mlambo and Hebinck 1996; Sears et al. 2007). For example, in a forestry project in Zimbabwe, rural farmers engaged in tree planting of their own accord through the "consolidation of local havens of knowledge through inter-regional and cross-border networks involving distant relatives and members of the family who are migrant workers. Such externally acquired knowledge is internalized, used and adapted to suit local conditions" (Hanyani-Mlambo and Hebinck 1996).

Because of both their understanding of ecosystem processes and the multiple sources of knowledge, some rural producers maintain resilient systems and are adept at adapting to environmental change and new market opportunities (Pinedo-Vasquez et al. 2002). Conventional extension education has tried to simplify their systems, and in effect destroying the resilience in the system. Sustainable development projects today, however, should consider ecosystem resilience, whether agro-ecosystems or natural ecosystems, as one of the goals. Rural producers have a great deal of empirical knowledge about resilience that can be useful to scientists and extension agents.

At the same time, it is critical for extension agents and rural producers to come to mutual understandings on emerging ecological concepts and market opportunities. Climate change, carbon sequestration, and payment for ecosystem services schemes are important concepts and opportunities, though they can be confusing to many. Extension agents must understand these well—the science, resource management, and the financial opportunities—in order to both explain the concepts and link rural people to the opportunities.

A Shift in Extension Providers: Enter NGOs and Field Researchers

Extension agents have traditionally been situated between the rural producer and the urban researcher, acting as a translator from the technical sciences to practice in the field. They also facilitate the movement of knowledge and information from the farmer to the technicians, helping to generate questions that can be investigated by science.

Conservation and development NGOs, as well as projects funded by international agencies that focus on rural poverty reduction, sustainable agriculture, natural resource management, and biodiversity conservation, have largely taken up the role of rural extension services (Pablo Eyzaguirre, 2 April 2009, personal communication). To alleviate pressure on national budgets and attract more resources, some countries, such as Mozambique, have moved toward outsourcing extension services, utilizing both public sector and private sector service providers (Geno and Rivera 2001).

NGOs employ multiple actors, including technical extension agents, scientific researchers, and local people. They operate under the sustainable development paradigm using approaches that are distinct from the classical diffusionist model of extension and development. They employ participatory methods to not only disseminate knowledge to rural producers, but also to access local knowledge to incorporate it into their materials and activities. For example, the “Farmer First” movement in the 1980s attempted to bridge the gap between development professionals and local people, training professionals to listen to local people and try to understand and integrate their local knowledge (Thompson and Scoones 1994).

To negotiate the challenges and opportunities posed by environmental change, emerging markets, and sustainable development policies, it is helpful if all groups can understand and communicate SEK and LEK. Ecological education opportunities related to agro-ecosystems, environmental markets, and ecotourism can help to bridge that knowledge gap.

Agro-Ecosystems

Some farmers simplify their systems to monocultures in response to market pressures, agricultural policies, credit opportunities, and extension education. These simplified systems are highly vulnerable to risks of failure from unpredictable changes in environmental conditions, such as predator infestation of a crop, and economic conditions such as highly variable price fluctuations. Farming systems that are ecologically complex, multifunctional, and diverse offer more security to farm families by providing a resilient system and income opportunities diversified over the seasons, products, and areas.

Complex agro-ecosystems are managed by farmers based on their understanding of ecosystem functions and services even though farmers may not explicitly manage with knowledge of these terms or concepts. Agriculture and resource management are linked processes for rural producers (Alcorn 1989; Padoch and Pinedo-Vasquez 2006). Rural producers focus on systems and processes within them, more than on structure and components of the system. For example, açaí palm extractors in the Brazilian Amazon know to keep some emergent hardwood trees in the stand to attract pollinators and seed dispersers (Brondízio and Siqueira 1997). Ecological research on agro-ecosystems helps to understand local production systems and the role of biodiversity and ecological processes in natural resource management (Altieri 1999; Gliessman 1990). Combining social and ecological research on these diverse systems, researchers can understand how socio-cultural and ecological factors inform decision making and management strategies by rural producers (Brookfield et al. 2002; Giampietro 1997; Jarvis et al. 2007). Results of these studies may be disseminated in both scientific and vernacular languages to be accessible to broader audiences.

Environmental Markets and Payment Schemes

Markets for ecosystem services (MES) and payments for ecosystem service (PES) programs are designed to provide economic incentives to landholders or resource stewards (a rural development opportunity) to conserve the natural environment (a conservation opportunity) in order to maintain critical ecosystem services (FAO and REDLACH 2004; Kremen et al. 2000; Pagiola et al. 2002) (see Chaps. 9, 10, 11, 12, 13 and 14, this Volume). Payment to producers or stewards of ecosystem services, such as carbon storage, watershed protection, and biodiversity conservation, can provide income to rural people. An example of this at the local level is downstream users of freshwater paying residents and resource users in the upper watershed to protect natural vegetation for its environmental service of providing soil stability and water flow mitigation (Creedy and Wurzbacher 2001). Maintaining ecosystem services such as these requires protection from land use activities that threaten the integrity of the ecosystem, such as deforestation. In other cases, increasing an ecosystem's capacity to provide the services may require active management of components of the ecosystem, such as re-vegetation, which will require relevant knowledge and information about that environment and species to be planted, which presents an opportunity for combining LEK and SEK in practice.

The success of PES schemes for conservation and poverty reduction depends on many factors, not the least of which is the ability to quantify and monitor the provisioning of the service, which requires a detailed understanding of ecosystem structure and function. It is difficult to establish a payment scheme for a service or product that is not accurately quantifiable using simple readily available methods (Corbera et al. 2007).

Other critical factors in the success of PES schemes include clear ownership of land or a resource (Unruh 2008), defining a legal framework and mechanism for the PES, and organizing institutional frameworks that connect providers and beneficiaries of the ecosystem service (Corbera et al. 2007, also see Estrada and Corbera, this volume). The rural poor who live in ecologically impoverished areas or without clear property tenure, which encompasses the majority of the rural poor, will have little, if any, opportunity to participate in these markets (Ebeling and Yasué 2008; Sunderlin et al. 2005).

Achieving the dual goals of conservation and rural poverty reduction through PES schemes requires continual education of resource stewards, extensionists, policy makers, and scientists in the realms of both natural and social sciences. Understanding how to become part of these markets requires advocates for the poor at the ecosystem marketplace, including advocates who speak both ecological languages, LEK and SEK, and advocates who understand the market and institutional arrangements.

Ecotourism

Ecotourism is cited as an approach to addressing both biodiversity conservation and rural poverty (Neto 2003). Because wealthy people are willing to pay to visit ecologically interesting places, there is a market incentive to local resource users or landowners to conserve biodiversity.

It goes without saying that ecotourism providers, and especially the guides, should know the natural history of the region, the species tourists want and might see, and salient points about the ecology of the area. Therefore, ecotourism guides must be well versed in LEK as well as SEK. Tourists also enjoy learning about the local livelihoods, local mythology, and local uses of natural products. To help the conservation cause, nature guides could also be versed in political ecology, so they can describe the pressures on local communities, households, and ecosystems that result in environmental degradation and threaten livelihoods and ecotourism opportunities.

Ecological education for ecotourism guides, therefore, should include not only both folk and scientific concepts about the natural environment, but also the links between ecosystems and human well-being. Educational activities should be geared toward capacity building for local residents and guides to be able to respond to tourist needs, but also to manage the ecosystem in such a way to maintain the ecological integrity of the environment. In ecotourism, there is value in both local and scientific ways of knowing.

Educational Goals and Opportunities

For rural people to engage in these three types of economic opportunities, they must be ecologically literate. Both LEK and SEK are valuable and useful forms of literacy.

Here, we discuss the educational opportunities and techniques for improving ecological literacy of all groups of actors.

Goals

Rural participation in emerging environmental markets, including agro-ecology, PES, and ecotourism, would be strengthened by a solid understanding of the ecological basis for both production and valuation of ecosystem services by all actors, including policy makers, rural producers, and extension agents, broadly including people from NGOs and public extension programs. Understanding the ecology is just half of the story, though. Being able to communicate this knowledge is equally important (Bekalo and Bangay 2002). Without access to information about production and trade opportunities, and the ability to communicate with buyers, rural people are limited to the crops and markets they know. Actors must understand also the rules governing the markets, the mechanisms for engaging in them, and have the power to negotiate equitable access to and participation in them.

In order to participate in environmental markets, all actors must clearly understand what ecosystem services are, how they are provided, why they are important, how they are quantified, and how other land uses may affect those services. Most of the rural people interviewed in a study on PES schemes in Brazil could describe the concept of ecosystem services and recognized those that were provided on their landholdings, though not in formal scientific terms (Veríssimo et al. 2002). Policy makers, on the other hand, may not be familiar with the term or the details of ecosystem function until they become engaged in these projects (ibid.). Ecological education for both actors, then, can help at least to break down the semantic barriers among actors engaged in these economic opportunities for conservation and rural livelihoods. For rural people, the ability to talk about the ecosystem in scientific terms can empower them to engage more equitably in market opportunities for development and conservation. The goal of ecological education for science literacy should be to expand local knowledge, not undermine it by denying its validity (Bekalo and Bangay 2002). For policy makers and market brokers, both scientific and local ecological literacy will help them make stronger legal frameworks and market mechanisms for negotiating these markets.

Development agents should be well versed in both local and scientific ecological knowledge, since, as field workers, they are in a position to disseminate information among scientists, policy makers, markets, and rural producers. It is particularly important today that extension agents understand environmental markets, and the design, policy, and science behind them.

Techniques

It is widely recognized that classical diffusionist models of technology transfer without capacity building are largely ineffective and certainly insufficient (Hanyani-Mlambo and Hebinck 1996). The key to environmental education initiatives aimed at poverty reduction is that they be responsive and relevant to the local socio-economic context as well as the distinct local environmental conditions. An essential element of both education and capacity building is to provide contextualized education using a problem solving approach (Bekalo and Bangay 2002). The contextualization of the curriculum and learning objectives, or situation in local context, roots learning activities in tangible issues and places that make the education relevant and more tangible to the learner. Environmental education in field settings, where learners are situated at the local site, either as visitors or as residents, is especially effective, especially by direct participation in demonstration projects (Salinger et al. 2005).

An opportunity to link local and scientific knowledge and technology is to incorporate local knowledge into technical extension programs, as was done in the global People, Land Management and Environmental Change (PLEC) program. PLEC projects enabled local expert farmers to host demonstration plots of a target species, system or technology using their own farming knowledge (Brookfield et al. 2002; Padoch 2002). The PLEC project sponsored demonstration events where farmers from neighboring villages and regions visited the farm of the expert, who would demonstrate her or his production system and farming methods. Participants shared not only knowledge gained in their own experiences, but also germplasm.

Similarly, based on a study of social networks of communication of environmental information in an article entitled "What you know is who you know," Crona and Bodin (2004) found that communication between fishers in a Kenyan fishing community was largely limited to those who use the same gear type. Limited communication between those fishers, other types of fishers, and influential non-fishers could be the reason for lack of effective collective action to manage the resource. Use of existing institutions for educational outreach, such as producers associations, religious organizations, and the local marketplace, can help facilitate participation of rural people in the activities, since commitment to formal education in poor, rural areas is often low (Bekalo and Bangay 2002). Participatory communication techniques, such as rural radio programs, are also useful for dissemination of information in rural areas (Chapman et al. 2003).

Another opportunity for ecological education for rural residents is through participation in field projects of NGOs and scientists. When ecologists and other scientists in fields related to the environment engage in field research or a project they often hire local field assistants. During this time, interacting with a research team, people who have otherwise had little scientific or research training are exposed to concepts, theories, and techniques for identifying, describing, and quantifying components, functions, and dynamics of ecosystems. For this exchange to be most effective, the scientist may have to first understand the local knowledge and understanding of the subjects of study. Understanding the local knowledge could also help to inform research questions, since the local knowledge is usually empirical and systems based.

Science and development are intertwined today. Many ecologists engage in applied research that contributes to the rural development agenda. An expected output of many ecological research projects today is a management plan for a useful or threatened ecosystem, species, product, or service. Specific educational modules for resource users can emerge from such projects, helping with broader dissemination of results. For example, research in a coastal Indian community on mangrove productivity and the effect of current fishing practices on both the fishery and occupational health yielded recommendations for a grass-roots public education module that can help resource users better manage their mangroves and prawn fisheries (Sarkar and Bhattacharya 2003). Evidence-based training about the fishery, from appropriate collection techniques to marketing, can help improve yield, quality of the product, and reduce ecological degradation of the fishery (Sarkar and Bhattacharya 2003).

In the following section, we explore the intersection of environmental education and poverty reduction, or pro-poor environmental education, through a case study of a community-based rural school. We examine the goals of the community-based school, their alignment with local and international agendas in sustainable development, the pedagogical model of the school, and challenges and inconsistencies of the educational opportunity.

Case Study: Escola Família Agroextrativista

Escola Família Agroextrativista do Carvão

The Escola Angela Família Agroextrativista do Carvão (EFAC) was founded in 1997 following a period of community organizing spearheaded by leaders of a local rural workers union, the Sindicato dos Trabalhadores Rurais do Mazagão (the Sindicate of Rural Workers of Mazagao). Leaders residing in Carvão gathered members of this and neighboring communities to discuss the state of rural education. Complaints ranged from poor, overcrowded condition of the buildings to the out-migration of rural youth to urban areas. Parents lamented that students had to travel to urban areas to finish high school, where they had to live with relatives. Many youth who finished secondary school in the urban areas would permanently leave rural communities. They also noted that there were no means for improving agricultural production in the rural areas, since the quality of rural extension services had steadily declined in the state during the 1990s.

The EFAC provides an example of an educational system that integrates local ecological knowledge with scientific ecological knowledge. The goal is to enhance local production systems with some technical training to address specific conservation goals (nutrition, health, education, and environmental sustainability). The pedagogy integrates the three perspectives defined by Sauvé, problem solving, interpretation, and social criticism. To date, the school has been successful in encouraging students to enroll. By 2007, there were over 100 students enrolled at EFAC from across the state; many of the students come from the várzea (floodplain) areas of Mazagão, traveling up to 2 days by boat to attend the 14-day sessions. In most of those communities, state schooling ends at grade four, so the EFAC has helped address the problem of the lack of educational services in the interior. The curriculum at EFAC has three components. One follows the national pedagogical model in which students learn traditional subjects of history, math, language, family health, and physical education. A second component focuses on recognizing the characteristics of the local environment, culture, and practices in the primary grades. These young students learn about the realities of their communities through interviews with their relatives to understand their cultural heritage, how their families make a living, and what aspirations their parents have for their farms. This establishes a home/community connection with the hope of building students' commitment to the improvement of their households and communities. The third component is the technical training in and practice of agricultural production and natural resource extraction. Students are taught a conservation paradigm based on the loose premise of valuing local biodiversity for sustainable development through project-based learning. Each project is designed to respond to a local challenge related to health and well-being of the household or community. For example, vegetable gardens students maintain at home include leafy greens and herbs in order to address the lack of fresh vegetables in the interior, which leads to common health problems such as anemia (See chapter on Ecology and Nutrition, Chap. 4, Vol. 1). An aquaculture project on designing and building storage tanks for shrimp and other species typically caught for household consumption addresses the shortage of wild food during the winter months in the varzea. EFAC is unique in the state and in Brazil because it was specifically designed to train students in the practices of both agricultural production and the extraction of forest products (hence the school's name), the typical production mode of the caboclo (indigenous or mestizo floodplain residents) farmers in Amazonia. The major extractive product in this region is the palm fruit açaí (*Euterpe oleracea*), which is a staple of the regional diet. Training at EFAC included methods of açaí production to augment the production of natural açaí stands as well as methods and principles of extraction. Of perhaps greater importance than the technical training, since most youth would have grown up harvesting açaí in their family landholdings, are the principles and practices of adding value to crude forest products and of marketing those products taught at the school.

Despite its novelty and success, the EFAC is facing difficulties remaining true to its desires to promote sustainable development in the region. Some of the obstacles facing EFAC are related to personal interests and goals of the rural youth. Many students do not have a personal interest in remaining on the farm; rather, they attend EFAC because it is the closest school to their community. Parents like to send their children to the EFAC because the teachers for the most part are better trained and more professional than those in the conventional state-run school, but they may not necessarily value the school's philosophy.

Thus, one of the major challenges of the school is keeping rural youth in rural locations, despite the educational model and pedagogy that encourage this. One of the reasons for this is that while the school administration has been able to obtain state funds from the Ministry of Education to ensure the operation of the school itself, it has not been able to procure funds to support its complementary yet equally important home-based activities to ensure the appropriate execution of on-farm projects. One solution to this problem is for school administrators to look for institutional support from government agencies and NGOs that share in the school's mission of promoting local sustainable development. Creating and maintaining these alliances would not only benefit the school and the students, but it would also help the agencies meet their goals of promoting rural development.

Furthermore, the administrators are facing challenges realizing the original objectives of EFAC due to students' lack of enthusiasm for the agricultural sector. While there are currently strong local urban markets for a number of native Amazon fruits (Brondizio 2008; Padoch et al. 2008), it appears that despite these opportunities to earn income through rural production, many youth still equate rural production systems with poverty. Youth are influenced by larger socio-cultural pressures to become educated professionals, which, in the Brazilian Amazon, has been historical route out of poverty. Thus, the challenge of EFAC is not only to show students a tangible means of prospering through rural work, but also to help shift the cultural stigma of the agriculturalist. We believe that when the agriculturalist enjoys a quality of life equal to that of an urban professional, cultural ideas and values will shift as well. Making these changes will result in a feedback loop—helping parents and students and the community value the school's goals and philosophy.

Conclusion

While ecological knowledge occupies an important place in rural development, and particularly in building the knowledge base critical for engaging in environmental markets, it is clear that a lack of other fundamental knowledge and capacities prevent the rural poor from climbing out of poverty traps.

Most of the world's poorest people do not have the capacity to act on the injustices from which they suffer, whether they be social, economic, political or environmental. They lack education, social organization, and clear resource ownership that would empower them to first understand their situation in a broader context, and second to take action. Many also lack basic human rights such as religious freedom and security, as well as basic development services, such as schools, healthcare facilities, and communication and transportation infrastructure. It is difficult under these conditions to engage at all, let alone equitably, in profitable activities based on environmental resources.

It is critical to examine the multiple ways that education can be used as a tool for empowerment, self-governance and access to economic opportunities based in natural resource management. Ecological literacy is a critical element of an integrated education that should help the rural poor to understand the relationships among economic, political, social, cultural and ecological systems. Education for sustainable development can help to build capacity in rural families and communities for empowerment, self-governance and communication to confront the environment-poverty links.

Educational initiatives, both informal and formal, that recognize the value of local knowledge and build on it; provide a contextualized curriculum and participatory pedagogy; and focus on building local empowerment and self-governance, are necessary to achieve poverty reduction goals. Only by understanding the broader systems and factors that contribute to poverty conditions can rural people and their advocates confront them. Education can play a major role in this, and ecological literacy is one important outcome.

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Chapter 4
Why Gender Matters to Ecological Management and Poverty Reduction

Isabel Gutierrez-Montes, Mary Emery, and Edith Fernandez-Baca

Introduction

Gender issues in conservation and rural development have been a topic of discussion within research and development institutions since the 1980s. Women have been excluded from many programs and projects both because of the traditional values of some cultures and because of the prejudice inherent in many development efforts of the time. Lack of participation in development programs has had long-term implications not only for the women themselves, but also for their children. Furthermore, focusing exclusively on men meant many programs failed to attain their goals for several reasons. In some cases, the information given to men was not communicated to women who were responsible for applying the information. In other cases, the focus of the project pertained to women's work and often the men who received the information or participated in the demonstration projects did not know what questions to ask. Some efforts targeted at men had adverse effects on women by changing agricultural processes in ways that negatively impacted women and their children. For example, the focus on cash crops often led to a decrease in subsistence farming and degradation of soils and, thus, increased food insecurity. Traditional views of development and of aid programs were based upon assumptions about who did what work and who made what decisions, which were not always reflective of reality. Thus, programs that intended to increase access to household resources were targeted at the male head of the household with the assumption that knowledge and information would trickle down to the rest of the household. These approaches also made unfounded assumptions about knowledge and knowledge transfer. A focus on gender has helped to broaden our understanding of how people learn and what skills and techniques are useful in effectively transferring information and expertise from one context to another.

I. Gutierrez-Montes (✉)
Centro Agronómico Tropical de Investigación y Enseñanza (CATIE),
Cartago, Turrialba, Costa Rica
e-mail: igutie@catie.ac.cr

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Women's demands that their voices be heard led to an understanding of the importance of recognizing and valuing local knowledge as part of an information exchange. The focus on gender encouraged a deeper look at equity issues, not just those related to gender, but also in relation to the role of technology transfer and how technologies can impact groups of people differently. In some cases, new technologies introduced and adopted by one group can lead to increased burdens for others. Thus, the introduction of motor bikes increased men's freedom and range of action, but increased the burden of work for those left behind. The critiques of development emerging from the analysis of gender have led to a deeper understanding of how societies function and how change occurs within particular societies. These analyses were important to formulating policies that support endogenous development.

Both ecological management and poverty reduction approaches have changed in response to gender-related critiques. Understanding the role of gender in any process is relevant and central both for human development and natural resource conservation. In this chapter, we will first explain the concepts of gender and poverty as they are used in the development field, provide some examples of how the concepts have been applied, and outline a systems approach to integrating gender, natural resource management, and poverty reduction.

Defining Gender in the Development Context

Gender is the assignment of masculine and feminine characteristics to bodies in cultural contexts (Grewal and Kaplan 2006). The term refers to the socially constructed differences and relations that exist between women and men. These differences vary according to local circumstances, context, and time. Using a gender focus can help researchers and practitioners understand other interrelated social variables such as class, race, and ethnicity (Poats 2000). All societies have a very well-defined division of labor between genders and, therefore, gender has a profound influence on the use of resources (Valdivia and Gilles 2001). In most societies, but not all, that division of labor is also connected to levels of status and access to resources. Thus, in contrast to men's activities, women's activities are often considered "duties" and not "work." Nevertheless in many cultures, women play a very important role in the creation and transmission of agricultural knowledge as stewards of natural and productive resources (Cabrera et al. 2001; Valdivia and Gilles 2001). Indeed, often because of this division of labor, women have more local knowledge about certain agricultural and conservation practices.

The relevance of gender began to surge in many development projects in the 1980s. For example, as labor allocation and management of resources became central to the question of how to improve the productivity of livestock in agro-pastoral systems and what conditions are necessary to facilitate adoption of different animal production systems at the household level, researchers and practitioners had to engage women to find answers (Valdivia 2001). According to Valdivia and Gilles (2001): "Women of the developing world contribute significantly to the three pillars of food security: food production, economic access to available food, and nutritional security." In many areas of the world, women are linked with nature in terms of having access to and responsibilities for the natural resources (for example, women are often responsible for finding and carrying water). Agarwal (1992, cited by Sachs 1996), using the example of rural Indian women, suggests that while women are often the primary victims of the destruction of nature, they are, as a consequence of their daily lives, owners of a specific knowledge about the environment. Agarwal goes further and argues that, because of this knowledge, women's perspective on the environment "may provide alternative visions for human relations with the environment based on material realities rather than symbolic connections" (Sachs 1996).

Gender and Natural Resource Management

The gendered division of labor in relation to natural resource management is critical to understanding ecosystems and developing strategies for sustainable management of those resources. Valdivia (2001) highlights the relationship between gender and resource management by stating that, "research experiences show the relationship between gender, resource management, and the ability to build livestock assets and security in different household production systems." The status of women, their income, and access to power determine their involvement in decision making related to conservation and the resources they may access to support sustainable management. These decisions and resources, in turn, determine the degree to which a woman can use natural resources to increase her resources, such as livestock, and, thus, increase food and financial security for herself and her family.

Many authors have specifically looked at the critical role of social capital in determining how households access and manage natural resources. Social capital is defined in terms of the networks in which people participate, both informally and formally, and the norms of trust and reciprocity that govern network relations. Jan Flora (1998) argues that social capital "places emphasis on the will and the capacity of people to solve problems and improve their lives in a joint enterprise." Cornelia Flora (2001) suggests that, social capital for sustainability depends on strengthening communities of interest[1] and communities of place.[2] Social capital is gendered in societies since men and women often belong to different social or community groups. Participation in these groups can determine access to resources, knowledge, and information. Social capital also has implications for natural capital. Natural capital here refers to the natural resources in our environment and is the focus of most stewardship work.

In addition to the gendered nature of social capital, women often lack access to educational programs and, thus, to knowledge about sustainable use of natural resources. In some societies, the basic work of natural resource management is divided among men and women, and in other cases, these male and female sets of knowledge rarely interact making effective natural resource management difficult, if not impossible. The inclusion of a gender perspective in biodiversity conservation and in the search of rational ways to manage natural resources is imperative since

[1] Group of people tied together by common activities or interests.

[2] Group of people sharing a geographical (physical) space.

gender is key to understanding the different ways people relate to natural resources and ecosystems. Women and men possess different types of knowledge, access, and control over resources; they have different perceptions and attitudes regarding their relation with resources and conservation; and, thus, their impact on natural resources is different as well (Schmink 1998). Conservation of natural resources requires the participation of all of the community, including men and women. Men and women's needs and interests in relation to nature can be very different, perhaps even in conflict with one another. Nevertheless, women's interests and voices are frequently silenced in public forums when decisions must be made regarding natural resources and their management. A good example of differences in perspectives and attitudes can be found in terms of water use and management. Often, for men, water is an important input within agricultural activities' placing particular attention on quantity, whereas for women, water is the starting point in food preparation for the family, and, therefore, quality rather than quantity is most important to them. These complementary perceptions of water needs, if acknowledged by both genders, can be seen not as a point of conflict, but of negotiation towards more holistic natural resource management to meet a range of livelihood needs.

Gender and Poverty

Many studies point out that key aspects of poverty specifically affect women including child nutrition and access to healthcare for women and their children. Globally, poverty among women is more acute compared to male poverty. Poverty among rural men has increased over the last 20 years by 30% while it has increased 45% among women, which has led to increased attention on what has been referred to as "the feminization of poverty" (Moghadam 2005). The number of rural women living in poverty in developing countries has increased globally by almost 50% over the past 20 years to 565 million with large increases of 374 million in Asia and 129 million in sub-Saharan Africa (Power 1993).

In terms of achieving rural poverty reduction, research indicates that projects that target women may have a greater likelihood of increasing nutrition and child welfare than those that target men. For example, "evidence from research at IFPRI shows that who receives the household income has a significant effect on the consumption and nutrition status of family members. Increases in the income of female headed households compared to male headed (or joint) households showed greater expenditure on food consumption with better nutritional outcomes for children" (CGIAR 1995) (see chapter 4 by Remans et al., Vol.1).

Gender, Poverty, and Conservation

The gendered nature of work and labor impacts a woman's status in nearly all societies and determines her access to resources, information, skills, and networks. Evidence indicates that because of a women's dominant role in procuring fiber,

Table 4.1 Gender division of labor in human societies that defines relationships with nature

Tasks and responsibilities	Men	Women
Productive	Production of goods and services: payments made for these goods and services	Home gardens and livestock (small animals): no visible payments made for these activities
Reproductive	Not socially recognized	Home labor and family health (pregnancy, child and elderly care, tasks and duties related to food, drinking and cooking water and firewood): no payments are made for this
Community	Key roles in decision making	Participation in community activities mainly in a supportive and often background role (organization, food preparation, taking notes, etc.)

(Adapted from Aguilar and Castañeda 2002)

firewood, carrying water, gathering fruits and food, and searching and preparing medicinal plants, women are often more directly interacting with the natural environment and are more directly affected by environmental degradation. Thus, women typically have a greater interest and investment in achieving environmental health through rational and sustainable use of natural resources. Some authors believe that this more sustainable use of resources results from the fact that they are responsible for reproductive activities (Oever 1991; Karremans 1994; Revelo et al. 1995; Aguilar and Castañeda 2002) (Table 4.1). That is, women's role in everyday life in regard to finding and serving food, childbearing and nurturing, finding and maintaining shelter, and linking their households to resources and support often gives them greater insight into the need for conservation. Their concern for their children's future is often a strong motivator for supporting conservation efforts.

The success or failure of poverty reduction programs is directly related to the relationship between poverty and environmental degradation. Since women and their children tend to be more affected by poverty and more vulnerable than men, a focus on their situation is critical to the success of any effort to protect or rehabilitate the local environment. The next section of this chapter outlines a systems approach for ecological management and poverty reduction that promotes gender inclusive approaches for mobilizing change and implementing projects.

Systems Approaches to Gender, Ecological Management, and Poverty Reduction

General systems theory emerged as a methodology for integrating natural and social sciences (Johansen 1991). Systems-based thinking is holistic; it allows us to search for ways to integrate research and practice from multiple perspectives and disciplines, to look at change across institutions and communities, and to embrace the complexity of interactions between levels and among elements of the system. Systems-based approaches focus on the situation as a whole, accepting the intricacy of the nature–people interface. Systems thinking has been used to address natural resource management issues in ways that take into account the stresses (out migration, climate change, decline in water quality, and access to fuel, for example) on rural populations (Bosch et al. 2007).

In order to understand why gender matters in ecological management and poverty reduction from a systems-based perspective, it is critical to consider the decisions that impact conservation and create pressure on natural resources. In many societies, these decisions emerge from both female and male decision-making arenas. Moreover, we have to consider that access, entitlement and control over assets, resources, or capital (with special attention to natural capital) are often gender based (Flora 2001). Therefore, understanding local knowledge systems, both male and female, and their dynamics is fundamental to discerning the complexities of natural resource management in any household or locality (Kelkar 2007). Analyzing the rural household and its assets and contexts requires seeing them as parts of a larger system. As Bosch found, "progress may be found in the application of systems thinking to understand and manage the 'natural' and 'people' systems associated with natural resource problems and solutions" (Bosch et al. 2007). With a systems-based approach that recognizes the gendered nature of relations with the environment, it is feasible to develop a positive feedback cycle where implementing environmentally friendly and equitable agricultural and natural resource strategies, in turn, contributes to reducing poverty, which in turn contributes to reducing pressure on natural resources. These actions replace the perverse downward cycle where inequality and environmental degradation contribute to greater poverty, which in turn leads to increased pressure on natural resources and, hence, increased environmental degradation (CATIE 2007). Two systems-based approach are particularly helpful for understanding the downward spiraling effect (and the possibilities for reversing the spiral) of increased pressure on resources leading to increased poverty: the Sustainable Livelihoods Approach (SLA) and the Community Capitals Framework (CCF).

Sustainable Livelihoods Approach and the Community Capitals Framework

The sustainable livelihoods approach provides a method of looking at household activities rather than simply the activities of people within households. Developed by sociologists interested in and working on both defining poverty and proposing solutions, SLA began as a conceptual framework at the Institute for Development Studies (IDS) at Sussex University (Scoones 1998; DFID 1999).

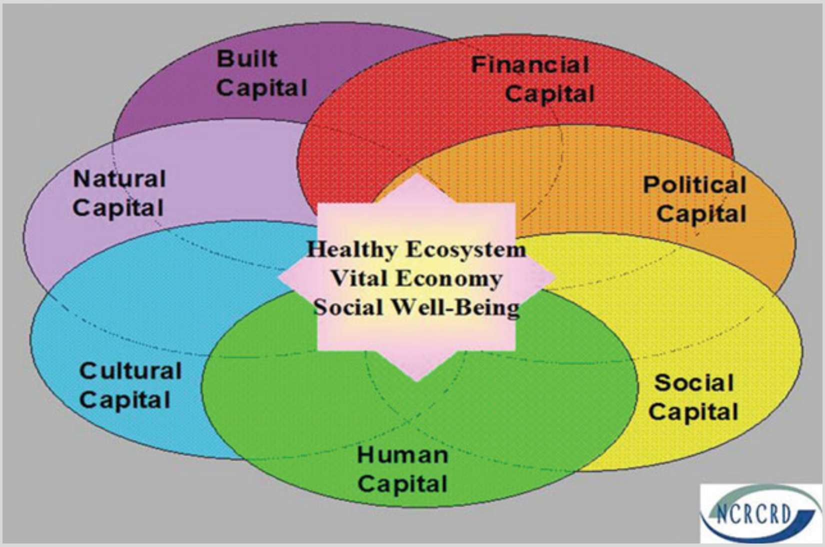
A striking feature of the framework was the articulation of a conceptual scheme that goes beyond the economic view that defines poverty as a simple lack of money.

The framework identified five types of livelihood assets that are critical for the poor and that require decisions and tradeoffs: human, financial, natural, social, and physical assets. In relation to the five types of capital, the sustainable livelihoods approach focuses on understanding how poor households find and manage resources in order to develop and reinforce strategies to reduce poverty. The strength of the livelihoods approach is that it encompasses identification of assets, capacities, and activities required to live a decent and sustainable life. Livelihoods are defined as sustainable when people can face and recover from stress and shocks and, more importantly, when households can maintain and even improve assets and capacities without deteriorating the natural resource base (DFID 1999). In the following years, the framework was used in a plethora of studies and as a result of the practice and field applications, improvements were proposed and new approaches developed. One such approach which emerged from work in the United States and Latin America by professionals from the Sociology Department at Iowa State University is the Community Capitals Framework (Flora et al. 2004). This framework redefined basic aspects of the framework, and broadened the resources, assets, or capitals to study from five to seven (Box 4.1). The seven capitals they proposed are human, financial, natural, social, built, cultural, and political. The inclusion of cultural and political capitals, absent from the SLA approach, filled the gap in the analysis by addressing the relevance of local knowledge and traditional uses of natural resources (cultural capital), as well as, the importance of power, relationships to decision-making structures, and community participation in the creation of agendas regarding natural resource use (political capital). Both capitals included within the CCF (cultural and political) are considered to be central when analyzing governance and governability of natural capital (Meinzen-Dick et al. 2004).

The introduction of the Community Capitals Framework offers a method of analyzing inputs and impacts from both (Gutiérrez-Montes 2005) within and outside the community that determine the success of sustainable livelihoods activities. The framework highlights interdependence, interaction, and synergy among the capitals, as use of the assets in one capital can have a positive or negative effect over the quantity and the possibilities of other capitals. On the one hand, the loss or degradation of assets within one capital will negatively affect one or more capital because when one of the community capitals is severely affected or depleted, the health and sustainability of the community is compromised (Gutiérrez Montes 2005; Emery and Flora 2006) leading to the domino effect (Gutierrez-Montes 2005). On the other hand, each capital has the potential to build or strengthen one or more of the other capitals. As Flora et al. (2004) point out, "each form of capital has the potential to enhance the productivity of the others." This multiplying effect among capitals can initiate an upward spiral and, ideally, a sustainable development process (Gutierrez-Montes 2005; Emery and Flora 2006). Finally, balance among capitals has proven to be important since one capital alone should not be favored at risk of the others. Given the interaction among the capitals, "When (Gutiérrez-Montes 2005) one capital is emphasized over all others, the other resources are decapitalized, and the economy, environment, or social equity can thus be compromised" (Flora et al. 2004).

Box 4.1 What Are the Capitals? (Flora et al. 2004)

Capitals are resources people and/or communities posses. These resources can be of different types and can and should be used (invested) to create more resources in the long term, for all endogenous community development processes. This concept is empowering because the starting point is what communities have (instead of what they need or are lacking) and it highlights the fact that these resources (capitals) a community has can be multiplied (through investment).



Natural capital refers to those assets that abide in a location, including natural resources, biodiversity, amenities, and natural beauty.

Cultural capital reflects the way people "know the world" and how to act within it. Cultural capital includes the dynamics of who we know and feel comfortable with, what heritages are valued, and collaboration across races, ethnicities, and generations, etc. Cultural capital influences what voices are heard and listened to, which voices have influence in what areas, and how creativity, innovation, and influence emerge and are nurtured. Cultural capital might include ethnic festivals, multi-lingual populations, or a strong work ethic.

Human capital includes the skills and abilities of people, as well as the ability to access outside resources and bodies of knowledge in order to increase understanding and to identify promising practices. Human capital also addresses leadership ability and the capacity to "lead across differences," to focus on assets, to be inclusive and participatory, and to be proactive in shaping the future of the community or group.

(continued)

Box 4.1 (continued)

Social capital reflects the connections among people and organizations or the social glue to make things happen. *Bonding social capital* refers to those close ties that build community cohesion. *Bridging social capital* involves strengthening weak ties that create and maintain bridges among organizations and communities and provide access information and resources.

Political capital reflects access to power and power brokers, such as access to a local office of a member of congress, access to local, county, state, or tribal government officials, or leverage with a regional company.

Financial capital refers to the financial resources available to invest in community capacity building, to underwrite businesses development, to support civic and social entrepreneurship, and to accumulate wealth for future community development.

Built capital refers to the infrastructure that supports the community such as telecommunications, industrial parks, marketplaces, water and sewer systems, roads, etc. Built capital is often a focus of community development efforts.

Community Capitals from a Gender Perspective

Gender aspects are key when integrating ecology into rural livelihoods and poverty reduction strategies. Labor division, decision-making, and income generation patterns of rural households typically are gender specific, as are other intra- and inter- household arrangements (Table 4.1). Strengthening the role, capacities, and decision-making power of women not only empowers them but also increases the livelihood security of rural households as a whole (CATIE 2007). According to Flora (2000), "communities of place and of interest have resources" and these resources can be consumed (used up), stored (not available for use), or invested to create new resources.

The focus on different types of capital emphasizes the relevance of place since the availability and significance of capitals are determined by where people are and live. Community capitals can be divided into two main groups or "factors": human and material factors. "Human" or "intangible" factors comprise social, human, cultural, and political capitals, whereas "material" or "tangible" factors comprise natural, financial, and built capitals. Women tend to be more focused on human factors, while men focus on the material factors (Chambers 1997, Figs. 4.1 and 4.2). Understanding how males and females interact and prioritize these capitals is essential for implementing projects aimed at ecosystem management and poverty reduction.

Embeddedness in place is important in achieving an upward trajectory that can lead to sustainability of a community. Communities have distinct world and life views that are grounded in common belief systems and physical places that translate into commitment to place and a relationship to the natural world that is meaningful, satisfying, and diverse (Curry-Roper 2000). According to Chiappe and Flora (1998),

Fig. 4.1 Community resource map drawn by men

Fig. 4.2 Community resource map drawn by women

sustainability embodies three imperatives: environment, economic, and social. Different actors, based on their social and geographic location, relate differently to these three imperatives. Understanding gender access to and control of resources contributes to our knowledge of the dimensions of sustainability (Flora 2001).

Differences in the way that men and women understand systems require us to put the two perspectives together to complete the picture of how the ecology of a place interacts with poverty and efforts aimed at fostering sustainability. Examples of the differences in gender perspectives can be observed when men and women separately map community resources. For example, Fernandez-Baca (1996) observed noticeable gender differences when men and women were asked to document natural resources in their community (Figs. 4.1 and 4.2). Men gave more importance to natural and animal resources (natural and financial capital), as well as to the natural boundaries surrounding the community, while human and institutional resources had a stronger presence in the diagrams drawn by women. In the example shown in Fig. 4.2, women drew themselves as an organized group in the main town plaza and indicated where their houses were in regards to the plaza. Chambers (1997) would call this the pervasiveness of the gender dimension. He argues that women are socialized more to deal with people (family, friends, neighbors), while men are socialized to deal with things (material).

Women often depend on informal relations and form strong relations of kinship and friendship. Informal networks based on everyday forms of collaboration, such as collecting water, finding wood for fuel, and sharing childcare, provide solidarity and access to household resources (Westermann et al. 2005). The relationships are reflected in the women's drawing where they show the strength of their relationship by having all of the women depicted as holding hands. While men put names on the map to indicate different crops, women used color and detail to indicate each crop. For example, different colors were used to depict potato plots of the different varieties, making it easy to identify the type and variety of potato (improved or native). The river that appears in the women's map (Fig. 4.2) also depicted not only water, but the fish (trout) found in it. Women were also more detailed in depicting biodiversity within crops. According to Flora (2001), women may focus more on biodiversity than men because their specific material responsibilities within the household give them some control over and access to different resources. Both male and female groups depicted water in the form of rivers and included both communal and private land.

While men's groups initiated their maps with rivers and streets, all the women's groups started their drawings with the church and main town plaza. Two possible explanations are the importance and relevance of these symbols. The church is seen not so much as a religious symbol, but as an institution that provided some form of aid to women's groups, though many times in return for something else (i.e. food, labor, or access to communal resources). Therefore a relationship exists between the community and the church based on mutual benefits. A second explanation of why women start by drawing the church and the plaza is that it was a way of locating themselves at an initial point that made it easier for them to position all other resources within the map.

Access and control over property and related resources also vary by gender. Flora (2001) observed in communities she studied in Ecuador, Burkina Faso, and the

Philippines that men are vested by law and custom with property rights, as well as the control of the labor of household members, while women are often seen as too weak or too emotional to have such control. Because women very rarely have control over or direct access to privately held resources – even if they own the resource – they are "more likely than men to be attuned to common resources and their conditions" (Chiappe and Flora 1998). Likewise, the lack of access and control over resources – financial, manufactured, human, social, and environmental – can limit women's ability to act effectively to translate their perception of an environmental threat into a concrete action (Flora 2001).

Globally, there are many examples on how women invest those capitals over which they have access and/or control and the effect of these decisions on ecological management and poverty reduction.

Natural capital: Women are often those who collect, use, protect, and nurture natural resources. They also play critical roles in the planting, cultivation, harvest, and storage of agricultural products (Howard 2003). Women's knowledge of seed selection for example, is important for maintaining biodiversity (Cabrera et al. 2001). Due to their interaction with the natural resources, women have developed vast local knowledge, skills, and technology regarding conservation, protection, use, and management of natural resources (Howard 2003; Adhikari 2001). Giving them a more relevant role in natural resource management programs is therefore necessary.

However, integrating gender into natural resource management projects is not always easy. Since access and control over resources is gender based (Flora 2001), decisions on how natural resources will be used usually depends on who has control over the asset that will be invested. Usually, women lack access to private property and must rely heavily on access to common property resources (Zwarteveen and Meinzen-Dick 2001). Even then, female access and control over those resources can be limited, especially in terms of land and water for agricultural purposes. Men often control land, and women depend on men to get access to it. Sometimes, women have access to common areas, or specific garden areas, where biodiversity contributes to household as well as environmental sustainability (Flora 2001). In some rural societies in the Andes region, women have little or no decision-making or negotiating power when it comes to developing/suggesting ways natural resources can be used more efficiently. Economic and social class and indigenous status are issues that exacerbate lack of negotiating power. Such is the case of water rights negotiations in South Asia. In Bangladesh, women have no involvement in agricultural water management. In the past, the 'right to water' was tied to the 'right to land,' which was usually in the hands of men (Faisal and Kabir 2005). According to Zwarteveen and Meinzen-Dick (2001), the insecure mechanisms that women have with which to negotiate water access can have a negative impact on the quality and distribution of this resource. Thus, the authors postulate that efforts to bring about greater gender equity in rights to water are more likely to succeed when irrigation systems are managed under a common property regime. As resource management and rights to resources are transferred from the state to local organizations, it is essential to ensure women's participation in control over these resources in order to strengthen the effectiveness of local organizations and improve compliance with rules and maintenance requirements (Valdivia and Gilles 2001).

Women in poor, rural households are disproportionately affected by the effects of environmental degradation on availability of natural resources that are key to women and their family's livelihoods. Fuelwood and other biomass sources are becoming increasingly inaccessible to women. Due to the increasing scarcity of energy sources, women have to walk greater distances to collect fuelwood, losing valuable time that could be allocated to more enterprising activities (Kelkar and Nathan 1996). Likewise, women are more vulnerable to ecological disasters that are now increasing in intensity and frequency due to the effect of climate change. Faisal and Kabir (2005) reported that women in Bangladesh identified a number of water-related factors that make them more vulnerable on a daily or seasonal basis. Collecting drinking water becomes extremely difficult during flooding as well as during droughts when women may have to walk several kilometers just to fetch a pitcher of water. At the same time, both floods and droughts can destroy homestead gardens and affect household food security. In many cultures, men and children are traditionally fed first and, so, women are the ones who will suffer more from poor nutrition. In situations of extreme poverty or vulnerability, where the goal is the family's subsistence, women will find ways to negotiate access to natural resources often at the expense of their time and limited resources (both human and natural).

Cultural capital: Women's ways of thinking and doing everyday chores (productive, reproductive, and community) determine how natural resources are used, how agriculture is undertaken in many societies, and how families and households cope with poverty (Table 4.1). Knowledge is gender biased as are the roles related to that knowledge. Gender-biased knowledge, therefore, varies according to the environment and gender roles. Women and men have access to different spaces and environments giving them different information about the local environment and biodiversity, often with women knowing the space around the home best and men being more familiar with the more distant areas. Older women often know most about medicinal uses of plants and are usually the most skillful seed savers. New high-yielding varieties of plants may not meet the full range of nutrients needed to assure food security within the household nor provide the by-products often used by women, such as straw for making mats and fodder and leaves for relishes. Thus, genetic erosion is tantamount to a form of cultural erosion that, ultimately, may result in loss of social status for women by reducing their ability to prepare traditional foods, and to make craft items (Momsen 2007). When women's ways of thinking and doing are not included, we have an incomplete picture of how a territory or watershed functions.

In agrobiodiversity research, gendered knowledge is often still considered to be abstract, uninfluenced by relations of power, culture, and context (Momsen 2007). Clearly, gender influences much of what we know and what happens in the developing world in regard to biodiversity. Furthermore, culture and power are determinants of the loss or preservation of natural resources in some fragile environments. In a study in a Kenyan village, Nyasimi (2006) observed that sexual cultural rituals and changes in them brought about by migration had a strong negative effect over soil, promoting erosion in already very fragile soils. In this case, the local culture dictated that certain sexual rites must be performed before fields are plowed. With men migrating out of the village in search of work, the rituals are not performed, the fields are not plowed and taken care of, and the downward spiral of poverty and environmental degradation continues and is exacerbated.

Human capital: Systems for knowledge and skills transfer also vary according to gender (Flora 2001). When women have access to additional resources, they use those resources to increase educational opportunities and healthy living strategies for their children. According to Flora (2001), women are traditionally responsible for human health in the family, especially in poor families, which gives them a different perspective on how they use the different forms of capitals. It also means that they might be the first to notice health problems associated with shifts in environmental quality. Very often, when women organize around poverty and sustainability at the community, regional, and state levels, access to education, healthcare, and social welfare support increases as does the focus on poverty eradication.

Quisumbing and Meinzen-Dick (2001) argue that investments in women's human capital "more than any other form of investment, increases women's capabilities, expands opportunities available to them and empowers them to exercise their choices." These authors assert that, "improving women's education is probably the single most important policy instrument to increase agricultural productivity and reduce poverty" (Quisumbing and Meinzen-Dick 2001). In some rural societies, such as those in the Andes, women might not have control over resources, but they still play a relevant role in deciding how those resources are managed. This is especially true with financial resources. Amaya (2008) observed that in the majority of households she studied in Bolivian watersheds, decisions about income and household management are made by men and women together. This is consistent with observations made by Fernandez-Baca (2006) in communities in the central highlands of Peru where although women have little control over natural resources, they share decision-making, power with their spouses when it comes to financial capital. At the same time, authors highlight that women's decisions have greater weight than men's when it comes to household management (Fernandez-Baca 2006). Additionally, Amaya (2008) reports that due to the increase of male migration, women now increasingly engage in productive activities that had previously been exclusive to men such as plot preparation and irrigation. Something similar has been happening in rural areas all over Central America, but persistently observed in livestock producing areas in Nicaragua and Honduras. Due to male migration in these areas, women are increasingly in charge of decision making and are responsible for deciding how investments of the remittances obtained from the migrant husband should be used, as well as, being responsible for the day-to-day decisions of farm management. For example, they increasingly make natural resource management decisions such as deciding whether or not to expand the agricultural frontier or adopt more environmentally friendly practices such as agroforestry systems.

Social capital: Many authors agree that, "Social capital may be one asset in which gender inequalities are not as pronounced, or where women even hold more and are at an advantage" (Quisumbing and Meinzen-Dick 2001). Women's advantage in terms of social capital is recognized by development programs for women, which often work through or involve existing women's groups and social networks providing women with services and increasing their access to and control over resources (land, water, livestock, and livestock products). Or, these programs may improve employment opportunities for women and, thus, go beyond providing direct benefits to become instruments for empowering women through their social capital (Dikito-Wachtmeister 2001).

In terms of women's social networks, a number of researchers have found that women often depend more on informal relations and, so, form stronger kinship and friendship relations than men, who tend to rely more on formal relationships (Westermann et al. 2005). Women's networks offer a mechanism to expand access to information and to mobilize community support for ecological practices and poverty reduction strategies. Historically, development agencies have focused on men rather than women, yet when women's networks support alternative practices, these practices are more likely to be employed in ways that result in real changes. Moreover, when women's groups are included, the incidence of conflict decreases. Inclusion of all groups within a community has been shown to be significant for the effectiveness of collective natural resource management. Westermann et al. (2005) found that collaboration, solidarity, and conflict resolution increase in natural resource management groups where women are present and that norms of reciprocity are more likely to operate in women's and mixed groups.

Political capital: In many communities, women lack access to formal power structures yet this is changing and, as a result, more women are entering these structures. Women's access to informal power structures, however, can determine the success or failure of projects related to sustainability and poverty reduction. Marginalization and discrimination are expressions of social exclusion, one of the multiple dimensions of poverty (Wagle 2002) and indicate a lack of political capital. Adato and Feldman (2001) point out that investments aimed at strengthening women's position within social units and empowering them as decision makers "have reduced inequality and improved well-being."

Research on the effects of ecological disasters (such as devastating forest fires) on rural communities showed that there is a direct relationship between a community with balance and synergy among the capitals and a functioning ecosystem that provides and supports the resources required for a healthy human community (Gutierrez-Montes 2005). In the current context, with the increasing likelihood of disasters and women's high vulnerability to disasters, the need to increase women's access and control over resources is critical. Akerkar (2008) analyzed the impacts of the 2004 tsunami that devastated communities throughout the Indian Ocean in terms of women's vulnerability and their access to resources and assets. The author found that women, already subject to existing institutionalized discrimination, were often prevented from participating in relief and recovery activities following the disaster, leaving them poorer and worse off (Akerkar 2008). Women's access to relief was highly constrained by various social and cultural factors. Widows, female-headed households, and migrants faced particular difficulties (2008). These groups of women are usually the ones with less voice within a community and, therefore, have fewer chances of negotiating access to resources. As Quisumbing and McClafferty (2006) emphasize, "gender considerations can affect allocation, targeting, and control of resources and, thus, policy and project outcomes."

Financial capital: Historically, development projects that include loans, grants, and access to agricultural inputs have focused on men. Nonetheless, when these resources are focused on women in the marketplace and women as producers, these resources have a stronger impact on family income and well-being. A similar situation is observed when resources related to ecological improvements focus exclusively on men, rather than on those whose everyday work interacts with the ecology. Under this scenario, we see less impact on ecological improvements. Studies all over the world have demonstrated the importance of property rights and control, not only over financial resources, but over land and the resources associated with the land, including water, forests, and biodiversity, as central in the improvement of women's access to credit, technical assistance, and information. This access to resources translates into women's well-being, but moreover into family, community, general society, and environmental well-being (Agarwal 1994; Chiappe and Flora 1998; Flora 2001; Fundación Arias Para la Paz y el Progreso Humano 2002).

The microfinance movement with a strong gender perspective has been proving all over the world its relevance to attain development goals and poverty reduction. Sharma (2001) points out: "Microfinance programs targeting women obviously have a strong potential to empower women whose daily lives are constrained by a pitiful lack of command over household and society resources."

Built capital: According to FIDA (2003), even though the lack of basic services and infrastructure (built capital) has an overwhelming effect on all rural inhabitants, women and girls are the ones who feel the most negative effects. As authors mention "despite the fact that women have been historically ignored by most of research and development programs," they "are vital to food security and family well-being and their need for labor saving and income generating technologies is acute" (Paris et al. 2001). The lack of access to basic resources such as water and fuel means that women and girls, who are responsible for the provision of these resources, must take charge of fetching water and wood for the household. These activities leave them little time for education and entrepreneurial activities that can improve their situation. When women participate in the planning of infrastructure such as water storage systems, they are likely to take over the responsibility of its maintenance and be very efficient in the administration of public goods.

Interestingly, women have been known to use technologies they have control over in innovative ways. Such is the case of telecommunications. With the advent of cellular phones, for example, there has been an increase in its use by women in the Andes to link their isolated rural dwellings to markets and other resources in rural or peri-urban areas where they can sell their products. Amaya (2008), in her study of access to markets for women in watershed communities in Bolivia, found that women participate significantly more in markets than men and that they made use of information tools and infrastructure to aid them to increasing their markets. Women use radio programs, social networks, and cell phones (their use has grown strongly in the last years) to access information, mostly about prices, volumes, and possible markets. Innovative use of technologies by women, make technology, not only information but also agricultural and postharvest technologies, as having "tremendous potential for enhancing women's welfare and their empowerment" and calling the attention towards "gender-sensitive participatory technology development" (Paris et al. 2001).

Summary and Implications for Ecological Management, Sustainability, and Poverty Reduction

The nature of natural resources management practices in many cultures requires an equally gendered approach to management and sustainability. Traditionally, development organizations have favored men, focusing their activities on financial and built capital and with detrimental and perceived loss of the social and natural capital. More recently, and with an explicit and clear gender perspective, environmental, development and educational organizations, as well as, micro-finance agencies base their interventions on human and social capital, and in some measure on political capital. According to Burchfield and colleagues (2002), women in Nepal who increased their literacy skills (human capital) become more active in community groups, social activities, and infrastructure-related activities. Women's groups that increased their literacy "have carried out community development projects such as repairing school buildings, installing water pipes and planting community forests."

Present ecological circumstances make women, who are already vulnerable due to existing social disabilities and discriminations, even more vulnerable and unable to counteract negative impacts on family, community, and society. If gender is considered critical to understanding current situations and options for change, then it is equally important to find ways to integrate that understanding into development and community change efforts and projects. Such projects adhere to three key principles:

1. Privilege local wisdom (tacit knowledge) over outside expert knowledge (explicit knowledge).
2. Focus on participatory methods that engage all those involved in ways that their voices are heard.
3. Support endogenous change that emerges from the people involved and includes a vision of the future and strategies to reach that future that are owned by the people impacted by the change effort.

Development approaches committed to these principles often invest current assets in social capital (links to outside knowledge, local networks, norms of trust and reciprocity) to engage community members in discussion and dialogue around assets, possibilities, and opportunities resulting in changes in cultural capital (local norms and beliefs) that empower them to identify resources and take action (political capital). Sustainable change comes when the community, both men and women and the networks in which they participate, is able to build its capacity to monitor and sustain the change effort. The process of community change often accompanies a redefinition of leadership. Emerging leadership structures tend to be more inclusive involving women and younger community members and more transparent in that decisions are shared with the community (see Chap. 3 this Volume).

Strategies that promote the engagement of women include:

1. Asset mapping that allows women and men to identify the assets in the community in each of the capitals, and based on that mapping, identify ways to invest these assets in sustainable and equitable change efforts.

2. Use of the four "D" cycle in Appreciative Inquiry (*discover* what is working; *dream* how it might work better; *design* how it will work better; and *deliver* the future) to facilitate discussion enabling community members (both men and women) to socially construct their vision of the future and their roles in that future.
3. Work with women's groups to discover, understand, and access existing networks. In some communities where gender differences are deeply engrained, participation may require working with men's and women's groups separately to encourage discussion and ensure that both voices are heard.
4. Insist on inclusionary processes. These processes tend to be more open to reflection and can lead to the transformation of organizations and practices. Groups that systematically learn from their experiences develop their capacity for sustainability. Inclusionary processes that revolve around a common vision or mission increase coherence and commitment among community members. Women are key users of natural resources and if they do not participate in planning how these resources are built upon or invested, then many development and/or conservation projects will continue to have little success. Women, for example, need to be further included in planning activities such as construction of water storage systems (built capital), given their proven capacity to take over responsibility of maintenance and efficient administration of public goods.
5. Support capacity building, particularly in human, social, political, and cultural capital by providing training programs that welcome women's participation in capacity building and project activities. Capacity building must also expand women's abilities to identify and access resources and to encourage mechanisms for reflection and assessment of what is working well and how it can work better. Use coaching and adult learning principles to engage women in discussions about how technology transfer and outside expertise can be successfully integrated into existing processes. These approaches provide opportunities for important discussions around balancing diversity with structure and process with action. Both coaching and adult learning approaches honor the wisdom of the learner and recognize that the ability to use new knowledge must be place based and shaped to local use.

Inattention to gender in the context of poverty reduction and natural resource management efforts limits and even prevents the possibility of sustainable change. Furthermore, the downward spiral that continually depletes natural resources as communities become poorer disproportionally impacts women and their children. By understanding the links among the social construction of gender, the nature of local access and decision making over different assets or capitals, the impact of environmental change on key assets, and the potential for poverty reduction, development efforts can support strategies to reverse the downward spiral. Strategies that help women to expand their role in the conservation of resources are critical. Successful development of these strategies requires the participation of local women in ways that build ownership and mobilize women to take action; action that will support their family and the environment at the same time.

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Chapter 5
Introduction: Population, Poverty, and Ecology

Alex de Sherbinin

This section includes chapters addressing population growth, migration, and urbanization as they relate to the ecology–poverty nexus. The work in the field of population–environment studies has been the province of demographers, geographers, sociologists, economists, and, perhaps pre-eminently, ecologists, through the seminal works of Duncan (1964), Hardin (1968), and Ehrlich and Holdren (1971). In the ecological contributions to this literature, population size, density, growth, and re-distribution (in the form of urbanization) are often presented as primary drivers of environmental problems, and the solution proposed is to reduce or reverse growth rates, or to set aside ecologically sensitive areas in parks. In the extreme, Garrett Hardin (1974a, b) proposed that poverty be allowed to run its course, unfettered by foreign aid, so that rapidly growing developing countries would better experience the "positive checks" on population growth of famine, misery, plague, and war postulated by Malthus 200 years ago. This he termed "life boat ethics."

While recognizing that Hardin hardly represents the mainstream of ecological thought, there remains an undercurrent of anti-population rhetoric in the environmental literature, with humans variously characterized as parasites or a spreading cancer, and population growth as a ticking time bomb. By contrast, these chapters by demographers and geographers seek, each in their own way, to illuminate the complexity and the context-specific nature of the links between population, poverty, and ecological problems, and to dispel some of the myths relating to population–environment interactions. In so doing, they direct scholarly inquiry away from hand-wringing about the negative impacts of population dynamics upon ecosystems, and instead present a more nuanced picture of the positive and negative aspects of

A. de Sherbinin (✉)
Center for International Earth Science Information Network (CIESIN),
The Earth Institute at Columbia University, New York, NY, USA
e-mail: adesherbinin@ciesin.columbia.edu

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human–environment interactions and what these interactions mean for poverty reduction. The authors recognize that we have entered the anthropocene – the age of human domination of the Earth's systems – and that the challenge for this generation is how to manage the environment in ways that both humans and ecosystems not only survive but hopefully prosper. Engaging in this management task means understanding how a variety of population dynamics – growth, migration, and urbanization – are related to poverty and the possibilities for ecological stewardship.

The first chapter by Jason Bremner, Jason Davis, and David Carr focuses on population growth and poverty and their relationship to terrestrial ecology (through land cover change), marine systems, and freshwater ecosystems. This chapter provides a valuable overview of population–environment theory, since theory both informs the research in this area, but also tends to direct attention to policy options for addressing environmental problems. The authors emphasize that because of the complex relationships among population dynamics, household poverty levels, and environmental change, disentangling the impacts can be difficult – though not impossible. The chapter points out that there has been an imbalance in the research, in that the impacts of population dynamics on forest ecosystems have been heavily studied, while relatively less attention has been given to marine and freshwater ecosystems.

The second chapter, by Susana Adamo and Sara Curran, addresses migration, suggesting that it can be a mediating factor facilitating or constraining the effectiveness of ecology for poverty reduction. The authors point out that migration relates to the environment in two ways: environmental changes, whether short-term disasters or longer term degradation, may instigate out-migration from some regions, while in-migration in other regions results in larger settlements and altered environments. Migration is also vital to poverty reduction, since people have traditionally moved to improve their well-being. Finally, remittances sent by migrants can improve well-being in home communities – though these communities may still be deprived of their most able-bodied or talented individuals. The authors adopt a migration systems perspective – one which incorporates settlements and flows in a holistic view – as a way of presenting the three-way linkages between poverty, ecology, and migration, and understand the regulatory mechanisms.

The last chapter in this section, by Peter Marcotullio, Sandra Baptista, and Alex de Sherbinin, focuses on urbanization, poverty reduction, and ecosystem integrity. Urbanization has traditionally been associated with the environmental and health impacts from polluted air and water, inadequate waste management and sanitation, toxic releases, and industrial brown fields, and generally not with the biodiversity, conservation, and energy savings traditionally embraced by ecology. Moreover, given the rapid expansion of slum settlements in and around developing world cities, decision makers are increasingly focusing attempts on limiting urbanization and urban growth as a way to improve their cities' chances for investment and economic development in the global economy. This chapter seeks to demonstrate that far from being monolithically bad for the environment and wealth generation, urbanization is perhaps the best chance that humanity has of concentrating productive power, alleviating poverty and reducing the impact of consumptive activities while sparing larger areas for conservation and sustainable management. The paper provides empirical evidence that rather than being unambiguously bad for the environment, cities have actually had beneficial impacts on their immediate environment and over larger regions. Furthermore, the chapter provides examples of actions at the local level that have addressed poverty through maintenance and enhancement of local ecosystem services. At the same time, however, these sanguine views are balanced by a recognition that many of today's developing country cities are under-resourced, in dire need of capacity building, and struggling to cope with the scale of contemporary population growth.

Across all chapters, there is recognition that spatial scale matters (i.e., not all findings pertain across all scales) and that markets and governance systems are critically important for ensuring equitable and ecologically positive outcomes. The authors do not deny in any way the central importance of population in the mix, but suggest ways that population dynamics can be "reimagined" so as to make the prospect of an urban, more populous, and increasingly mobile world less threatening and the possibilities of positive outcomes for the environment more plausible.

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Chapter 6
Population Growth, Ecology, and Poverty

Jason Bremner, Jason Davis, and David Carr

Introduction

The world's population of nearly one billion in 1800 has grown to approximately 6.9 billion today, and population projections suggest that the world population will fall somewhere between 8 and 10.5 billion by 2050, depending on changes in national level fertility and mortality rates (UNPD 2010). Nearly all of the world's net population growth over the coming 40 years will occur in cities in less developed countries.

At the same time, the ecosystems that support people's livelihoods and well-being are being rapidly degraded. The Millennium Ecosystem Assessment examined 24 critical ecosystem services upon which humans depend for their well-being and found that 60% were being degraded or used unsustainably (2005). The impacts of degraded ecosystem services are being disproportionately borne by the poor, are a principal factor contributing to poverty, and are a barrier to achieving the Millennium Development Goals (MEA 2005). Population growth is identified as one of the key indirect drivers of the degradation of these ecosystem services. Population growth itself, however, remains an insufficient explanation of relations between population, ecosystems, and poverty. Changes in population composition and population distribution also have important impacts on ecosystems. For example, models show that the aging of populations over the next several decades could result in significant changes in carbon dioxide emissions even in the absence of any technological change (Dalton et al. 2008). In some contexts, the number of households in a society is as important as population size in determining a population's impact on ecosystems (Liu et al. 2003).

J. Bremner (✉)
Population Reference Bureau, Washington, DC, USA
e-mail: jbremner@prb.org

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Nonetheless, current trends in population growth and ecosystem health suggest a challenging future for the world's poorest. More than 1.4 billion people live in extreme poverty (less than US $1 per day) (Chen and Ravillion 2008), and many of them depend on degraded ecosystems. Furthermore, the poor are more vulnerable to further declines in ecosystem services (MEA 2005). The goal of this chapter then is to further describe the complex relationships between human population growth, ecosystems, and poverty. The chapter begins with a discussion of several theories on the relationships among population growth, ecosystems, and the impact on human well-being. Poverty is then discussed in more detail as both a contributing factor to and consequence of population growth and environmental change. Empirical examples related to land-cover change, water, energy resources, and climate change are examined to illustrate the complexity and diversity of these dynamics. The chapter concludes with a brief discussion of the limitations of current knowledge on the links among population growth, ecosystems, and poverty and the implications for future research and policy.

Theory: Population Growth, Ecosystems, and Poverty

The connections that bind human and natural systems are innumerable, but arguably, one of the most discussed through human history has been the ever increasing size of the human population and its relationship with the natural resources upon which it depends. Modern theories on the association between population growth and the environment date to 1798, with Thomas Malthus's statement that, "The power of population is indefinitely greater than the power in the earth to produce subsistence for man," (Malthus 1986).

Malthus envisioned an impending doomsday scenario where excessive human population growth would overtax a limited supply of natural resources (Malthus 1986). He argued that agricultural production grows geometrically and arable land is finite while population growth is exponential. He hypothesized that as human numbers grew, food supplies would be insufficient to feed humankind and human numbers would be pushed back below the carrying capacity of agricultural systems by "positive and preventative checks." Positive checks would encompass increases in mortality due to outbreaks of disease, famine, higher infant mortality, malnutrition, and war. Preventative checks would include lowering of fertility through delays in marriage, contraception, abortion, and infanticide.

The agronomist Ester Boserup countered Malthus' contentions and described an alternate response of humans and their agricultural systems to increasing population growth (Boserup 1965, 1981, 1990). Boserup argued that humans would respond to the food demands of a growing population by intensifying land use, increasing agricultural yields, and developing new agricultural technologies. Examples of agricultural intensification include multi-cropping, increased labor to land ratios, and the development and use of better tools, irrigation systems, and soil amendments. Boserup thus argued that there are no limits to human population growth assuming sufficient changes in agricultural systems. Boserup, however, largely overlooked Ricardo's law of diminishing returns, did not discuss poverty as a barrier to intensification, and ignored other natural systems such as forests, oceans, rangelands, and freshwater ecosystems upon which humans depend.

The idea of a multiphasic response, developed by Davis (1963) and later adapted by Bilsborrow and Ogendo (1992), describes demographic and economic responses communities and households take to maintain their standard of living as increasing population growth and population density result in land scarcity. The theory suggests that agrarian heads of household, unwilling to decrease their standard of living, make multiple and phased behavioral changes to land use, livelihoods, mobility, and fertility.

Bilsborrow and Ogendo (1992) theorize that in the first phase of response, idle agricultural lands in a community are distributed to new households, and when no additional lands are available, existing lands and land tenure arrangements begin to fragment as lands in a community and rights to use resources are parceled to married children as dowry and inheritance. In a second phase, as households become increasingly reluctant to decrease their standard of living and further divide lands to their married children, young adults seek new lands beyond the control of their community, either by claiming nearby lands, which often results in conflict, or by out-migration to seek land along expanding frontiers. At the same time, adults may seek wage employment to compensate for limited agricultural prospects or may intensify their use of existing parcels of land with labor, technology, and fertilizers. In the final phase, both heads of household and young adults may make behavioral changes that decrease fertility. Couples, may take measures to reduce fertility including using traditional and modern contraceptive methods. Furthermore, young adults may delay onset of marriage as they decide to out-migrate or seek education to increase their prospects for wage employment, both of which tend to decrease fertility. The theory of multiphasic response was an important advance from Malthus and Boserup because it better reflects the complexity of pressures on households' standard of living and the multiple responses that households might employ to maintain those standards.

Theoretical Underpinnings of Micro-level Population–Poverty–Environment Relationships

Recent research on demographics, livelihoods, and the environment has suggested the use of a livelihoods approach as an organizing framework to examine population–environment relationships. The livelihoods framework characterizes households as making decisions regarding livelihood activities based on available natural, social, human, physical, and financial capital (Ellis 2000). The examination of different types of capital allows for a more complete understanding of population, poverty, and environment relationships. DeSherbinin et al. (2008) have suggested that the livelihoods framework be used to assess a vicious circle model (VCM) of population,

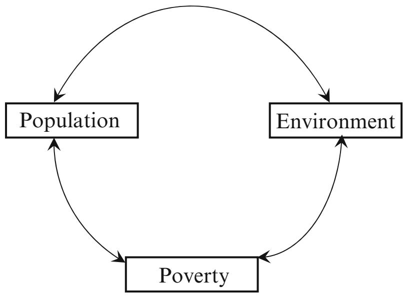

Fig. 6.1 The vicious circle model

poverty, and environment. According to the VCM, positive feedbacks at the household level among population growth, poverty, and environmental degradation lead to a downward spiral for poor households. The VCM concept of multiple feedbacks is useful and encourages examination of not just how population growth impacts on the environment, but also how population growth affects poverty, poverty affects population growth, poverty affects environmental degradation, and environmental degradation affects poverty (Fig. 6.1).

In many instances, fertility may contribute to the poverty of households through several mechanisms: health and educational needs of large numbers of children generally reduce household savings rates and reduce investments in production activities; high fertility lowers female labor force participation and thus tends to decrease household income; and finally, population growth due to high fertility may exacerbate resource pressures in areas where a large proportion of the population already relies on natural resource-based livelihoods, including agriculture, grazing, forest products, and fishing for income and subsistence on marginal lands and less productive natural ecosystems (MEA 2005).

Poverty may also limit the responses that households have to environmental change (Carr 2008). Impoverished households may be less likely to have adequate land resources to parcel to offspring, and have fewer resources to be able to obtain new land. They may also have little access to the financial capital necessary to intensify resource use through technological or physical inputs, or invest in new agricultural products in response to changing markets. Finally, poor households are less likely to have the financial and human capital necessary to migrate elsewhere in search of land or employment in response to limited local opportunity.

Poverty may contribute to higher fertility as well (Birdsall et al. 2001). Infant mortality in poor households tends to be higher than national averages meaning that poor families may perceive the need to have more births in order to achieve desired family size (Palloni and Rafalimanana 1999). Furthermore, women in poor households are less likely to have knowledge of and access to means of preventing unwanted pregnancies (Dreze and Murthi 2001). Finally, young women from poor households are more likely to marry early and have less education, both of which are associated with higher fertility in most contexts (Davis 1963).

In all of the relationships discussed above, attention to spatial and temporal scale is important, as relationships often do not hold across changing scale. For example, numerous multi-country studies have professed a strong connection between population growth, deforestation, and other forms of land-cover change at the national level (Amelung and Diehl 1992; Rudel and Roper 1997), but when viewed at finer scales (e.g. regional, community, and household levels), fewer studies have identified a strong linkage (Rindfuss et al. 2004; Carr 2002). Temporal factors may also result in mixed findings as evidence of changes in ecosystems may take years or decades to develop while demographic events are usually quickly apparent.

Underlying political, economical, and institutional factors may also contribute to population growth, poverty, and environment relationships. While proximate population growth can drive the expansion of natural resource extraction from local ecosystems, multifaceted underlying factors that exert their influence may actually be significant drivers of ecosystem degradation (Geist and Lambin 2002). For example, a frontier farmer may expand his agricultural holdings deeper into primary forests to properly provide for his family, but outside factors such as displacement from other regions, facilitating land distribution policies (or lack thereof), improved access due to expanding roads, and global demand for agricultural and forest resources also underlie farmers' decisions (Geist and Lambin 2001; Carr 2004, 2006). All together, the theories and challenges suggest few simple statements regarding population growth, poverty, and environment relationships. Understanding linked human–natural systems demands specific knowledge of local patterns, processes, and underlying factors.

Empirical Observations of Population Growth, Poverty, and Environment Interactions

Population Growth, Poverty, and Land-Cover Change

The shift of hunter–gatherers to agriculture launched a cycle of change that has had the largest impact on land cover in the history of humankind (Turner et al. 1990; Myers 1991; Parsons 1994; Foley et al. 2005; Davis 2006). Today, more than 40% of the world's surface is under agriculture (Ramankatty and Foley 1999), and forest clearing for agricultural expansion in the tropics is currently the most significant land conversion happening on Earth (Geist and Lambin 2002). Population growth is generally recognized as an important contributing factor to land-cover change though the importance of this relationship has been the subject of some debate (Houghton 1991; Myers 1991; Vanclay 1993; Wibowo and Byron 1999). Some have declared population growth and poverty to be the primary causes of global deforestation (Allen and Barnes 1985; Amelung and Diehl 1992; Mather and Needle 2000), while others recognize population growth and poverty as just underlying factors (Rudel and Roper 1996; Lambin et al. 2001; Geist and Lambin 2002). In a review of 152 case studies of tropical deforestation, Geist and Lambin (2002) report that population growth is just one of numerous factors that act synergistically to cause tropical deforestation. Population growth and poverty interact with a host of economic, environmental, political, and sociological factors to effect land-cover change (Lambin et al. 2001; Turner et al. 2001). Economic inducements include the basic desire for products of consumption (timber, fuelwood, and agricultural products), but they also include market failures, the desire of national governments to generate capital, and the lack of economic disincentives to prevent deforestation (cheap land, labor, and fuel). Other factors of import include political inducements to colonize forested lands and cultural factors that deemphasize the intrinsic value of these habitats (Geist and Lambin 2002).

A number of case studies of tropical deforestation in the Amazon have examined population, poverty, and land-cover change relationships. In most of these cases, the localized population growth has been due to migration of colonists from other regions of the country. For example, in the 1970s, the Brazilian government offered large tracts of land in the Amazon Basin to inner-city poor and facilitated their migration to the Amazon as a means to reduce land pressure and growing political unrest (Eakin 1998). This colonization led to large increases in slash-and-burn agriculture and eventual consolidation of large tracts of once biologically rich land into cattle pasture and soy farms. Similarly, in Ecuador, land inequality and land pressure in the highlands led to government policies encouraging migration to the less populated Amazon, resulting in large scale deforestation (Pichón 1992). High fertility and resultant population growth were not the local proximate drivers of land-cover change, but in origin areas, they did contribute to land pressures, poverty, and political unrest that contributed to out-migration to the Amazon. Locally, the contribution of high fertility to population growth and deforestation in the Amazon has been more difficult to ascertain.

At the household level, positive correlations between high fertility, poverty, and deforestation are often assumed, but the relationships have not been found to be so clear-cut (DeSherbinin et al. 2007). In the Ecuadorian Amazon, lower fertility among colonist households was associated with larger plots of cleared land, secure tenure, and more wealth (Carr et al. 2006a), while in other settings, negative and neutral associations between household fertility, poverty, and land holdings are observed (Carr et al. 2006b). In most cases, these mixed results don't seem to support the VCM's predictions of positive feedbacks leading to spiraling poverty and deforestation, but rather indicate the complexities of local context in determining population, poverty, and environment relationships.

In spite of these mixed findings, Demographic and Health Surveys in the region do reveal high rates of fertility throughout the remote rural Amazon (Bremner and Dorelien 2008), and it is likely that high fertility in the Amazon will contribute to deforestation for years to come as the children of colonists create new households, clear land, and migrate to new areas of the frontier (Barbieri and Carr 2005). Few studies have looked at how fertility and the migration of children are related to local land availability or perceptions of land availability, and this represents one logical next step in understanding relationships between fertility, poverty, and the environment in this context (de Sherbinin et al. 2008). In addition, indigenous populations manage 25% of the remaining forests of the Amazon and are characterized by high fertility and extreme poverty. Relatively little is known about indigenous resource management institutions and their resilience to population growth and poverty (Bremner and Lu 2006). Future research may fruitfully explore relationships between land use and other natural resources among these important and rapidly changing populations (Bremner et al. 2009; Gray et al. 2008).

Population Growth, Poverty, and Coastal and Marine Environments

Coastal and marine resources are not immune to human pressures and concomitant poverty. Coastal ecosystems and coastal cities are often destination areas for migrants, and many of the world's largest cities are located along coasts. In this context, global fisheries data indicate a decline in fisheries and a trend towards fishing down food webs or shifting from species at high trophic levels to low trophic levels (Pauly et al. 1998). Thus, it is unsurprising that migration and population growth are often cited as causes of coastal resource and fisheries degradation (Curran et al. 2002). Coastal population growth, poverty, and environment relationships differ markedly, however, from the land-use relationships, principally because coastal resources tend to be common pool resources (Curran and Agardy 2002). Pauly (1997) describes the population, poverty, and environmental change responses occurring in many parts of inland Africa, Asia, and South America where high population density in upland areas creates landlessness and out-migration. He cites examples of Filipino rice farmers, Peruvian Highlanders, and Senegalese pastoralists who, due to a lack of land or pasture access, are pushed to coasts where fishing is unrestricted in order to meet subsistence needs.

Increases in the number of resource users along coasts can have varied impacts on common pool resources, and the relationship between population and coastal ecosystems is largely mediated by the institutions (either public or common property institutions) and social relations that govern local resource use (Curran and Agardy 2002). In areas in which institutions are weak, an increase in the number of users or new fishing techniques introduced by migrants may lead directly to degradation of coastal resources through overharvesting (Cassels et al. 2005). In areas where institutions do play an active role in resource management, an increased number of users may nevertheless result in decreased per capita income and a situation where institutions and social bonds break down as some users employ deleterious fishing methods to maintain their existing income (Pauly et al. 1989; Curran and Agardy 2002). Even in areas where institutions governing coastal resources are strong, such as in marine protected areas, a growing number of resource users may result in a growing constituency demanding changes to existing regulations. In the Galapagos, for example, the dramatic increase in the number of fisherman relying on lobster and sea cucumber fisheries, and their growing political presence, as witnessed through regular protests, influenced catch quotas, season duration, and changes in protected area staff during the 1980s and 1990s (Bremner and Perez 2002).

Conversely, in Mexico's Sian Ka'an Biosphere Reserve, the need to protect lobster harvest sustainability was cited as a primary reason for nearly 100% contraceptive prevalence rates, suggesting an understanding of the impacts of human fertility on marine resources and local livelihoods (Carr 2007).

A growing coastal population may also have indirect impacts on coastal resources through land-cover change for development, increased sediment and pollutant runoff, and increased market demand for coastal resources (Roberts 1993). Many of these impacts will be on local resources, but growing demand for fisheries resources due to population growth and consumption preferences in distant markets can drive local resource impacts as well (Curran et al. 2002).

Population Growth, Poverty, and Freshwater

In the twentieth century, global water consumption grew sixfold—twice the rate of population growth during the same period (WMO 1997). Much of the increase in human water consumption was made possible through construction of dams and reservoirs, affecting nearly 60% of the world's major river basins (Revenga 2001). Water is used for a plethora of human needs including: direct consumption, household uses, crop irrigation, transport of human sewage, hydroelectric energy, the production of aquatic food resources, and manufacturing. Securing access to clean water is a key aspect of development for the world's poorest countries, and the Millennium Development Goals (MDGs) set the challenge of halving by 2015 the proportion of people without sustainable access to safe drinking water and basic sanitation. This access is vital in the prevention of diarrheal diseases, which account for 1.5 million deaths annually, the majority among children less than 5 years of age (Prüss-Üstün and Corvalán 2006). In areas with little access to clean water and sanitation facilities, improving access can be among the most-cost effective means of reducing morbidity and mortality (World Bank 2006) (see Chapters 6, 7, 8 and 9, Vol.1, on water resource management challenges).

One of the great challenges to meeting the growing water demands of the world's population is that freshwater is distributed unevenly across the world's surface. The world's arid regions, for example, only receive 2% of the world's rainfall despite covering 40% of the world's surface (Revenga 2001). This fact coupled with projected population increases augurs poorly for fresh water ecosystems and human well-being. Research on population distribution and water scarcity indicates that 2.3 billion people live in "stressed" water basins—areas with per capita water supply of less than 1,700 m^3/year, and that 3.5 billion people will live in stressed water basins by 2025 (Revenga 2001). Furthermore, many of the world's countries with the poorest access to clean water and sanitation are among those with the fastest growing populations, and several are expected to double their populations in the next 20–30 years (PRB 2008) (Table 6.1).

These aggregate indicators, however, tell us little about population growth, poverty, and water relationships at the household level, and unlike land-use research,

Table 6.1 Population size and projected growth in ten countries with the worst access to improved water sources

Country	Population w/improved water source (%) 2006	Population (millions) Mid-2008	Mid-2025
Afghanistan	22	32.7	50.3
Somalia	29	9.0	14.3
Papua New Guinea	40	6.5	8.6
Ethiopia	42	79.1	110.5
Mozambique	42	20.4	27.5
Niger	42	14.7	26.3
Equatorial Guinea	43	0.6	0.9
Congo, Dem. Rep.	46	66.5	109.7
Fiji	47	0.9	0.9
Madagascar	47	18.9	28.0
Nigeria	47	148.1	205.4

Source: Population Reference Bureau (2008). World Population Data Sheet

there are few studies that examined these local-level relationships. Large parts of the Sahel region of sub-Saharan Africa, including Niger, which has the highest fertility rate in the world, have endured long periods of drought with subsequent losses of major crops systems and declines in livestock, resulting in declining food security and chronic and acute malnutrition (Batterbury and Warren 2001). A few studies have looked at the relationship between access to water, labor requirements for women and girls, and fertility and this topic could be a fertile research avenue in the coming years.

As with coastal resources, freshwater resources are common pool resources and factors such as institutions and social relations that govern these resources similarly mediate population growth and poverty relationships. For instance, research indicates that in areas where women and girls are responsible for obtaining water, increases in the time it takes to obtain water negatively impact female labor force participation (Ilahi 2000) and girl's education (Levine et al. 2008), both of which are related with early onset of childbearing, high fertility, and poorer maternal and child health outcomes.

Freshwater is also vital to natural ecosystems, but humans are appropriating an increasing portion of it, primarily for agricultural irrigation (Postel et al. 1996), and in many areas, natural ecosystems are no longer receiving sufficient water supply to maintain them. Barring major changes in water use, these diversions are likely to increase. In both developed and developing countries, new diversion projects will provide water and food for some people while negatively affecting the populations and ecosystems that rely directly on downstream freshwater ecosystems for their health and livelihoods. How these trade-offs are negotiated and to whom the deleterious repercussions fall will be increasingly pertinent questions as pressures inherent in the demographic-ecological-development nexus unfold in the coming decades.

Conclusions

In this chapter, we have tried to illustrate the complexities of population growth, poverty, and environment relationships. Understanding of these relationships has progressed greatly from the original Malthusian roots; yet even today, few generalizations can be made unambiguously and VCM scenarios of downward spiraling poverty, population growth, and environmental degradation appear over-simplified given the complexity of empirical cases. Research has demonstrated across multiple scales that population–environment–poverty dynamics tend to be non-linear, ecosystem specific, and involve multiple pathways among population and environmental change, population and poverty, and poverty and environmental change. Furthermore, in most cases, population growth's relationship to the environment is mediated by various types of capital available to households and institutions, culture, and social relations.

In general, however, poverty, both as a result and as a contributing factor to population and environment relationships, has only recently been systematically addressed in the population–environment literature. De Sherbinin et al. (2008) opine that the livelihoods framework is a good starting point for incorporating poverty into population–environment research and for organizing disparate conceptual frameworks (also see Chap. 4 for a further discussion of livelihoods frameworks). Research using the livelihoods framework already suggests that livelihood and demographic decisions are interlinked with households managing risk through livelihood diversification and migration and responding to culture-specific norms regarding appropriate and desirable activities and demographic responses (de Sherbinin et al. 2008).

Community development efforts, however, still remain largely sector specific along lines of livelihoods, biodiversity conservation, and health interventions. Greater attention to local population, poverty, and environment relationships would likely improve community development efforts and ensure that interventions would decrease vulnerability to poverty while improving people's health and protecting local ecosystems. Over the last decade several conservation and development organizations have experimented with integrated community development approaches that seek to address diverse priorities related to population growth, reproductive health, environmental change, and livelihoods in areas of high biodiversity. These integrated projects (often termed population, health, and environment projects or PHE projects) are reaching underserved and impoverished populations that are highly dependent upon natural resources in priority conservation areas (D'Agnes and Margolius 2007). These projects have proven successful in extending health services to remote communities and setting up conditions for sustainable use of local resources (Pielmeier et al. 2007). The efforts, however, still remain largely unproven over the long term in terms of improving livelihoods and maintaining ecosystems. Integrated PHE projects have also proven challenging due to the additional capacities organizations need to implement integrated projects.

Integrated PHE approaches depend greatly on a detailed understanding of local population, poverty, and environment relationships; therefore, there is a need for interdisciplinary research teams to diversify the ecosystems, geographies, and social systems studied in population–environment research. While there has been a great deal of household level population and land-use research in tropical forest settings, understanding of population and poverty dynamics in dryland, coastal, and freshwater ecosystems remains inchoate. In particular, relationships among population growth, common pool resources, and common property institutions require further study in both aquatic and terrestrial ecosystems.

Despite research findings and conceptual changes, Neo-Malthusian perspectives and linear associations still largely dominate public dialogue and professional development narratives concerning population growth, environmental change, and concomitant poverty. This gap between research findings and public knowledge suggests the need for strengthened efforts in communication of research to policy-makers and the public. These communication efforts will ensure future support for development policies and funding priorities focused on integrated development approaches and interdisciplinary research.

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Chapter 7
Alliances, Conflicts, and Mediations: The Role of Population Mobility in the Integration of Ecology into Poverty Reduction

Susana B. Adamo and Sara R. Curran

Introduction

> For among three elements, each one operates as an intermediary between the other two, exhibiting the twofold function of such an organ, which is to unite and to separate (Simmel 2006:367)

The poverty–environment–migration triad sustains a complex relationship characterized by contradictory and ambiguous effects and multiple feedbacks (Locke et al. 2000; Adger et al. 2002). Population mobility[1] may both directly or indirectly contribute to poverty reduction (e.g., through remittances, diversification of livelihoods, improved living conditions, access to social and other services, and expansion of networks), or to the exacerbation of poverty (e.g., accelerate aging in sending communities, brain drain, community weakening, impoverishment of displaced populations, or raising unemployment in receiving communities). Similarly, migration may have environmental impacts (frontier settlements change land use patterns; growing population density in sensitive and/or hazardous areas exacerbates ecological deterioration; depopulation facilitates forest re-growth; remittances may accelerate adoption of green technologies) and in turn migration can be a demographic response to environmental change and deterioration, as in the case of environmentally induced displacement. Finally, migration may be a mediating factor between poverty reduction and ecosystem health – either facilitating a positive feedback (e.g., remittances may lessen the income demands for resource extraction in origin communities) or exacerbating a negative feedback relationship (e.g., remittances may provide the technological investments to accelerate resource extraction).

[1] In this paper, we use the terms population mobility and migration interchangeably.

S.B. Adamo (✉)
Center for International Earth Science Information Network (CIESIN), The Earth Institute at Columbia University, New York, NY, USA
e-mail: sadamo@ciesin.columbia.edu

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In this chapter, we examine how population mobility may be implicated in the ecology-poverty relationship using a migration systems perspective. Doing so provides a holistic view of population mobility from the basic components of settlements and flows to the contexts shaping these components – such as households' and communities' characteristics – and the regulatory institutions in the realms of policy, social networks, markets, governance, and culture. This perspective emphasizes that movements are not automatic, drawing attention to the regulatory mechanisms that affect the initiation and perpetuation of population movements as well as the selection of specific destinations (Faist 2000).

Migration Systems: Settlements, Flows, Contexts, and Regulation

> Like many birds, but unlike most other animals, humans are a migratory species. Indeed, migration is as old as humanity itself (Massey et al. 1999:1)

Population mobility is a multi-dimensional and multi-faceted phenomenon, with multiple levels of analysis (individual, household, community, nation, and globe), embedded in social and other contexts and further defined by temporal and spatial dimensions (Massey et al. 1993; Portes 1997; Brettell and Hollifield 2000; Faist 2000; White and Lindstrom 2005). The deceiving simplicity of early push–pull models masks a richness of reciprocal effects, multiple causation, and rapid changes that demand a dynamic conceptual perspective to understanding migration at various scales (Kritz and Zlotnik 1992). A systems approach provides such a dynamic perspective, considering population mobility as a lively, constantly evolving process (Boyd 1989; Fawcett 1989; Kritz and Zlotnik 1992; Faist 2000).

Through a migration system approach, the traditional linear view of movement as independent (individual decision making), singular (once-in-a-lifetime event), and unidirectional (from origin to destination) phenomenon is incorporated into a much broader perspective where movement is conceptualized as circular and interdependent, leading to 'self-modifying' systems where changes in one part affect the whole system (Faist 2000:51). Both migrants and non-migrants in each location (sending and receiving areas) are included in the analysis because both are part of a migrant, family, social, and other networks. They are nodes interconnected by different flows. And, it is the migrant nodes and ties that knit the migration system together.

Migration systems are basically composed of settlements and flows: population mobility involves a place of origin, a place of destination, and a flow between them (Lee 1966). Movements can be long or short term, long or short distance, can cross international borders, or stay within the same country. These characteristics define different types of mobility: internal, international, permanent, and circular. According to the type of settlement in origin and destination, basic categories of flows are rural–urban, rural–rural, urban–urban, and urban–rural. Large migrations flows are behind the urbanization of the world since the industrial revolution, with urban–urban flows becoming more common as developing countries undergo the urban transition. In addition to ‘people’ links, other links related to migration exist between places, the most common example being remittances. Linkages evolve over time, influenced by feedback mechanisms between places, among flows, and with the contextual macro-factors (Kritz and Zlotnik 1992). For example, the World Bank forecasts a decline between 5% and 8% for remittances flows due to the effects of the global financial and economic crisis (Ratha and Mohapatra 2009).

Migration is a highly selective process. The main selection variable is age (Rogers et al. 2002). Young adults (roughly 15–25 years old) are far more likely to move than individuals in other age groups. In terms of gender, overall men are slightly more likely to leave, but the participation of single young women has increased steadily in more recent decades (Deshingkar and Grimm 2005; United Nations. DESA. Population Division 2008a, 2009). In general, women tend to favor urban destinations, while men are usually the majority in flows to rural areas.

Education attainment is another important selection variable. More educated people and those with “marketable” skills or professions – which depend on and vary by labor market demands in destination – are more mobile. In recent decades, skilled and high skilled migration – the so-called ‘brain drain’ and ‘brain gain’ – is on the rise: in 2000, about 20 million migrants aged 25 and older with tertiary education were living in OECD countries, compared with 12 million in 1990 (United Nations. DESA. Population Division n/d). In 1990, 50% of these high-skilled migrants came from non-OECD countries, representing 57% in 2000 and were expected to reach 62% in 2007 (Docquier and Marfouk 2004; Dunnewijk 2008).

The contexts within which individuals are embedded are also fundamental components of migration systems. Settlements and flows are affected by multiple levels of embedding contexts – from the individual to the global level, including households, community, national, political, social, economic, and cultural conditions in both origin and destination. Households in particular have a triple role, as units that (a) sustain and socialize individuals, controlling and regulating resources and cultural values; (b) link to kinship networks and provide the social glue that binds geographically dispersed people; and (c) make strategic decisions about who stays and leaves from a household (Boyd 1989:643; Faist 2000).

The local context in origin areas may constrain or facilitate migration in several ways, through the local structure of opportunities (e.g., the characteristics of labor and financial markets or access to land) and by providing norms and values about migration. Is it an accepted or encouraged behavior? Or are migrants considered outcasts? (Hugo 1981). Over time, a reciprocal effect of migration can develop in the sending community, whereby migration changes the social, political, and economic conditions (Massey 1990b; Taylor and Wyatt 1996). Similarly, macro social and economic conditions in receiving areas – for example, exchange currency rates, or labor market conditions – influence migration decisions: to leave, to stay, to return (Parrado and Cerrutti 2003).

A final component of migration systems is the regulatory institutions, both formal (legal or contractual) and informal, that are found in markets, governance and policy (national and global), social networks, relationships, and cultures shaping norms, expectations, and traditions (Hugo 1981; Massey 1990a; Massey et al. 1993;

Roberts et al. 1999; Faist 2000). In this globalized era of increasing connection and mobility – when goods, capital, and services circulate with increasing ease – the formal regulation of international migration flows fluctuates between openness, selection, and restriction (Freeman 1995; Hollifield 2004). Internal migration can also be formally regulated, as in the case of China's household registration system (hukou) (Chan et al. 1999).

In terms of informal regulations, migrant networks connect migrants (permanent, long term, temporary, and return) and non-migrants across space and time, in origin and destination. These networks contribute to the self-sustenance of the movement through the interchange of information, assistance and obligations, and to the self-perpetuation of migration beyond the permanence of the conditions that originated it (Massey et al. 1987; Boyd 1989; Kritz and Zlotnik 1992; Roberts et al. 1999). By lowering the economic and emotional costs associated with migration and reducing the risks, networks also increase the pool of potential migrants beyond the classical filters of age, education, or skills. However, migrant networks seem to work differently in international and internal systems, and for male and female migrants (Curran and Rivero-Fuentes 2003; Fussell and Massey 2004).[2]

Poverty and Migration

It is estimated that between 1995 and 2005, international migrants increased from 165 to 190.6 million people, in both dates representing 2.9% of the world population (U.N. DESA. Population Division 2009). Current international migration systems (which include different types of mobility, i.e. long- and short-term mobility, circular migration, etc.) cluster around five major destination regions: North America, Western Europe, Asia and the Pacific, the Gulf region, and the Southern Cone of South America (Massey et al. 1999). Around half of the international migration flows are between developing countries, particularly among countries that share a border (Ratha and Shaw 2007).

Internal migration is assumed to involve a much larger number of people – for example, 309 million in India and 140 million in China according to last censuses – although it is almost impossible to find reliable world estimates, mainly because of comparability issues across countries. Internal rural–urban flows make a significant part of the accelerated urban transition in the developing world, particularly in Asia and Africa (UNFPA 2007), although a slowing trend has been recently detected in the case of Africa (Potts 2009). Rural–rural movements are also relevant, although less

[2] For example, and for the case of Mexican migration systems, the cited studies found that: male migrant networks have a significant effect on male migration to the United States but not on female migration; female migrant networks increase female migration, particularly within Mexico, but decrease the odds of male migration; and that the effect of social networks on first migration to the United States is stronger on rural sending communities than on large urban areas.

visible in terms of policy concerns (Lucas 1997), and they are a key element in understanding environmental change in remote and frontier areas (Bilsborrow 1992, 2002).

How many of these migrants are poor, or from poor areas, or both? It is difficult to say, since the nature of the interactions between poverty and migration is rather ambiguous and heavily influenced by contextual factors in origin and destination areas (Waddington 2003; Waddington and Sabates-Wheeler 2003). "In some part of the world and under certain conditions, poverty may be a root cause of migration, whereas in other parts, under different conditions, the poor may be among the last to move. Equally, in some areas, migration may be an avenue out of poverty while in others it contributes to an extension of poverty" (Skeldon 2002:1).

It is recognized that deep poverty is actually a deterrent for leaving: migration is a costly enterprise, and the poorest households and people may be less likely to migrate long distance, long term, or both because of the lack of resources (Skeldon 2002). However, individuals and households that are not able to afford long-term and long-distance migration may still engage in temporary and short-distance mobility, for example circulation and seasonal migration (Craviotti and Soverna 1999; Rain 1999; Skeldon 2005; Hugo 2006). As another example of "survival migration" (Skeldon 2002:71), people moving to frontier areas in search of affordable land may be among the poorest in the original settlements (Amacher et al. 1998; Carr 2009:370).

The diversification of household livelihoods[3] is a critical step for increasing social resilience, particularly among poor and vulnerable households (Moser 1998; Adger 2000; Ellis 2000). Diversification reduces vulnerability, risk, and uncertainty, although its effectiveness depends on the local context, including the structure of opportunities, degree of isolation, and distance to markets (Dirven 2004). Population mobility contributes to diversification by better allocating a household's labor resources across other (non local) rural or urban labor markets. Mobility may adopt different forms (circular, short-term, long-term, internal, or international); could entail the departure of just some family members while the rest of the family continue to work in the area of origin; or could consist of extra income earned during the off-season involving all the family (Craviotti and Soverna 1999; Adamo 2004). Circular labor migration, in particular, has been extensively studied in relation to diversification of livelihoods in rural areas of the developing world (Hugo 1982;

[3] Household livelihoods consist basically of three elements: (1) capabilities, (2) assets or resources, and (3) activities or strategies (Chambers and Conway 1991:6; Scoones 1998; Long 2001). Livelihood strategies, including economic participation and demographic behavior, mobilize, or actualize resources or assets to secure material and biological reproduction, with the purpose of achieving and retaining a certain standard of living, variable across societies and over time (Schmink 1984; Stark 1991; Bilsborrow 1992; Massey et al. 1993; Forni and Neiman 1994; Hugo 1998; Ellis 2000). The development and organization of social networks of relatives, friends, or neighbors, for purposes of exchange and solidarity, are important elements when considering the feasibility of particular strategies (De Dios 1999). The concept of household strategies places the domestic unit as the mediation between individual behavior and macro or structural socio-economic conditions, helping to explain different outcomes when facing a similar environment (Arguello 1981; Schmink 1984; Forni et al. 1991; Hugo 1998).

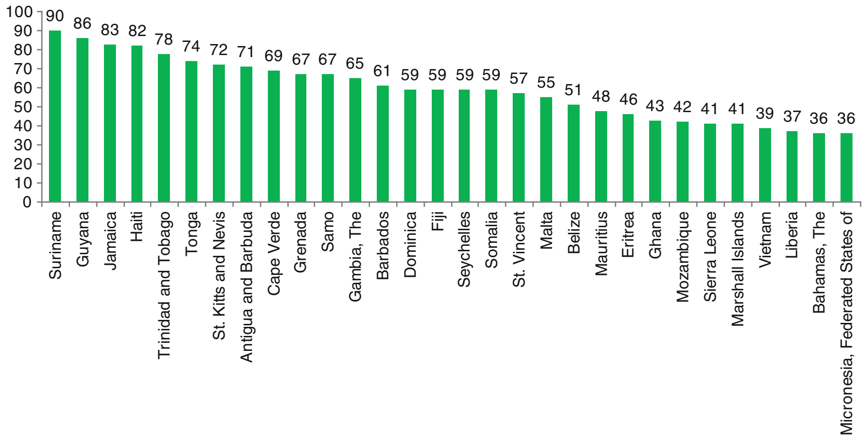

Fig. 7.1 Top emigration countries of tertiary educated, 2000 (as % of total tertiary educated) (Source: Ratha and Xu 2008)

PISPAL et al. 1986; Cordell et al. 1996; Rain 1999; Hugo 2006), and findings indicate that it has a positive effect on poverty reduction in sending areas (see for example De Brauw and Harigaya 2007). The selection of the destination (internal or international, urban, or rural) is crucial in this strategy (Wouterse and Taylor 2008), and migrants and their families frequently rely on family and other networks for information about non-local opportunities (Adamo 2003:112).

Still, population mobility can have negative effects on poverty in original settlements because of its selective character, which can result in accelerated aging, brain drain, and community weakening, all of which could eventually exacerbate poverty (Skeldon 2002, 2005; Usher 2005; Tacoli et al. 2008). Rural areas and small urban centers in developed and developing areas have depopulated due to out-migration, in relative and absolute terms. Because young adults are more likely to leave, there has been a rapid aging process in many of these places, with an age structure polarized between children and the elderly. At the community level, community weakening due to selective depopulation is also possible, as well as feelings of relative deprivation (Skeldon 2002).

“Brain drain” refers to this emigration of highly educated individuals, generally to developed countries but also to large metropolitan areas of developing countries. It is defined by a significant loss of the highly educated population (Fig. 7.1); and by adverse economic consequences following the loss (Lowell 2003). The selective emigration of skilled professionals could seriously affect development efforts, resulting in a further decline of the quality of life in sending areas, for example in terms of health care (Taylor et al. 1996; Usher 2005).

Migration flows are usually from settlements where the prevalence of poverty is higher if compared with receiving areas (Skeldon 2002): from developing to developed countries, or from rural to urban areas within the same country. The larger the gradient of poverty and development between sending and receiving areas, the more intense the movement (Hugo 2006). A lower prevalence of poverty in receiving areas, however, may not hold true at the level of migrant households and individuals. After arrival, migrants may experience a decline in their living conditions including unemployment and poverty, although in general, conditions tend to improve over time.[4] For instance, risking urban poverty could still be an improvement from harder living conditions in rural areas, in terms of access to public and social services (mainly health and education) and more diversified opportunities of employment.

The transfer of poverty from rural to urban areas through migration is known as the 'urbanization of poverty' (Ravallion 2001; Ravallion et al. 2007). This is frequently associated with a mismatch between employment creation and labor supply in destinations areas (see for example Cai and Du 2006 for the case of China), work insertion in the informal sector[5] (Tacoli et al. 2008), and higher cost of living in urban areas (Ravallion et al. 2007). Urbanization of poverty has been frequently associated with the growth of urban slums (UNFPA 2007). However, it would be misleading to assume that rural migrants are the only or in some cases even the main group populating urban low-income settlements in developing countries (Tacoli et al. 2008:39). Work by APHRC in Kibera slum in Nairobi shows that most residents have lived there all of their lives – migrants make up only a small percentage.[6]

As regulatory mechanisms, family and other social networks influence the choice of destination. Counter-intuitively, they may lead migration to areas where the incidence of poverty or unemployment is high, because they are able to provide support for new migrants independently of the harsh contextual conditions. This may include the introduction in the informal sector or connections for entering specific labor market segments like construction work. For example, family networks encourage migration from Paraguay's rural areas to Argentina's urban centers regardless of macro-economic or labor markets considerations (Parrado and Cerrutti 2003).

One of the ways migration contributes to poverty reduction in sending areas is through remittances, the funds that migrants send back to their places of origin (Sana and Massey 2005; Usher 2005; World Bank 2006). Between 1995 and 2005, international remittances increased from US \$101.5 million to US \$225.8 million worldwide. In more developed countries, they increased from US \$46.5 to US \$80.8 million, representing 0.2% of the GDP. In less developed countries, remittances increased from US \$55 to US \$145 million, representing 1.7% of the GDP in 2005

[4] For example, a study found that, between 1993 and 2001, unemployment and poverty were higher among recent internal migrants in the Buenos Aires Metropolitan area (Cortés and Groisman 2004), while another study concluded that immigrants from outside the European Union were exposed to a higher risk of poverty than the EU native population (Lelkes 2007).

[5] The term "informal sector" seeks to capture the large share of the global workforce that remains outside the world of full-time, stable, and protected employment. This also called "informal economy" includes informal enterprises as well as informal (i.e. not registered) work in informal and formal enterprises (International Labor Office 2002:11–13).

[6] Alex de Sherbinin, personal communication, May 2009.

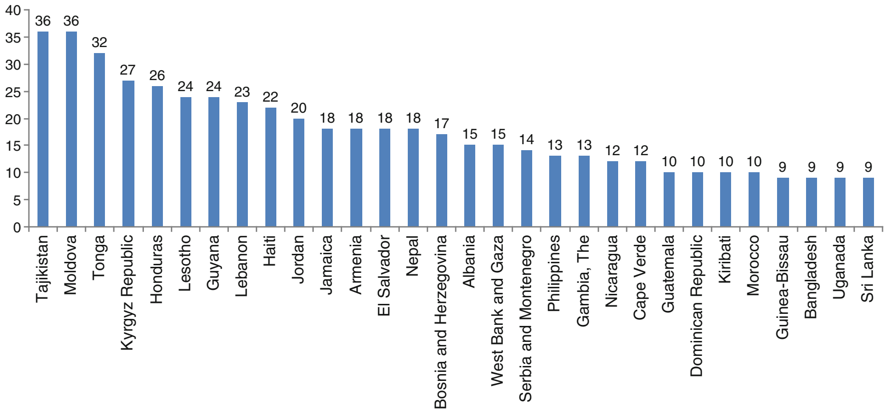

Fig. 7.2 Top remittance-receiving countries, 2006 (as % of GDP) (Source: Ratha and Xu 2008)

(U.N. DESA. Population Division 2009). Remittances represent more than 30% of the GDP in countries such as Tajikistan, Moldova, and Tonga (Fig. 7.2).

Although mainly studied in relation to international migration, remittances from urban areas to rural households are also relevant in poverty reduction (Skeldon 2005). It has been estimated, for example, that remittances from out-migrants of the Chinese province of Sichuan represented 7% of its GDP in 1995 (US $ 2.4 billion) (Wang and Cai 2007).

Remittances flow through the social networks and also fuel network growth by reinforcing the connections between non-migrants, and temporary, return and long-term migrants in origin and destination communities (Massey et al. 1987; Usher 2005). Remittance behavior is strongly regulated by social norms about family obligations and gender roles, but also by economic conditions in destination and origin. Those who remit are still considered members of the household and have a place in the community (VanWey 2004; Sana and Massey 2005; Sana 2008). Remittances can be used for everyday necessities – food, shelter, and clothing, improving living conditions of households in need and reducing consumption poverty (Tacoli and Okali 2001). They can be used for investments (land, health care, education, property, business) or for non-essential consumption.

Pros and cons of remittances at the community and national level have been extensively debated (e.g. Waddington 2003; Goldring 2004; Sana and Massey 2005; Adams 2009; Gupta et al. 2009). Their effect in alleviating individual and household poverty is amply recognized, and still in some cases remittances can lead to increasing inequality and inflationary pressures within sending countries and communities (Taylor et al. 1996; Skeldon 2002:4). Their long-lasting contribution to lower poverty would depend on their volume, frequency, regularity, the time horizon in which the effects are considered, the local structure of opportunities, and how the funds are used – for survival, consumption, or investment. Long-term effects on poverty reduction and local development are linked to remittances being invested in real estate, farmland, capital goods, or business initiation.

Ironically, while remittances have a significant role in alleviating poverty in sending areas and particularly in rural ones, they may also contribute to the increase in the incidence of poverty among migrants in destination areas. A study on poverty reduction in China found that the incidence of poverty among rural migrants in urban areas increased from 17.5% to 27.8% after remittances (Cai and Du 2006).

Environment and Population Mobility

Environment and population mobility affect each other in several recursive ways (Unroh et al. 2004; Curran 2002). Migration flows can be triggered by sudden or slow-onset environmental changes and deteriorating living conditions in areas of origin. In turn, migration flows may contribute to environmental change and deterioration in receiving areas, for example in the case of frontier settlements (for mining, agriculture, or ranching), increasing population density in sensitive ecological areas as around conservation areas (see for example Wittemyer et al. 2008; Joppa et al. 2009), or rural depopulation leading to land abandonment and forest re-growth (Aide and Grau 2004). Remittances from developing countries or from urban to rural areas can have a positive effect on the local environment, for example, by reducing resource dependence through the substitution of purchased goods by locally produced goods, or through investments in resource conservation. But they may also have deleterious effects, for example if they are used to expand investments in damaging practices such as transforming agricultural lands into peri-urban real estate (Meyerson et al. 2007; de Sherbinin et al. 2008).

Secondly, environmental conditions are part of the embedded contexts of migration systems in sending and receiving areas. Availability and quality of land, water, forest and other natural resources,[7] and the presence of natural amenities – sea beaches, mountain landscapes, or a warm, dry, sunny climate – may act as pull factors, while deterioration processes – land degradation, pollution, desertification, deforestation, or the recurrence of extreme events such as cyclones or other natural hazards – may push people out.

Finally, institutional regulations of migration systems also influence environment–migration interactions. Planned displacement and resettlement often follow large infrastructure works impacting the environment, as in the case of dams. Conservation projects (national parks, reserves of biodiversity, etc.) may block access to natural resources, notably land and water, and generate migration flows. International markets influencing changes in land use may also be related to in- and out-migration flows, for example farm workers going to work on new cassava fields – a product of growing international demand for the crop – in Thailand (Curran and Cooke 2008). Migration networks, offering support and advice, often facilitate leaving areas

[7] The livelihood approach to rural poverty identifies environment endowments as natural capital, one of five categories of household assets (Ellis 2000). Equally important is the issue of access, closely related to land/water distribution and tenure issues.

affected by recurrent environmental hazards, while adequate government disaster prevention and relief measures may facilitate the option to stay (Saldaña-Zorrilla 2008). In other words, migrant and migration impacts on ecosystem health are frequently mediated by the contexts that regulate migration. For example, if similar numbers of migrants arriving in a place are integrated into a community and its surrounding ecosystem through kinship, marriage, or other socially integrative means, then their impact on local environmental conditions is considerably lower than if their arrival is unregulated and non-integrative (Cassels et al. 2005).

Environmentally induced population displacements and mobility represent one of those environment–migration interactions, a demographic response to environmental change that results in threats to livelihoods and deteriorating living conditions (see for example Richmond 1995; Hugo 1996; IOM 2007; Renaud et al. 2007; Adamo 2009; Adamo and de Sherbinin forthcoming). The IOM (International Organization for Migration) defines environmental migrants as "persons or group of persons who, for compelling reasons of sudden or progressive changes in the environment that adversely affect their lives or living conditions, are obliged to leave their habitual homes, or choose to do so, either temporarily or permanently, and who move either within their country or abroad" (IOM 2007:1). This broad definition indicates that the attribution of population movements to environmental factors is still a delicate task. There are ongoing discussions on what constitutes an environmentally induced move, and the statistics available to assess the magnitude of the phenomenon show a large diversity. It is difficult to estimate future trends, and there is only modest agreement about the mechanisms linking environmental change and migration.

In any case, environmental factors do not act alone, and other contextual elements influence the relationship between environmental stress and population mobility. According to Wood,

> …modeling complex ecosystems and mobile populations is difficult: cause and effect relationships between environmental variables and migration are hard to quantify and are tied to economic, political, and cultural factors… (Wood 2001:44)

Migration is just one among several possible responses and adaptations to environmental change. Environmental deterioration may count among the factors, but it is not the main reason for emigration, except in cases of environmental disasters, since "people are more likely to migrate toward opportunities than away from problems" (Meyerson et al. 2007:108). Individual factors also influence the relationship, particularly people's subjective view and perception of the hazard and of their own vulnerability (Izazola et al. 1998; Adamo and de Sherbinin forthcoming).

Environmentally induced mobility can be shaped into a continuum from forced to voluntary migration. Three concrete points in the continuum would be: (a) refugee-like situations: very low level of control over the whole process and very high degree of vulnerability; (b) environment-driven displacement: compelled but voluntary, more control over timing and direction and less vulnerability than refugees, but less control and more vulnerability than migrants; and (c) migrant-like situations: greater control over the process and less vulnerability, even if people are moving in response to deteriorating conditions (Hugo 1996; Bates 2002; Renaud et al. 2007). The type of event (sudden or slow-onset), its intensity, duration, predictability, and magnitude influence environment-induced migration. Also important is the degree of vulnerability of the affected populations, which is closely related to poverty status. As a rule, the poor show a higher exposure to environmental hazards, including climate change-related events, with fewer alternatives to settle in safer places, and endure more severe and long-lasting consequences (Blaikie et al. 1994; Adger et al. 2007. See Chapters 19, 20, 21, 22 and 23, Vol. 1 on slow and fast-onset disasters).

Displacement of vulnerable populations has already been pointed out as one of the major potential impacts of climate change, and poor people make a disproportional percentage of vulnerable populations in developing countries (OSCE Economic and Environmental Activities 2005; Adamo and de Sherbinin forthcoming). Drought and water availability, desertification, extreme weather events, and sea-level rise are expected to be the climate change events with the higher potential for triggering population displacement, which is likely to show large heterogeneities across and within countries. The IPCC's Fourth Assessment Report highlighted the significance of already established migrant networks and patterns as part of the inventory of adaptation practices, options, and capacities available to face climate change impacts (Adger et al. 2007). Current concerns about the consequences of global climate change for human populations originate in the recognition that migration may be one of the most viable adaptation strategies as climatic changes begin to be felt, but that such population movements will undoubtedly have security implications[8] (Campbell et al. 2007; WBGU 2007; Adamo 2009).

Migration flows may have an impact on the environmental conditions of areas of origin and destination, particularly in rural regions. About 56% of the population in less developed regions still lives in rural areas, and this proportion reaches 73% in Least Developed Countries, but the growth rate has declined in recent decades (United Nations. DESA. Population Division 2008b) (Fig. 7.3). Absolute declines have been observed in certain areas, for example in South America, as the counter face of the urban transition that is taking place elsewhere in the developing world (Bilsborrow 2002; UNFPA 2007). Agricultural population[9] shows a similar increasing but slowing trend (Table 7.1), while for the same period, agricultural land expanded 8.7% worldwide, 5% in Africa, and 19% in South America.

Intense out-migration to urban areas may lead to an absolute decline of rural population (depopulation) and even to the complete abandonment of the rural area when critical services such as education, health centers, and post offices cannot be

[8] Security implications are twofold. On one hand, they refer to climate change impacts being triggers or concomitant factors in the emergence or aggravation of conflict situations. On the other hand, they reflect concerns about human security challenges, including the security of individuals, households, and communities, and about their coping and adaptation capabilities (Adamo 2009).

[9] Agricultural population is defined as all persons depending for their livelihood on agriculture, hunting, fishing, and forestry. It comprises all persons economically active in agriculture as well as their non-working dependents. It is not necessary that this referred population exclusively come from rural population. (FAO. Statistical Division. http://faostat.fao.org/site/379/DesktopDefault.aspx?PageID=379. Accessed May 31, 2009).

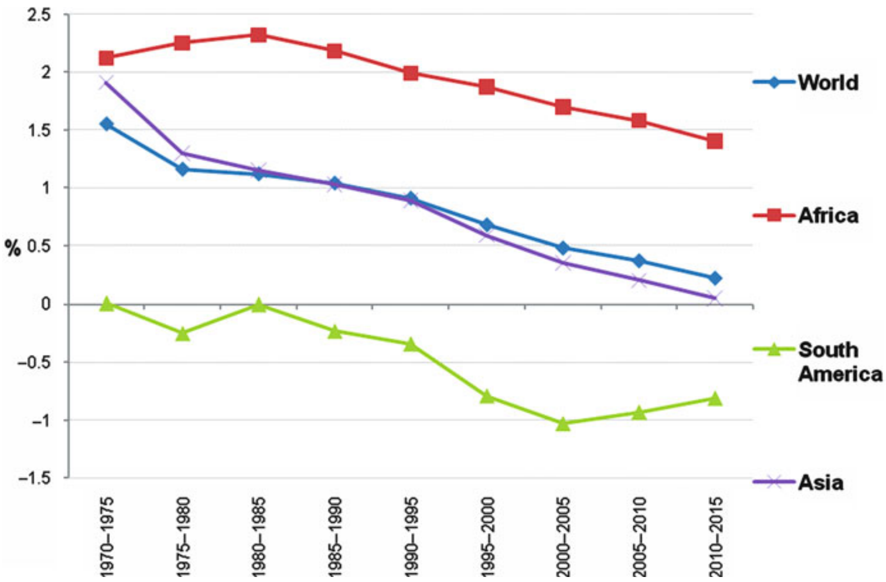

Fig. 7.3 Rural annual population growth rate (%) (Source: Population Division of the Department of Economic and Social Affairs of the United Nations Secretariat, *World Population Prospects: The 2006 Revision* and *World Urbanization Prospects: The 2007 Revision*, http://esa.un.org/unup)

Table 7.1 Evolution of agricultural population and area, 1970–2000

Year	World	Africa	South America	Asia
	Agricultural population (1,000)			
1970	1,994,271	270,515	78,711	1,435,744
1980	2,222,931	320,175	79,863	1,635,338
1990	2,455,154	387,315	70,113	1,830,787
2000	2,597,073	456,438	63,291	1,958,757
	Agricultural area (1,000 ha)			
1970	4,563,306	1,064,926	484,902	1,106,208
1980	4,666,114	1,078,863	525,287	1,158,599
1990	4,858,103	1,102,549	551,992	1,313,716
2000	4,960,102	1,126,222	576,022	1,678,061

(Source: FAOSTAT/© FAO Statistics Division 2009–31 May 2009)

maintained (Bustos Cara 1990). In Latin America, land abandonment of rural areas has been associated, among other things, with macro-economic changes in agriculture (e.g., the growing worldwide demand for commodities and biofuels, trade agreements, etc.) due to socioeconomic globalization. In some cases, general ecosystem recovery – particularly forest recovery and transition – has followed rural depopulation and land abandonment (Aide and Grau 2004; Rudel et al. 2005). However, the social consequences of these processes, including their interaction with poverty and inequality, have been less studied (Meyerson et al. 2007; Baptista 2008).

Rural-to-rural flows redistribute population among rural areas, often responding to land scarcity, land deterioration, or both in the area of origin, as well as to the 'opening' of new lands, i.e. areas where land is available and accessible, or where new crops offer opportunities for farm workers. Areas of origin are frequently located in marginal lands that combine population growth, land scarcity, poverty, and lack of alternatives to agriculture (Bilsborrow 1992, 2002; Amacher et al. 1998; Curran and Cooke 2008; Carr 2009).

Land extensification refers to "the expansion of the agricultural frontier by rural-rural migration" (Bilsborrow 1992:4, 2002). This process can have significant impact on the environment of receiving areas, particularly in fragile environments, as well as, implications for rural development. Poor, landless farmers are a large part of the flows going to frontier areas (Carr 2009). Intense and rapid deforestation is one possible outcome. This has been observed, for example, in the colonization of the Amazon regions of Ecuador and Brazil (Barbieri 2008), the upland forests of Philippines (Amacher et al. 1998), and Guatemala (Bilsborrow and DeLargy 1990). In-migration and expansion of agriculture frontier in dry lands, usually with the use of irrigation can lead to land degradation and desertification, as has happened in some Sahel countries (Bilsborrow 1992; Geist et al. 2006). Magnitude of the flows as well as their composition – in terms of characteristics of the individuals, households' lifecycles, timing of the settlement and type of activities (agriculture, ranching, lodging) are all important factors for understanding the possible impact of frontier settlement on the environment (Bilsborrow 1992; Barbieri et al. 2009; Carr 2009). However, cross-national comparisons and case study comparisons suggest that these patterns are not uniform and that the institutions regulating migration can be highly influential. When migrants are embedded within common property resource management institutions, their impact on environment is minimal, but when they are not, their impact exacerbates environmental degradation and heightens the negative feedback loop between poverty and environment (Curran 2002; Curran et al. 2002; Curran and Agardy 2004).

A last example of interactions between environment and population mobility is the case of involuntary displacement and resettlement due to large infrastructure projects, in which institutional regulations (formal and informal) are present. Development-induced displacement refers to planned and spontaneous displacement and resettlement due to large infrastructure works that modify environmental conditions, which has became a global problem (Robinson 2003:10; Castro et al. 2009). A large proportion of this displacement is due to the construction of dams and other water-related works (Table 7.2). Urban infrastructure and transportation are also relevant. Other categories include energy (power plants, oil exploration and extraction, pipelines), mining, agricultural expansion, parks and forest reserves, and even population redistribution schemes (Robinson 2004).

Large infrastructure projects basically alter the existing local land and water uses, modifying entitlements and access to resources, severely disrupting local livelihoods. Very frequently, this results in the impoverishment of the affected populations, bringing to the forefront the latent conflict between national development needs and local populations' rights (Bartolome et al. 2000; Cernea 2000; Morvaridi 2004; Stanley 2004).

Table 7.2 Distribution of displaced populations by cause of displacement in World Bank projects (active in 1993) with resettlement

Cause	Projects	Percentage	People	Percentage
Dams, irrigation, canals	46	31.5	1,304,000	66.4
Urban infrastructure, water supply, sewerage, transportation	66	45.2	443,000	22.6
Thermal (including mining)	15	10.3	94,000	4.8
Other	19	13.0	122,000	6.2
Total World Bank	146	100	1,963,000	100

(Source: Stanley 2004)

Forced, coerced and voluntary migrations have different consequences in terms of poverty. Force migration and unplanned relocations usually result in the impoverishment of migrant populations, which generally lose their livelihoods and their support networks. Cernea (2000:20) identifies different but interconnected 'impoverishment paths' by decomposing the risk of impoverishment in eight components: landlessness, joblessness, homelessness, marginalization, increased morbidity, food insecurity, loss of access to common property, and social disarticulation.

It has been strongly recommended that, if large infrastructure projects are to take place, their negative effects need to be mitigated (since they cannot be completely avoided). A key component of this mitigation is a resettlement program that "includes a focus on means of livelihoods instead of assets, understands the dimensions of the relationships between people and their assets, and is open to a negotiated definition of just compensation" (Bartolome et al. 2000:v) Active involvement of the affected population in the elaboration of the resettlement plan is a critical component.

Conclusions: Population Mobility as Mediating Factor in the Integration of Ecology into Poverty Reduction

> The people who first built a path between two places performed one of the greatest human achievements (Simmel 1994:6)

The application of ecological knowledge for better management of local ecosystems – for example, watershed management and restoration, forestry management and recovery, control of disease vectors, increase of crop yields for hunger relief while maintaining the resource base, management of biomass for fuel needs – aims to improve the living conditions in rural low income and marginal areas (Rumbaitis del Rio et al. 2005; DeClerck et al. 2006).

Population mobility may play a critical role in this integration. It has been already recognized that, although not explicitly included in the Millennium Development Goals framework, migration may have a significant influence on poverty reduction, education, gender equality, health, and environmental sustainability (Usher 2005; Skeldon 2005). Still, it is difficult to establish clear-cut positive, negative, or neutral effects of population mobility in integrating ecology for poverty reduction.

In this chapter, we have illustrated different ways in which population mobility interacts with poverty and the environment, using a migration systems approach in order to consider the positive and negative effects of flows, contexts in sending and receiving areas, and formal and informal regulatory institutions in shaping the pull and push forces that move people around.

Migration may be a facilitator between poverty and ecology in several ways: reducing population pressure in sensitive areas; expanding access to financial and other resources, for example through remittances; and facilitating the adoption of new and environmentally friendly technologies. All this could lead to the reduction of poverty while also contributing to environmental recovery and management.

However, these advantages can be offset by the selective character of migration, which may remove young and more educated population from the community (Grau and Aide 2007). It can also be the case that positive 'alliances' between poverty and migration in sending areas – for example through the diversification of rural livelihoods – result in environmental conflicts in receiving areas, as in the case of deforestation in frontier settlements, or the growth of urban poverty and urban slums in cities. Similarly, positive interactions between environment and migration, for example forest recovery due to land abandonment, may hide the impoverishment of the remaining populations.

Finally, the heterogeneity and multiplicity of possible interactions among poverty, environment, and migration components (settlements, flows, context, and regulation) warn against one-size-fits-all explanations. The existence of general frameworks for analysis and intervention should not disguise the fact that each case is unique and attention has to be paid to local conditions, specificities, and determinations to ensure optimum outcomes for communities and the environment.

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Chapter 8
Urbanization, Poverty Reduction, and Ecosystem Integrity

Peter Marcotullio, Sandra Baptista, and Alex de Sherbinin

Introduction

Cities and the urbanization process are often portrayed as environmental villains (Odum 1991; Srinivas 2000; White 1983; Brown and Jacobson 1987; Marshall 2005; Wackernagel and Rees 1996). For example, analysts focused on the "green agenda" of biodiversity conservation often suggest that urbanization is a major driver of environmental harm (Millennium Ecosystem Assessment 2005; York et al. 2003). Cities have been depicted as dystopias of poverty, chaos, and confusion (Linden 1996). By some estimates, almost one billion urban dwellers are living in poverty in today's cities (Satterthwaite 2007), and the numbers are predicted to continue growing (Davis 2007; UNFPA 2007). Hence, when it comes to both environmental harm and poverty, the reputation of cities and the urbanization process itself are considered as suspect at best.

This chapter argues that the associations between urbanization, poverty, and ecology are much more complex and much less well understood than often portrayed. We join those attempting to confront and dispel the fallacies and myths surrounding contemporary urbanization (see for example, Martine et al. 2008). In doing so, we critically examine simplistic notions that cities should be 'blamed' for environmental degradation and poverty. We acknowledge that there are many examples of cities that are poorly managed and that generate significant negative environmental externalities. We recognize that there are legitimate concerns about what an urban future means for long-term sustainability. Yet, there are many aspects of the critique of urban areas that have little to do with urbanization *per se*, but rather reflect concerns about patterns of production, accumulation, distribution, and consumption and the scale of economic activities and waste generation that occur in cities. Urbanization is not the cause of these problems considering that, in a world of over six billion

P. Marcotullio (✉)
Department of Geography, Hunter College, City University of New York, New York, USA
e-mail: peter.marcotullio@hunter.cuny.edu

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economically differentiated people, there will necessarily be large-scale production systems and differences in consumption preferences whether populations were evenly spread out over the planet's surface or concentrated, as they are now, with three billion people living on just 2–3% of the planet's land surface (McGranahan et al. 2005). In fact, it can be argued that evenly spreading out populations – were it possible – would be far more detrimental to the environment (Martine 2001). Urbanization is not the driver of ecological impacts, but rather a mediating variable. Urban governance systems, access to technologies, and civic mindedness of urban residents can make profound differences on environmental outcomes.

A more helpful approach for public, private, and civil society actors engaged in decision-making arenas recognizes the reality of current urbanization and the inevitability of further urban growth and seeks to identify pathways of urbanization that raise living standards and improve economic opportunities of the poor while being less ecologically damaging. We present a case for urbanization as part of the solution for improved well-being and ecological integrity. In terms of improved well-being, cities are engines of economic growth, offer enhanced opportunities for community participation, and provide institutional leverage to remedy poverty. In terms of ecological integrity, urban residents often have fewer environmental impacts than their suburban, exurban, and rural counterparts and the concentration of populations in urban areas can free up land for conservation and enhancement of ecosystem services.

That said, the challenges confronting urban areas in developing countries, where annual rates of urban growth often exceed 3%, are significant. Although industrialized countries saw similarly high rates of urban growth in the nineteenth and early twentieth centuries, often with even worse environmental and health consequences than those experienced by low-income countries today, there are still some important differences. First, the scale is different; the absolute number of people involved in today's urban transition in developing countries is 2–3 orders of magnitude larger. Second, owing to complex changes in the global economy, including the recent economic downturn, the prospects for development among the poorest countries are not nearly as bright as the situation faced by today's industrialized countries 150 years ago. Third, the distribution of material benefits and health and well-being within urban populations is highly skewed, and given the scale of urbanization, this distribution creates daily challenges for much larger numbers of people than in the past. Lastly, the global environmental context has changed dramatically, and today there is real concern about the impacts of further industrial development – a cornerstone of the urban economy – in the face of climate change and the rapid loss of ecosystems and biodiversity.

Thus, these dynamics cannot be understood without an examination of the scale, types of challenges, inter- and intra-urban differences in well-being and environmental conditions and the development context within which the urbanization–environment–poverty relationships are unfolding. Examining these relationships under these different lenses suggests four findings: (1) urbanization is not the only force involved in transforming the environment at any scale of impact from local to global; (2) the impacts we find are not *a priori* associated with urbanization, meaning, cities and the urbanization process are not inherently good or bad for the environment; (3) cities and urbanization can actually be beneficial for the environment at various scales; and (4) urbanization is a necessary, but not sufficient condition for the alleviation of poverty and economic wealth generation. Given projected population increases, it is difficult to show how global development could proceed in a more environmentally benign manner without urban settlement patterns.

We make these arguments in the following four sections. In the next section, we argue that urbanization is associated with positive socioeconomic growth and that urban growth helps to reduce poverty and enhance human well-being. In the third section, we present a framework that explores the environmental impacts associated with urbanization to demonstrate the complexity and variety of relationships that exist between urbanization and ecological and environmental change. In the fourth section, we present specific examples in which urbanization has been accompanied by ecological benefits and reductions in poverty. In the final section, we conclude by summarizing our main points.

Urbanization and Poverty Reduction

Urbanization is part and parcel of modern industrialized development (Bradshaw and Fraser 1989; Annez and Buckley 2009), the process being inseparable from economic growth and development (World Bank 1991, 1999). There is simply no counterfactual of an economically developed country that has not experienced urbanization. Satterthwaite (2007) has pointed out that around 97% of the world's GDP is generated by industry and services located in and around cities and around 65% of the world's economically active population works in these sectors. Moreover, as the world has urbanized, the proportion of each of these sectors in total global production has increased.

There are many reasons for these associations. Fundamental aspects of urban economic growth are related to communications and transportation, returns to scale and agglomeration economies, and advances in technology. All of these activities bring substantial benefits for most industries, generate economic activity, and increase wealth (Anas et al. 1998; Glaeser 1998; Montgomery et al. 2003).

Although relative costs of communication and transportation have declined sharply with advanced technologies, cities retain their vital role for several reasons. First, technology and personal exchanges may often be complementary, forcing complementary growth (Glaeser 1998). Second, personal contact enhances levels of trust and ensures confidentiality, among other important business fundamentals. These principles cannot be erased by technological advances (Montgomery et al. 2003). Third, agglomeration proceeds and overcomes decentralization trends associated with advances in telecommunications because the sources of agglomeration remain in the production process, albeit in the service economy (Sassen 2001).

These arguments strongly suggest that despite advances in technologies and shifts in economic activity from production of goods to production of services and consumption dynamics, cities continue to provide firms with increasing returns to scale. Global forces converge on cities creating conditions where speed of transactions, transnational decentralization of firms, new technologies and the increasing institutional complexity for carrying out business facilitate urban agglomeration economies. Cities remain the centers of economic growth, because they are the places where the work of globalization gets done through the provision of specialized business and producer services for today's international business environment (Sassen 2001).

There are other related positive economic aspects of urbanization. Dense settlements enhance access to labor for firms and facilitate the provision of public services such as water supply, roads, and electricity, all of which reduce business costs and help to make cities attractive (Bairoch 1988). Urban densities and populations differentially concentrate economic activities among city systems distributing good and service provision (Christaller 1966: originally published 1933; Losch 1954: originally published 1939) and increase sector productivity (Ciccone and Hall 1996). The resulting higher incomes in cities provide governments with the means to further extend investment in public services and education. Rising incomes within cities also support the development of specialized private markets (Montgomery et al. 2003). Given these dynamics, economists have long suggested that urbanization has a strong positive correlation with economic activity (Williamson 1965; Annez and Buckley 2009).

All this suggests that urbanization is linked to increases in wealth and, therefore, helps to reduce economic poverty. The notion is further bolstered by new research on the spatial patterns in wealth and poverty (World Bank 2009). Estimates suggest that approximately 75% of the world's poor are located in rural areas, which is higher than expected given the current speed of urbanization and growth of slums (World Resources Institute 2008). Even applying different urban and rural poverty lines that account for the higher cost of living in urban areas, most countries have a lower percent of the population below the poverty line in urban than rural areas, though in some African countries, the absolute numbers of poor people are higher in cities (Fig. 8.1). At the global level, approximately 30% of all rural residents live in extreme poverty ($1 per day), and 70% live in poverty (less than $2 per day), while approximately 13% of urban residents live on $1 per day and 34% on $2 per day (Ravallion et al. 2007a, b). These trends are a far cry from the prediction that 75% of the total global poor would live in cities by 2015 (Piel 1997). Furthermore, on average, those living in cities tend to live longer than those living in rural areas (Montgomery et al. 2003).

According to some poverty experts, the difference in predicted versus observed trends is explained by the economic activity that accompanies urbanization. Simply put, urbanization is bound up with processes that create wealth and reduce economic

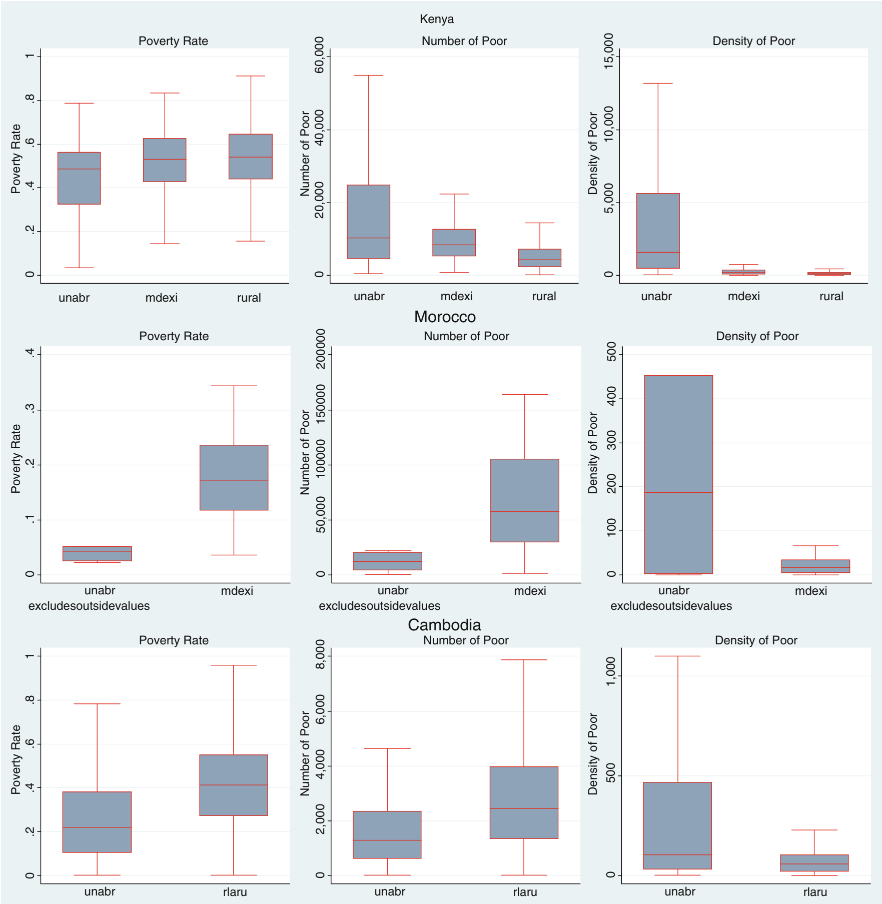

Fig. 8.1 Urban and rural poverty using separate urban/rural poverty lines *Left hand column y*-axis is poverty rate, middle column *y*-axis is number of poor, and right hand column *y*-axis is density of poor (Source: Maria Muñiz, CIESIN, Columbia University)

poverty and, therefore, one of the most important ways to reduce global poverty is to urbanize (Ravallion 2007).

Urbanization is a dynamic and often messy process, which is, at least initially, associated with unsafe, unhealthy, and seemingly chaotic environments, particularly for the poor. That is, there is significant economic stratification within cities. Disparities in income and health are often more sharply defined in urban areas than in rural areas, with upper and lower income neighborhoods often located within blocks of one another (Tacoli et al. 2008; Stephens and Stair 2007). These spatially proximate disparities, when combined with the concentration of poor in urban slums, lead some to conclude that poverty – or at least the character of poverty – is worse in urban than in rural areas. These perceptions, however, tend to overlook not only the economic gains but also the positive non-economic aspects of urban living.

Although social stratification is often a consequence of such wealth generation, it has been proven difficult if not impossible to lift large populations out of poverty based on a purely rural economic development strategy (World Bank 2009). Cities spur technological innovation (Jacobs 1969). Advances in science and technologies and in systems of law and government progress through the aggregation of educated people (Montgomery et al. 2003). Cities are centers of education and artistic expression and, therefore, new sources of knowledge. They are sites of cultural change and political mobilization, serving as focal points for accommodation between diverse ethnic and religious groups.

Population concentration makes it cost effective and efficient to provide infrastructure and services to the urban poor and makes it easier to govern society and facilitate political empowerment (Satterthwaite 2007). Dense settlements lower the costs per household and enterprise for infrastructure provision (roads, pedestrian paths, piped water, sewers, drains, and electricity) reducing tax burdens and providing better social services (day care, schools, sanitation, health care, and emergency services) than rural areas. Access to health and other services and basic infrastructure is on average better in urban than in rural areas (UN-Habitat 1996). Much of the problem of current urban poverty lies in the lack of planning and provision of infrastructure and services for those populations along with low levels of public participation (UNFPA 2007).

It is not the agglomeration of people that is causing and maintaining poverty, but the social and political organization of societies within and outside of cities that determines differential access (Myllylä and Kuvaja 2005). If poverty also includes the lack of access to education, public services, secure land tenure, adequate housing, and political empowerment, the characteristics of dense urban settlements provide opportunities for poverty reduction.

Urbanization and the Environment

We argue that rather than having a straightforwardly negative environmental impact, urbanization has a complex relationship to the environment. To explore this claim, we review the findings of four aspects of the urban–environment relationship, which we argue cover most of the studies in this area. First, and perhaps foremost, using urban environmental transition theory (McGranahan et al. 2001; McGranahan and Songsore 1994), we address the scale of environmental impact, arguably associated with the average level of income of an urban center. Second, we include studies that examine local ecological impacts associated with urbanization. Third, we distinguish between ecological impacts in urban and non-urban locations. Fourth, we consider the experiences of urban environmental transitions during different historical periods. In each case, we question the notion that urbanization, defined as population concentration, is the only or even the most important factor involved in loss of ecosystem services or environmental impact. Moreover, we suggest that there are instances where urbanization is relatively beneficial to local ecology and has the potential to mitigate negative environmental outcomes that accompany development.

Urbanization and the Scale of Environmental Impact

The direct urban footprint has had significant impact on the environment. For example, approximately 29 of the world's 825 ecoregions have over one third of their area urbanized, but this imperils 8% of the terrestrial vertebrate species on the IUCN Red List (McDonald et al. 2008). At the same time, however, it is also important to understand that human activities within cities have impacted the environment at larger scales (de Sherbinin et al. 2007; Grimm et al. 2008). If we consider the environmental impact of activities that occur in cities, they spread way beyond political borders. For example, while urban land use accounts for less than 3% of globally available land, about two thirds of the world's energy is consumed in cities (IEA 2008).

Different types of environmental harms are associated with urban activities at different periods in history, across levels of income and at different spatial scales (Tarr 1996; Melosi 2000). These shifts in the scale of environmental challenges for cities have been associated with growing affluence (McGranahan et al. 2001; McGranahan and Songsore 1994; Hardoy et al. 2001). The theory of environmental transition suggests that cities undergo a series of shifts in the type and geographic and temporal scale of environmental burdens as they grow in wealth. Specifically, the theory suggests that low-income cities predominantly experience localized, "brown" agenda challenges such as limited access to fresh water supplies, sanitation, and drainage services and high incidence of indoor air pollution. Middle-income cities experience geographically regional environmental burdens including air and water pollution. Affluent cities predominantly experience "green" agenda or sustainable development burdens such as high levels of consumption and associated impacts and high levels of emissions (e.g., greenhouse and ozone depleting gases) while expanding their footprints through resource extraction in distant regions.

We highlight two important messages from urban environmental transition theory. First, cities are associated with several different types of environmental conditions that emerge at different scales. Therefore, understanding the urbanization–environment relationship requires the examination of different geographic and temporal scales within which the urban center is located. Second, because these trends shift with income, all cities contribute to some form of environmental harm (Satterthwaite 1997). Even if the local characteristics of an urban center appear healthy and livable, the impact of activities in that center is creating larger scale problems. For instance, it is difficult to argue that a low-income city (e.g., Dhaka, Bangladesh) is more or less sustainable than an affluent city (e.g., New York, London, or Tokyo), which has a far larger ecological footprint. These points make the urban environmental transition theory a powerful tool in exploring changes in the urban environment.

Importantly, however, in this model, changes in environmental impact are not directly related to urbanization, defined in terms of increase in number of urban areas, growth of urban extent, or densification of urban areas. Instead, wealth generation is the main axis for which to view transitions. Rather than the urbanization process, this theory uses changes in income to examine environmental burdens related to cities. Indirectly, it suggests that with increasing urbanization, wealth rises and, therefore, the relationships hold. On the other hand, there is not an explicit connection between urbanization and environmental impact. Hence, the theory is deficient in explaining the number of cities at the lower end of the income scale, such as Curitiba and Porte Alregre, that are well managed and provide water supply and sanitation services and infrastructure to most of their populations (Hardoy et al. 2001). Nor does it account for estimates suggesting that per capita greenhouse gas emissions are higher in low-income and rapidly developing cities (e.g., Beijing and Shanghai) than high income cities (e.g., Tokyo) (Dhakal 2004). So, while the concept is powerful because it identifies the shift in scale of environmental impact with increasing urban affluence, it is less useful as (and not intended to be) a theory relating urban growth or urbanization to environmental impact. One could use this theory to examine the shifts in environmental impacts and the scale of these impacts associated with growing wealth at the household, regional, or national levels. While it is crucial to understand the multi-scaled changes associated with growing wealth in urban areas, the urban environmental transition does not directly correlate urbanization, *per se*, to environmental impact.

Urbanization and Local Ecological Change

Notwithstanding the loose connections between urbanization and urban growth to environmental impact in urban environmental transition theory, the perspective also is deficient in its coverage of local ecological concerns. Therefore, in order to more fully explore how urbanization might impact local ecology, we must look elsewhere. In doing so, we demonstrate that in most instances, the local ecological impact is significant, but also variable among and within cities.

The urban ecological literature is rich and growing. We identify two major research strands: ecology in cities and the ecology of cities (Grimm et al. 2000; Pickett et al. 2001). Research of ecology in cities has a long history, beginning in Europe in the seventeenth and eighteenth centuries, but the number of studies expanded dramatically after the 1940s (Sukopp 2008). Studies on ecology in cities focus on habitats, individual species, or organisms and the relationship between abiotic and biotic entities within urban environments. Several texts have elaborated on the general attributes of local ecology in cities (Gilbert 1989; Gill and Bonnett

1973; Hough 2004). There are also specific literatures on the association of urbanization and hydrology, soils, air quality, and biodiversity. Work on the ecology of cities is more recent, emerging at the turn of the twenty-first century and growing rapidly. The ecology of cities uses a systems approach to focus on socio-ecological linkages and how these relationships control flows of energy and materials into, within, and out of urban systems. We briefly present a summary of research in each area.

Perhaps, the first extensive and systematic set of global ecology in cities studies examined changes in urban hydrology with changes in land use and land cover (UNESCO 1977; Ward 1981). Urbanization-related activities of removing vegetation, constructing houses, streets, and culverts, paving tracts of land, channelizing waterways and constructing storm drains have dramatic effects on local hydrology (for a review, see Paul and Meyer 2001). Typically, the water cycle in urban areas demonstrates lower general evapo-transpiration, infiltration, and interception levels, lower groundwater tables, increased stream sedimentation and storm flows, decreased base river flows, and earlier and higher runoff concentration peaks with increased volume of runoff that provides local relief but aggravates those downstream (Rodgers 1994; Dunne and Leopold 1978). Urbanization also impacts water quality and fluvial geomorphology (Ellis 1999; Graf 1975; Klien 1979). Further detailed analysis is presented in Chap. 8, Vol. 1, Balancing Human and Ecosystem Needs for Water in Urban Water Supply Planning, which discusses options for balancing human and ecosystem water needs in urban water supply management.

Urban ecological studies of soils identify differences between urban and non-urban conditions. Typically, soils in urban areas are more compacted, have less distinct boundary layers, higher ph and temperatures, lower water drainage, more developed water repellent crusts, and higher levels of contaminants when compared to rural soils (Craul 1985, 1992; Marcotullio et al. 2008). In some cases, soil biodiversity changes dramatically with urban growth (Pouyat et al. 1994; Steinberg et al. 1997). Given the differences between urban and non-urban conditions, urban soil scientists have introduced new soil classification systems to distinguish urban soils (Hollis 1991). In New York City, for example, a new soil survey identified a number of different names for soils formed on human constructed landforms with names such as Shea, Central Park, and Big Apple (New York City Soil Survey Staff 2005). However, not all soils within urban areas have been similarly impacted. Recent studies suggest significant variation in some physical and chemical characteristics of soils within cities and point out the important influences of underlying geology, land use, and land cover and the potential for management practices in creating differences (Pouyat et al. 2007). Hence, not all aspects of soil disturbance and degradation are uniform throughout urban areas.

Some of the earliest and most intensive work on the urban atmosphere focused on the urban heat island effect (UHI), the difference between urban and rural temperatures (Oke 1973). These differences, and specifically the higher temperatures in the city compared to rural areas, vary by time of day (they are greatest right after sunset) and seasonally (in the United States, they are greatest during the winter). Moreover, the UHI varies by region, spatial heterogeneity of the urban landscape, city size, and population density (for a review, see Arnfield 2003). Research on UHI is increasingly important, given the predicted rise in global temperatures and the climate effects in cities that have arguably contributed to the recent devastating impacts of heat waves on urban residents in Europe (2003) and the United States (1995). Moreover, while no evidence that collective UHIs have a direct impact on global warming has been found (Parker 2004; Peterson 2003; Alcoforado and Andrade 2008), as cities increase in size and number, the UHI effect may play a role in regional climate (see for example, Kaufmann et al. 2007).

Atmospheric scientists such as Landsberg (1981) also observed that urban climatic conditions include lower radiation, more cloudiness, higher precipitation, higher temperatures and more particulates, gaseous admixtures and other contaminants than non-urban climates. These characteristics arguably make urban climates unique (for reviews, see Shepherd 2005; Souch and Grimmond 2006). Particularly adverse impacts include high local ozone concentrations, which reduce photosynthetic processes and have significantly negative effects on crop yields, forest growth, and species composition at the regional level. Within areas of rapid urbanization rates such as Asia, ozone concentrations are increasing, particularly around larger cities (Ashmore 2005).

Finally, a recent review of over 105 studies on the association of animal and plant diversity and urbanization performed in developed, developing, and transition economies suggests that, in central urban core areas, species richness tends to be reduced (McKinney 2008). Most of the studies of plant species variation indicate that suburban areas have greater species richness than urban and non-urbanized areas. This increase in biodiversity is also seen in some studies of animal diversity. For example, in Seattle (United States), bird species diversity peaks in intermediate areas of human settlement due to greater rate of the colonization of settlements by synanthropic species (i.e., species that are ecologically associated with humans) than extinction rates of indigenous species (Marzluff 2005). That is, within urban areas, synanthropic species combined with abundant food sources create local species diversity and abundance levels equal to or greater than surrounding landscapes. At the same time, studies suggest that as cities globalize, floral and faunal biodiversity becomes more homogenous as "urban exploiters" expand their geographic range globally (McKinney 2006). The result has been a loss of local endemics, which is contributing to a global biodiversity crises. Protecting biodiversity will, therefore, necessitate coordinated planning among neighboring municipalities so as to create different landscapes and parkland types, which would help to increase biodiversity locally and regionally (Marzluff 2005) and reduce global homogenization.

Another interesting trend is occurring in some large cities, particularly in the developed world, with the return of medium-sized and even some large mammal species. In New York City, for example, there have been documented returns of white-tailed deer, red-tailed hawks, raccoons, and beaver (see for example, Mittelback and Crewdson 1997; McCully 2007). These sightings are probably related to two effects: (1) the re-expansion of species' ranges due to growing rural and suburban species populations as they respond to successful regional and state conservation efforts; and (2) environmental improvements within the city. New green design in Europe is having a significantly positive impact on urban biodiversity (Beatley 2000). Admittedly, these examples are anecdotal and the density of infrastructure and fragmentation of habitats by roads has a significant impact on large terrestrial mammals. At the same time, by concentrating people within cities, we have a greater opportunity to protect large tracts of rural land (and thereby reduce habitat destruction and fragmentation) than we do without urbanization (Trzyna 2007).

Integrated approaches to cities as ecosystems attempt to incorporate human dimensions in ecological studies in a systems framework. Systems analysis suggests that one cannot fully understand these complex phenomena by examining component parts separately. The new urban ecology attempts to bring together systematically a wide range of theories, models, and findings from multiple disciplines in examining the dynamics of urban (eco)systems in a holistic fashion. Integrated ecological studies focus on understanding the complex linkages among the various components that make up the urban ecosystem (see for example, Machlis et al. 1997). Studies that understand cities as ecosystems shed light on the differences between ecosystem functions within dense settlements and those in other systems (Alberti 2008). For example, long-term ecological research in the Baltimore Ecosystem Study (United States) suggests that stream water quality in urbanized areas is not correlated with density and that nitrogen retained in metropolitan suburban soils is quite high. The results are surprising since ecologists typically associate urbanization and built-up areas with higher levels of degradation than rural and natural areas. These studies have also demonstrated that socioeconomic impacts exhibit a lag effect. That is, previous income levels are more highly correlated to neighborhood greenness than current levels (Pickett and Cadenasso 2006). Together, studies of ecology in cities and the ecology of cities demonstrate that urban ecosystems are different from "natural" ecosystems, but not in ways commonly understood.

The question often overlooked in the studies mentioned above is whether the transformations identified are only related to population concentration. The answer is not as clear as one might think. Complicating our understanding of these relationships is that within the urban ecology literature, there are inconsistencies in the application of the definition of urban (McIntyre et al. 2000). In most cases, both "ecology in" and "ecology of" studies define urbanization by land-use change and the decrease in pervious surfaces (McMahon and Cuffney 2000). Hence, an increase in imperviousness, or shifts in land use from agricultural to built-up areas, translates to higher levels of urbanization. Some studies are based upon space-for-time substitutions where urban-rural gradients represent changes in urbanization level (McDonnell and Pickett 1997). Moving from areas of high imperviousness in core parts of the city to areas with fewer impervious surfaces suggests to some researchers that they are moving back in time to periods of smaller and fewer cities. The direct causes of building development or the underpinning of urban land-use change are influences rarely explored in any detail. The point is that there is a set of complex economic, social, political, and biophysical processes that leads to decisions of land-use change and infrastructure investment and ultimately helps to explain urban growth. Pointing to the growth of cities as the source of the problem misses these essential parts of the puzzle and ultimately will lead to policy misspecification.

Urbanization affords opportunities to enhance ecosystem services both within core urban areas and around cities. More importantly, environmental degradation is not only related to dense settlement and land-use change. Rather, there are a number of complex drivers associated with urban expansion that need to be better understood if we are to reduce the ecological impacts. Blaming cities for local ecological degradation may even exacerbate current challenges.

Spatial Variation in Environmental Conditions Between Urban and Non-urban Areas

Here, we compare the environmental and ecological impacts and spatial arrangements of urban, suburban, exurban, and rural areas. In doing so, we make two caveats. First, dynamics along urban–rural ecosystem gradients shift with development. That is, as presented before, at a low level of development, there may be more ecosystem damage associated with urban areas than rural areas, while in more developed societies, these differences may be attenuated or may even reverse. Second, we also understand that individual cities are extremely diverse with varying levels of ecosystem challenges within the urban fabric.

In the comparison of regional and global ecological impacts, we find that those in urban areas are not only less detrimental, but in some cases, they also provide environmental benefits. One of the best-known examples of different levels of impact is energy use. Cities in the developed world are typically more energy efficient than their suburban or exurban counterparts. Compact forms and more accessible transit opportunities lower consumption. New York City, for example, has lower per capita energy consumption (2,000 kWh) than that of the United States (4,000 kWh). New Yorkers also use less energy (300 kW per month) than their suburban Westchester neighbors (450 kW per month) (Ascher 2005). In developed countries, per capita energy use in metropolitan areas is lower than national averages (IEA 2008). While it has been argued that those in developing world cities eat more varied diets and more meat than their rural counterparts, recent evidence suggests that these differences are more related to income than access to foods. Stage et al. (2009) have demonstrated that for several developing economies, rural and urban families of similar income have similar expenditures on meat. Alternatively, in the United States urban residents eat less meat than rural residents (Davis and Lin 2005). Meat consumption has enormous impact at regional and global scales (see for example, Steinfeld et al. 2006). Lower meat consumption translates into lower greenhouse gas emissions, lower air and water quality degradation, and more efficient personal energy budgets.

Furthermore, urban densification can help to improve nearby ecosystems in rural locations. It is often suggested that cities are located close to productive agricultural land. The productivity of this land, however, may be a function of the human improvements and amendments facilitated by proximity to markets and transport infrastructure as well as the type and quality of soil. Or urban demand can lead to better rural well-being and lower ecosystem stress. In Niger, for example, urban demand for fuel wood, combined with re-greening efforts in rural areas, has created new opportunities to earn income in expanded and diversified rural economies and has saved shrinking natural forests from further destruction (World Resources Institute 2008). In the New York City region, because of the need for fresh water, the local government engaged in a long-term effort to maintain and preserve large tracts of land in the upstate community to provide fresh water (Pires 2004). Current efforts include an ecosystems approach to managing the 5,180 km^2 that comprise the city's reservoirs and surrounding lands (World Resources Institute 2001).

Urban Environmental Variation Among Cities That Develop During Different Historical Periods

The conditions under which the developing world is currently growing are significantly different from the context within which the developed world grew (Held et al. 1999). The prospects for economic development of the poorest countries are not nearly as favorable as during the urban transition of the developed world a century ago, and the global environmental context has also changed dramatically with a concern for greenhouse gas emissions from industrial development that didn't exist even 20 years ago (de Sherbinin and Martine 2007). Economic globalization, structural adjustment, and growing debt have produced conditions that impede major investments in environmental remediation, sanitation infrastructure, and enforcement of environmental regulations. Simultaneously, some technologies are cheaper and being transferred to middle- and low-income economies. Trade is expanding the availability of goods, services, and natural resources, creating both opportunities and challenges for developing countries.

These conditions shape new environmental experiences for cities. When compared to their developed world counterparts, contemporary cities in the developing world are experiencing multiple environmental burdens that change at faster rates and emerge at lower levels of income (Marcotullio 2007). As such, the context for addressing environmental damage is different. Increasing environmental burdens at lower income levels, for example, force local officials in low- and middle-income cities to address more complex sets of environmental conditions than previously addressed by their developed world counterparts.

Therefore, the lessons learned in the developed world are less applicable to current low-income, rapidly developing cities. The economic, institutional, political, and environmental conditions are often too different to transfer seemingly beneficial policies and technologies across the board to developing cities. For example, a recent review of experiences in water and sanitation provision to developing world cities suggests that "the current 'high-income nation' solution will not work in most urban centers in Africa and Asia or many in Latin America and the Caribbean" (Satterthwaite and McGranahan 2007, pp. 28). While 'compact city' policies may make sense for reducing energy demand in developed world cities, they may fail to improve environmental conditions in developing world cities and instead create more problems related to congestion and health risks associated with air pollution.

Despite the differences in urban environmental experiences, the picture is not bleak for developing world cities. Many developing countries have thriving informal recycling sectors that dramatically reduce waste streams. In the transport sector, despite the rapid motorization in the developing economies of the Asia Pacific, all have lower per capita road carbon dioxide emissions than the United States and several have emissions levels equivalent to or lower than European nations at similar levels of income (Marcotullio and Marshall 2007). Public transportation usage rates are far higher than the developed world. Furthermore, most developing nations have increased their energy efficiency more rapidly than the United States. This efficiency has translated into lower carbon dioxide emissions at similar levels of economic development (Marcotullio and Schulz 2007). Moreover, health conditions in the rapidly developing world are better than those experienced by the developed world during its rapid industrialization (for conditions in the USA, see Preston and Haines 1991). In general, human well-being, defined by the UNDP Human Development Index, is much better in the developing world now than it was in the developed world at comparable levels of development (Crafts 2000).

While some gains are due to the transfer of efficient and beneficial technologies and knowledge to cities of the developing world, there are also solutions emerging from the cities of the South that are tailored to their particular circumstances. Some of these technologies have been identified in workshops given by the Academy of Sciences for the Developing World (TWAS) and the United Nations Development Programme (UNDP) special unit focusing on South–South knowledge transfer (see http://tcdc.undp.org/SIE/sharingsearch.aspx).

In summary, it is difficult to generalize about the association of urbanization and ecological conditions across development ranges. Conditions experienced in the past or present by the cities of the developed world cannot be correlated to those experienced by developing world cities. Therefore, lessons learned and policies that work in the developed world may not be universally applicable to current developing world cities.

Urban Strategies for Poverty Reduction, Ecological Integrity, and Sustainability

As we have argued, the relationships between urbanization, poverty, and ecosystem services are misunderstood. Cities are not the villains in this development triangle, but may indeed be the heroes. We now turn to examples of conditions within cities in different parts of the world where actions are both reducing poverty and maintaining or enhancing ecological integrity. In most of these cases, the quality of urban governance determines the effectiveness of program implementation.

Forward looking urban environmental management strategies in a number of cities demonstrate that urbanization and high levels of well-being are not necessarily linked to harmful environmental impacts. Actions in cities such as Malmo (Sweden), Freiberg (Germany), Kalundborg (Denmark), Curitiba and Porte Alegre (Brazil), Bogota (Columbia), Tokyo (Japan), New York City (United States), and Singapore suggest a number of large-scale innovative environmental programs that have had tangible environmental benefits. Many of these and other programs have been highlighted in recent publications (Sheehan 2007; Wheeler and Beatley 2004; Satterthwaite 1999). Here we mention two (see Boxes 8.1 and 8.2).

There is widespread evidence of economic sectoral change that accompanies urbanization. As countries urbanize, there is a switch between agricultural and industrial and service industry development. As such, there is much concern over the loss of agricultural and other types of ecosystems to urbanization (Balstad Miller and Small 2003; Hara et al. 2005; Redman and Jones 2005). Research claims that encroachment has resulted in significant impairment of ecosystem functions and, for some, loss of rural livelihoods (Adeboyejo and Abolade forthcoming; Guneralp and Seto 2008). Evidence suggests, however, that the concern over urban lands reducing agricultural output may be misunderstood. Over the last 25 years, in China and Thailand for example, productivity advances in agriculture accompanied strong performance in other sectors while these countries rapidly urbanized, leading to increased output across the board (Annez and Buckley 2009). A recent study suggests that there is no indication that higher urbanization rates have been linked to reduced agricultural productivity (Stage et al. 2009). Moreover, since the 1970s, urban agriculture has expanded and is today ubiquitous around the world (Halweil and Nierenberg 2007; United Nations Development Programme 1996). Agriculture and urbanization are not necessarily conflicting land uses, but can occur simultaneously, each enhancing the other. For example, during the 1980s, despite high density, Hong Kong produced 40% of its fish demand (Smit and Nasr 1992). Urban agriculture provides economic and health benefits to urban residents. Some researchers suggest that urban gardening can reduce the risk of obesity, heart disease, diabetes and occupational injuries (Halweil and Nierenberg 2007). Urban agriculture also enhances a number of different ecosystem services within cities. Work in this area has the potential to significantly improve local provision of food and maintain or enhance ecosystem services in both developed and developing world cities (Box 8.1).

As mentioned previously, urbanization can have beneficial effects on proximate ecosystems. For example, forest recovery has accompanied urbanization in southern Brazil's Florianópolis city region (see Box 8.2). Although the planted and secondary forests are less diverse than previously existing forest formations, these renewed forests provide an array of valuable ecosystem services (Baptista 2008; Baptista and Rudel 2006). Similar forest transitions associated with urbanization have occurred in Puerto Rico and the Dominican Republic (Aide and Grau 2004).

Box 8.1 Urban Agriculture

Urban agriculture is defined as "food and fuel grown within the daily rhythm of the city or town, produced directly for the market and frequently processed and marketed by the farmers or their close associates" (Smit and Nasr 1992, pp. 141). Urban agriculture is a diverse and growing aspect of agricultural development and includes aquaculture (in tanks, ponds, rivers, and coastal bays), livestock production (in backyards, along roadsides and streams, and in poultry sheds), apiaries, orchards, street trees, community gardens, backyard trees, and vegetables and other crops grown in backyards, vacant lots, and elsewhere. Urban agricultural production is significant and growing in developed, emerging, and poor economies. Estimates suggest that the world population activity engaged in urban agriculture is 800 million (United Nations Development Programme 1996). In many cities, in addition to backyards and vacant land, urban farmers are increasingly considering rooftops and even vertical farms as locations for food production (Despommier 2009).

There are multiple benefits from urban agriculture including job provision, nutrition improvement, and reduction of food insecurity. Local food production also helps to enhance several local ecosystem services. Some of the most significant ways that urban agriculture enhances services in cities is through lowering food imports into cities and reducing waste exports. Local food production can be used for bio- and phyto-remediation and help to prevent erosion and landslides (Halweil and Nierenberg 2007). Locally grown food systems help to develop community environmental awareness, enhance urban park management, prevent the development of "urban deserts," and can reduce greenhouse gas emissions (Pollan 2006; RUAF 2006).

For centuries, Asians have been using aquaculture ponds enriched with human wastes to grow plants, rear fish, control floodwaters, and remove local pollutants. In some cities, this is a major industry. For example, in Bangkok, production of morning glory, water mimosa, and freshwater fish within the city is meeting current demand. Indeed, nearly a third of the nation's intensive urban aquaculture production comes from around Bangkok, generating approximately US$75 million annually. Wastewater from sewage systems flows into Beung Cheung Ek Lake, Cambodia, where this nutrient facilitates the growth of water spinach for thousands of families. At the same time, this production helps to enhance the regulation services of the lake. Similarly, in the wetlands of East Kolkata, the city dumps its wastewater and over 250 wild and farmed fisheries produce approximately 18,000 tons of fish for the city, supporting 60,000 residents while helping to improve the water quality (RUAF 2006; Halweil and Nierenberg 2007).

Nevertheless, challenges exist for farming in cities. Not only is finding land to cultivate plants difficult, but in many cities, livestock are also prohibited. Moreover, the soils within urban areas are typically highly polluted. Urban wastewater used to produce food contains a whole range of pathogens that can survive for weeks after being applied, posing public health threats. Given that urban farming is typically informal, there is an urgent need for government regulation, education, and investment in this growing area (Halweil and Nierenberg 2007).

Box 8.2 Forests in Latin America

Greater Florianópolis, in the coastal zone of southern Brazil, provides an example of rapid urbanization occurring in concert with forest recovery and associated social-ecological benefits in a globalizing middle-income city (see Baptista 2008). Regenerating forests near urban areas have gained value because of the multiple ecosystem services they provide including their recreational and landscape amenity values. Developers and the more affluent segments of the society have invested in real estate, infrastructure, and services in the city's most desirable locations. The poor, on the other hand, have been left with limited alternatives to meet their housing and other basic needs and thus have established and expanded *favelas* on public lands as well as informal settlements on private lands. *Favelas* are often situated in areas, such as hillslopes and mangroves, that are legally protected by environmental legislation. The case of Florianópolis highlights the importance of understanding and shaping the role of policies, institutions, and the quality of governance to achieve poverty reduction, promote social inclusion and direct democracy, avoid extreme levels of social inequality, and maintain healthy ecosystem functions. Over the twentieth century, rapid urbanization in Brazil unfolded with minimal government intervention at the federal, state, and municipal levels, contributing to a number of serious socioeconomic and environmental problems. Brazil's City Statute (*Estatuto da Cidade*; Brazilian Federal Law No. 10257, enacted in 2001), which regulates and extends Articles 182 and 183 in Brazil's 1988 Federal Constitution, recognizes the Lefebvrian principle of the "Right to the City" and elaborates on the principle of the "social function of property and of the city" to address property rights, land tenure, urban planning, and development as human rights issues (see Fernandes 2007; Macedo 2008). Now that this legislative-institutional framework is in place, stakeholders are convening in cities across Brazil to develop participatory municipal master plans. The evolving inclusive participatory urban planning process in Brazil has the potential to provide effective arenas for developing land-use and resource management strategies that are in compliance with environmental legislation and that also respect the basic needs as well as aspirations of the poorest and most vulnerable populations in the city. Further research is needed in the various Brazilian cities involved to evaluate how the participatory municipal master plans are unfolding.

Conclusion: Reconsidering the Relationships Between Urbanization, Poverty and the Environment

At a recent workshop designed to produce positive future visions for the planet in approximately 50 years, a participant expressed horror at the thought of an urbanized world. This person voiced a common hesitation; a recent UN report suggests that decision makers are increasingly concerned about urbanization levels in their respective countries (UNFPA 2007). Certainly, future predictions based upon current trends in urbanization and slums and squatter settlement expansion, with associated health and social implications, are disturbing. Yet, even given this improbable scenario, we question whether the material conditions and chances for those predicted 2.9 billion new urbanites can improve while their negative per capita environmental impact can be reduced by spreading them out more evenly throughout their respective country landscapes? Our answer is no. We believe that they would be worse off, their chances for a better life would be lower, and collectively they would have greater environmental impact. Urbanization *is* part of the answer to our greatest future challenges.

Underpinning our discussion about the entire debate, however, has been an unresolved definition of terms. Urbanization is seen by those in the anti-urban group as a forcing influence, as if the tendency for populations to move to cities can be easily teased out from growing wealth and economic structural and cultural change, among a host of other processes that accompany development. We see urbanization as intimately intertwined with a number of social and biophysical processes, playing the roles of both an impact and a driver of change. As Lowry (1990, pp. 149) has suggested, "most of the ills that are blamed on urbanization can be more accurately attributed to population growth, industrialization and prosperity. Although these factors have in different ways encouraged urbanization, they would continue to cause economic dislocation and ecological disaster even if global urbanization were to halt and reverse."

Urbanization, like many of the factors bound together in the development process, is a mediating variable and should not be singled out as the environmental or economic villain (de Sherbinin et al. 2007). Urbanization, inherently, is neither good nor bad for local, regional, or global ecosystem services. The difference between those cities that have major environmental health and ecological burdens and those that have reduced impacts is in the design and governance of the city (McGranahan et al. 2005). As in the historical cases of investment in cities in the eighteenth and nineteenth centuries, governments have a critical role to play in making cities healthier and less environmentally damaging (Annez and Buckley 2009).

Finally, we strongly believe that blaming urbanization for poverty or for ecosystem damage can lead to policies that exacerbate these challenges. Attempts to prevent urbanization, if possible, will not only lead to lower economic growth and lessen poverty reduction but also potentially create greater environmental harm, placing more demands on agricultural land, wilderness, and lands used for forestry (Martine 2001) and make environmental governance and service provision more difficult and expensive. We attempt not to over or underplay the role of urbanization in poverty reduction and environmental damage, but rather, see the process of population concentration as a necessary, but not sufficient, feature in creating sustainable local, regional, and global societies.

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Chapter 9
Introduction to Innovative Financing: The Role of Payments for Ecosystem Services in Poverty Reduction

Jane Carter Ingram

The goal of balancing biodiversity conservation with poverty reduction has challenged conservationists and development practitioners for years (Adams et al. 2004). Efforts such as Community Based Conservation, Integrated Conservation and Development Programs, and sustainable forest management have all attempted to do this, but the linkages between conservation and economic benefits for communities have often been too indirect or vague for these approaches to achieve both goals. Furthermore, in many of these cases, the tradeoffs among conservation and poverty reduction have often outweighed the synergies (Wunder 2007). For these reasons, innovative financial mechanisms, such as Payments for Ecosystem Services (PES), have emerged as a more efficient, and direct way to balance conservation and development (Ferraro and Kiss 2002). While many functioning PES programs have been implemented in developed countries, the idea of PES has become attractive in poor, rural areas of tropical counties, where there are high concentrations of biodiversity that support a range of ecosystem services and where payments may help reduce poverty of poor, rural ecosystem service managers. However, there are few examples and analyses of the enabling conditions needed to establish PES programs in developing countries and what their success has been for supporting livelihoods and conservation. Thus, it is difficult to judge if these mechanisms are as promising as they seem to be for achieving both conservation and poverty reduction.

In this section, Michael Jenkins introduces the concept of PES, the status of ecosystem service markets, particularly carbon, and outlines key challenges to the future of PES where ecologists can specifically contribute. These challenges include matching the spatial and temporal scale of ecosystem provisioning to scales that align with management practices and the marketplace. Finally, he points to the need to better understand how the 'bundling' and 'stacking' of ecosystem services and payments

J.C. Ingram (✉)
Wildlife Conservation Society,
2300 Southern Boulevard, Bronx, NY 10460, USA
e-mail: cingram@wcs.org

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may work. Getting this right ecologically and economically, to benefit small-holder farmers, is by no means a straightforward undertaking.

There are few cases where PES schemes have been employed as a strategy for conserving wildlife in developing countries; however, Sachedina and Nelson explore how market-based approaches can help achieve wildlife conservation goals and influence livelihood diversification in ways that are compatible with conservation. They describe a functioning conservation easement that has been established outside of Tarangire National Park in Tanzania whereby private tour operators pay villagers not to convert grasslands important for wildlife grazing into agriculture. The authors describe the importance of ecological science for determining the design of the program and the importance of relationships and trust in implementing the scheme. While the program is not generating large revenues, the PES funds are the only source of discretionary funds that can be used for community development projects that are decided upon by community members/leaders. Due to the communal nature of the revenue, the funds have helped support the development of local governance institutions as the community must collectively decide how to allocate and spend the income. A key lesson that emerges from this case study is that the local economic opportunity costs upon which PES agreements need to be negotiated are shaped not only by theoretical land values or productive potentials, but by social and political factors as well.

Fisher discusses the concept of social traps and how PES programs are an attempt to overcome such snares. Through an analysis of PES potential in the Eastern Arc of Tanzania, he illustrates the challenges of using one policy tool, such as PES, to achieve more than one objective. Fisher demonstrates that tradeoffs will be inevitable when trying to balance two goals such as poverty reduction and ecosystem service conservation. However, the role that PES can play in reducing poverty is a nuanced one, founded upon understanding the complex nature of poverty, ecological functioning and the interactions between the two systems.

Corbera and Estrada discuss the growing carbon market and the expectations that have been put upon it to deliver large amounts of money to stem emissions from land-cover change and to reduce poverty in places where the majority of land-based emissions are produced. Their chapter reveals that forest-based emissions reductions projects show mixed, nuanced, and context-specific outcomes regarding their contribution to climate mitigation, biodiversity conservation, and poverty reduction. For example, in their review of voluntary forest carbon projects, they found that the most common benefits for poor farmers and communities are an increase in disposable income, a diversification of livelihood systems into reforestation activities or more profitable conservation of existing forests, local organizational strengthening, the provision of collective goods, and legal recognition or reinforcement of land tenure systems. However, existing projects have also produced some negative impacts and experience is showing that even the most well-intended, pro-poor projects can exacerbate inequalities due to uneven access to carbon funding, as poorest households' participation may be constrained by limited land assets and available labor. They emphasize the importance of understanding the ecological context when developing forest-based carbon projects; without this, projects can have neutral to disastrous ecological consequences. Finally, the authors state that it will be critical to ensure that existing national and international support to the rural poor will be complemented – and not replaced – by carbon market funds, and that their rights and livelihoods will not be jeopardized by the use of new and/or reformed carbon mechanisms.

Finally, Estrada and DeClerck discuss the use of ecological tools to assess the potential of a landscape to generate multiple ecosystem services in Costa Rica. With a case study from the Volcan Central Talamanca Biological Corridor, they demonstrate the importance of understanding the ecology and spatial distribution of ecosystem services across a landscape to improve the design and increase the efficiency of PES schemes. The authors also emphasize the potential benefits that farmers could derive from selling multiple ecosystem services through bundling or stacking, as Jenkins also discusses in this section, and from managing a landscape to generate myriad ecosystem services, many of which may not be marketable but add value to farm practices.

Thus, it is clear that while PES has a lot of potential for reducing poverty and conserving ecosystem services through both large-scale mechanisms such as the global carbon market or smaller scale, local initiatives such as the conservation easement in Tanzania discussed in this section, the design, implementation, and benefits from PES are not straightforward and often involve compromises. Nevertheless, evidence, some of which is presented in this section, demonstrates that when sound ecological science is combined with an understanding of local social, political, economic, and cultural contexts to develop these programs, PES or PES-like approaches can deliver promising results in terms of protecting ecosystems and supporting rural livelihoods with monetary and non-monetary benefits.

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Chapter 10
An Overview of Payments for Ecosystem Services

Michael Jenkins

Introduction

In the twenty-first century, we are challenged to dramatically transform the way we view, value, and manage our planet's ecosystems. Healthy ecosystems have long been understood to produce 'goods,' such as food and fiber, that have a market value within the prevailing economic paradigm; however, the 'services' of these ecosystems have either been undervalued or not valued at all. These are ecosystems that provide trillions of dollars worth of clean water, flood protection, fertile lands, clean air, pollination, and disease control. These services are essential to maintaining livable conditions and are delivered by the world's ecosystems, in effect, the world's largest 'utilities.' Yet, over 60% of these 'utilities' are on the verge of collapse or are being used in ways that cannot be sustained (Millennium Ecosystem Assessment 2005).

In response to these growing pressures, markets and market-like instruments are emerging for ecosystem services all around the world. As the chapters in this section further describe, formal markets, many voluntary and others mandated by regulation, now exist for a wide spectrum of ecosystem services related to greenhouse gases (as discussed by Estrada and Corbera), water (as discussed by Fisher), and biodiversity (as discussed by Sachedina and Nelson). Additional individual Payments for Ecosystem Services (PES) are being forged to invest in restoration and maintenance of particular ecological systems and the services they provide. A parallel explosion of interest in the science and economics of ecosystem services has taken place particularly in the last decade. Governments have begun to experiment with 'cap-and-trade' mechanisms and other market-based incentives for ecosystem services, in an effort to mobilize private sector efforts to advance public goals. Major financial institutions have created environmental markets and carbon

M. Jenkins (✉)
Forest Trends, 1050 Potomac Street NW, Washington, DC 20007, USA
e-mail: mjenkins@forest-trends.org

J.C. Ingram et al. (eds.), *Integrating Ecology and Poverty Reduction: The Application of Ecology in Development Solutions*, DOI 10.1007/978-1-4614-0186-5_10,

trading units. Businesses have been early pioneers voluntarily engaging in carbon, water, and biodiversity market-like instruments preparing for the future. Development agencies and resource economists have realized that new funding for ecosystem services could be a very major supplement to international aid for sustainable development and a way to finally reach the scale of investment commensurate with the scale of our global environmental issues. Increasing attention is also being given to how these new environmental service markets relate to indigenous and rural communities as instruments to address poverty. In the middle of this are conservation biologists, ecologists, and foresters scrambling to provide the foundational science that helps us measure ecosystem service function and struggling to weave ecosystem services into the global economy in ways that will harness markets toward the stewardship of land and sea.

The potential scale of the emerging PES market instruments and the rapid pace of their development should give us both great hope and cause for concern. While still young or emergent, changing rapidly, and spread out over geography, they have nevertheless created great expectations around the globe. The most dramatic example of this is carbon markets and forests. The contribution of deforestation and degradation emissions to climate change has been quantified, and the international community is now hastening to create incentives for avoided deforestation. At the United Nations Framework Convention on Climate Change (UNFCCC) Conference of the Parties-COP 13 in Bali in December 2007, international consensus grew on the feasibility, indeed necessity, of including *the reduction of emissions from deforestation and degradation in developing countries* (REDD) as a strategy for meeting the climate change and sustainable development objectives of the Convention. Since COP 13, developing countries and project developers have been encouraged to invest in REDD-related activities, and several major REDD initiatives have begun to move forward. In addition to the Government of Norway's landmark commitments to REDD, a variety of governmental and non-governmental actors have also launched important initiatives. Australia established an International Forest Carbon Initiative to research and sponsor REDD pilot projects in Oceania, particularly Indonesia and Papua New Guinea. The World Bank launched, in late 2007, the Forest Carbon Partnership Facility to build REDD capacity and sponsor a series of pilot projects to test the viability of compensated reduction schemes in developing countries. Just recently, they have added the Forest Investment Program (FIP). The United Nations' Agencies' REDD Initiatives are planning to provide support to readiness activities. Many international NGO's have also developed REDD initiatives. In parallel, brokers in the voluntary carbon markets, seeing the potential for REDD as a source of real emissions reductions with important, and marketable, co-benefits, have also begun to move aggressively in this space, including significant investments from banks such as JP Morgan, the Macquarie Group, Morgan Stanley, and others (Ecosystem Marketplace 2011).

While in many ways the COP 15 in Copenhagen in December 2009 was a failure, forests were the bright spot in the negotiations. Cancun (COP 16) in Mexico in late 2010 was another step towards cementing the role of forests and other agriculture and land use-terrestrial carbon into any future international climate mitigation framework; developing a forest/terrestrial carbon financing mechanism; and building the capacities that can lead to real emissions reductions and local pro-poor benefits in the years ahead.

While carbon markets dominate the discussions today, there are growing examples of payments and markets associated with water, as well as, biodiversity that will become as important in years to come. Some believe water will be the next carbon market in terms of scale and scope. Combining current estimates of government-mediated watershed PES (in New York and China, for example), voluntary watershed management payments (Vittel, Coca-Cola), and compliant water quality trading schemes (Hunter River in Australia, and Pennsylvania, United States), the Katoomba Group's Ecosystem Marketplace estimated that the total of actively traded water markets from 1994 to 2005 was $373,655,000.00 (Ecosystem Marketplace 2011). While carbon markets will help us move away from a carbon economy, we will never be able to find substitutes for water.

Biodiversity markets including conservation easements, regulated wetland, stream and endangered species mitigation markets are also growing. The United States is the largest market today, estimated to be over $4 billion. Australia has become the leader in innovation, and new programs and interest in Brazil, Colombia, South Africa, and Uganda are emerging. Voluntary biodiversity offset markets are also growing and are being driven by businesses (oil, gas, mining, agriculture) that are anticipating increased regulation in the future (Ecosystem Marketplace 2011).

Yet, this rapid growth of PES presents significant challenges and very legitimate concerns. While there is enormous potential in these new 'market-like' approaches to managing ecosystem services, there is no guarantee that this potential will be realized. Structuring incentive schemes around ecosystem services is a complex process, and pilot schemes have shown that agencies must not only get the science, economics, and institutional frameworks right for ecosystem service payment schemes to work, but also be able to tailor them to the social, political, and cultural realities at hand. In Chap. 3 of this section, Sachedina and Nelson, discuss at length the importance of designing PES programs to fit within local, social and cultural contexts.

PES, Rural Communities, and Poverty

Perhaps, the greatest of challenges to these new environmental markets is the expectation that they can address not only environmental problems, but also global poverty in rural areas. Without question, the future of many of these ecosystems (such as forests and coral reefs) and the future of millions of the world's poorest people are inextricably linked. Rural poverty is high in many areas where the world's biodiversity is most threatened. More than a billion people now live within the world's 19 forest biodiversity "hotspots" and population growth in the world's tropical wilderness areas is 3.1%, over twice the world's average rate of growth.

> ***Communities and small and medium landowners have a crucial role to play in reducing emissions***
>
> Nearly 50% of deforestation and degradation is driven by subsistence activities (Blaser and Robledo 2008)
>
> Reducing deforestation from slash-and-burn agricultural conversion could deliver ~2 $GtCO_2e$ per year in low-cost emissions reductions (McKinsey & Company, 2009)
>
> 26% of the remaining forests in the developing world are under indigenous or community ownership and management (Sunderlin et al 2008).

Indeed, in many areas, ecosystem services and products are the principal assets of the rural poor and the most proximate opportunity for poverty reduction. At the same time, many indigenous peoples' organizations are now expressing strong concerns over their lack of participation in international REDD negotiations and the potential impacts REDD mechanisms may have on their rights to land, resources, and cultural survival. At this early stage of REDD development, there is an opportunity to aggressively address the high transaction costs for communities and concerns about carbon property rights in areas with collective or customary tenure. These issues are not likely to be unique to REDD, but will also emerge in other growing ecosystem service markets.

Global mitigation efforts and finance to combat climate change will need to have significant engagement of communities and smallholders, as agents of deforestation and/or as stewards of forests, if this abatement potential is to be realized (see box). Solutions must simultaneously address global needs to combat the risks of climate change and respond to priorities in the developing world of sustainable livelihoods and economic development.

Unfortunately, while the costs of climate change are easily socialized and fall dramatically on the world's poorest and most vulnerable, finding equitable mechanisms for engaging these same rural populations in reducing deforestation or increasing reforestation is far more challenging.

Key Forest Carbon Community Challenges and Opportunities

Among the key barriers to implementing PES programs in developing countries are a critical deficit of capacity for developing these projects, a shortage of working projects demonstrating results on the ground, the lack of policy and legal frameworks to recognize and stimulate opportunities, and a paucity of business models that facilitate market access for communities and small and medium producers. If some of these challenges can be effectively addressed, there will be opportunities with great potential for conservation of the world's ecosystems and support for local livelihoods. To illustrate these key points, it is useful to explore how they pertain to REDD as an example within a burgeoning global ecosystem services market:

- *REDD and indigenous peoples*: Indigenous peoples own roughly 22% of the world's tropical forests and have historically proven to be effective stewards, with average deforestation rates significantly below those of areas dominated by non-indigenous agricultural and ranching populations. However, processes of cultural and economic change are likely to increase pressures on these forests in the next two decades – a process which might be exacerbated perversely by leakage effects from effective deforestation reduction activities by other populations. Proactive mechanisms are needed to ensure the protection and sustainable use of these indigenously managed standing forests, an imperative accentuated by the growing political voice and concerns raised by indigenous peoples and their allies in international negotiations.
- *Community management of public forest lands*: Currently, 5% of tropical forests are under devolved community administration, allowing for local management and benefits. Under current trends, the area under community administration is likely to double (to 700–800 million hectares) between 2000 and 2015 (Molnar et al. 2004). While mechanisms have been put in place around the world to stimulate community resource management, benefits received are often not sufficient to offset local management and opportunity costs. Positive REDD incentives, combined with adequate local capacity and a supportive national policy framework can potentially tip the scales in favor of local stakeholders' forest conservation. However, a lack of adequate incentives and management capacity will make these forests highly vulnerable to deforestation and degradation. Effective project approaches could be scaled up more quickly through better, more effectively designed policy mechanisms in many countries.
- *Aggregation strategies for reforestation*: Smallholder activity accounts for nearly 30% of reforested areas to date, which indicates the scope in this sector for significantly contributing to activities that support the achievement of the 1.4 GtCO2e per year of abatement potential estimated for reforestation of degraded lands to 2030 (McKinsey & Company 2009). Key challenges in this sector include legal issues around tree and carbon tenure; cost-effective methods for planting, monitoring, and verification; and, fundamentally, the question of scale: how can large numbers of small producers effectively get individually small volumes of emissions reductions to market? This scale challenge is not unlike those successfully resolved by small-holder produced commodities supplying global markets or by microfinance institutions. Working to explore possibilities for layering carbon finance for smallholder reforestation onto existing platforms for cocoa and coffee production and for delivery of rural micro-credit holds promise.

Getting the Science Right- and Linking to Policy and Market Development

Even before facing the obstacles of tenure, ownership, and benefit sharing comes the necessity of building a solid PES foundation from science that links ecosystem function and services to land-use practices. Science and ecology will play a critical role in getting the ecological relationships right in these new markets. Linking this science to the policy development will be critical to ensure credibility. Additionally, linking carbon markets to rural producers will depend on our ability to create the 'instrumentation' of these markets – the measuring and monitoring tools that are accurate, efficient, and that can work for small-scale producers.

While the overwhelming conclusion of the 1992 Rio Earth Summit was the need to combine the three goals of social, environment, and economic development, 25 years later, true sustainable development remains elusive. Although some gains have been made, conditions of the rural poor and our natural ecosystems have continued to deteriorate and our institutional capacity to take on more integrated approaches has been clearly inadequate.

Markets and payments for delivering ecosystem services may well be the final element to help bring about this transformation to properly value our precious ecosystems. It represents a major transition from the traditional donor paradigm of providing development assistance through grants with conditionality to a model of a business 'contract' between partners. The key characteristic is the focus on maintaining a flow of a specified ecosystem service such as clean water, biodiversity habitat, or carbon sequestration capabilities – in exchange for something of economic value. There are layers of complexity around this new scenario including resource access and ownership and institutional capacity and transparency. Equally as central to this 'contract' is ensuring that the ecological service is indeed maintained – as buyers will expect and demand this as a return for their investment. These linkages will require ongoing verification that the service or ecological function is indeed being delivered as a result of the natural resource management practice implemented.

Ecologists and other conservation scientists and practitioners will have a fundamental role to play for ecosystem service markets and payments to reach their full potential; yet, they themselves and the institutions they represent face the major challenge in integrating more effectively with other actors in these PES markets. To ensure that the science and conservation goals are relevant and adopted by policy makers and project developers, it is equally critical that ecologists expand their own boundaries and reach out to other types of institutions (rural development agencies, farmer organizations, NGOs) to practically ground their work and to private sector companies, financial institutions, and multilaterals to ensure this research is informing market activity and policy. Classically trained ecologists, foresters, and biologists, for example, will be forced to learn new 'languages' to speak to these other stakeholders if they will have an impact on the development of these markets.

Two looming PES challenges that ecologists can help tackle:

1. Ecosystem functions and services are best delivered at landscape scales. Using landscapes or watershed frameworks for planning and implementation gives us flexibility to optimize other co-benefits like sustainable development and energy. For example, an integrated approach to landscape-level carbon accounting for application in heterogeneous, dynamic landscapes is needed to account for diverse land-use practices and to enable rigorous but more cost-effective monitoring of large area carbon sequestration, rather than individual accounting of multiple small holder-owned parcels that comprise the landscape. At present, efforts by small-scale agricultural and forestland smallholders to gain access to the carbon market are hampered by high transaction costs associated with institutional difficulties, such as lack of aggregation of buyers and sellers. Challenges also include the different time scales at which various stakeholders act and make decisions, a variety of expectations on investment returns, risk management, and terms of contracts. For these reasons, current land-based carbon investments in the voluntary market are predominantly comprised of relatively small numbers of farmers. Alternative approaches that enable participation of larger numbers of farmers are needed to really have an effect on a landscape scale – the scale at which diverse types of activities could contribute to addressing total greenhouse gas sequestration or emissions reductions while maintaining the provisioning of a range of ecosystem services.
2. Many people point to the ‘bundling’ or ‘stacking’ of ecosystem services and payments as the ‘holy grail’ of these environmental markets. Simply put, bundling is the way that a range of ecological/natural resource values can be integrated into one unit or credit. In principle, a bundled credit would be valued for all of its ecological values and demand a higher price. There are a number of examples of pioneering work towards this bundled credit approach, such as the work of the Mexican NGO Sierra Gorda. ‘Stacking’ is a similar concept where natural resource values are sold as credits (carbon, wetland, water quality) independently. The financial attraction for stacking is that you get multiple revenues from one parcel of land and risks associated with one market can be spread across several “products.” A major hurdle for stacking to gain traction with markets is the lack of experience of mixing different metrics (carbon tons, water liters, biodiversity habitat, etc.) and dealing with the issue of additionality.

Both of these promising opportunities suffer from few rigorous assessments of the ecological impacts of these market-based approaches. Practical cross disciplinary/sector experience is urgently needed, and this needs to be a rallying cry amongst the conservation community, if work is to be relevant and successful. Ultimately, it will be the private and financial sectors that will determine the scale of these emerging PES markets, but policy makers will play an important role in shaping these markets with regulation, and the community of ecologists, conservationists, and development practitioners have the opportunity to help ‘get the rules right’ ensuring real conservation outcomes and benefits to local communities.

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Chapter 11
The Potential of Carbon Offsetting Projects in the Forestry Sector for Poverty Reduction in Developing Countries

Manuel Estrada and Esteve Corbera

Introduction

The international carbon market – comprising both the regulated national, regional, and international markets resulting from the implementation of the Kyoto Protocol and the voluntary trade of carbon offsets by individuals, companies, NGOs, and governments outside the Kyoto framework – is currently considered the most important new and additional source of development finance, valued at US$126 billion in 2008 and potentially exceeding USD$50–120 billion/year in the long term (Capoor and Ambrosi 2009). Given the great potential for the implementation of alternatives to mitigate carbon emissions in the Land Use, Land Use and Forestry (LULUCF) sector in the tropics and the fact that over 70% of the world's poor are located in rural areas, great expectations have been put on the capacity of this innovative source of funding to support rural poverty reduction initiatives in developing countries.

In fact, according to the Intergovernmental Panel on Climate Change (IPCC), in the short term (2008–2012), the potential area available for afforestation and reforestation (AR) activities under the Clean Development Mechanism (CDM) is estimated to be 5.3 million ha in Africa, Asia, and Latin America together, with Asia accounting for 4.4 million ha (IPCC 2007). What is more, the cost of carbon sequestration projects in such countries appears to be competitive (ranging from 0.5 US$ to 7 US$/t$CO_2$) when compared to that of similar projects in developed countries (1.4 US$ to 22 US$/t$CO_2$), to the market price of Certified Emissions Reductions (CERs)

M. Estrada (✉)
Independent Climate Change Consultant, Mexico
e-mail: mporruacop9@gmail.com

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generated through the CDM (€7 to €22[1]) and to the price of European Union Allowances (EUAs) (€7 to €27[2]) in 2009. Important mitigation potential also exists in other LULUCF activities, particularly through avoiding emissions from deforestation, which currently is not an eligible projects activity under the CDM, but that may enter the regulated carbon market after 2012 and is currently allowed in the voluntary offset market. It has been estimated that in a short-term context (2008–2012), 93% of the total mitigation potential in the tropics corresponds to avoided deforestation. Looking at the long term, for 27.2 US$/t$CO_2$, deforestation could potentially be virtually eliminated (IPCC 2007). Additional mitigation options available to low-income communities and individuals include, for instance, the sustainable use of biomass (e.g. by using efficient cook stoves) and agricultural carbon conservation and sequestration practices (e.g. no tillage).

However, the mitigation potentials and cost-effectiveness figures outlined above should be assessed carefully and not be taken for granted. The experience of many developing countries in designing and implementing forest governance and conservation policies has shown that this is a costly process, with recurrent and often increasing costs, as a result of changing land-use dynamics over time and the multiplicity of actors involved and dependent upon the LULUCF sector (Corbera et al. 2009a). For carbon offsetting activities in the LULUCF sector to actually contribute to rural poverty reduction in developing countries, a demand for such carbon emission reductions must exist and enough capacities and resources should be in place to allow for smallholders to design and implement projects able to respond to their needs while resulting appealing to carbon buyers. What follows is an analysis of the current state and trends of the regulated and voluntary carbon markets, with particular focus on the situation of small-scale LULUCF projects (the most common size of community projects), as well as a review of experiences and lessons learned from the implementation of carbon sequestration projects in low-income communities in developing countries. In the concluding remarks, a brief reflection of what the future of small-scale LULUCF could be based on recent market and policy developments is presented.

Small-Scale LULUCF Projects in the Regulated Market

The CDM – a projects-based market approach aimed at assisting developed countries to cost effectively reduce their emissions of Greenhouse Gases (GHG) to the levels and within the timeframes required by the Kyoto Protocol (5.2% from 1990

[1] These figures reflect a higher price volatility in CERs than compared to previous years, when prices for primary market forward transactions were in the range of €8–13 in 2007 and €17–22 in 2008. Prices in the higher end of that range typically rewarded projects that were further along in the CDM process (such as registered projects), projects that were being developed by experienced and established sponsors (low credit risk and performance risk), and/or for projects with high expected issuance yields (Capoor and Ambrosi 2008; December 2009 and 2008 Market Report, www.sendeco2.com).

[2] This figure refers to the maximum and minimum price for EUAs during 2009 (December 2009 Market Report, www.sendeco2.com).

levels during the period 2008–2012) while promoting the sustainable development of developing countries – has proved to be a successful vehicle for funding carbon mitigation activities, mobilizing over US$ seven and six thousand million in 2007 and 2008, respectively (Capoor and Ambrosi 2009). In contrast, its value as a sustainable development tool is still to be demonstrated. So far, buyers in the CDM market have favored options resulting in large volumes of credits, with low costs per ton of carbon mitigated and reduced risk profiles, such as mitigating emissions of industrial gases or methane from landfills and animal waste. Consequently, projects with high sustainable development benefits – normally less massive, more costly and risky –, particularly in the least developed countries (i.e. Africa), are almost non-existent in the regulated market (Boyd et al. 2009; Ellis et al. 2007).

The potential of the CDM to alleviate rural poverty was severely reduced at the very inception of the mechanism, when LULUCF activities other than AR activities, such as Reducing Emissions from Deforestation and forest Degradation (REDD) and agricultural soil carbon management were not allowed during the first commitment period of the Kyoto Protocol. This potential has been further minimized in practice by policy decisions and market preferences. As of January 2010, only 13 AR CDM project had been registered by the CDM Executive Board – the entity in charge of overseeing the operation of the mechanism – of which six are small-scale activities, out of a total of 2,029 registered projects (UNFCCC 2010).

One can identify at least three main causes behind the lack of AR projects in the CDM: first, market-design issues and participants' considerations regarding these projects; second, landholders' and project developers' financial constraints; and third, lack of technical knowledge to design and manage such type of projects. The exclusion of CERs from AR projects from the European Union Emissions Trading Scheme (EU ETS) – the largest source of demand in the regulated market so far – undermined the attractiveness of this mitigation option, due to some EU countries' concerns regarding the reversibility of biological carbon sequestration, the displacement of carbon emitting activities (leakage), and the potentially large amount of forestry credits that could enter the EU ETS (arguably postponing the implementation of domestic policies and measures and, thus, delaying the development and competitiveness of climate friendly energy technologies) (Streck 2008). There has also been a relatively negative perception of forestry projects and credits by buyers, which can be attributed to the likely challenges involved in collecting data about carbon stocks and flows, high risks and transaction costs associated with working with communities, securing and enforcing property rights over land and the carbon sequestered, and the future permanence of forests and plantations. For example, a survey carried out among European and Japanese companies and major carbon funds indeed showed that some of the main perceived obstacles to investments into forestry projects are the temporary character of AR CDM CERs or "tCERs" (they must be replaced before their expiry date with an equal amount of credits) (27% of respondents), the high perceived risk of this type of projects (16%), and the need to provide upfront payments for late credit deliveries (5%) (Dannecker 2005).

On top of the adverse policy and market circumstances noted above, in order to qualify for the use of simplified modalities and procedures (aimed at reducing transaction costs), small-scale projects (as well as bundles of such projects) were originally limited to a maximum size of 8,000 tCO_2 per year – a figure that was the result of a political decision during the negotiations but which is considered many times lower than the minimum size required to cover the typical transaction costs associated to these projects (for instance, based on its operational experience, the World Bank has recommended increasing the limit to 32,000 tCO_2/year) (FCCC/SBSTA/2007/MISC.1). In December 2007, the limit was revised upward, to 16,000 tCO_2e/year (Decision 9/CMP.3), but the effects of this change – if any – on projects' feasibility remain to be seen. In addition, small-scale projects suffer a competitive disadvantage in the regulated market, since compliance purchasers usually demand volumes over 100,000 CERs per year per project, so as to cover the transaction costs implied, for example, by the negotiation of Emissions Reductions Purchase Agreements (ERPAs) with project developers. This situation is somehow less common in the voluntary market, where offset purchasers generally have far smaller volume requirements (Tyler 2007).

A recent paper analyzing the design and early implementation of four CDM AR projects has been able to identify additional financial and technical constraints to the development of forestry projects under the CDM (Thomas et al. 2009). On the one hand, its authors emphasize that "the most critical financial constraint on the development of CDM AR projects is the length of time it takes to gain revenue from a CDM A/R project" (Thomas et al. 2009, 2), as newly planted trees take a number of years to yield net sequestration benefits and, therefore, compromise buyers' interest in short-term and large volume credit revenues. According to the paper, this issue has been aggravated by project proponents' difficulties to access finance for project design and development, including transaction costs. Financial constraints are further aggravated by lack of technical knowledge as project proponents have also encountered difficulties in proving additionality (that emission removals would not occur without the project), establishing baselines (as historical data is often not available), and accounting for leakage. Furthermore, project proponents have also faced the difficulties to deal with the complexities of the forestry and land-use sectors, which include multiple interests, problems of accessing land for plantations, negotiating rights and establishing participatory (and time-consuming) processes to deal with direct and indirect beneficiaries, as well as with surrounding populations.

Small-Scale LULUCF Projects in the Voluntary Carbon Market

Voluntary schemes have traditionally prioritized the social and environmental benefits of projects, which have actually represented a large portion of the emission reduction product demanded in the market. This purchasing motivation and lower volume demand find a perfect fit with carbon projects that demonstrate a high value for sustainable development (Tyler 2007). The voluntary market reflects consumer demand for action on climate change and, in contrast with the regulated market, represents an immediate resource for a wide variety of poverty reduction projects in rural areas (for instance, voluntary transactions in avoided deforestation have occurred since before 1990, while their inclusion in the UNFCCC framework is just beginning to occur under the evolving REDD framework and the World Bank Forest Carbon Partnership Facility and UN-REDD initiatives).

Nevertheless, in the last few years, the share of forestry projects in the voluntary market has declined (Corbera et al. 2009b). In 2007, forestry projects represented 15% of the transacted volume in the voluntary Over The Counter (OTC) market[3] (of which 10% came from AR projects and 5% from REDD activities), less than half of their share in 2006 (37%) (Hamilton et al. 2008). It appears that carbon forestry deals do not enjoy the same level of support that they had in the early years of these markets, and ex ante deals may also be falling out of favor. Generally, OTC market consumers are orientating to less controversial and "charismatic" project types that have public appeal. However, not all OTC market consumers are driven by these motivations. Some companies (representing 29% of the volume supplied in 2007), particularly those in the USA, are also investing in carbon offsets with the hope of potentially selling them for compliance purposes (Hamilton et al. 2008).

Even declining, the share of forestry projects in the voluntary market still represents more credits than those generated through AR CDM projects. The 13 AR CDM projects registered to date are expected to generate 32 thousand tCO_2e per year on average over the first crediting period, with substantial differences between small-scale (4.9 $ktCO_2e$) and large-scale (55.3 thousand tCO_2e/year) (UNFCCC 2010). Meanwhile, the 2007 AR and REDD projects in the voluntary market produced a combined total of 6,315 million tCO_2e. Moreover, micro (less than 5,000 tCO_2e/year) and small-scale (5,000 to 15,000 tCO_2e/year) forestry projects accounted for 4% and 8% of the transacted volume in the voluntary carbon market in 2007, respectively, whereas no credits from small-scale AR CDM project have been issued to date. In any case, it is important to note that most of the forestry projects existing in the voluntary market are being implemented in the US, Canada, New Zealand, and Australia, and therefore, the contribution of this market to rural poverty reduction in developing countries may be quite less than the market value of the 6,315 million tCO_2e previously noted.[4]

[3] The voluntary carbon market is divided into two main segments: the voluntary, but legally binding, cap-and-trade system that is the Chicago Climate Exchange (CCX); and the broader, nonbinding, over the counter (OTC) offset market, commonly referred to as the voluntary offset market. Almost all carbon credits purchased in the OTC market originate from project-based transactions. Credits from the OTC market are often generically referred to as Verified Emissions Reductions (VERs).

[4] The price of forestry projects in the voluntary market, in particular those involving afforestation/reforestation, have remained some of the highest priced project types across 2006 and 2007 with weighted average prices of \$6.8 to \$8.2/tCO_2e. Credits from REDD projects have averaged \$4.8/tCO_2e (Hamilton et al. 2008).

Experiences to Date in the Implementation of Carbon Forestry Projects with the Participation of Low-Income Smallholders

There is scant independent research conducted on existing forestry offset projects from regulated and voluntary markets. Existing analyses reveal mixed, nuanced, and context-specific outcomes regarding their contribution to climate mitigation, biodiversity conservation, and poverty reduction. For example, an examination of an AR CDM-registered project operating in China highlights that the project can be considered both a success and a failure (Gong et al. 2010). Since 2006, the project has been developing 80 reforestation plantation sites, covering 4,000 ha and combining tree species like Pinus massoniana and Eucalyptus sp. It has developed a unique share-holding system, which involves villagers, a public forest development company, two local forest companies, and the World Bank Biocarbon Fund, which acts as a buyer of tCERs for a 30-year crediting period. On the one hand, the World Bank Biocarbon Fund signed a contract with the Luhuan Forestry Development Company from Huanjiang County, who represents all sellers under the share-holding system. In turn, the Luhuan Forestry Development Company signed individual contracts with further intermediaries, the other two forest companies, Kuangyuan and Fuyuan forest farms from Cangwu County. For communal lands, all three forest companies signed contracts with natural village leaders who then determined how revenue obtained from timber, resin, non-timber forest products, and carbon would be shared among their community members and the companies; for individual lands, the forest companies signed the contracts directly with household heads. The project has been successful in setting such a unique carbon share-holding system but its mitigation and social expectations may need to be re-assessed: only 45% of the planned area has been replanted and villagers are withdrawing from their original agreements. This can be explained by several factors. Firstly, several individual farmers are trying to renegotiate the terms of the contract with the local forest companies, as the value of timber and other products has recently increased in local markets. Secondly, some areas have seen a rise of tree planting costs because their level of degradation is higher than originally expected. And thirdly, several communities face now internal disputes over how timber and carbon revenues should be shared due to recent changes in local leadership.

Voluntary carbon forestry in developing countries also shows mixed results. The most common benefits for poor farmers and communities include an increase in disposable income, a diversification of livelihood systems into reforestation activities, or more profitable conservation of existing forests, thereby conserving biodiversity, local organizational strengthening, the provision of collective goods, and legal recognition or reinforcement of land tenure systems (Alban and Argüello 2004; Boyd et al. 2007a, b; Corbera et al. 2008; Jindal et al. 2008). But existing projects have also produced some negative impacts. Corbera and colleagues have shown that even the most well-intended pro-poor projects can exacerbate inequalities due to uneven access to carbon funding, as poorest households' participation may be constrained by limited land assets and available labor (Corbera et al. 2007). Furthermore, carbon projects, like any other development intervention, may end up prioritizing and benefitting some political allegiances and community groups over others. A review of 23 carbon forestry projects in Africa (Jindal et al. 2008) shows that some projects developing plantations in grasslands have changed ecosystem dynamics and impacted negatively on community groups who have lost access to resources. Indeed, the ecological implications of planting trees in certain ecosystems need to be carefully assessed, moving away from the presumption that forests are always good for poverty reduction and the overall local environment. Research on a carbon project located in the Ecuadorian Andes has shown that Eucalyptus plantations have a detrimental effect on available water down the catchment and contribute to reductions in soil quality (Granda 2005). In this regard, planting native, locally selected, species is key for project development, although ensuring seedlings' availability may be costly and require existing local knowledge and capacity.

Implementation challenges are also an important issue of concern. Pro-poor carbon forestry should be suited to local needs and complex socio-ecological systems. Projects require careful, inclusionary planning, as well as constant monitoring and adaptation to shifting political and social conditions of rural farmers and communities. Boyd et al. (2007a, b) emphasize that the historical context and institutional dynamics or relationships between different stakeholders are important to consider. This will require addressing the roles of stakeholders, project ownership, and, explicitly, differences in objectives, needs, and priorities. Seemingly, Tshackert and Tappan (2004) suggest that developing carbon forestry systems requires putting the farmer in the center of decision making, in order to identify suitable technical options and, if possible, a multiplicity of carbon sequestration practices, which can suit their needs, capacities, and adaptive strategies to cope with risk and uncertainty. Even if a community or farmer-oriented integral approach is pursued, it is important to bear in mind that not all poor communities and farmers may be willing to participate. Kosoy et al. (2008), for example, show that community participation in carbon projects in Mexico has been influenced by the level of trust with project developers, efficient and transparent communication procedures, as well as political views, livelihood strategies (i.e. more or less prone to forestry), and conservation values.

The cost per unit of supplying sequestration is also an important determinant of the returns producers will attain from the adoption of carbon sequestration practices, as well as their competitiveness relative to other potential suppliers (Lipper and Cavatassi 2003). The costs can be divided into two components: the actual cost of generating the sequestration (e.g. the abatement cost) and the cost of getting the sequestration to the carbon market (e.g. the transaction cost). For instance, Cacho et al. (2003) estimate the production costs per ton of carbon for four agroforestry systems on degraded lands in Sumatra and found that systems associated with smallholders were more competitive than oil palm plantations in carbon productivity. However, they also note that smallholders may indeed be less competitive than single large properties as a result of the fixed transaction costs involved in project design and implementation, such as organizing participant smallholders, designing the baseline, demonstrating additionality, preventing leakage, and measuring carbon stocks on-site. Another very detailed study of the opportunity and production costs associated with carbon sequestration in a variety of land-use systems in Sumatra also concludes that smallholder systems are competitive in sequestration production but casts doubt on their competitiveness with large-scale oil palm plantations in terms of private profitability (Tomich et al. 2001).

High transaction costs associated with the development of sound and marketable carbon projects represent a major barrier to smallholder participation in carbon markets. These costs arise from the small-scale and isolated conditions under which poor land users operate, as well as a higher degree of uncertainty in their rights to land-based property. Clearly, the costs associated with identifying, negotiating, contracting, and enforcing sequestration payments are much higher when dealing with small and geographically scattered producers, operating under heterogeneous agro-ecological and institutional conditions (Lipper and Cavatassi 2003). In this sense, it has been suggested that project developers may need to explore the possibilities of working with existing and legitimate organizations at the community level, which can act as information channels and conflict mediation agents (Corbera et al. 2007), and to enhance collaboration between stakeholders through, for example, community-based monitoring and evaluation systems (Skutsch 2005). Transaction costs can be reduced significantly by coordinating and consolidating sequestration supply among groups of poor landholders. Carbon projects which consist of a coordinated group of land-use activities such as community forestry may be conducted through local-level organizations, which are already in place, such as local governments, farmers' associations, or NGOs. However, participating under these schemes implies an additional cost to sellers that should be lower than the expected benefits in order to maintain the attractiveness of the scheme. In some cases, these costs are subsidized by the intermediary, particularly those that are interested in promoting overall development objectives, rather than pure carbon market transactions. It is also possible that future buyers may be willing to subsidize such costs by paying higher prices for sequestration credits that carry a "sustainable development" certification (Lipper and Cavatassi 2003).

Finally, another important issue to be addressed while designing carbon sequestration projects involving smallholders is the problem of complex and unclear property rights. In some cases, assistance in managing a common property may be all that is required, while in other situations, cadastral surveys and titling could be involved. The costs associated with these different options vary considerably, but can be very substantial, certainly large enough to make sequestration supply very expensive and non-competitive (Lipper and Cavatassi 2003). A carbon project in Mexico found that those communities with intractable internal conflicts were economically unfeasible for the supply of sequestration services, while those which had already achieved successful community management of resources were found to be competitive. The differences in costs for establishing community capacity for joint forest management ranged from \$52/ha in the communities with high levels of social capital, to over \$325/ha in those where conflicts were prevalent (de Jong et al. 2000).

Looking into the Future

From the review presented above, it seems evident that the current conditions of the regulated global carbon market offer little room for the implementation of small-scale LULUCF projects in developing countries, which are arguably the best (and sometimes unique) option for the rural poor to access carbon funds. However, a number of recent developments seem to indicate that this situation may be changing in the near future. The recently enhanced public awareness on the issues of climate change, forests, and poverty has certainly influenced the negotiations under the UNFCCC toward a wider consideration of the LULUCF sector in developing countries, in particular, the establishment of an incentives mechanism to address deforestation and forest degradation, as well as the conservation and enhancement of forest carbon stocks (REDD+) in developing countries.

The Eliasch Review – an independent report to the UK government aimed at providing a comprehensive analysis of international financing to reduce forest loss and its associated impacts on climate change – has noted that, without tackling forest loss in developing countries, it would be highly unlikely to achieve the stabilization of greenhouse gas concentrations in the atmosphere at a level that avoids the worst effects of climate change. It has also stressed that a deal providing international forest financing could also benefit developing countries, support poverty reduction, and help preserve biodiversity and other forest services. Rural smallholders, and particularly some indigenous groups in tropical countries, still hold large tracks of forests in common, which are often threatened by external agents and drivers. REDD^{+} could thus be used to support ongoing conservation efforts of the rural poor.

Public awareness may also positively impact the perception of buyers in the carbon market, thus increasing the demand for forestry credits. In addition, recent studies have shown that, if adequate rules to ensure enough domestic action in EU countries are established, admitting forest credits into the EU ETS should have little or no impact on the EU carbon market price (Eliasch 2008), which might alleviate the concerns of EU countries regarding a potential delay in the adoption of new mitigation technologies and thus influence a future change in the position of the EU about the inclusion of such credits in the EU ETS.

The emergence of a number of independent protocols, standards, verification procedures, and registries to guarantee the quality of the offsets traded in the voluntary carbon market may also have positive effects on the future of LULUCF projects. Most of these standards – for instance, the Voluntary Carbon Standard (VCS), or TÜV Süd's VER^{+} Standard – focus mainly on the demonstration of the climate benefits of projects, while a few others – like the Climate, Community, and Biodiversity Standard (CCB) – address and guarantee their social and biodiversity benefits. The adoption of any of these – or, even better, the combination of carbon and social-biological performance standards – by project developers will help to minimize projects' negative impacts and risks and maximize their benefits, thus contributing to increased commercial value of the carbon credits they generate by guaranteeing their quality.

The use of social performance standards may be particularly relevant for projects developed by low-income communities, as they may increase their attractiveness in the market even if their costs per ton of carbon are not competitive. Moreover, the entrance of small-scale projects into the carbon market may be further facilitated by the recent incorporation of programmatic approaches to both the CDM and the voluntary market (i.e. the VCS), which are expected to reduce transaction costs through project bundling.

Important experiences have been gained during the last decade by development agencies, governments, NGOs, researchers, and farmers participating in forestry carbon projects in developing countries. These lessons, if widely shared and supported by adequate capacity building efforts, will be vital to improving and accelerating the access to carbon funds by low-income communities. Finally, it will be critical to ensure that existing national and international support to the rural poor will be complemented – and not replaced – by carbon money, and that their rights and livelihoods will not be jeopardized by the use of new and/or reformed carbon mechanisms.

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Chapter 12
The Development of Payments for Ecosystem Services as a Community-Based Conservation Strategy in East Africa

Hassan Sachedina and Fred Nelson

Introduction

Payments for ecosystem services (PES) are increasingly considered an important approach to solving global environmental challenges (Daily 1997; Ferraro and Kiss 2002). PES approaches provide individuals or communities with financial incentives for resource use decisions that increase the provision of ecosystem services such as water purification, flood mitigation, or carbon sequestration (Jack et al. 2008). Intense pressure on ecosystems has catalyzed the development of such market-based tools to seek to influence environmental behavior. The rationale is that incentives reduce costs for 'producers' (or stewards) of ecosystem services and prescribe more realistic values to ecosystem services, costs which, in theory, are borne by consumers (Engel et al. 2008).

For millennia, pastoralists have shared landscapes with wildlife throughout much of Africa (Homewood and Rodgers 1991; Little et al. 1999; Pilgram et al. 1990). During the twentieth century, this co-existence has been in decline as conservation policy has excluded people and livestock from protected areas, and demographic growth and expanding agriculture have displaced wildlife populations (Serneels et al. 2001; Ellis and Swift 1988; Pagiola et al. 1998; Little et al. 2001; Western and Gichohi 1993; Ottichilo et al. 2001; Homewood et al. 2001). Furthermore, many pastoral systems across the globe, including those of Maasai pastoralists in northern Tanzania, are under unprecedented pressure to diversify livestock-based economies (Little et al. 2001; Fratkin 1993; Fratkin et al. 1999). Yet, the presence of unfenced and uncultivated rangelands adjacent to PAs is critical for providing the total range of resources needed by wildlife for long-term survival as predicted by island bio-geographic theory (Western and Ssemakula 1981). In Kenya, for example,

H. Sachedina (✉)
Wildlife Works, Nairobi, Kenya
e-mail: hassansachedina@googlemail.com

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an estimated 70% of wildlife populations are dispersed outside protected areas (PAs) on land which overlaps pastoral land (Western and Gichohi 1993). Thus, the lands outside of protected areas are subject to competing claims due to their importance for multiple uses by people and wildlife.

Significant wildlife population declines have been registered throughout Tanzania and Kenya over the last 20 years, with some notable exceptions: Serengeti National Park (NP) in Tanzania and Laikipia District, Kenya. Wildlife populations are generally stable in Serengeti NP, but there have been major declines of wildlife in surrounding reserves in both Kenya and Tanzania. A significant portion of the land used by Serengeti's migratory wildlife is located within the NP, while much of Laikipia are privately owned, former large-scale cattle ranches that are now managed for conservation. These models have distinct advantages from a wildlife management perspective: they have a sole 'owner' (either the State or a corporate entity), employ dedicated PA management strategies, and usually possess a single mission of conserving wildlife and maximizing its value. Incentivizing conservation on communal land in key wildlife dispersal areas with thousands of human residents, contested land tenure, and multiple land uses is more complicated and requires different approaches in order to balance development needs with those of conservation.

There are few cases where PES schemes have been employed as a strategy for conserving wildlife in developing countries (Nelson et al. 2010). The trade-offs between conservation and development mean that only a small subset of integrated conservation and development project (ICDP) opportunities exist that really achieve environmental, economic, and social sustainability (Inamdar et al. 1999). The economic effectiveness of community-based conservation (CBC) schemes, which compensate rural people for trade-offs, such as the loss of access to resources in return for wildlife utilization revenues, often fail to live up to expectations (Warner 2000; IIED 1994; Rutten 2002; Metcalfe 1995). In addition, conservation interventions in Tanzania need to deal with a historical legacy of pastoral land alienation in the region, and decades of resentment directed at conservation efforts.

This paper explores the development of a pilot PES scheme in the Tarangire ecosystem of Tanzania in response to specific wildlife declines and policy constraints. It charts the development of this initiative from its genesis based on PES experiences in Kenya. This paper specifically explores the questions of whether the utilization of free-market enterprise tools to achieve conservation goals influences Maasai livelihood diversification in ways that are compatible with conservation. If provided with more options for diversifying their income through wildlife and livestock herding, will Tanzanian villagers protect wildlife corridors and exhibit behavior that is more conservation friendly?

The Tarangire ecosystem of northern Tanzania provides fertile ground to examine this. It is renowned for its large-scale seasonal migration of large, grazing ungulates (Kahurananga 1981, 1979; Lamprey 1963b, 1964). Of particular importance are grazing and calving areas in the Simanjiro Plains, where thousands of wildebeest (*Connochaetes taurinus)* and zebra (*Equus burchelli*) congregate during the wet season, driven largely by phosphorous-rich soil, which is deficient in Tarangire

NP. Conservation of the ecosystem's migratory wildlife populations largely depends on maintaining these habitats on communally owned lands (Borner 1982, 1985; Kahurananga 1997; TCP 1998). The progressive conversion of pastoral rangelands to large-scale farming and permanent subsistence agriculture is contributing to the insularization of Tarangire (NP) (Lamprey 1964; Borner 1985; Kahurananga 1981, 1997; TCP 1998; Kajuni et al. 1988; EcoSystems Ltd. 1980b; Peterson 1978). Continued isolation of Tarangire NP is likely to result in increased wildlife declines in the ecosystem (TCP 1998; Voeten 1999), which could threaten tourism revenues.

Local Communities and Wildlife Conservation in Simanjiro: The Historical and Institutional Context

In Tanzania, PAs cover 167,602 km^2 including national parks, the Ngorongoro Conservation Area (NCA), Game Reserves (GR), Game Controlled Areas (GCA), Wildlife Management Areas (WMA), and Forest Reserves (FR). GCA's conservation value as a PA is hazy; people can live and farm in GCA's and it seems to be more of an administrative construct to allocate hunting blocks. Approximately 30% of Tanzania's land surface is strictly protected in which cultivation and settlement are prohibited (Brockington 2006). The global goal of the 1982 World Parks Congress in Bali was to protect 10% of specific habitats (Jepson 2001: 191). Interestingly, approximately 30% of the Tarangire Ecosystem is strictly PA land, in which people are excluded.

Despite Tanzania's apparent strong record in establishing PAs, there have been some human costs. The Maasai have probably been the most severely affected group of people by PA establishment in East Africa (Neumann 1998) and are wary – even hostile in places – to conservation policies. Tarangire was gazetted as a game reserve (GR) in 1957, which caused unease in Simanjiro, as people had relatives who had recently been evicted from the Serengeti (Igoe 2004: 61). Gazettement of Tarangire NP in 1970 remains a painful memory as people were evicted forcefully by the State (Igoe and Brockington 1999). Access to valuable dry-season water and pasture resources in Tarangire was lost.

Other than exclusion from Tarangire's resources, other factors affected changing pastoral economies in the area: increasing human populations, static livestock populations, and livestock disease all contributed to weakening pastoral food security and encouraged diversification into farming (Sachedina and Trench 2009). Additionally, regional politics fomented the anti-conservation rhetoric. Farming restrictions in the Ngorongoro Conservation Area (NCA) caused some Ngorongoro Maasai to emigrate to Simanjiro District to seek farms and improved livelihoods. They warned that any process termed 'conservation' would weaken and impoverish herders in Simanjiro.

In 1982, the Frankfurt Zoological Society (FZS) proposed a multiple land-use authority covering the entire Simanjiro area modeled after the NCA (Borner 1982: 9).

The proposal for the "Simanjiro Conservation Area" cited threats to conservation from commercial farming and livestock grazing, and called for a total ban on farming within the area (Borner 1985). Subsequent government proposals called for the Simanjiro plains to be strictly protected and farming restricted (Kajuni et al. 1988). District authorities even proposed a new game reserve of 3,822 km² in the Simanjiro and Sanya Plains (URT 1993). Herders unsurprisingly opposed these schemes to appropriate more land and resources for conservation in the face of their weakening pastoral economy and declining land base (Igoe 1999, 2000, 2004; Igoe and Brockington 1999). To counter the perceived risk of Simanjiro's land appropriation, the Simanjiro Maasai became more politically aware and active, with the struggle against conservation interests serving as a rallying cry.

Tension toward conservation was fueled by national policies promoting private investment, including efforts by the Tanzania Investment Centre, to establish district-based "land-banks" comprising village lands earmarked for outside commercial investment. Herders were afraid that rangeland looked like unused "wilderness" to policymakers (WWG 2004). Villages in the Simanjiro Plains decided to sub-divide the plains to individuals to hedge against the potential threat of land appropriation. Poorer pastoralists or enterprising individuals leased land to commercial farmers who ploughed vast swathes of the plains. Villagers were partly motivated by the desire to "brand" the land; land that is ploughed is likely to be seen as owned by someone and it is also less valuable to conservation. Commercial farmers were drawn by the ease of farming in the plains; it had no trees and could be ploughed easily using tractors (Fig. 12.1).

It is important to note how relationships between pastoral communities and conservation non-governmental organizations (NGOs) soured, as it affected the future roll-out of a PES scheme. In 1985, Tanzania National Parks (TANAPA) established a Community Conservation Service (CCS) termed "Ujirani Mwema," in Kiswahili for "Good neighborliness" (Dembe and Bergin 1996; Bergin 1995). From the Maasai point of view, good neighborliness should mean access for livestock to natural resources inside Tarangire (just as wildlife graze outside the park). A key TANAPA partner was an international conservation NGO, the African Wildlife Foundation (AWF). The intention of CCS and AWF was to engage local people in conservation. AWF subsequently advocated that communities should establish wildlife corridors and limit farming, which was seen as an attempt to block peoples' herd recovery strategy, and a covert mechanism for extending the park. Community meetings in the late 1990s broke down with the threat of violence, ending with AWF physically withdrawing from the Simanjiro area and local debates over land use. AWF has come under criticism for supporting central interests at the expense of local communities and prioritizing its own organizational growth over community interests (Sachedina 2008; Goldman 2006; Igoe and Croucher 2007). Such tensions most likely contributed to large mammal declines of over 50%, except for buffalo and elephant, during this 15-year period. When local communities felt abandoned by 'community-based' organizations that claimed to represent them, they resorted to more aggressive tactics to defend their land, such as defensive farming. Commercial poaching of wildlife was ignored by local people, and in some cases willingly

Fig. 12.1 Aerial view of farms in the Simanjiro Plains

engaged in by villagers, who felt that eradication of wildlife would remove the attraction of their land. Tensions between government, NGOs, and local communities over conservation practices and land-use patterns, combined with the history of pastoralist alienation due to conservation, ultimately created a context in which conservation and development had become starkly polarized.

The Ecology of the Tarangire Ecosystem

The Tarangire ecosystem is considered to have high global biodiversity value; it contains the second highest concentration of large migratory mammals in East Africa, after the Serengeti-Mara ecosystem (Reid et al. 1998). The ecosystem covers an area of approximately 22,200 km² in geographic scope. It includes two national parks, Tarangire NP and Lake Manyara NP, National Forest Reserves (Marang and Essimingor), Mkungunero Game Reserve, and the Northern Highland Forest in the NCA. The parks constitute the core resource 'anchors' in the ecosystem. TNP is 2,850 km² and LMNP covers 330 km².

TNP was established in 1970 and was designed to protect a range of African wildlife species such as wildebeest, zebra, elephant, lions, and buffalo. TNP serves

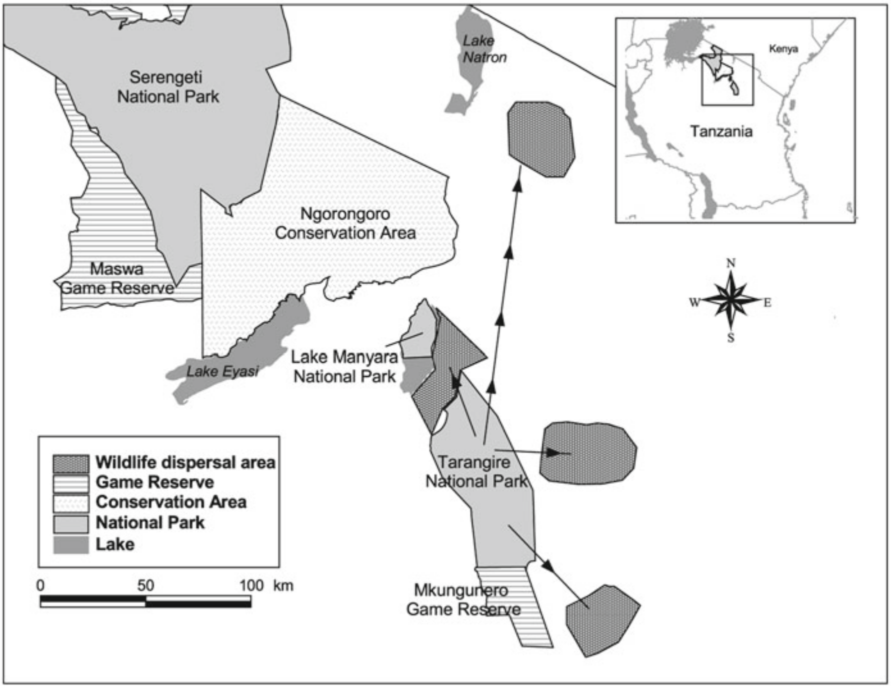

Fig. 12.2 Geographical overview of the Tarangire ecosystem (From Nelson et al. 2010)

as important dry season wildlife habitat, but the park comprises only 2,850 km^2 out of roughly 22,000 km^2 in the overall ecosystem (Fig. 12.2). For approximately 6 months a year, wildlife disperses into the Simanjiro Plains to the east of Tarangire on lands under the jurisdiction of Maasai pastoral communities. The plains are heavily utilized by zebra and wildebeest as they migrate between wet and dry season pastures, and are shared by pastoralists (Borner 1985; Kahurananga 1997). There are two primary ecological drivers for the migration. TNP's soils are phosphorus deficient (Voeten et al. 1999) while the Simanjiro Plains are higher in phosphorus, an essential mineral needed by lactating wildlife. During the long rains, wildlife move onto the plains to calve for several months, then migrate back into the park during the dry season to access the Tarangire River, the main perennial water source in the ecosystem.

Monitoring of the area's wildlife populations by air and by road has occurred since the 1960s (Lamprey 1963a, 1964; EcoSystems Ltd. 1980a; Kahurananga 1981; Foley 2004; TAWIRI 2004; TCP 1998; TWCM 1999, 2000). Recent data reveal differences in species abundances across species found within the TNP (Fig. 12.3). These data reveal a considerable drop in wildebeest and zebra populations when compared to other species found in TNP, probably related to poaching and habitat change.

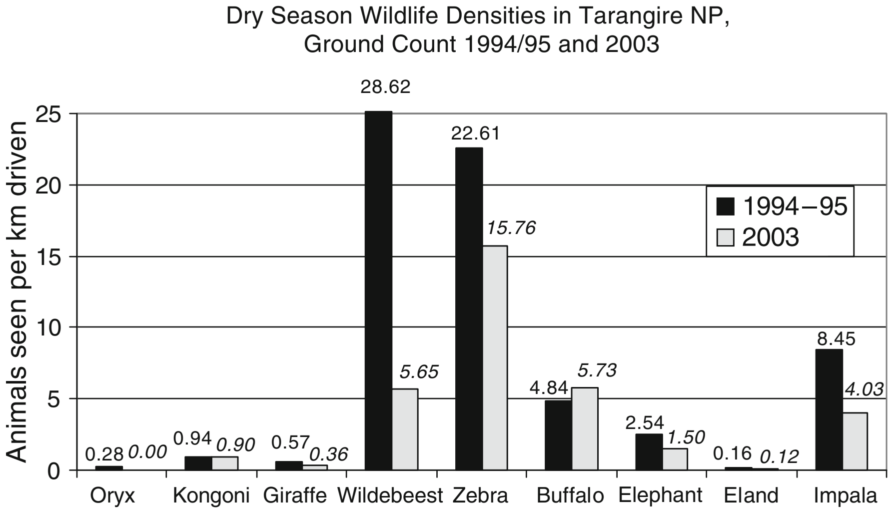

Fig. 12.3 Dry season road counts of wildlife densities in TNP in 1994/1995 and 2003 (Source: Foley and Foley 2005)

The Economics of Wildlife in Tanzania

The potential for wildlife to contribute economically and alleviate poverty in Tanzania is significant. Tourism represented 25% of export earnings in Tanzania in 2002; by 2008, this had grown to US$ 1 billion for the first time in Tanzania's history. In 2006, tourism accounted for 17.5% of GDP, a year in which foreign visitor numbers had increased to 644,000 tourists compared to 583,000 in 2004. Tanzania's 14 NPs generated US$ 51.7 million in 2006 from 657,000 foreign and local visitors. This suggests that at least 23,000 Tanzanian nationals visited NPs in 2006, which suggests that local value exists for NPs although these 'local' visitors are almost entirely tour guides who pay the entry fee price for nationals. Demand, therefore, is clearly skewed toward foreign visitation.

The economic value of the wildlife industry in and surrounding the Tarangire and Lake Manyara NPs may exceed US$ 30 million per year. Seventy-five percent of international tourism to Tanzania is based in the 'northern circuit', which includes TNP, LMNP, Serengeti NP, NCA, Kilimanjaro NP, and, to a lesser extent, Arusha NP (CSF and TANAPA 2004; Woien and Lama 1999), the backbone of a tourism industry valued at US$ 1.3 billion per year (Sumba et al. 2005: 3) (Fig. 12.4). Revenues from Tarangire and Lake Manyara NPs subsidize several lesser performing parks and are one of the few parks to generate an operational funding surplus, so these parks are of strategic national importance to the Tanzanian State (Otto et al. 1998).

The majority of tourism receipts are generated from photographic tourism. However, an important component of Tanzania's wildlife industry is tourism hunting. In 2006, Tanzania earned US$ 13 million from wildlife hunting, up from US$ 9.9

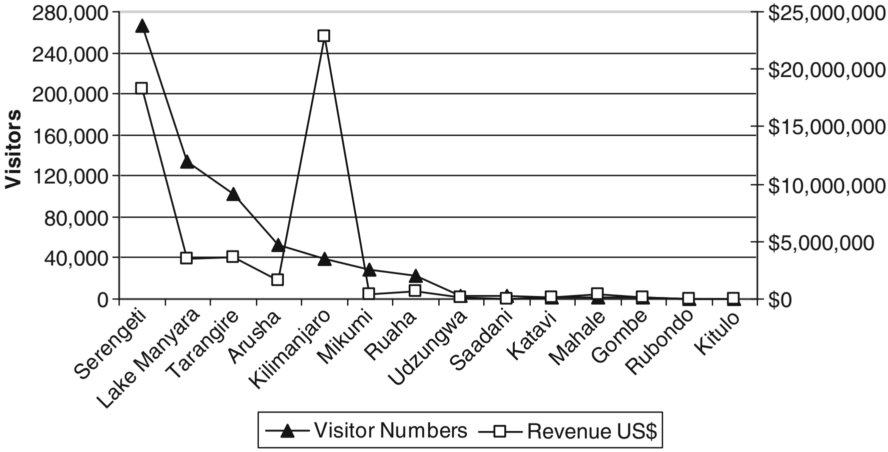

Fig. 12.4 Number of visitors and revenue to National Parks in 2006 (Source: TANAPA)

million in 2004, (an increase of 32%). Noteworthy successes have occurred in southern Africa, where sport hunting has supported devolvement of management rights and increased local livelihoods (Barnett and Patterson 2005: iii; Murphree 2001; Bond et al. 2004). A substantial portion of tourism hunting concessions are located on village land in Tanzania. This suggests that tourism hunting has the potential to contribute meaningfully to local livelihoods. However, given the market value of the tourism industry in Tanzania, both hunting and photographic tourism have yet to play a significant role in poverty reduction or supporting sustainable land-use outcomes at a local level, although substantial potential exists.

Adaptive Innovation: The Emergence of the Terat 'Easement'

The underlying barriers to wildlife conservation on community lands in northern Tanzania are not limited to the Tarangire Ecosystem, but reflect governance problems facing wildlife, and natural resource management more generally, throughout Tanzania. By 2004, after close to 15 years of work and international and national investments of approximately US$ 50 million for wildlife management in the ecosystem and most populations of large mammals in protracted decline, the extant situation suggested a different approach was necessary. Conventional wildlife management that focused on fairly efficient park-based enforcement was not working; Wildlife Division and District Council investments in wildlife management and community outreach outside the parks were sparse, militant, and dogged by rumors of corruption (Sachedina 2008); and community-based tourism operated in a murky legal environment and was constrained by inefficient distribution of revenue at a household level.

The impetus for actually developing an experimental PES scheme on the ground in Simanjiro came from a proposal put forth by the author, who at the time was working for AWF. The proposed project was termed the Enterprise Linkages to Conservation and Development (ELAND). The basic premise was to more directly involve the private sector (photographic and tourism hunting operators) to improve conservation management on lands outside Tarangire and Lake Manyara NPs. The private sector was seen as a funding source that could be more directly linked to supporting conservation on community lands. Realizing its own organizational constraints, AWF saw ELAND as an opportunity to improve its reputation amongst the private sector, harness a new source of funding, and catalyze an innovative community–private sector partnership approach.

Up until that point, the tourism sector had largely paid their fees and taxes but was somewhat powerless in contributing to wildlife conservation. Thus, the challenge was to convince private sector stakeholders that it was rational for them to be more actively involved in conservation, in order to ensure the sustainability of their businesses in the long term.

Tour operators were unlikely to dip into their narrow profit margins to capitalize ELAND; but they had access to clients who could be donors. Key to the strategy of ELAND was getting photographic and tourism hunting operators to collaborate, which had not occurred on any significant scale up until that point. By contrast, photographic tourism and hunting were generally seen as mutually exclusive activities, all the more so complicated by CBT investments in hunting concessions, which hunting operators opposed. In theory, collaboration between these two sectors should have been straightforward since they both depended on the same resource, but the centralized nature of tourism hunting and wildlife policy effectively pitted CBT operators against hunting concessionaires.

The ELAND concept was to create a basket fund from a combination of tourism company contributions (e.g. through a $1 per night special levy on all clients staying at lodges in Tarangire NP), supplemented by funds raised by international NGOs, such as AWF or WCS. The ELAND proposal led to a meeting of NGOs and private tourism and hunting companies active in the Tarangire ecosystem in July 2004 in Arusha. There was general agreement at this meeting between photographic and hunting operators about the threats to the ecosystem, and to the sustainability of their businesses. A working committee was established to further the conceptual idea of ELAND, which included Ujamaa Community Resource Trust (UCRT), a local NGO with strong Simanjiro community ties, Dorobo Tours, which had long-standing involvement in the area through a CBT concession in Emboreet village, a village which also happened to be responsible for much of the ongoing agricultural expansion onto the Simanjiro plains (Sachedina 2006). UCRT had worked with Emboreet since the late 1990s to develop a land-use plan and village by-laws, and was collaborating with the Sand County Foundation (SCF) on community legal training seminars in Simanjiro and other regions of northern Tanzania. A fourth partner, Wildlife Conservation Society/Tarangire Elephant Project (WCS/TEP), had worked with a private tour operator and villages to the north of the Simanjiro plains, in Lolkisale and Makuyuni villages, to zone areas for wildlife, tourism, and livestock.

Initial discussions amongst these collaborating organizations recognized the fundamental problem in Simanjiro: wildlife needed to generate economic returns to local communities, but the continuation of centralized conservation policies undermined this aim and continued to fuel negative local attitudes toward wildlife conservation. An additional practical problem was that while community-based tourism ventures had enabled the protection by villages of much of the habitat immediately bordering Tarangire National Park, wildlife tourism was not viable on the Simanjiro plains. The main problem on the plains is that during the wet season, when most wildlife is out on the plains, the area becomes difficult to access due to the plains' black cotton soils; during the dry season, access is easier but wildlife more sparse. Alternatives needed to be found, and initial discussions emerged among those collaborators of the possibility of designing a PES-type framework, or a community-based 'conservation concession.' At the time, it was not clear where the financing for such a scheme would come from or what the scale of such an initiative would be.

Follow-up discussions amongst Dorobo, UCRT, TEP, and SCF identified the ELAND concept as having potential for mobilizing financial resources to create a local 'conservation concession' according to PES principles in Simanjiro, based on its novel idea of pooling financial resources from tourism operators whose businesses depended in part on Tarangire NP. Concerns emerged, however, on two key points. The ELAND proposal envisioned creating a new legal trust with a range of trustees representing private sector, government, and NGO representatives. While this might be inclusive, it seemed cumbersome and had the potential to invest large amounts of time and energy in creating new organizational structures rather than focusing efforts at the village level, where local governance structures already existed. ELAND thus seemed in danger of becoming yet another top-heavy initiative in a region where a great deal of money had been spent on community-based conservation with limited on-the-ground impact. Second, there was a fundamental problem with linking any new initiatives in Simanjiro designed to build community incentives for wildlife conservation with international conservation organizations such as AWF, as a result of the locale's historic tensions between external conservation interests and local communities' land rights and livelihood concerns (Igoe 2004; Sachedina 2008).

For these reasons, Dorobo Tours, as the private sector actor with the longest history in Simanjiro and with strong experience both in community negotiations and collaborative conservation processes, took the lead in building support among a core group of private operators for a village-based PES scheme in Simanjiro. At the time (the second half of 2004 and first half of 2005), this included not only tourism operators but also Tanzania Big Game Safaris, a hunting company that leased the hunting concession in Simanjiro that overlapped part of Terat and Sukuro village's lands, as well as other villages to the south.

Dorobo built consensus among the operators for investing a small amount of financial resources into a pilot PES scheme, but the decision was made to de-link the initiative from the original ELAND proposal due to the concerns about costly bureaucracy and formal links with AWF which might raise concerns about land appropriation by external conservation interests at the village level. Furthermore, in hindsight, the use of a wildlife species name in English for a community-based PES scheme amongst Maa and Kiswahili speakers in an area of conservation conflict was ill-conceived. Few people in the area spoke English, and linking a PES scheme to an animal suggested that wildlife, not people, came first in the initiative; a subtle but important consideration given the conservation history in the area.

By early 2005, momentum was building for an experimental PES scheme in Simanjiro, but it was not yet clear exactly what shape this would take or what its coverage or cost would be. Initial discussions revolved around the seven key villages to the east of Tarangire NP, and later focused on the three – Emboreet, Terat, and Sukuro – which contain virtually all of the short grass plains which are the critical wildlife calving areas.

Finally, a decision was made to initiate a 'conservation concession' with Terat village based on set annual payments financed by annual contributions from a small group of tourism operators, with Dorobo Tours taking the lead in presenting the initiative to the village and brokering the deal. In exchange for the payments, the community would protect its portion of the short grass plains. The village of Terat was chosen out of the three as the site to pilot this concession for a few important reasons related to opportunity costs, local land-use preferences, and community capacity to manage natural resources and exclude outsiders, all of which are important ingredients for establishing PES programs.

Emboreet village was the source of much of the agricultural expansion onto the plains from the west, but also had a strongly antagonistic outlook toward wildlife conservation initiatives (see Sachedina 2008). Because so much agricultural conversion was occurring, it seemed like Emboreet would be potentially the most difficult village in which to initiate a PES scheme for protecting the plains, as there would be substantial opportunity costs to villagers and the scheme would likely encounter local political resistance. Both financial and political considerations thus did not favor Emboreet as a place to pilot the PES scheme, although this was where the problem of habitat loss/land-use change was most pronounced. Terat village, by contrast, had a history of excluding agricultural expansion from the short grass plains, which made up roughly a third of their village land area, and maintaining the plains for livestock grazing. In 1997, an incursion of outsiders with high-level regional political connections had invaded the plains in Terat and started cultivating land. The village mobilized rapidly and evicted these settlers, both physically and legally, through a subsequent court case. Farming had been effectively excluded from Terat's portion of the plains since then, and this incident demonstrated the enduring vitality of Terat's *collective* land and resource management institutions.

It is important to emphasize that the decision to start with Terat rather than Emboreet was explicitly a 'thin end of the wedge' strategy. The aim was to initiate the easement in a village that seemed most conducive to such an agreement, and by establishing a successful and mutually acceptable pilot initiative, to create the opportunities to later expand to other villages, including the more challenging context of Emboreet.

The PES Mechanism

The basic PES concept was that, although the plains were already protected by Terat as a seasonal livestock grazing reserve (used mainly July–October as a dry season reserve), an added financial payment could serve (a) as an extra incentive to prevent any future moves by individuals or the community to convert the plains to agriculture; and (b) provide incentives for the community to not only tolerate but actually conserve wildlife by protecting it from bushmeat poaching by outsiders. Beyond these direct impacts in Terat, the initiative would hopefully provide a new and locally acceptable PES framework applying community-based conservation linked to private tourism revenues, which could later be scaled up to include other villages in key dispersal areas.

The basic proposal put to Terat was as follows: the tour operators would pay the village an annual fee in exchange for the village agreeing to prevent agricultural cultivation, charcoal production, and illegal hunting on their portion of plains. Dorobo proposed a sum of five million Tshs (roughly $4,500) – a small enough amount that it would be feasible for the operators to contribute every year, but large enough to provide a meaningful incentive at the village level.

The implementation of the proposed initiative was led by Dorobo Tours and UCRT. Dorobo continued to organize the tour operators, securing pledges of financial support from four other operators. Three of these operators – Sopa Lodges, Tarangire Safari Lodge, and Asilia Lodges – own permanent tourism facilities inside Tarangire National Park. The main initial motivation for them was to contribute resources to an initiative that would improve the status of the wildlife populations in the park that their businesses relied upon, although non-financial conservation motivations were also an important factor. Notably, as the negotiations moved forward, the one hunting company involved, Tanzania Big Game Safaris, dropped out of the operator consortium. The hunting company was concerned about the deal being a mechanism for tourism activities to expand into its hunting block, was concerned about formally recognizing village land rights in its hunting concession, and, lastly, simply did not want to spend the money.

UCRT worked in their role as a local capacity-building facilitator organization to broach the concept locally. UCRT first reached out to several local elites, including Ilaramatak Lorkonerei, a local development organization based in Terat with a long history of land rights advocacy in Simanjiro, including opposition to wildlife conservation interests. The discussions were gradually expanded in August and September, 2005, from the village leadership to all the sub-village leaders, and finally endorsement by the Village Assembly. In October, the tour operators and village leadership met in Terat, and in December, the final contract was signed.

No significant changes were made to the written contract from the proposal initially brought to the village, with the deal providing five million Tshs. (about $4,500) paid to the village annually in exchange for the easement area being managed under the following conditions: agricultural cultivation and charcoal production would be prohibited, and the village would seek to prevent illegal hunting as well. All livestock-based uses would continue per the community's traditional practices. The one addition that was made, informally, was that the village requested the operators to also fund four village game scouts who would work to protect the wildlife and other natural resources in the village and, thereby, enforce and monitor the easement's provisions. This was agreed to in principle by the operators, although WCS/TEP later agreed to fund these game scouts, with UCRT administering their salaries and provision of equipment.

Several points need to be emphasized with regards to how the proposal was received at the village level, and the relatively harmonious negotiation over establishment of the easement. First, a key to the easement is that it is based on supporting traditional land-use practices, and that pastoralist communities in Terat and elsewhere face their own internal trade-offs with respect to maintaining land as livestock pasture or allowing land to be converted to agriculture. In Terat, the short-grass plains have always been managed as a dry season grazing reserve for livestock, and agriculture has been excluded and limited to other portions of the village land. For the village, agreeing to a formal contractual prohibition on agriculture in this area bore no immediate costs, and in fact served to reinforce the community's existing land-use practices.

Second, the main potential barrier to the easement agreement was not the potential opportunity costs to the community in adopting it, but rather the entrenched suspicion of wildlife conservation interests as a threat to local land rights and livelihoods throughout Simanjiro. This barrier was addressed strategically, by introducing the proposal first to several elite leaders from Terat, including the director of Ilaramatak Lorkonerei, an organization which had in the past been at the forefront of mobilizing opposition to conservation initiatives. Ilaramatak not only supported the idea, but assisted UCRT in facilitating the village-level meetings to discuss the proposal, which led to its fairly expeditious endorsement.

Third, an important factor in the community's acceptance of the deal was the long-standing existence of the village-operator tourism contracts and concessions in neighboring villages, particularly Emboreet. It was also significant that Dorobo Tours had been practicing tourism in Emboreet for nearly 15 years and was therefore well-known throughout the area. The community's familiarity with these tourism ventures made the easement proposal easily understandable, and helped allay possible fears about hidden wildlife conservation agendas. As Dorobo emphasized during the crafting of the initial easement proposal, a key strategy was to present the easement as a business proposal based on the tour operators' financial stake in the health of the Tarangire-Simanjiro wildlife populations, so as to ensure the community understood the rationale of the easement and to dispel fears of hidden conservationist agendas. This was a rationale for limiting the easement fund, at the outset, to contributions from tourism companies only.

Following signing of the easement contract, a management board was established at the village level consisting of five villagers elected by the Village Assembly every 5 years. This is the organizational mechanism for communication between the operators and the village, as well as the village-level institution responsible for overseeing receipt and use of the annual payments. In addition, four village game scouts were selected by the village; two permanent scouts and two who rotate every 6 months. These scouts are paid 60,000 Tshs (~$50) monthly, using funds provided by WCS/TEP and administered by UCRT. The scouts report to the village easement management board, which in turn reports to the Village Assembly. TEP has recently trained the scouts in the 'event book' system of monitoring wildlife populations used in Namibia's community conservancies (Stuart-Hill et al. 2005). This will provide data on wildlife trends at the village level, which will provide valuable information on the impact of community conservation measures in Terat, and also may help to mitigate human–wildlife conflict. This will also represent the piloting of community-based wildlife monitoring in Tanzania, where almost all data is collected at large spatial scales by government wildlife authorities.

In response to institutionally rooted wildlife governance problems prevalent in northern Tanzania, an informal group of individuals and organizations began working in 2002 on creating a new type of local organization that could integrate conservation, economic development, and governance issues and thereby build the kinds of long-term strategies necessary for addressing such complex institutional problems. This organization evolved into the Tanzania Natural Resource Forum (TNRF) by 2006. Key initial players in creating this organization were Dorobo Tours, Ujamaa-Community Resource Trust (UCRT), Wildlife Conservation Society (WCS) (through the Tarangire Elephant Project- TEP), and Sand County Foundation (SCF). Collaboration amongst these same organizations was also a key to the emergence of the Terat easement.

The Terat easement has been in place for about 6 years now. It has provided a formal mechanism for communities to protect approximately 9,300 ha of critical habitat on the Simanjiro plains and an incentive to work toward preventing illegal use of wildlife in this area (Nelson et al. 2010). It formalizes traditional land-use patterns and rules, which effectively serve as a barrier protecting the Simanjiro plains from the expanding agricultural frontier coming from Arusha to the north. The easement places a remunerative financial value on the ecological services that traditional livestock and land management practices provide in Simanjiro in terms of the maintenance and conservation of wildlife habitats. The easement therefore provides a model for correcting the 'market failure,' which drives wildlife declines in East Africa, in that wildlife valuable over large scales (e.g. the national tourism industry) is not valuable to local communities, which traditionally conserve habitats (Nelson et al. 2010). The impacts of the easement are both in terms of its formally protecting a large area of the Simanjiro plains as well as in providing incentives for communities to improve local protections of wildlife, which is traditionally treated as an 'open access' resource due to the weakness of centralized law enforcement mechanisms and the rule of law in Tanzania more generally. For example, village game scouts have arrested several groups of poachers, and use mobile phones to communicate with other anti-poaching forces such as hunting companies and Tarangire NP game scouts.

The village has received roughly 20 million Tshs to date (about $17,000), investing the bulk of these funds in primary school construction in one sub-village, as well as supporting a new secondary school in Terat village center. Although the total annual communal revenues from the easement, at about $4,500, are relatively small in relation to the total support for social services that the village receives from other sources such as the District Council and charitable NGOs, the easement funds are one of the few sources of discretionary revenues received by the village government. This small amount of village revenue gives community governance institutions greater flexibility in terms of supporting new or existing development projects. It also contributes to the development of local governance institutions as the community must collectively decide how to allocate and spend these revenues. Individual benefits are received by the four village game scouts, whose salaries of $50 per month, while modest, are nevertheless significant in a context where monthly per household cash expenditure is only around $10, and opportunities for employment are highly limited.

The initiative enjoys broad local support although it has faced one notable obstacle, revolving around a conflict between Terat village and one farmer who is also a former village council member. This farmer, an Iraqw (Mbulu ethnic group) immigrant to the area but a long-time resident, has a large farm (several hundred acres) in the northern part of the easement area, along the Terat-Loiborsoit border. The farmer claims that he was given the land by neighboring Loiborsoit village, and therefore Terat has no authority to remove him. Terat has since re-affirmed their village boundaries and obtained a Certificate of Village Land (as required by the Village Land Act of 1999), and involved government land officers in clarifying the location of the surveyed boundaries. Terat has also since removed the farmer from membership of the village council and successfully prosecuted a court case, using some of the funds from the easement payments, to remove this individual from the village's land. This demonstrates the additionality of the easement beyond existing land-use practices in terms of providing formal incentives for the village to secure the boundaries of the easement area and effectively confront sources of encroachment.

Beyond the immediate conservation and financial impacts at the village level, an equally important outcome of the Terat easement is the emergence of a new, locally acceptable, and cost-effective (approximately $.48/ha) framework for wildlife conservation on village lands in Simanjiro (Nelson et al. 2010). While the Terat easement is, to a large degree, identical to the framework for village-private tour operator, wildlife tourism concessions in nearby parts of Simanjiro, the structure of the Terat agreement is quite different since the tourists do not actually use the lands of the Terat easement. The plains are conserved to enhance the wildlife value of the park. As a result of the generally good reputation of the easement agreement in Simanjiro, in 2008, neighboring Sukuro village expressed interest in adopting a similar arrangement to cover its portion of the Simanjiro plains. In addition, Emboreet, while not yet embracing an easement on its portion of the plains, has appointed six village game scouts, which UCRT is overseeing and TEP is funding.

The potential for these easements or ‘conservation concessions’ to spread throughout the system in the next few years suggests that PES arrangements may provide a realistic framework for reconciling community interests with conservation objectives and providing local-level incentives for conservation of the wildlife in the Tarangire Ecosystem.

Lessons Learned

Creative Collaboration

The Terat easement arose from a collaborative effort among a diverse set of conservation, tourism, and rural development interests, all of whom were searching for solutions to wildlife population decline and continued conflicts between various stakeholder groups (e.g. between tourism and hunting companies, and villages and central government) over land and natural resource management in Simanjiro. The easement emerged because those collaborators recognized that existing institutional constraints, such as the reticence of the Tanzanian government to implement the 1998 wildlife policy and decentralize management to the local level, demanded creative new mechanisms for channeling benefits to communities if the decline of wildlife outside protected areas was to be halted. The collaborators also recognized that existing community-based conservation efforts in Simanjiro were fundamentally top-down and not sufficiently based on local livelihood interests and land tenure concerns.

The impact of the Terat easement cannot be fully measured by the area set aside by the village or the financial returns to the community. An additional and important impact of the easement is its establishment of a framework for community-based conservation that brings together local community, private sector, and conservation interests. The easement has forged common ground and produced a working example of community-based conservation in an environment that has been characterized by conflict between local communities and wildlife conservation for much of the past 30 years. The easement has resulted in new organizational relationships and common aims, which provide essential human and organizational capital for scaling up further collaborations and community-based conservation efforts throughout the Tarangire Ecosystem. The establishment of collaborative relationships and mutual understanding is a key outcome of the easement experiment, and potentially more important than its immediate ecological and economic impacts.

The easement has also resulted in leveraging other forms of external support for community-based conservation in Simanjiro, mainly in the form of collaboration between TEP and UCRT. TEP not only funds the village game scout salaries, equipment, and monitoring training, but also additional activities carried out by UCRT to support natural resource management in Terat, such as the surveying and formalization of village land rights. In 2007, the resources invested in the area by TEP amounted to about $11,000, and is expected to increase to $30,000 in 2008 as the program potentially expands to Sukuro village and land-use planning will be carried out as a precursor to an easement there. Thus, the operators' financing of the easement contract itself has been able to leverage additional resources to further support community-level natural resource conservation activities in Simanjiro, and also helped cement the collaboration between TEP and UCRT, which, in turn, provides a range of services supporting the easement itself and absorbs most of the transaction costs associated with the deal.

Local Champions

It is worth emphasizing that in the case of the Terat easement, as in so many other innovative conservation or development projects, businesses, or social movements, a handful of key individuals and organizations played a pivotal role. In particular, the long-term experience of Dorobo Tours and its directors in Emboreet village and the Tarangire Ecosystem more generally, was critical. Dorobo brought extensive experience with community-level negotiations, collaborative processes, and deep social and ecological knowledge of the region to the initiative. Equally, UCRT is a uniquely skilled facilitator of community-based natural resource management in Simanjiro and northern Tanzania more broadly. The organization had key contacts with local political elites in Terat, which were vital to introducing the idea of the easement in a suitable manner and ensuring it was not perceived as a conservationist 'land grab.' Without these two unique organizations, the easement idea would not have gotten off of the ground, and scaling it up further in Simanjiro is heavily dependent on their skills, relationships, commitment, and resources.

PES on the Margin

Wunder (2007) notes that PES arrangements will often be "best suited to scenarios of moderate conservation opportunity costs on marginal lands and in settings with emerging, not-yet-realized threats" (Wunder 2007). These conditions apply in Terat, where the key to the easement's successful implementation is the fact that it builds on traditional livestock-based livelihoods, and the incentives the community already possessed for limiting the expansion of agriculture into grazing lands (Nelson et al. 2010). Because the community had already worked to limit agriculture's spread onto the plains, adoption of the easement incurred very low opportunity costs. The easement serves to bolster the incentives the community possesses for limiting the future spread of agriculture into the plains and restricting agricultural cultivation to other parts of the village, which are less important habitats for wildlife and for livestock. The easement therefore serves to increase the marginal benefits of livestock versus agriculture as a local land-use choice on the plains, by enabling the community to capture additional economic benefits from wildlife as a complement to pastoralist livestock production.

In Emboreet, by contrast, land farmed on the plains is estimated to be bought and sold for up to $350/ha. However, as Sachedina (2006) describes, cultivation of the plains in Emboreet is also driven by the fears in that village that their land will be taken over by conservation interests – hence the 'defensive' strategy to cultivate and displace wildlife. A key lesson that emerges is that the local economic opportunity costs upon which PES agreements need to be negotiated are shaped not only by theoretical land values or productive potentials, but by social and political factors as well. The short-grass plains in Emboreet and Terat have the same nominal productive potential for agriculture, but different social and political contexts in the two communities result in very different relative land-use patterns and valuations at the village level.

Adaptive Strategies

The social, institutional, and ecological complexity of a large and variable ecosystem such as the Tarangire Ecosystem is considerable. Conservation strategies and interventions can only be effectively developed by (a) improving practitioners' understanding over time of how and why social and ecological change is occurring and (b) experimenting with new approaches that can be monitored and themselves used as opportunities for learning. Such an adaptive approach – or as Lindblom (1959) called it, 'muddling through' – focuses on gradually making iterative progress toward an ultimate goal, but recognizes that strategies to reach that goal must be altered as both surprises and learning occur (Lindblom 1959).

The 'muddling through' or adaptive management approach aptly describes the process that led to the emergence of the Terat easement. By 2004/2005, there was a nascent effort among a core group of experienced collaborators to devise alternative strategies toward the ultimate goal of creating community-level incentives to conserve wildlife on the Simanjiro plains. However, it was not until the unforeseen ELAND proposal that the impetus was given to crafting and implementing an operational land easement initiative. The ELAND initiative produced both a danger – the threat of increased suspicion of external conservation interests at the local level – and an opportunity by bringing a group of tour operators together to begin a collective dialogue among this group of conservation challenges in the ecosystem. Thus, both threat and opportunity catalyzed the core group of collaborators to re-shape the ELAND proposal into an operational PES scheme, which was experimentally piloted in Terat village.

PES as a Model for Community-Based Conservation

A key lesson from the experience of the easement is that PES can provide a simple and highly cost-effective model for community-based conservation of wildlife and wildlife habitats outside state protected areas (Nelson et al. 2010). In savannah ecosystems where wildlife regularly ranges far outside protected area boundaries, finding effective mechanisms and incentives for communities to promote wildlife conservation as a form of land use outside of protected areas is a critical issue affecting the long-term persistence of many species. In eastern Africa, wildlife populations are widely declining as a result of the lack of local economic incentives for conservation (Norton-Griffiths 2007). For example, the Loita plains wildebeest population of Kenya's Maasai Mara system declined by over 80% from the mid-1970s to mid-1990s, largely as a result of conversion of communal rangelands to farming and fenced individual properties (Homewood et al. 2001). PES arrangements such as the Simanjiro easement may provide an alternative framework for creating local incentives for wildlife conservation in contexts where alternative sources of economic incentives (e.g. tourism revenues) are not sufficient and in many cases non-existent.

In Tanzania, the easement model appears to be widely suitable for protecting key dispersal areas and migration corridors outside state-protected areas. It is important to emphasize the cost effectiveness of the easement framework in Simanjiro, in a context where millions of dollars have been spent on promoting community-based conservation to limited effect. This supports arguments that PES may be more efficient and effective than alternative methods for integrating conservation with rural development such as so-called integrated conservation-and-development projects (ICDPs) (see Ferraro and Kiss 2002). However, it is also important to highlight the complimentary nature of the Simanjiro easement and other community-based conservation models such as village-private ecotourism ventures. These different models are not zero-sum options nor are they mutually exclusive, but should be promoted according to context and practical challenges and opportunities.

Future Challenges

Several notable challenges face the Terat easement moving forward. The easement arose because Tanzanian wildlife management institutions have failed to put in place a legal and policy framework that encourages community-based conservation, based on local capture of wildlife's economic value, on village lands. Conflicts between central wildlife authorities and local communities, particularly over the matter of hunting concession allocations on village lands, continue in Simanjiro. The Terat easement has operated as a direct contract between private operators and the village, supported by a range of NGOs. Conflicts between the village and higher levels of government over land tenure and resource use remain a challenge for conservation practitioners in the Simanjiro area. The easement could be undermined by central appropriation of village lands in the Simanjiro plains, which has been a threat to the communities for over 20 years, or by continued inflammation of local attitudes toward wildlife by top-down conservation initiatives by government or external NGOs. Although communities have clear rights to their land, the rule of law in Tanzania remains weak and central and external appropriation of communities' resources, either through de jure or de facto measures, is common.

Another challenge is financial. The tour operators who are the contracting parties to the easement are, in a way, subsidizing the benefits captured from Tarangire's wildlife for other groups with a broad range of interests. These include other private operators, but particularly government agencies such as TANAPA, which earns millions of dollars from park gate fees and lodge concessions in Tarangire NP. The operators note that an underlying assumption of the Terat initiative since its inception has been that by catalyzing a successful model for conservation on village lands, their financial contributions to the easement would be able to leverage external conservation funding to expand the model to other villages, or even perhaps take over the financial support of the Terat easement. While the operators' investment has been able to leverage significant additional resources, mainly through the TEP support to UCRT, which underpins the easement's administrative costs and the village game scouts' activities, it remains unclear how willing the operators will be to scale up their existing level of financial contributions. Mechanisms for scaling up the easement model using other sources of funding, such as a long-term endowment raised by conservation interests, have not been delineated, and the overall financial strategy for scaling up the easement to other villages has not been clearly articulated. This will be a key area for future collaborative efforts in the ecosystem in order to successfully build off of the catalytic experiences of Terat for conserving wildlife populations.

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Chapter 13
Poverty, Payments, and Ecosystem Services in the Eastern Arc Mountains of Tanzania

Brendan Fisher

Introduction

A social trap is a situation where the short-term benefits of a decision are at odds with long-term optimal outcome (Cross and Guyer 1980). Some poverty traps are social traps. For example, rampant clear-cut deforestation may have short-term payoffs but is, in many cases a long-term net loss for both the agents of deforestation and wider society. With regard to socio-ecological systems, these traps may be spatial as well as temporal – where the actions of some in one locale may adversely affect others elsewhere (economists call these externalities). In a world of rapidly changing environmental quality, our inability to solve social traps across time and space affects the immediate welfare of millions of people living at the margin, as well as the long-term welfare of society and wildlife populations writ large.

Pressing and interrelated problems including large-scale conversion of ecosystems and the subsequent loss of biodiversity (MA 2005); increasing poverty and water scarcity (Rosegrant et al. 2003); potentially dangerous alteration in the climate system (Schneider 2001; Mastrandrea and Schneider 2004); and global fisheries collapse (Myers and Worm 2003; Worm et al. 2006) drive an urgency for integrated solutions to escape social traps which pit the consumption of one beneficiary against the livelihood of another or force decisions where rational short-term gains (e.g. agricultural extensification) undermine ecosystem services critical to long-term welfare (e.g. climate stabilization). Solutions require a deeper comprehension of the environmental infrastructure upon which human existence and social welfare depend (Sachs and Reid 2006; Schroter et al. 2005). Of primary importance is the immediate need to address the welfare of those already marginalized by regional and global economic systems and falling environmental quality.

B. Fisher (✉)
Woodrow Wilson School of Public and International Affairs,
Princeton University, Princeton, New Jersey, USA
e-mail: bpfisher@princeton.edu

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Doing so will require an explicit acknowledgement of (1) the complex nature of human poverty, (2) the complexity of ecological processes that deliver ecosystem services and welfare benefits, and (3) the interrelatedness of these two complex phenomena.

Here, we briefly describe these three complexities through looking at the Eastern Arc Mountains in Tanzania, one of the world's most important areas for biodiversity. We focus on human poverty in the region, the vital ecosystem services provided by the ecological systems, and the relationships between the two. We use potential payments for ecosystem services (PES) program for water to motivate a discussion on if and how such an intervention can overcome some of the social traps inherent in a system where so many people live on the margin. The trap described here is basically one where local use of resources negatively affects the regional and/or global provisioning of that service. The major complicating factor of escaping this trap is that the main agents of land degradation are acting rationally, based on their position of extreme poverty.

The Eastern Arc Mountains: Biophysical System

The Eastern Arc Mountains (EAMs) consist of 13 distinct mountain blocks stretching from Southern Tanzania to Southern Kenya (Fig. 13.1). The highest peaks in the EAMs are Lukwangule Plateau and Kimhandu Peak (>2,600 m). Northern and central blocks show two wet periods: the short wet season peaks in November and the large peaks in April with a yearly average rainfall of roughly 1,500 mm. In the southern blocks, there is one main wet season, peaking in March and April (~2,000 mm/year) (Lovett 1996). From the basins up the altitude gradient, there are transitional forests, sub-montane, and montane forests, and at the highest elevations (>2,000), there are closed evergreen forests (Burgess et al. 2007). However, montane and closed forests only cover about 3% of the EAMs. Most of these forests are protected in forest and nature reserves. The dominant land cover is miombo woodland, covering approximately 34% of the area. The woodland exists in various degrees of degradation and has lost more than 40% of its area since the 1970s (Mbilinyi et al. 2006). Roughly 8% of the EAM basins are under agriculture – mainly maize, cassava, and paddy. The remaining major land cover is a mix of bushland, grassland, and mixed cropping mosaic.

The EAMs are typified by having high biodiversity and endemism. This is reflected in their status as one the world's hottest hotspots (part of the Eastern Afromontane; Mittermeier et al. 2004) and as a globally important eco-region (Olson and Dinerstein 2002). There are over 4,000 plant taxa of which over 800 are endemic. There are at least 96 endemic vertebrate species including several endemic primates such as the sanje mangabey and kipunji monkey. The status of invertebrate species in the EAMs remains to be determined, but there are at least 43 endemic butterfly species (for a full treatment of biodiversity importance of the EAMs, see Burgess et al. 2007).

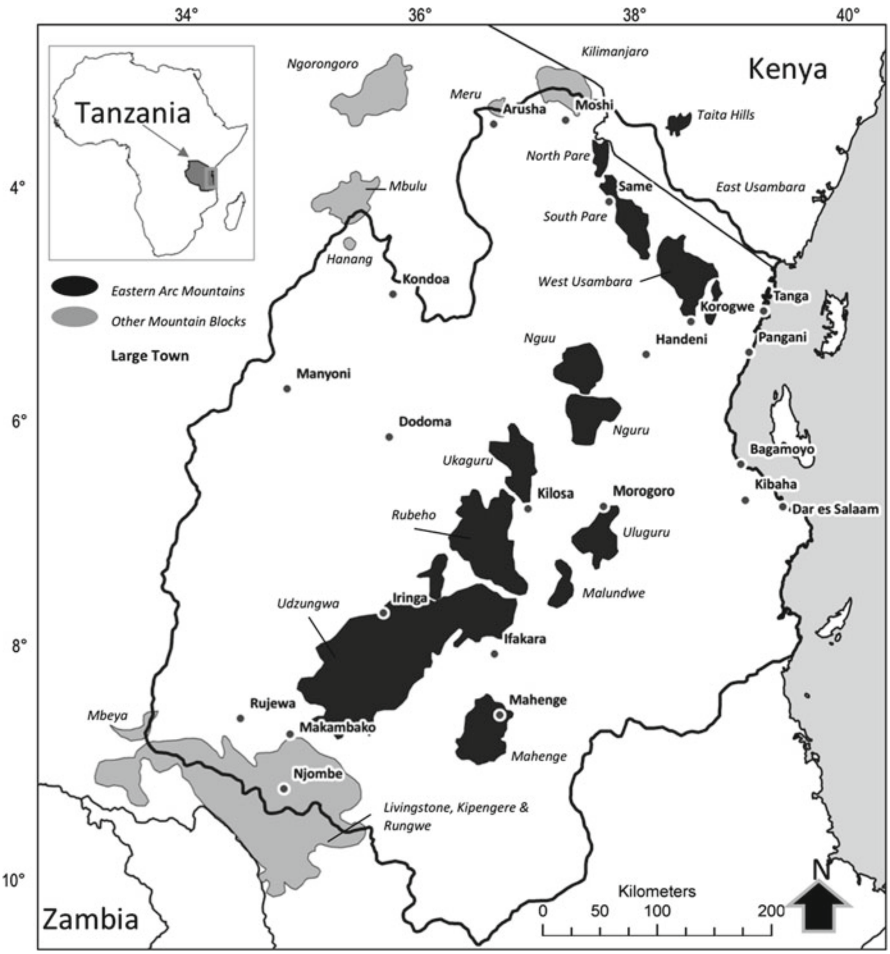

Fig. 13.1 Tanzania and the Eastern Arc Mountain blocks

Poverty in the Eastern Arc Mountains

It is well recognized that poverty is a complex phenomena (Myrdal 1957; Sen 1985; World Bank 2001). Insufficient income generation, food insecurity, water scarcity, inadequate shelter, child mortality, and access to health care are just a few indicators of impoverishment. To really address human welfare, we need to look at a suite of these indicators since meeting some minimal level of one of these indicators does not typically substitute for meeting another. The basic human "needs and satisfiers," as Manfred Max-Neef (1992) calls them, all need to be addressed to ensure that people are realizing a decent life. Nobel laureate Amaryta Sen and Martha Nussbaum based their "capabilities approach" upon this premise and specified that a person's

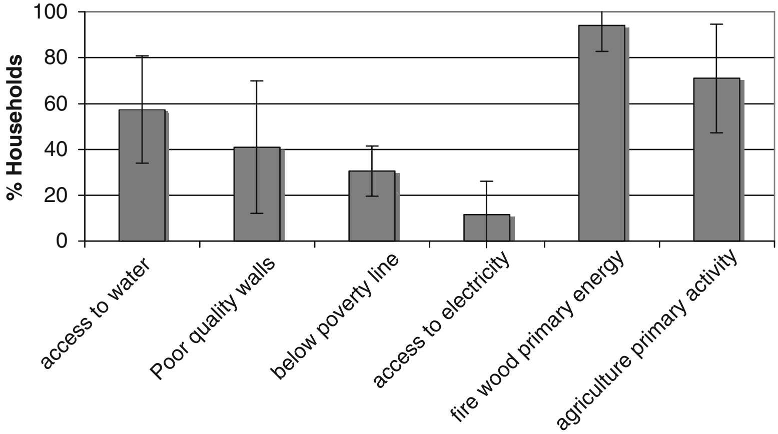

Fig. 13.2 Levels of poverty in the EAMs depicted by various indicators

ability to function in society is predicated by a capability to not only actively engage in society but also have choices as to how to develop their lives (Sen 1985).

Tanzania is a sub-Saharan country labeled as having "low human development" by the UNDP *Human Development Report* (2008). Typified by low life expectancy at birth (51 years), low formal education rates (50%), and low GDP/capita ($744 purchasing power parity (PPP)), Tanzania ranked 159 out of 177 countries in the latest Human Development Index (HDI). These country-wide statistics set the foundation for any smaller scale conservation-development work. Current research on understanding the ecosystem services produced and provided in the country is focused on the Eastern Arc Mountains and their drainage basins (see www.valuingthearc.org), and therefore sub-national poverty indicators are required recognizing that poverty is spatially heterogeneous.

Figure 13.2 shows the levels of poverty for a suite of indicators across the EAMs (bars represent ±1 SD data based on district level averages). On average across the EAM districts, 31% of the people live below the national poverty line; over 40% do not have access to an improved water source; 41% live in poor quality shelters; and less than 12% of people have access to electricity. Additionally, firewood is the primary energy source for over 90% of the people, where more than 70% of people make their livelihoods on small-scale agriculture. Based on these statistics and the national scale statistics, it is easy to say that Tanzania faces huge challenges to alleviate poverty. At the same time, there is a large variation on these averages (see SD bars). For example, although on average, 12% of people in a district have access to electricity, in rural areas (much of the region) this figure falls to around 2%. If statistics were available at higher spatial resolutions, this type of variation would appear in other indicators both across and within districts. Understanding the variation at the finest grain available is important for any type of development intervention.

What about correlations across indicators? For example, are areas with poor access to an improved water source the same as areas where there are many people

Table 13.1 Pearson correlations for various poverty indicators in the EAMs

	Poverty line	Poor quality shelter	Electricity	Health clinic access	Improved water source	Infant mortality
Poverty line	1	–	–	–	–	–
Poor quality shelter	.238*	1	–	–	–	–
Electricity	−.546**	−.423**	1	–	–	–
Health clinic access	−.164	.132	.044	1	–	–
Improved water source	−.486**	−.620**	.677**	.123	1	–
Infant mortality	.297*	.177	−.463**	−.028	−.409**	1

*Significant at the .05 level (one tailed)
**Significant at the .01 level (one tailed)

living below the economic poverty line? Table 13.1 shows the Pearson correlations for these poverty indicators, showing that while there are weak associations across some indicators, there are no strong correlations. The signs of the significant associations make intuitive sense. For example, as the number of households in a district with access to electricity goes up, the number living below the poverty line goes down. The strongest associations are cases where infrastructure for one indicator is likely to lead to the conditions to meet another. For example, as the number of households with access to electricity goes up, so does access to an improved water source – possible due to a physical connectedness by roads facilitating connectedness by water pipe and powerlines. However, based on the statistical power of these relationships, we cannot draw conclusions across different dimensions of poverty. Knowing whether poverty indicators are well correlated across a landscape is important because it allows for proxies to be used in cases where data is scarce. For example, food insecurity and average caloric intake are well correlated at the national level, which means when one indicator is missing for a country, we can use the proxy to understand the other phenomena (Fisher and Christopher 2007). When poverty indicators are not well correlated, it just adds more credence to the fact that poverty is a complex and multifaceted phenomena. It also means that interventions have to be targeted at fine scales and based on the most appropriate indicator.

This drives a question: how does the magnitude and variation of the different dimensions of poverty affect our ability to create mechanisms and interventions to avoid social traps? In order to answer this, we need to look at the socio-ecological system that defines how human welfare is affected by and affects ecosystem functioning.

Ecosystem Services in the EAMS

Understanding complexity of socio-ecological systems has quickly become a global research and policy priority (MA 2005; Sachs and Reid 2006; Sachs 2008). Under the relatively new badge of "ecosystem services," issues linking the integrity of ecological systems to human welfare have been acknowledged as critical research priorities (Carpenter et al. 2006). As discussed in other sections of this book, the

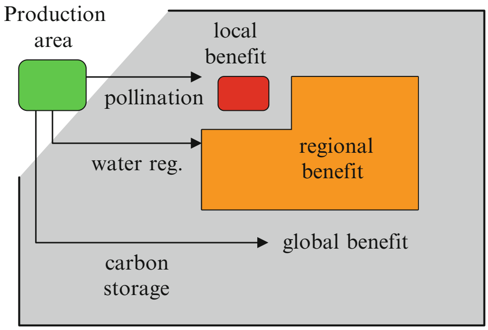

Fig. 13.3 Spatial relationships between where ecosystem services are produced and where the benefits are received

concept of ecosystem services is a multifarious concept incorporating micro-scale processes such as nutrient cycling to global scale processes such as regulation of the climate system. One way we can delineate between ecosystem services is to look at the spatial and temporal relationships between where services are produced and where the benefits are received (Hein et al. 2006; Fisher et al. 2009). This is in acknowledgment of the fact that services typically "flow" from one area to another. Some of these service flows are global in nature (e.g. climate regulation), while others are more localized (e.g. pollination). Figure 13.3 is a simple construct of a single locale producing diverse benefits across scales. Here, we look at some of the global, regional, and local ecosystem services and their benefits produced in the EAMs in order to motivate a discussion on ecosystem management, PES, and its potential relationship with poverty reduction.

Global Scale Services in the EAMs

The EAMs provide several ecosystem services and benefits at the global scale. Two of the most recognizable are carbon storage and the benefits that flow from the preservation of biodiversity. The latter provides opportunities for cultural, aesthetic, and spiritual fulfillment, as well as more instrumental benefits like opportunities for ecotourism and potential pharmaceutical discoveries. From above, we see that the biological importance of this area is nearly unparalleled. The benefits based on biological diversity are global in nature since there is the *potential* for people from all over the world to realize these benefits (actual benefits are highly dependent on access, wealth, education, etc....).

As far as the carbon storage and sequestration services are concerned, the EAMs store carbon in the majority landscape of miombo woodlands and the various other forest types. Quantification of the magnitude of this carbon store has only recently commenced, but research suggests that African tropical forest systems store a large portion of terrestrial carbon and have likely been increasing their rate of sequestration (i.e. becoming a bigger sink) to the point that this increase may represent a large portion of the world's "missing" carbon sink (Lewis et al. 2009).

Regional Scale Services

Producing and modulating water flows across the EAMs is a service provided by well functioning ecosystems in the region. Upslope vegetated landscapes throughout the EAMs help to attenuate the river flows in the wet season, providing water flows throughout the year (Doggert and Burgess 2005). Water regulation is critical for several benefits experienced across the region. In a typical year, over 60% of Tanzania's electricity generation comes from hydroelectric power plants on EAM rivers (The Economic Survey 2007). Most of this is utilized far from catchment forests, with the bulk of it supporting the coastal urban centers, mainly Dar es Salaam. Water regulation is also important for the current irrigated croplands of Tanzania, which are optimistically forecasted to expand from 200,000 ha to a million hectares by 2025 (The Economic Survey 2007). Again, timing of water flows is critical to irrigated agriculture supporting staple crops such as rice, as well as export cash crops such as coffee.

Another benefit which flows regionally from point of production and regulation with regard to water is drinking water. Most Tanzanians get their drinking water (either improved or unimproved) from rivers or shallow wells, as boreholes and deep aquifer extraction are cost prohibitive (Kulindwa et al. 2006). There is anecdotal evidence that the EAMs and their cloud capture are actually areas of water production in addition to regulation, but this assertion needs further measurement and modeling (Kulindwa 2005). Without water storage capabilities, hydroelectric power, irrigation and domestic use water all rely on ecological systems to deliver a more regular flow of water throughout the EAM basins.

Another benefit provided regionally by ecological functioning in the EAMs is fuel wood and charcoal. The production of biomass, particularly in the woodlands, is an ecosystem service utilized for charcoal production. Once produced, it is then trucked to regional centers across the country. There is some indication that up to 50% of the roughly 24,000 bags of charcoal used in Dar per day comes from the EAMs and associated basins (each bag being approximately 56 kg) (Van Beukering et al. 2007). This represents a critical role that woodland systems play in supplying urban centers with services. Charcoal is the main cooking and heating energy source for urban areas.

Local Services and Benefits

Much of the ecosystem service benefits utilized in the EAMs could be considered to be both produced and utilized on the local level. Unlike in the cities, the main energy and cooking source in rural areas is firewood. Most districts in the region show that greater than 90% of households use firewood as the main (or only) fuel source for cooking and heating. Firewood, produced in woodlands, scrubland, bushland, and farmland, is extensively collected and provides a currently un-substitutable resource for daily livelihoods.

Additional, locally produced benefits include poles for building and construction, raw materials and fibers for mats, roofing and fencing, wild fruits and vegetables, and medicinal herbs. All of these are critical to local welfare and livelihoods throughout the EAMs (Ndangalasi et al. 2007). The magnitude of such dependence is suggested in the fact that on average, 41% of houses in the EAM districts are made with poles and other natural building materials, and in over 12 districts, this figure is greater than 75%. Also, preliminary results from household surveys suggest that a large percentage of households in the Usambaras and the Udzungwas collect wild vegetables, fruits, or mushrooms from surrounding forests and woodlands. Local pollinators are also likely to play some role in pollinating mixed crops of subsistence farmers; however, the most important staple crops, such as maize, rice, and cassava, do not rely on insect pollination.

All of the services provided by well-functioning ecosystems in the EAMs embody complex relationships between beneficiaries and those actors who are most likely to affect the provisioning of these services. Because of this, management and policy mechanisms must respond to these complexities to ensure that net societal benefits of land-use change are positive. One such mechanism that is increasingly used for managing ecosystem services is the Payments for Ecosystem Services (PES) schemes pioneered in Costa Rica and discussed elsewhere in this text.

PES as a Policy Tool in the EAMs

In Fig. 13.3, we see that an individual parcel can deliver local services such as house pollinators for local crop pollination; it can deliver regional water regulation services by attenuating water flows in regional rivers for example; and it can deliver carbon storage where the beneficiaries are global in distribution. It is at the regional and global scales of delivery where systems such as PES are designed and expected to overcome social traps, or externalities. The traps exist because what is rational and necessary for local people (i.e. resource utilization and extraction) may be a net cost to wider society. In fact, preliminary evidence in a range of systems from across the world suggest that we have reached a point where the conversion of most remaining natural/semi-natural systems is like to be a net cost to society (Balmford et al. 2002; Turner et al. 2003; Naidoo and Ricketts 2006). While the decision to convert a parcel is likely to deliver dis-benefits to those living downstream of the benefit flows, the converse is also true – conserving the parcel for the benefit of downstream beneficiaries is likely to impose dis-benefits on those local actors in the form of foregone opportunities. PES systems are an attempt to overcome this trap ensuring that short-term decisions do not turn out to be long-term losses.

In these cases, the spatial distribution of costs and benefits of conservation is of critical concern. In the EAMs, it is safe to assume that forested landscapes deliver water regulation services to the district basins. Here, one of the key benefits

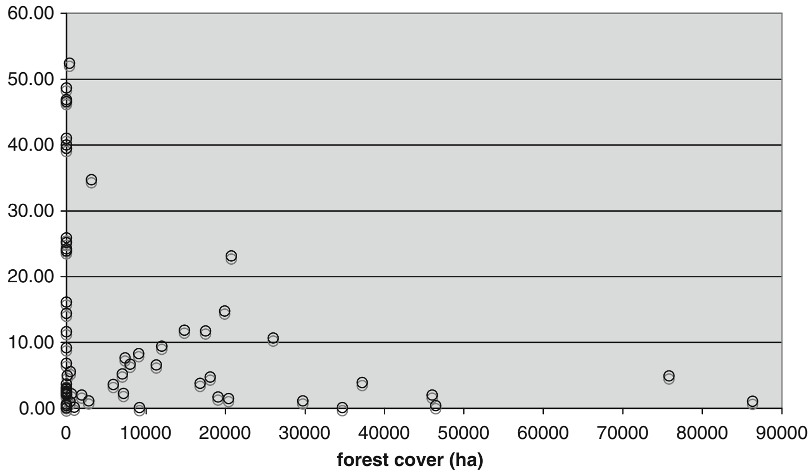

Fig. 13.4 Forest cover and household access to electricity in the EAMs (by district)

of regulating water flows is to ensure that wet season rains flow in rivers throughout the year. This is essential for Tanzania's electricity supply. The cost–benefit questions that arise are the following: (1) "Who benefits" from this provision? (2) Where is the provision generated? and (3) Who pays the cost of provision? Most of the electricity use and benefits accrue in the coastal cities where in a typical year, 60% comes from hydroelectric dams on EAM rivers. The current evidence to date supports the idea that the water regulation is a function of the forested inland districts. We can begin to dissect this relationship by plotting the percentage of households in a district with access to electricity against the area of forests in the district (Fig. 13.4). This result is intuitive, showing districts with high electricity access have low forest cover. In fact, there is a significant difference in the average access between districts with greater than 5,000 ha of submontane, montane, and upper montane forests, and those districts with less than 5,000 ha forest cover – mean difference 9.73 (±3.4) [two sample *t*-test, $t=3.10$, 43 df, $p < .003$]. Despite the obvious relationship, it reinforces an important consideration for environmental management – areas where forests are standing are providing a service to typically urban electricity users. This is likely to remain the case for the foreseeable future since the forecast is to rely on hydroelectiric generation for at least 50% of grid supply for the next two decades. There are several reasons that this disparity exists (historical infrastructure investment, remoteness, slope, etc....). However, that does not change the fact that any conservation of forest cover could impose an opportunity cost on those who live near the general lands and reserves where the forests stand. In short, the rural upstream people pay a cost for downstream beneficiaries, but are not compensated directly for any opportunity costs. This is the type of situation that PES is designed to overcome, but how can these payments be operationalized?

PES and Poverty Reduction

There is much debate about how and if PES schemes can be pro-poor (Pagiola et al. 2005; Corbera et al. 2007a, b; Zilberman et al. 2008; Bulte et al. 2008). However, in economics, there is a long tradition recognizing that it is very difficult to try to meet two policy objectives with a single policy instrument. A general rule of thumb is the "Tinbergen Rule" where for each policy objective, society should have a single mechanism to attempt to meet that objective. In order for PES systems to be pro-poor, defined as targeting poverty reduction, several very special circumstances need to apply. These circumstances are discussed below, but the main difficulty arises in trying to "maximize" across two objectives. If we are trying to combine the provision of ecosystem services with poverty reduction, we must have a primary policy goal. First, we have to make the assumption that forest and woodland cover help regulate water flows in rivers across the region. As mentioned above, this assumption is relatively certain, although robust measurement and modeling are still to be undertaken. Water regulation is of critical importance in an area typified by wet and dry seasons. Without the regulation functions, all of the water delivered in the wet season flows through the system in a relatively short time frame. The forests and woodlands attenuate those flows and aid in the slow release over the year. For this illustration, we also need to assume that all forest cover is equal, meaning that any forest cover delivers the same service as any other. This we know to be untrue, but the reality of the heterogeneous value of forests for water regulation is not likely to confound the example below.

So what is our primary policy goal? Prioritizing for either poverty reduction or conservation of landscapes that deliver water regulation gives us two different outcomes and therefore suggests different prioritization strategies. Figure 13.5a shows a cumulative density curve of the number of people living below the poverty line in the EAMs by district. We can see that 51% of the poor live in just 19 districts. Also, in Fig. 13.5a is the cumulative area of forested woodland represented by the districts that were optimized for income poverty. If PES is a *poverty-first* tool, then to alleviate 50% of the poverty, we need to work in 19 districts and pay some compensation to the poor under the stipulation that the associated forest cover (32% of the total) is protected for the delivery of water regulation services.

On the other hand, if we are prioritizing for water regulation (read forest cover) and hoping to get poverty reduction as a side benefit, then we are in a situation closer to Fig. 13.5b. This graph shows the cumulative density function for woodland forest cover in the EAMs. Here, we prioritize different districts based on an ecological criterion: delivering landscapes that are likely to regulate water. Any payments to protect forest cover may have knock-on poverty effects. If the payment goes toward covering opportunity costs of the poor or providing in kind services (schools, roads) then the poor may benefit. Alternatively if the payment go toward land enclosure, then the poor are likely to lose out.

However, in this case, it is not poverty driving the intervention but water regulation. In order to conserve 50% of forest cover in the EAMs, we need to work in only

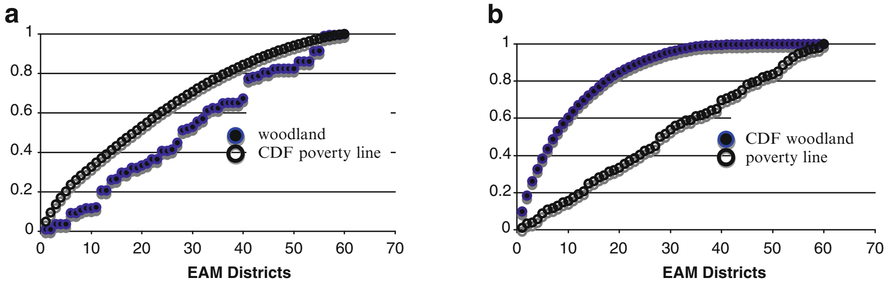

Fig. 13.5 Cumulative density curves for poverty and forest cover in the EAMs (by district)

eight districts. The associated (potential) poverty effects in these eight districts represent only 13% of those in the EAMs living under the poverty line. Back to *panel a*, if we wanted to intervene in districts and conserve 50% of forests based on *poverty-first* rule, we would need to work in 28 districts. This difference could be a critical consideration under a fixed budget where it can be assumed that the more areas you need to contract with and monitor, the greater the administration and transaction costs. Therefore, the higher the number of districts, the less the funding that goes toward ensuring ecosystem service delivery.

Both panels in Fig. 13.5 use the poverty line as the indicator of poverty. However, as highlighted in the discussion above, one poverty indicator does not necessarily correlate to other indicators. The complex and heterogeneous poverty–landscape relationship has particular consequences for any PES system. Would the same districts be prioritized if we used "poor quality houses" as an indicator? The Pearson results (Table 13.1) tell us the answer is "no." Therefore, an intervention prioritization is a function of the indicators chosen for analysis.

PES Compensation Mode

There are strong arguments that direct payments for conservation may be effective and efficient (Ferraro and Kiss 2002), but in what form should these direct payments take? In our case, providing hard currency into the areas important for regulating water flow may be affective in slowing forest degradation, but only if there are alternatives to the main drivers of forest degradation. Is cash a substitute for fuel wood collection, pole cutting for building, and agricultural encroachment? If the payment is "in-kind" such as schools and roads… we still have the same question: In an area where more than 90% of households rely on firewood for cooking and heating, does an in-kind payment cover this lost cost? In both the cash and in-kind cases, payments would have to go hand-in-hand with alternative cooking and heating energy options. The dynamic considerations include what would "cash" payments be spent on? This is an important consideration, as scenario research in the Amazon revealed that market gains from more sustainable Brazil nut collection would likely be invested in increasing land and cattle stocks by nut collectors. If that were to be the case, then more sustainable nut collection would eventually generate increased deforestation (Evans et al. 2006). These types of feedbacks would need to be considered for any long-term sustainability agenda with regard to payment modes.

Again, looking at the complexity of poverty in the region, over 40% of households rely on poles and forest products for their house construction and building needs. Here, we can see that the local benefit from forest lands (raw materials) is likely to trade off against more regional and global ecosystem services of water regulation and carbon storage. The magnitude of the lost opportunities for local agents is often considered to be the critical payment level for any PES (Naidoo and Iwamura 2007). However, a simple cash transaction ignores the complexity and necessity of meeting needs where many such actors live outside the market and few substitutes exist (Hyden 2007). In the EAMs, again, what is the substitute for the primary building resource used by the majority of households? What is the cost of obtaining this substitute?

These types of questions and considerations are complex and require context-specific investigations. Field research at different sites attest to the complex relationship between poverty and ecosystems service provision as well as the role that a PES scheme can play in modulating this relationship (Pagiola 2008; Wunder et al. 2008). Some general conclusions emerge. First, the difficulty in measuring, modeling, and monitoring the provision of any ecosystem service means that assessing the effectiveness of payments is extremely difficult (Ferraro 2008; Wunscher et al. 2008). Second, the degree to which market incentives can motivate "additional" stewardship behavior also remains an empirical question (Bowles 2008). Third, before we assume a positive relationship between payments and poverty benefits, three critical criteria need to hold (see Bulte et al. 2008; Wunder 2008 for reviews):

1. Rights to land and resources – PES systems are typically founded upon clear property rights. This ensures that payments can be made to a manager (and/or owner) of the land that delivers a stream of ecosystem services (Corbera et al. 2007a, b; Wunder 2008). In the EAMs, most of the closed canopy forests are under government control, where most of the woodlands are considered general lands. This lack of individual property rights on the general lands could be considered an impediment to PES schemes due to lack of a defined service provider. However, from an equity standpoint, this may foster a situation where any benefits from a PES could be distributed to villages attached to the general lands and therefore not dominated by large landowners. Good governance is of course a requirement for any potential equitable distribution of benefits. However, even in schemes where institutions are strong, the capture of payments by wealthy landowners has been demonstrated (Zbinden and Lee 2005). For those schemes dedicated to environmental outcomes, this is unlikely to pose a problem, but will raise flags when poverty and equity are of concern in implementation.

2. Land of high ES value – Again if the priority goal is to deliver an ecosystem service (carbon, water, etc.), any PES scheme should focus on providing the service cost effectively by seeking out areas of high service value at low costs (Ferraro 2008). This consideration has implications for how a PES can affect poverty levels. First, in order for the poor to receive any payment, they must be associated with lands that actually deliver the service of interest (Zilberman et al. 2008). To assess this, we need to understand correlations between the service of interest and poverty levels. Where there is a positive relationship, there are likely to be win-win situations. Understanding this relationship requires both the social data for assessing poverty and the ecological-economic modeling necessary to highlight areas of high ES value. While our understanding of poverty throughout the EAMs is increasing, we are still on the early part of the curve in the spatiality explicit and robust models of ES provision. However, there may be rules of thumb that already provide insight into areas of high ES provision. The literature is replete with rules of thumb as to where carbon is stored, where water capture occurs, and what areas are important for pollinators. How rules of thumb replicate reality is an outstanding empirical question, which needs to be addressed in the EAMs and in other areas where conservation development tradeoffs are likely to be the norm.
3. Opportunity costs – Another criterion for any PES to work for the poor necessitates that compensation levels are adequate to meet the opportunity costs of the ES providers. Simple monetary valuation of the opportunity costs, and monetary compensation, may work in some cases where there are fluid markets and substitutes for the activities that reflect the opportunity costs. For example, if forest degradation is driven by livestock ranching, and there are market substitutes for the nonmonetary benefits of raising livestock (milk, meat, leather), then paying ranchers their opportunity cost for not increasing their ranch lands may be an adequate mechanism. However, as is the case pointed out above, infrastructure, schools, and cash payments are unlikely to alleviate the need of the majority of households to collect firewood and poles in the EAMs. Here, the opportunity costs need to be met not in the monetary sense, but rather by understanding and providing livelihood and resource substitutes. In line with this, the poor must also not be so vulnerable such that limiting opportunities actually increases their vulnerability (Asquith et al. 2008).

There are also critical criteria for both the ecosystem service buyers and the institutional structures necessary for effective PES. These include the perception of the buyers that the ES will be provided by poor (Wunscher et al. 2008), a willingness to pay the necessary opportunity costs, and evidence that this provision is additional to what would be provided in the absence of a PES mechanism (Wunder et al. 2008). These criteria also include monitoring, verification, and adjudication structures – in short, high-capacity institutions and governance, the very criteria that are unlikely to exist in areas where there are high poverty–ecosystem service tradeoffs.

Conclusion

The role that PES can play in reducing poverty is a nuanced one, founded upon understanding the complex nature of poverty, the complexity of ecological functioning, and the interaction of these two systems. These relationships will certainly be context specific, for example, in some places, the poor are already living on degraded lands and landscapes providing few ecosystem services, in other instances, people might be income poor but have good living standards because of their local environmental condition. Here, we show that in our study area, poverty is multifaceted and that different indicators of poverty cannot be assumed to be collinear. This means that understanding any poverty–ecosystem service relationship requires a clear definition of poverty and an explicit indicator of such poverty. We also show that in areas typified by multiple dimensions of poverty, the dependence on ecological systems is likely to be very large; therefore any mechanism which has the potential to foreclose resource use options must carefully consider the actual mode of payment in PES scheme. In some cases, monetary compensation may be the most efficient. However, in cases where substitutes for essential goods are not available through the market, monetary compensation may not deliver the anticipated ES flows. We have seen that prioritization for poverty reduction determines a different intervention strategy than if we were to prioritize for ecosystem service delivery. This is an obvious result; however, it is unclear in the literature if all parties involved in the poverty-PES debate understand that there is a difference between "PES as a poverty alleviation tool" and "PES potentially delivering some poverty reduction co-benefits." Economic theory embodied in the "Tinbergen Rule" would suggest that the former stance is inefficient, that is, if you were to design a poverty reduction mechanism, would it be a PES scheme?

As has been pointed out elsewhere, there is no "silver bullet" for overcoming the very real tradeoffs between ecological conservation and human development (Ostrom et al. 2007). However, a careful understanding of the nature of any poverty–ecology relationship in a specific spatio-temporal context can highlight, if not win-win situations, at least instances where PES can help society to avoid deepening social traps.

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Chapter 14
Payment for Ecosystem Services for Energy, Biodiversity Conservation, and Poverty Reduction in Costa Rica

Natalia Estrada Carmona and Fabrice DeClerck

Introduction

Interest in ecosystem services and the development of financial mechanisms to incentivize their protection have rapidly expanded over the past decade. The notion of ecosystem services and their value was described by (Daily 1997) with the publication of the book "Nature's Services," which highlighted the notion that ecosystems provide human society with a variety of important needs. The valuation of these services was brought to light by (Costanza et al. 1997) who estimated that ecosystems provide humanity with \$33 trillion dollars per year in services, which was higher than global Gross National Product, \$18 trillion dollars, when the analysis was conducted. The notion of ecosystem services was further popularized by the Millennium Ecosystem Assessment (MEA 2005) that identified four classes of services: provisioning, regulating, cultural, and supporting. In reality, the public and private interest in ecosystem services has surpassed our ecological knowledge regarding how ecosystems and biological communities interact to provide these services (Daily, personal communication). As ecosystem services continue to gain in popularity and in demand, it is critical that ecologists and economists continue to collaborate in understanding how to correctly value the provisioning of services, and how to ensure that the service purchased by the buyer is provided by the landowner.

Payments for Ecosystem Services (PES) is a market-based instrument that recognizes the economic value of ecosystem services and strives to provide financial compensation to land owners who implement interventions that maintain, or improve the provisioning of these services through specific land management practices. In many cases, the goal of this payment is to encourage a change in land management by compensating for the costs of adopting natural resource management strategies that

F. DeClerck (✉)
Centro Agronómico Tropical de Investigación y Enseñanza (CATIE),
Cartago, Turrialba, Costa Rica
e-mail: fdeclerck@catie.ac.cr

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result in conservation, restoration, or increase of an ecosystem service of interest (Wunder 2006). Wunder (2006) compares PES with other instruments used to guarantee and generate ecosystem services, such as subsidies, taxes, Integrated Conservation and Development Projects (ICDP), and sustainable forest management initiatives. He suggests that payments are a more direct and efficient strategy for achieving conservation goals (Wunder 2006), which translates into a larger number of conservation units or services provided per dollar invested (Wendland et al. in press). Goldman et al. (2008) compared the success of The Nature Conservancy (TNC) projects that only include conservation goals to projects that combined conservation and ecosystem services goals (water purification, carbon sequestration, and opportunity for ecotourism) and concluded that: (1) projects including ecosystem services obtained greater revenue overall and more funding support from corporate sources with a wider variety of financial incentives; (2) projects including ecosystem services were better able to incorporate private lands and private landowners; (3) the ecosystem service projects played a more significant role in reducing projected agricultural land expansion; and (4) success of the conservation goals was critically dependent on monitoring and evidence of additionality (monitoring to provide evidence that the intervention had improved the condition beyond baseline measures).

Within PES schemes, it is essential that the ecosystem service or services that are being sold are well defined. Payments may be made directly for each unit of service supplied (ton of carbon) or for a set of practices and land uses that guarantee or improve the supply of the service sold (Wunder 2006). This involves understanding how the specific combinations and arrangements of species provide services at the plot scale (Naeem et al. 1996), to how specific combinations and arrangements of ecological communities provide services at landscape scales. For example, Bunker et al. (2005) show that within a single forest, changing the combination of species included can alter the amount of carbon stored by over 600%. At landscape scales, Chan et al. (2006) demonstrate that different portions of the landscape are important for different sets of ecosystem services. When payments are involved, the buyer must remain confident that they are receiving the service for which they have paid.

It is important that the payments help achieve conservation goals that are *additional* to what would happen in the absence of the payment. Clarity in conveying the achievements of PES is crucial to publicize this economic tool and to involve more stakeholders who are able to pay to contribute to enhancing the ecosystem service but are not sure of the effectiveness of their payments with respect to conservation outcomes (Wunder 2006).

It is also essential that ecosystem services be well defined in terms of what service is being provided, which ecosystem or ecological community is capable of providing the services, and in the case of multiple services, which portions of the landscape are providing services of interest. This spatial component of ecosystem services, which is well recognized in the ecological literature, has been less appreciated within ecosystem service markets. Some ecosystem services are relatively independent of location such as carbon that can be sequestered and stored on any portion of the landscape. Some services are very dependent on location, such as pollination, which is strongly tied to landscape context (Steffan-Dewenter et al. 2002) or distance to the nearest forest fragment (Ricketts et al. 2004). PES schemes that target biodiversity conservation often have not been very spatially explicit; however, when fragmentation is a concern, and the goal is to increase biological connectivity, the strategic placement of connectivity routes for biodiversity is very much spatially explicit. Farms that are located between two large forest patches are going to have a much more important role to play in maintaining connectivity than farms that are surrounded by a matrix of sugar cane, or located directly adjacent to urbanizations. PES schemes whose aim is to protect wild biodiversity will make a much better investment in the former than in the latter.

PES is evolving to take on multiple different forms and to include a range of different services. In this chapter, we recognize certification as a kind of PES mechanism because certification schemes aim to compensate farmers who apply environmentally friendly practices, which help conserve ecosystem services, in growing their crops through differentiated payments for their products. Various certification systems have been created to transmit information about production processes and their environmental and social impacts to consumers willing to pay a higher price for a good quality product and for compliance with certain environmental and social standards (Heidkamp et al. 2008). One of the sources of payments that we consider in this case study is the certification of coffee (and other crops) by organizations such as Rainforest Alliance and Starbucks in an effort to promote biodiversity friendly cultivation of coffee. The purchaser ensures the producer a premium price, or greater access to the market in exchange for on-farm interventions that conserve habitat for wildlife. This can include increasing tree density within the agricultural system through agroforestry as is common with coffee production, or it can include setting aside land on the farm for forest protection.

One of the most common forms of PES programs involves payments for land-use management practices that help conserve water regulation and/or purification services for energy production, drinking water, and other purposes. In Costa Rica, hydroelectric dams have become one of the most popular sources of clean energy providing more than 82% of the country's energy needs. The role of dams has become more important as the Arias government races to be the first country to be carbon neutral before 2021 (as a critical component of Costa Rica's "Peace with Nature Program"). While dams have enormous environmental and social costs in terms of extensive riverine ecosystems lost and the loss of connectivity for aquatic biodiversity, there is little doubt that hydroelectric dams rank amongst the highest in terms of clean and renewable energy production.

Landscape ecology is instrumental for designing PES schemes that aim to support conservation and poverty reduction, particularly in landscapes that generate multiple ecosystem services and, thus, the potential to attract multiple payments. All landscapes consist of numerous land uses, stakeholders, and stakeholder interests. As such, they are typically managed for multiple services simultaneously. Identifying common interests between stakeholders in terms of landscape management strategies presents opportunities for selling multiple ecosystem services from the same landscape. For example, increasing tree cover in a coffee farm can result in carbon sequestration and storage (Beer et al. 2003), may act as a buffer between monocultures and protected areas (Moguel and Toledo 1999; Perfecto et al. 1996), can provide pest control services (Perfecto et al. 1996; Varón et al. 2007, can help regulate hydrological services (Swift et al. 2004), and may provide habitat for biodiversity (Beer et al. 2003; Swift et al. 2004). Chan et al. (2006) demonstrated the multiple values provided by a landscape by mapping how different portions of the Salinas Valley in California, USA provide seven different services and the spatial distribution of these services in the landscape. Landscapes provide multiple services that have potential economic value, and the provisioning of these services is temporally and spatially explicit, such that not all portions of the landscape have the same ability to provide ecosystem services consistently throughout time. For these reasons, landscape ecological tools are especially useful for identifying areas in the landscape that have the greatest potential for generating services that could be conserved through compensation generated from a PES program.

In this chapter, we use a case study from the Volcan Central Talamanca Biological Corridor (VCTBC) in Turrialba, Costa Rica to show how the use of ecological tools and ecological modeling can be used to develop PES so that dollars invested are targeted in those portions of the landscape best able to provide specific services. We use two spatially explicit ecosystem services that have known markets and for which payments are actually being made within the VCTBC, namely (1) erosion control for its impact on energy production and (2) biological connectivity, which is the primary ecological objective of the biological corridor. The spatial targeting of these payments not only is essential in improving the efficiency of PES, but the targeting is also essential in improving the poverty reduction ability of PES by getting payments to farmers who may not otherwise know about the availability of payments. Furthermore, not all areas within the biological corridor are prone to high erosion rates, and certain areas are of greater concern for reducing erosion to decrease the cost of energy production.

Most of the poor tend to be concentrated in rural areas, particularly in locations such as the steep slopes of the upper watersheds (CGGIAR 1997; Heath and Binswanger 1996) where the natural ecosystems are more fragile but are of major importance to well-being. However, these communities do not have the economic capacity to reverse the effects caused by the overexploitation of resources in response to market demand, or to adopt environmentally sustainable production systems that would enable them to adapt to these changes or even to cover the transaction costs required to access at the PES. We present this case study with the goal of addressing four important questions: (1) how can landscape ecology help to identify target regions for the provisioning of ecosystem services, (2) what is the potential for payments for interventions that produce multiple ecosystem services, (3) how do we ensure that services that are paid for are actually received, and (4) how can these concepts be integrated in order to have a greater impact on poverty reduction?

The Volcan Central Talamanca Biological Corridor

In order to explore how ecological modeling can improve energy production, ecological connectivity, and poverty reduction, we use the case study of the Volcan Central Talamanca Biological Corridor (VCTBC) in Costa Rica (Fig. 14.1).

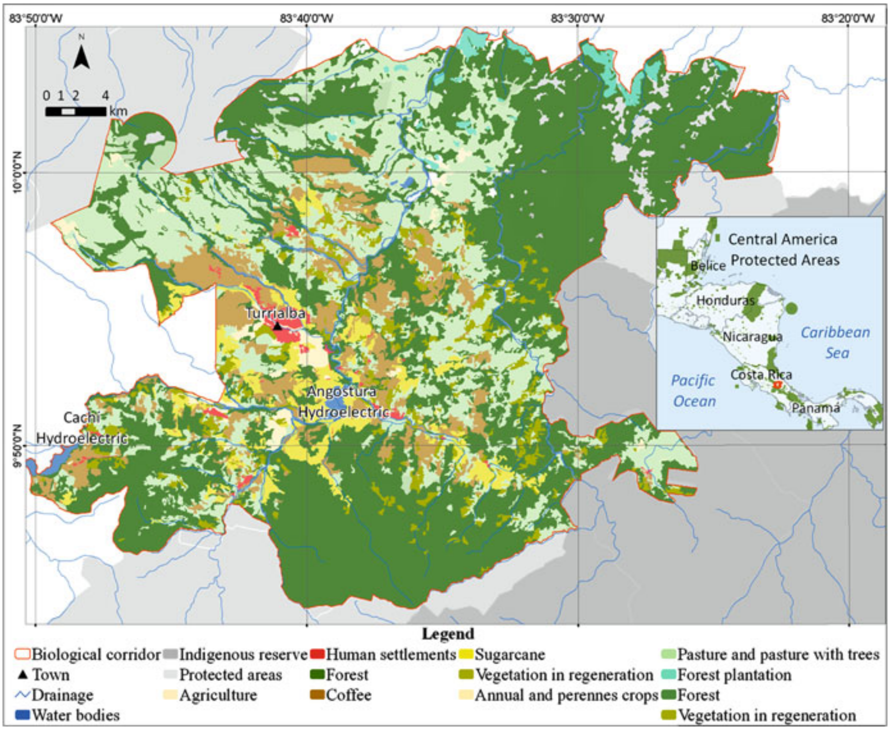

Fig. 14.1 Map of the Volcan Central Talamanca Biological Corridor and its land uses. The Cachi and Angostura hydroelectric dams are indicated. A new dam is slated for the northeastern portion of the biological corridor

The biological corridor covers an area of approximately 114,617 ha and connects nine protected areas and an indigenous reserve via an agricultural landscape dominated by forest patches. At a regional scale, the corridor is one part of the much larger Mesoamerican Biological Corridor that spans southern Mexico to Northern Colombia. The VCTBC unites Costa Rica's central cordillera to the north with the Talamanca Mountains to the south. The primary goal of the biological corridor, just as the names implies, is to provide sufficient natural and semi-natural habitat within the landscape to ensure biological connectivity, or the movement of organisms and genes, throughout the landscape. From a conceptual point of view, a forest-dependent species such as a jaguar should be able to cross from the Talamanca Mountains to the Central Cordillera with relative ease. Connectivity can be provided by maintaining forest cover, though it can also be maintained through wildlife friendly land uses such as tree plantations, multistrata agroforestry systems, or the conservation or riparian forests for example. As such, maintaining biological connectivity is one of the primary ecosystem services of interest in this region. The status of "Biological Corridor" is federally recognized and prioritizes federal PES programs in the region.

In addition to the important role that the corridor plays in conserving biodiversity, it is also one of the most important energy-producing regions in Costa Rica due to the combination of steep topography, and regular, abundant precipitation. The corridor has an important altitudinal feature with elevations ranging from 160 m at the mouth of the Reventazón River up to 3,330 m at the top of the Turrialba volcano. The area has an average annual rainfall of 4,400 mm largely distributed between the months of May through December and is situated in the middle of two important watersheds for hydoelectrical energy production: the Reventazón and Pacuare river basins. The Reventazón includes two operational dams, Angostura and Cachí, which are operated by the Costa Rican Electricity Institute (ICE) in addition to three privately operated hydroelectric tunnels, Tuis, Las Lajas, and La Joya. Together, these structures generate 25% of the country's energy needs, producing 2,367 GWh of Costa Rica's annual 9,400 GWh annual consumption. ICE is currently building at least two additional hydroelectric structures within the biological corridor, a hydroelectric tunnel tied to the Angostura dam, and a third dam in the lower stretches of the Reventazón. ICE estimates that these structures will produce an additional 2,100 GWh by 2014, or 37% of the projected national energy need.

One of the greatest costs of maintaining these structures has been the expense of removing excess sediments that accumulate behind the dam. Sedimentation is considered a cost both from the point of view of reducing the efficiency of the turbines, as well as in the cost of removing these sediments from behind the dams, which is typically accomplished by opening the floodgates to flush the sediments out. This implies an important loss of energy production potential. To address and reduce the effect of sediments, in 2000, ICE established the "integrated management plan for the Reventazón River basin." In this plan, three priority watersheds were selected to implement 6 year pilot projects testing the effectiveness of agroforestry systems, land management, sediment control, channel rehabilitation, education, and rural extension for reducing sediment loads.

ICE projected that investment in these interventions would decreases sediment loads by 21% in 6 years, and 55% in 10 years (Gómez, Cajiao, and Associates 2000). Currently, after 2 years of implementation, ICE has established six tree nurseries mostly managed by women. They have also provided loans and extension services to farmers in the watersheds to intensify cattle production decreasing the amount of land in open pasture, collecting manure for biogas production (27 fermenters installed), and decreasing agrochemical loads. This initial effort to change practices and improve ecosystem services has been well received in the community (ICE 2002). These targeted investments supporting land-use management practices are complemented by direct payments to farmers for forest protection and the management of agroforestry systems. Payments are made through Costa Rica's federal PES program organized by the National Forestry Financing Fund (FONAFIFO), which is supported by a gasoline tax.

FONAFIFO pays farmers for (1) reforestation, (2) forest protection, (3) water resource protection, (4) planting trees in agroforestry systems, and (5) natural regeneration as eligible projects for PSA. Each activity has a different amount of payment, restrictions, and requirements (FONAFIFO 2007). In the VCTB, which includes the cantons of Turrialba, Paraiso, Jimenez, Alvarado, and Siquirres, there were 124 PES contracts made in 2009. Forest protection represented the greatest number of PES contracts and represented the greatest area under payment as well (86%; 14,578 ha). Reforestation came second with 20 contracts and 402 ha covered, followed by forest management with three contracts over 175 ha. The contribution of PES to agroforestry included the planting of 25,732 trees between 14 contracts (FONAFIFO 2007). In 2009, FONAFIFO paid \$320 ha^{-1} for forest protection and \$980 ha^{-1} for reforestation with payments spread over a 5-year period. In contrast, agroforestry-based interventions were paid \$1.30 per additional tree planted with payments spread over 3 years (FONAFIFO 2007). Currently, FONAFIFO is developing agreements with private hydroelectric companies to encourage them to provide capital for payments made to farmers within targeted watersheds. The rationale is that payments that result in land-use change that help reduce the quantity of sediments in the river can decrease ICE's operational costs.

The second PES system we consider in the VCTB is environmental certification of coffee by the Rainforest Alliance, in particular. This certification system requires that farmers follow a norm developed by the Sustainable Agriculture Network, (2009) based on ten principles: (1) social and environmental management system, (2) ecosystem conservation, (3) wildlife protection, (4) water conservation, (5) fair treatment and good working conditions for workers, (6) occupational health and safety, (7) community relations, (8) integrated crop management, (9) soil management and conservation, and (10) integrated waste management. These principles are evaluated through pre-defined indicators. To obtain and maintain certification, farms must comply with at least 80% of all applicable criteria and 50% of each principle's applicable criteria (Sustainable Agriculture Network 2009). Two farms within the VCTBC have been certified since 2003 (Quispe 2007), including Aquiares, the largest coffee farm in the corridor.

According to a landscape analysis conducted by CATIE in 2009 (Estrada 2009), the corridor is comprised of 52% forest cover immersed in an agricultural matrix of pastures (24%) and coffee (8%). The fourth dominant land use in the corridor is sugar cane, occupying 4% of the total land area. Of these land uses, coffee has been promoted as a conservation friendly land use capable of providing multiple ecosystem services. As a shade-tolerant perennial crop, coffee cultivation is largely managed as agroforestry systems with varying degrees of tree cover. Depending on the structure, composition, and distribution of trees on coffee farms, these systems are promoted as capable of providing a wide range of ecosystem services including buffering the effects of monocultures on adjacent protected areas (Moguel and Toledo 1999; Perfecto et al. 1996), reducing soil erosion, reducing water contamination from agrochemicals (Perfecto et al. 1996), providing habitat for important pollinator species (Klein et al. 2003), and control of crop pest populations (Perfecto et al. 1996; Phillpott et al. 2009). In addition, coffee agroforests have been demonstrated as important sites for migratory birds (Komar 2006), capable of providing a diversity of timber and non-timber products (Beer et al. 2003), including carbon sequestration and storage (Swift et al. 2004).

However, according to estimates from the Costa Rican Coffee Institute (ICAFE), coffee production in the corridor has declined, with the coffee growing area declining from 11,912 to 10,006 ha between 2001 and 2005. A study carried out in the

VCTBC (Cerdán 2007) likewise found that 64% of local coffee producers have reduced the area planted with coffee and 3% were considering eliminating this crop. This same study identified the main factors responsible for this change as: (a) falling coffee prices, (b) proximity to urban areas, (c) the increase in land values and prices, and (d) farms located in marginal zones for coffee production. The greater portion of the coffee-growing area is being converted into pastureland and sugar cane.

Modeling Soil Erosion

Reducing erosion for energy production is directly related to four primary factors: (1) the intensity, duration, and quantity of rainfall a region receives; (2) the steepness of the slope, or even the location on a slope; (3) whether the soil type is easily erodible or not; and of course, (4) the characteristics and type of vegetation (forest, tomato field, sugar cane, coffee agroforest) covering the soil surface. Note that of the four variables mentioned above, only vegetation type can be easily manipulated, but that the others are very much tied to geography. Payment for erosion control therefore aims to change or maintain less erosive cover types, on the most erosive parts of the landscape in order to maximize the amount of ecosystem services received, per dollar paid. In addition, to these biophysical parameters, erosion control as an ecosystem service will have much greater value in valleys with hydroelectric dams, than on those that do not. Identifying which portions of the landscapes should be prioritized to maximize the provisioning of ecosystem services can be achieved through geographic information systems and remote sensing (see Chan et al. 2006; Naidoo and Iwamura 2007 for examples).

We used the well-known Revised Soil Loss Equation (RUSLE) (Wischmeier and Smith 1978) to identify critical erosion hotspots that should be prioritized for erosion control interventions. This equation quantifies the potential sediments produced from laminar erosion via a linear regression that relates five factors: (1) type of land cover management and the protection it offers the soil, (2) rainfall-runoff erosivity factor, or the capacity of the rainfall to remove soil particles based on rainfall intensity, (3) erodability of the soil based on its texture, organic matter content, and structure, (4) the combination of slope length and slope steepness or gradient, and, finally, (5) land management practice (Fig. 14.2). This last factor is generally assessed as a unit since the information available for the landscape at this scale and on these practices is generally very limited or nonexistent.

Geographic Information Systems (GIS) have increasingly proven useful for modeling environmental processes at landscape scales. Even RUSLE, which predates GIS now, can be run with standard GIS software such as ArcGIS through specially designed scripts and toolboxes such as N-SPECT (Eslinger et al. 2005). We ran the RUSLE model for the VCTBC and successfully identified the areas that produce the greatest amounts of sediment, or that have the greatest erosion. These areas were then classified in terms of their importance where areas with high sediment production being those areas where interventions must be made to reduce the

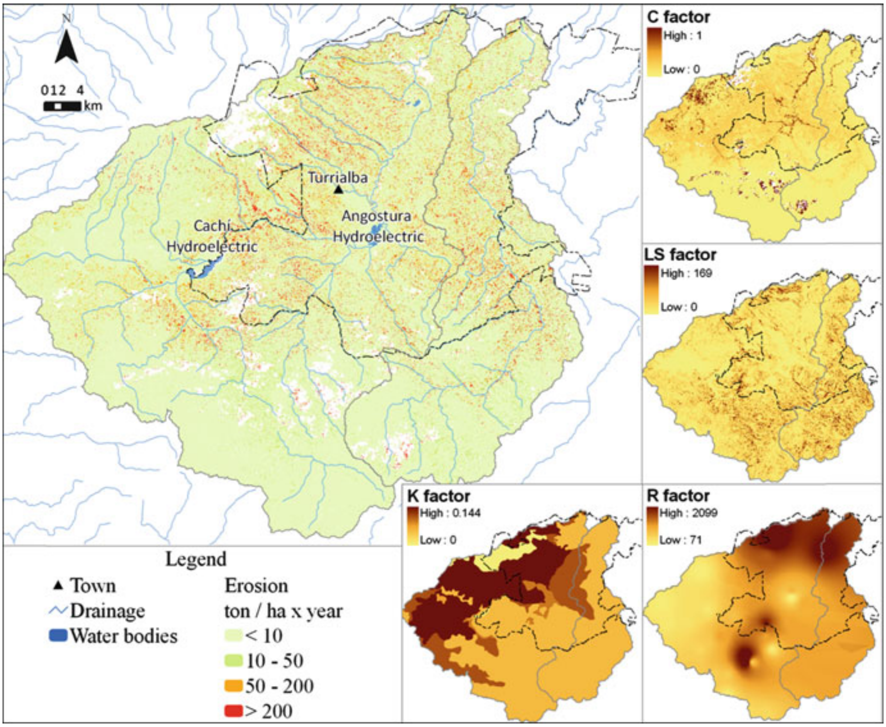

Fig. 14.2 Estimation potential erosion based on land management (C factor), slope length and steepness (LS factor), rainfall erosivity (R factor), and soil erodability (K factor)

amount of soil erosion (Fig. 14.3). These areas typically have specific topographic and soil characteristics that make them particularly vulnerable if inappropriate agricultural practices are used.

Modeling Functional Connectivity

The second service we model here is functional connectivity. Although biodiversity per se is not an ecosystem service, numerous certification standards (Rainforest Alliance, Starbuck's C.A.F.E. Practices, Utz Café, Smithsonian Bird Friendly to name a few) reward farmers for conservation interventions made on their farms through either better coffee prices, or guaranteed access to a market share. Most Mesoamerican conservation efforts have focused on the conservation value of different land uses particularly coffee and cacao production systems. However, these efforts fail to take into consideration the spatial location of the intervention and its contribution to landscape scale connectivity. In light of the VCTBC's goal of maintaining landscape scale connectivity, the strategic placement of conservation efforts

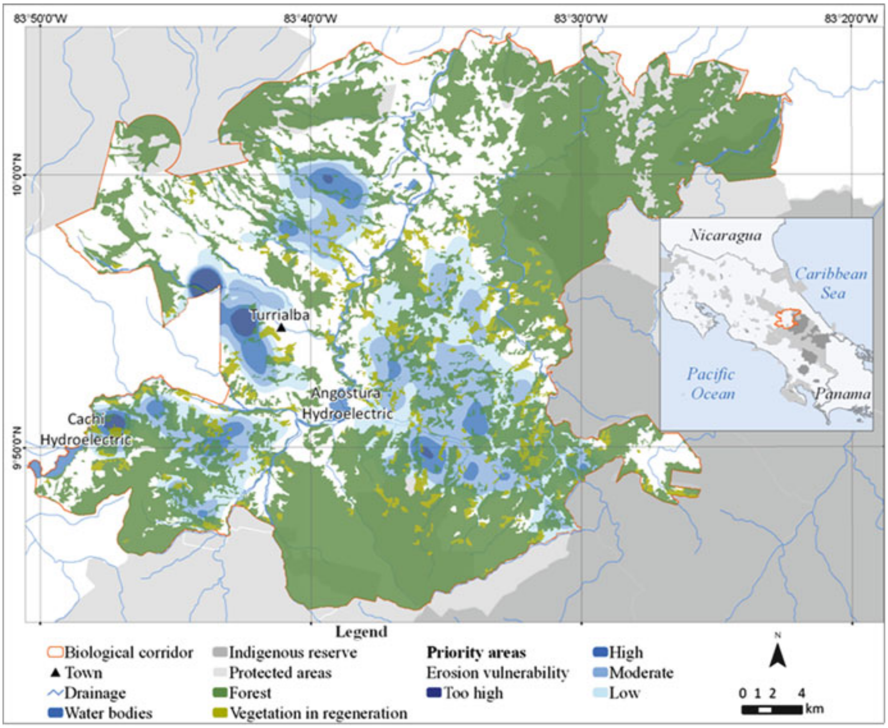

Fig. 14.3 Map of areas of high erosion potential within the VCTBC. These are the same areas which should be targeted for PES programs where erosion control is of interest. Areas with erosion above 200 t ha^{-1} year^{-1} are classified as "too high"; areas with erosion above 200 t ha^{-1} year^{-1}are classified as "high"; areas with erosion above 50 t ha^{-1} year^{-1} are classified as "moderate"; and areas with erosion above 10 t ha^{-1} year^{-1} are classified as "low"

within the landscape should be of critical importance. As such, modeling can be used to identify connectivity hotspots, or areas that should be specifically targeted for conservation efforts because of their contribution to maintaining biological connectivity. In the case of modeling functional connectivity in the VCTBC, we used a three-step process consisting of (1) selecting an appropriate indicator species for identifying critical corridors, (2) determining the potential distribution of the selected species within the VCTBC, and (3) modeling potential corridors or connectivity routes for these species.

In the case of the VCTBC, we selected the ochre-bellied flycatcher (*Mionectes oligeaneus*) as our indicator species for three primary reasons. First, this species is widespread throughout the biological corridor. Second, abundant biological data on the bird species exists, particularly, through a multi-year mist-netting effort that included mark and recapture data. This aspect was included given the need to have a minimum number of observations of the species for the correct functioning of the modeling tool to identify potential ecological niches. This data provided the basis for the connectivity modeling indicating the species affinity for forest and other habitats. Finally, the species is forest dependent; however, it also demonstrates an

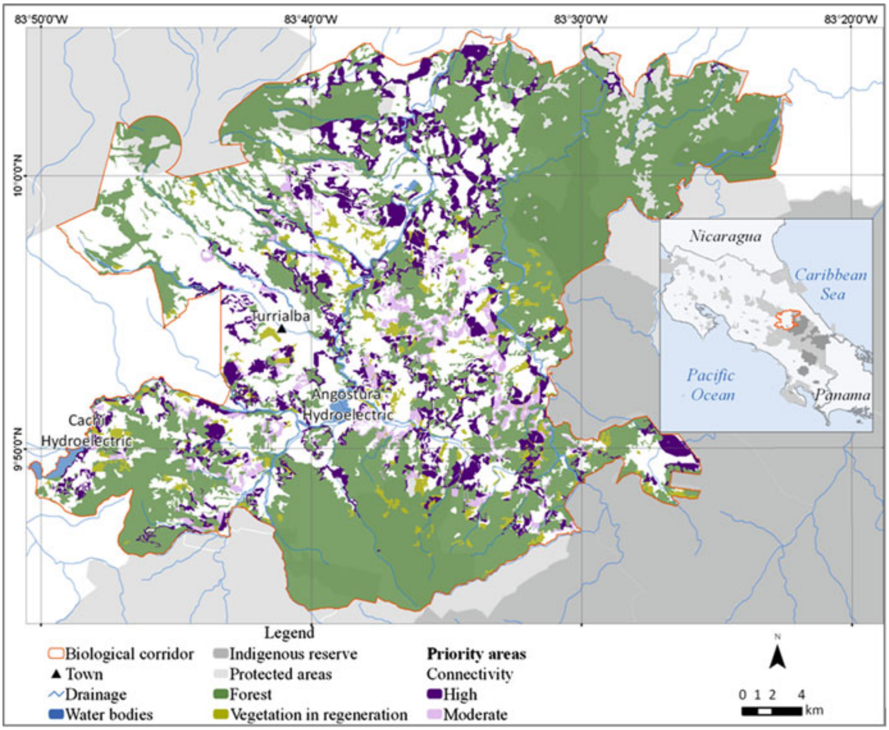

Fig. 14.4 Map of functional connectivity for the Ochre-bellied flycatcher. The *green* represents areas of quality habitat for the species. The *purple* colors depict critical gaps in functional connectivity for the species, which we propose should be targeted for ecosystem service payments or other conservation strategies

ability to move through semi-natural land uses – thus, it is a species that benefits from PES schemes such as Rainforest Alliance certification of coffee farms – the PES scheme that is the target of this modeling exercise.

To determine the potential distribution of the ochre-bellied flycatcher, we identified landscape scale environmental niche of the species representing the species regional range by associating different environmental layers in GIS including forest cover, precipitation, temperature, slope gradient, and orientation with a layer of reported sightings of the species throughout Costa Rica. We entered this data in the program "Desktop GARP[1]" generating a set of rules for predicting the location of the species within the VCTBC. The reason for developing this layer is to first identify those areas that potentially serve as habitat for this species within the corridor effectively placing a boundary on our connectivity model as it would not make sense to model connectivity of the species outside of its potential niche.

Next, we generated functional connectivity networks (Fig. 14.4) within this potential range using a graph theory-based GIS modeling tool called FUNCONN

[1] Desktop GARP: software package for biodiversity and ecologic research to predict and analyze wild species distributions. http://www.nhm.ku.edu/desktopgarp/.

(Theobald et al. 2006). Functional connectivity is distinct from structural connectivity in that the model includes species-specific parameters such as species perceptions of habitat quality, the permeability of, or the ability to cross, different land uses while moving across landscapes, the minimum size of suitable habitat patches, and species perceptions of edges. This added species specific parameterization permits the development of models that integrate the current spatial structure of the landscape, the distance between forest patches, the effect of various adjoining uses, and the size and shape of habitat patches and corridors.

These two maps permit the two primary buyers of ecosystem services to identify those areas within the corridor where interventions are most needed to either preserve, or improve on the provisioning of these two services. The combining of these two maps can be used to assess options for stacking water and coffee ecosystem services and can also contribute significantly to poverty reduction as we discuss below.

Access to PES Payments

Another benefit of the spatial targeting of ecosystem services is in increasing access to ecosystem service providers. Although Costa Rica's PES scheme is well-known in development and environmental circles internationally, it is less known to the farmers who could actually have access to these payments. In a workshop presented by these authors to the farmers of the VCTBC in 2009, we were struck by how few of the farmers we spoke to, knew of PES or were aware that they could apply for these funds. There were two problems. While farmers may know that including trees on their farms can reduce erosion, and recognize that their management practices affect their neighbors (just as they are impacted by their neighbors practices), few take this several steps further to consider the impacts of this erosion on energy production. Even fewer recognize that ICE or the Costa Rican government would be willing to pay them to decrease this erosion potential (see the chapter by Sears and Steward, on ecology and education for a greater discussion on this subject). The identification of ecosystem service hotspots permits the service buyer to "buy at the source" so to speak. This increases access of payments to the rural poor, reduces the cost of capacity building (since efforts can be targeted rather than diffuse), while at the same time increases the efficiency of PES.

The delimitation of the most important areas for each type of service, or for a set of services, immediately identifies the target population and its socioeconomic characteristics. This will help to define feasibility of PES, the most appropriate payment strategies, the required practices, the key actors and will also facilitate monitoring. This selection of priority areas to enroll in payment schemes does not necessarily exclude farmers who are willing to participate but whose farms are not located in strategic areas, but it does imply that they would have reduced access to payments, in the same way that farmers have access to different markets as a result of the crops they cultivate. The targeting of regions where service are most needed, or areas that are most capable to provide a specific service helps to ensure that the limited economic resources available for PES are invested in areas where the investment is really important and where the intervention will really be effective.

Managing Multiple Ecosystem Services

Many providers of ecosystem services have indicated that payments made are rather small, and often do not cover the intervention cost. One means of increasing payments has been to "stack" one or more services and receive revenue from multiple services generated from the same landscape or to "bundle" additional services into a primary service that is being sold (e.g., the selling of carbon sequestration/storage services with biodiversity benefits). Country-level PES programs generally tend to include various ecosystem services because of the broad range of users and/or consumers at the local, national, and international scales, who have different investment interests (carbon, biodiversity, water quality, quantity, and regulation, for example). Therefore, a general strategy in ecosystem service supply is to promote the conservation of forest remnants that provide multiple services such as protecting biodiversity, preserving landscape beauty, and delivering watershed services, as has been done in Mexico and Costa Rica (Porras et al. 2008). The selling of multiple ecosystem services through stacking or bundling is gaining in popularity, both in the ecological literature, as well as, in the voluntary markets for ecosystem services. For example, in terms of the voluntary markets that are often made with increasing public relations in mind, private companies have expressed greater interest in paying for carbon storage/sequestration, which is "bundled" with biodiversity and poverty reduction benefits over those schemes that provide one service (Note: Poverty reduction is a social service rather than an ecosystem service).

If we combine the erosion control and connectivity for movement of biodiversity models from Figs. 14.3 and 14.4, we can easily pinpoint the areas that are important for each individual service as well as those areas of overlap between both services (Fig. 14.5). With these models, we identify that 36% of the total area of the VCTBC contains strategic areas for the provisioning of both ecosystem services evaluated. Of this area, 62% is important for controlling and reducing erosion; 22% is important for providing functional connectivity within the corridor; and 16% of the area provides both services. Currently, forests predominate within these areas, occupying 31% of the total area prioritized. These are areas where PES or other interventions could be used to maintain and/or improve the contribution of forests for providing these ecosystem services. In contrast, 28% of the area is in pastures and 23% is in monoculture coffee. In these areas, PES or other conservation interventions could be used to either change the land use and/or to promote systems such as agroforestry, which has the specific aim of improving both erosion control and biological connectivity.

From an ecological point of view, the importance of stacking or bundling comes with the realization that the value of biodiversity increases with the number of services

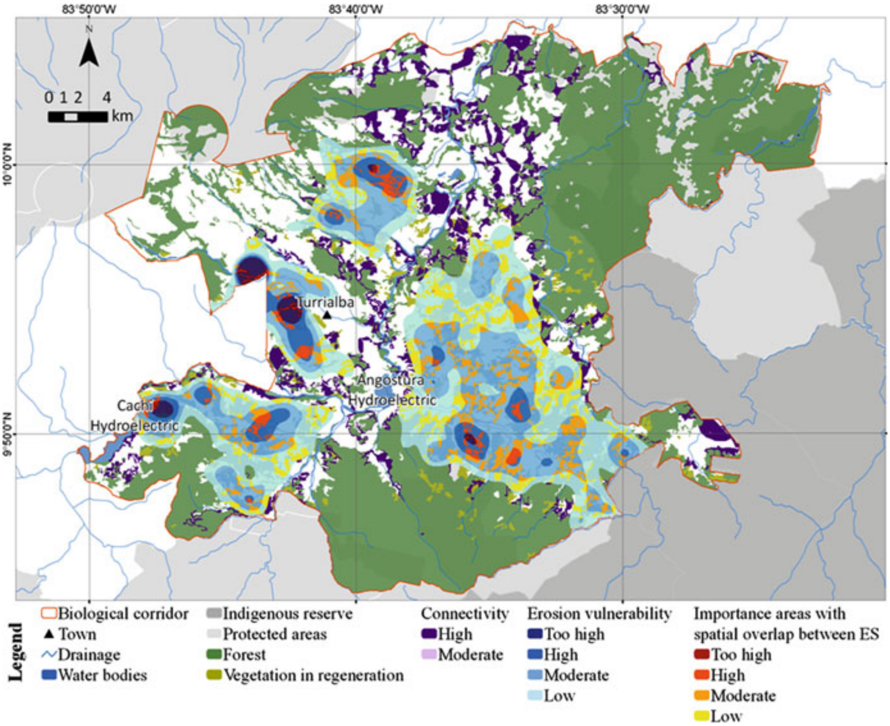

Fig. 14.5 Prioritization of areas in terms of the type and level of ecosystem service supplied with the overlap between erosion control services and biological connectivity

that are sought or generated as a result of payments. If we use the example of hydrological services for the VCTBC, ICE pays farmers for reducing erosion by planting native trees on their farms, maintaining forest patches, and/or using agroforestry technologies. While these interventions help to reduce erosion, they frequently also contribute to, or can be managed for carbon sequestration, biodiversity conservation, and scenic value for example. There is also increasing evidence that these interventions may contribute to on-farm services such as pest control (Perfecto et al. 2004), pollinator services (Ricketts et al. 2004), or soil quality (Perfecto et al. 1996), not to mention a host of services that may not have been identified or quantified to date even though erosion control is the only service being paid for. Note that erosion control, carbon sequestration, and scenic value are all services provided by the farmer that, respectively, reduce the cost of energy production for all energy customers, offset the impact of climate change, and potentially increase tourist revenue. Because their benefits are received off-farm by different types of consumers and may require slightly different land-use interventions, they each have the potential of being included in a different PES scheme. In contrast, pest control, pollination, and soil quality are services received by the farmer, and should in practice either reduce management costs, or increase farm productivity over time. A key challenge with stacking or bundling is that the same service should not be "double

Table 14.1 Ecosystem services and commodities. Examples of watershed services and associated proxy commodities (data from Indonesia, India, Bolivia, El Salvador, Central America, Mexico, Ecuador, and Costa Rica)

Services	Commodity
Water quality and water quantity	• Improved land practices through soil conservation and zoning • Conservation of existing forests and reforestation
Water quality, regulation of water flow, biodiversity protection, carbon sequestration, reduction of landslide risks, and scenic beauty	• Protection and restoration of existing forests • Improved land practices through combining trees with agricultural production (agroforestry, silvopastoral practices, shade-grown coffee, live fences)
Water quality, regulation of water flow, and reduction of landslide risks	Improved land practices in agricultural land (mulching, low tillage, live barriers, conservation works)
Water quantity and regulation of water flow	Mostly conservation of paramo and natural forests, but also some improved agriculture measures
Regulation of water flow and reduction of sediments in lakes	Mostly conservation of existing forests and prevention to conversion

Adapted from Porras et al. 2008

counted," which means it cannot be sold twice. When attempting to sell single or multiple services from the same landscape, sellers (or farmer, in this case) will have to demonstrate that the payment received is resulting in the provisioning of an ecosystem service that is *additional* to what would have been conserved or enhanced in the absence of payments from single or multiple PES schemes. It will be important for sellers and buyers to decide at an early stage if it is more advantageous for them to stack (sell multiple services from the same landscape individually to different markets) or to bundle (to sell a primary service, such as carbon storage, with bundled services, such as biodiversity, included in the sale for a slightly higher price than would be paid for the sale of one service alone).

The ecological notion of complementarity can be integrated into PES schemes as well. An ecosystem service provider may offer one or several services, depending in large part on the site's location within the landscape, which will be a determining factor in terms of the quality, quantity, and type of services supplied. Furthermore, certain land-use or management practices may complement the provision of different ecosystem services, whereas some land-use practices may be rival with respect to generating different multiple ecosystem services. For example, plantations of fast-growing tree species can be used for carbon sequestration, but are poor habitat for biodiversity, often do not improve the provisioning of water and are not appealing to tourists (Wunder 2006). The factors of complementarity and competition between uses and practices as related to different ecosystem services reinforce the idea of landscape scale planning for prioritizing areas for targeted payments, where ecosystem service buyers can be assured that their payments are promoting practices that provide these services while not undermining other important ecosystem services (Table 14.1).

Landscape Planning for ES

As we have demonstrated here, many ecosystem services of interest require landscape scale planning efforts and are associated with the structure, composition, and design of agricultural landscapes. Thus, a strategic plan or design based on sustainable management and arrangements or reforestation efforts from farm to landscape scales can contribute to landscape configurations that guarantee or increase the supply of the more established ecosystem services and of those that are newer to the market. This vision is totally different yet complementary to the traditional view that regards natural and planted forests as suppliers of marketable ecosystem services. The inclusion of agricultural landscapes as a source of ecosystem services (see the chapter by Smukler et al. on agriculture and ecosystem service) recognizes various factors: (1) the high proportion of the terrestrial landscapes under agriculture or other managed land use in comparison to forested or natural land, (2) the need to restore the quality of natural resources and the supply of ecosystem services in environmentally vulnerable areas where agricultural activities are essential to the food supply and the local economy, (3) the impact that sustainable management of different land uses has on various ecosystem services, and (4) the contribution that sustainable management systems can make to reducing the pressure and degradation of forest areas.

Well-Defined Services Permits Well-Defined Payments

An important factor is to ensure that service providers are familiar with the location and role within the landscape of each ecosystem service supplied. For example, with biodiversity conservation, the quantity of wildlife conserved will not be the same in an isolated area when compared to a farm that is located between two forest patches. This begs the question, should the payments be equal as well? Similarly, the investment made in practices to control soil erosion are not the same for a farm located on a high slope as for one located in a flat area. This localization or contextualization of service providers in the landscape enables them to gauge the extent to which their improved practices contribute to conservation. This differentiation of the ecosystem service supply at landscape level, both in natural and semi-natural managed systems, may also serve to include recommendations regarding price differentiation, thereby incorporating the different opportunity costs for conservation or for the adoption of certain practices, along with the quantity and quality of the ecosystem service supplied for which payment is being made.

How much to pay remains a central question of most PES schemes. Many such schemes have used studies on the opportunity costs of conservation or users' willingness to pay for services, the cost of adopting sustainable management practices, or have simply divided the monthly budget by the area to be conserved. (Porras et al. 2008) found that the most common strategies for determining payments, are: (1) based on administrative terms, for which national prices are established using one of the methods mentioned or a combination of several methods to identify the optimum price; and (2) based on negotiations (direct or through intermediaries). However, some studies (Wünscher et al. 2008; Wendland et al. in press; Robalino et al. 2008) have shown that PES programs are more efficient when the price is differentiated in terms of (a) the quality and quantity of the ecosystem service delivered, (b) the risk of losing the service, and (c) providers' participation costs (transaction costs, opportunity costs of conservation, and/or restoration). In this case study, we propose that landscape ecology and modeling can and should play a bigger role in establishing price differentiations in terms of the landscape's capacity to provide the service or services of interest. Hotspots should receive greater payments than areas that are not as critical, with intermediate payments in those areas that are intermediate. Price differentiation, according to the recommendations of Wünscher et al. (2008) and Robalino. et al. (2008), makes it possible to select PES areas with low opportunity costs, thereby increasing the size of the area contracted, as well as, targeting the payments in areas of high additionality. Finally, such differentiation should help focus the markets and insert a degree of competitiveness in the market to ensure that the buyers are getting the service they pay for.

The bundling or stacking of services are additional mechanisms that can be used to help increase revenue to farmers or land-use managers who are helping maintain or generate several ecosystem services, particularly when the interventions required for providing multiple services are complementary. In the case of the VCTBC, overlaying the maps of erosion hotspots and connectivity hotspots identifies those areas where there may be an opportunity for farmers to receive additional or higher payments if they are implementing interventions that result in the generation or maintenance of multiple ecosystem services. However, it is important to ensure that payments are resulting in ecosystem service provisioning that is *additional* to that which would occur in the absence of a payment so that double counting can be avoided.

The spatial distribution of ecosystem service providers and the socioeconomic characteristics of those providers influences the quantity of ecosystem services that can be offered at the landscape scale. In most existing PES schemes, the ability to participate depends on the management capacity of the producer or producers (cooperatives); on the ability to cover the transaction costs; on legal land tenure; and on knowledge of these instruments and their benefits. For these reasons, payments tend to be distributed, dispersed, or even targeted in areas where the threat to conservation is nil or very low, but the socioeconomic conditions are more conducive to PES. In some cases, payments have even included recreational farms, where the degradation or elimination of forests is very unlikely (Miranda et al. 2003). This situation once again underscores the need to target payments in areas where the supply of ecosystem services is really being threatened, and, therefore, where external incentives are most needed to guarantee the protection and improvement of natural and semi-natural systems. This approach also promotes a more equitable participation by providers in payment schemes.

Conclusions: Can PES Ultimately Contribute to Poverty Reduction?

It is important to bear in mind that the main purpose of PES is to guarantee the supply of different ecosystem services. However, based on its design and structure, PES may also contribute to poverty reduction. The surge in discussion of payments for ecosystem services in development circles has generated many expectations in terms of rural farmers receiving significant income contributions from PES interventions they implement. Many of these expectations have not come to fruition, and are unrealistic. Rural communities who are aware of these payments often do not understand the ecological mechanisms that drive PES (See the chapter by Sears and Steward), and do not realize that, as with any market, payments will be driven by demand. Nevertheless, PES can serve as a platform and a foundation for other development projects.

FAO (2007) recommends the inclusion of four important steps in the design of an effective PES system: (1) identify what should be paid, (2) who should be paid, (3) how much should be paid, and (4) what payment mechanisms should be used. However, in line with what has been mentioned previously, a fifth factor should be included, (5) *where* payments should be spatially targeted. It is also important to recognize that the goals of PES programs must be consistent with national policies and multilateral commitments.

Payments scheme designs are complex and should integrate the environmental conditions needed for their establishment (social, economical, political) as well as the technical design (demand – offer stakeholders and areas and costs), implementation, evaluation, and monitoring (Madrigal and Alpizar Rodriguez 2008; Jack et al. 2008). Payment schemes will have to be tailored to the different environmental, economic, social, cultural, political, and human contexts unique to each country and region. However, the growing body of literature on different payment strategies facilitates the adaptation and adoption of this tool, so that its impact and efficiency can be improved, especially in relation to poor rural populations, who have suffered and continue to suffer the most from the degradation of natural resources and who have few economic resources for adapting to ecological degradation and disturbances. Fortunately, there are now powerful tools such as geographic information systems, remote sensing, and landscape modeling that help prioritize areas for each ecosystem service supplied and to fine-tune payments strategies so that PES schemes are developed efficiently with respect to landscape context.

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Chapter 15
Introduction: Ecosystem Governance for Conservation and Poverty Reduction

Jane Carter Ingram and Caleb McClennen

As all of the chapters throughout this volume have discussed, increasing amounts of pressure have been exerted on natural resources by a wide variety of user groups. In the absence of effective governance institutions implemented at the appropriate ecological scale, natural resources and the environment are in peril from these increasing pressures (Dietz et al. 2003). As Ostrom (2009) outlines, resource collapse is more likely in large, highly valuable, open-access systems when the resource harvesters are diverse, do not communicate, and fail to develop rules and norms for managing the resource. Establishing effective institutions and processes for managing multiple resources in such situations is often a highly complex undertaking (Wilkie et al. 2008). For example, many globally valuable natural resources, including a wide variety of flora and fauna species, fisheries, minerals, fossil fuels, timber and water are also connected to ecosystem components or functions that are critical for rural livelihoods. Managing multiple, simultaneous, and often interacting endogenous and exogenous demands on these resources can be especially challenging when tenure or resource rights are non-existent or unclear; governance structures to enforce rights or ensure equity in natural resource decision making are weak or non-existent; and/or when the extent of a resource or system transcends state or national boundaries as is often the case with fisheries or river basins, for example. This section is concerned with how some of these management challenges manifest themselves, the implications for ecosystem integrity and poverty reduction, and some of the solutions that have arisen for managing those challenges.

McClennen begins this discussion with one of the most difficult natural resources to manage, fisheries, in the face of ecological complexity and scientific uncertainty. He reviews the disastrous management failures associated with many fisheries and the devastating consequences similar, future failures could have in developing countries, where 90% of all fishermen/women are located. McClennen describes a broad

J.C. Ingram (✉)
Wildlife Conservation Society,
2300 Southern Boulevard, Bronx, NY 10460, USA
e-mail: cingram@wcs.org

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shift away from single fisheries management towards an ecosystem-based management (EBM) approach, which focuses on management of the whole system to ensure that a variety of ecological functions and species upon which both commercial and small-scale fisheries depend are conserved. In order to ensure multiple fisheries are managed to support livelihoods of small-scale fishing communities and to sustain fishery stocks at ecologically functional levels, McClennen outlines the importance of clear resource rights in fisheries management, such as Individual Transferable Quotas for regulating fish catches; recognition and support for community-level governance of fishery resources, particularly in the face of international fishing fleets; international policies that reduce perverse subsidies that have led to overfishing historically; and market-based approaches that encourage sustainable fishing practices.

Naughton-Treves explores in depth the challenges of managing different rights and uses of resources through participatory zoning practices that aim to balance conservation and poverty reduction. Through several case studies, she outlines the difficulties in zoning a landscape for multiple purposes important to various stakeholder groups and the political obstacles to doing this, which may last for many years. From her review and analysis, Naughton concludes that scientists involved in ecological zoning are more likely to be effective if they are transparent in their work, incorporate local ecological knowledge, and clearly communicate the benefits of different zones for neighboring communities. Recent advances in Geographic Information Systems (GIS) technology and participatory mapping present opportunities for better communication and collaboration.

Zoning often involves setting aside land or coastal waters for protected areas (PAs), but the question of how much to set aside to sustain ecologically functional populations and to support neighboring, human communities is an ongoing topic of debate. The chapter by Holland discusses the evolution of PAs as a means for conserving biodiversity and how the roles with which they have been charged have changed throughout their history, particularly with respect to their role in poverty reduction. Despite a dramatic increase in PAs over the last few decades, it has been difficult to assess their effectiveness at biodiversity conservation or for supporting poverty reduction. In light of this knowledge gap, Holland discusses the importance of the scientific community for providing numbers and concrete objectives relating to what, where, and how much area is required for effective biodiversity conservation and protected area establishment. As these objectives are matched alongside those of poverty reduction goals, ecologists can more effectively inform the development of common sets of metrics for analysis and modeling across scales. Additionally, as more is understood about what should be protected and where, ecologists can help move the policy discussions beyond that of "minima" and into an improved understanding of how conservation can best be achieved.

These chapters illustrate several challenges of governing and effectively conserving biodiversity and multiple ecosystem services from land or sea-scapes in developing countries while ensuring benefits generated are equitably distributed. In each of these chapters, clear role for ecologists exists not only in fostering scientific understanding about biodiversity and ecosystem functioning to inform natural resource governance and decision making, but, also in helping to communicate complex information about ecosystem dynamics in ways that are palatable, transparent, and meaningful for policy makers and many different resource users.

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Chapter 16
Sustainable Fisheries Production: Management Challenges and Implications for Coastal Poverty

Caleb McClennen

Wild-caught fisheries and aquaculture constitute a critical sector for fishing communities and developing coastal economies throughout the world. They contribute a nominal value of $170 billion to the global economy, while supporting the livelihoods of an estimated 520 million people (FAO 2008). While a significant portion of the value of production is through large-scale or high-value stocks such as tuna, shrimp, and anchovy, demographically, 90% of all fishermen work at a small scale and live in developing countries (McClanahan et al. 2008; World Bank 2008). At the household level, fisheries provide not only significant employment income but also critical food security as the main source of protein for numerous small island states and coastal countries such as Bangladesh, Cambodia, Equatorial Guinea, French Guiana, the Gambia, Ghana, Indonesia, and Sierra Leone (FAO 2008). On a national scale, fishery contribution to Gross Domestic Product (GDP) ranges from around 0.5% to 2.5%, but may be as much as 7% in some coastal countries, such as Senegal, and upwards of 20% on small Pacific islands (Zeller et al. 2007).

Despite its critical importance in coastal economies, global wild-caught fishery production has stagnated since the mid-1980s between 80 and 90 million metric tons (MT). Temperate fishery production has declined to half its historic production since the mid-1980s due to systematic failures in fisheries management including the collapse of several of the world's largest greatest fisheries. Concurrent to this, catch supplied from tropical, developing countries has more than doubled, enabling a steady contribution of wild-caught fisheries from these areas to the global economic system (Fig. 16.1). In the past several decades, the world's net fishery production has avoided collapse by the exploitation of new untapped

C. McClennen (✉)
Wildlife Conservation Society, 2300 Southern Blvd., Bronx, NY, USA
e-mail: cmcclennen@wcs.org

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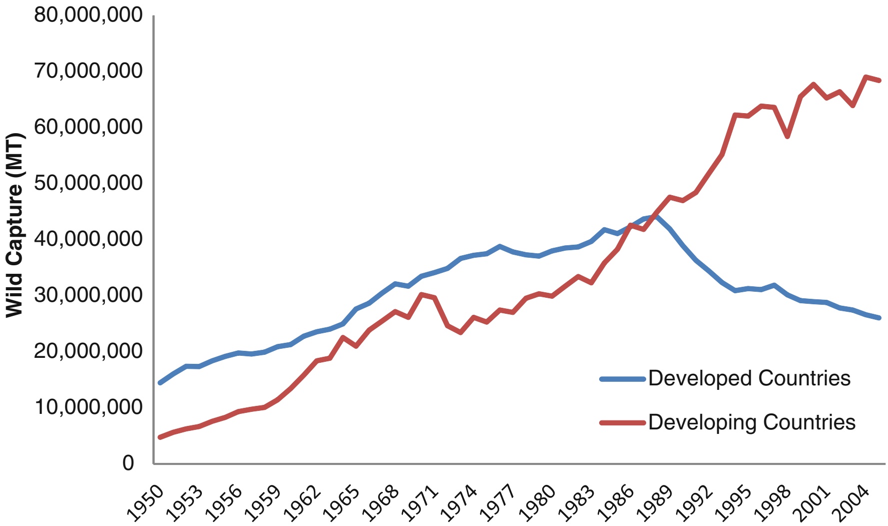

Fig. 16.1 Global wild capture fisheries production in developed and developing countries. 1950–2006 (Data source: FAO 2008)

resources as a substitution for declines in historically productive fisheries. Leading fisheries scientist Daniel Pauly (2003) has labeled our historic production of fisheries as "a series of serial depletions, long masked by improved technology, geographic expansion and exploitation of previously spurned species lower on the food web." Global output has been sustained only by the substitution of newly discovered or newly exploited fisheries for depleted, historically robust, fishery resources (Pauly et al. 1998). Asian sea urchin fisheries are a classic example of this transition. Confined to Japanese waters in the 1940s, by the 1990s, the fishery had expanded worldwide, while domestic Japanese catches had become negligible (Berkes et al. 2006). The waters off the United States, biologically some of the most productive on the planet, provide another powerful example as US consumers now import more than 85% of their seafood due to lack of adequate domestic supplies (NOAA 2008).

This chapter documents the complexity of fishery management from an ecological perspective, exploring a variety of driving factors that have led to the ubiquitous difficulties in sustainable fisheries management. Many of the challenges described are from well-developed fisheries, with an implication that globalized fishery production creates the unfortunate potential that history may continue to repeat itself throughout the developing world with significant consequences for the livelihoods of coastal communities and developing economies. The challenge presented to the world's coastal poor is acute, as integrated markets and globalized trade provide not opportunities, but also concomitant threats to the viability of fisheries as the bedrock for long-term sustainable and secure growth of developing coastal economies.

The Challenge of Fisheries Management

Until the twentieth Century, humankind's understanding of the world's oceans rested in a paradigm of inexhaustibility. Articulated at the turn of the seventeenth century by Hugo Grotius, in his historic treatise on the freedom of the seas, *Mare Liberum*, "For everyone admits that if a great many persons hunt on the land or fish in a river, the forest is easily exhausted of wild animals and the river of fish, but such a contingency is impossible in the case of the sea." Famously, the British ecologist, Thomas Huxley, stated in a fisheries conference in the mid-1800s that "…the cod fishery, the herring fishery, the pilchard fishery, the mackerel fishery, and probably all the great sea-fisheries, are inexhaustible; that is to say that nothing we do seriously affects the number of fish. And any attempt to regulate these fisheries seems consequently… to be useless" (Ellis 2003). Notably, this paradigm was neither universal nor timeless, as a number of societies through the millennia have engaged in successful marine resource management (Johannes 1982). By the early 1900s, the perceptions of an inexhaustible ocean, as reflected by Grotius and Huxley above, were beginning to erode by mounting commercial and ecological extinctions of the great whales, crashes of oysters, California sardines, Atlantic cod, and Pacific fur seals (Jackson et al. 2001). It is now estimated that there has been a 90% drop in the abundance of top predators, such as tunas, marlins, and sharks from pre-exploitation levels throughout the world's oceans (Myers and Worm 2003), and thus, the long-standing paradigm of an inexhaustible sea has now long been shattered.

In response, the science of fisheries management emerged. The science was founded upon population biology theories of varying population growth rates, and the concept of surplus production. The concept is based on the assumption of an s-shaped population growth curve, where small populations reproduce slowly due to scarcity and mid-sized populations reproduce most rapidly until their growth rate diminishes as a carrying capacity is approached. Using this theoretical basis, if a population could be maintained at an appropriate high growth level, significant surplus production could generate a theoretical maximum yield (Fig. 16.2). The concept of a Maximum Sustained Yield (MSY) thus emerged in the early half of the

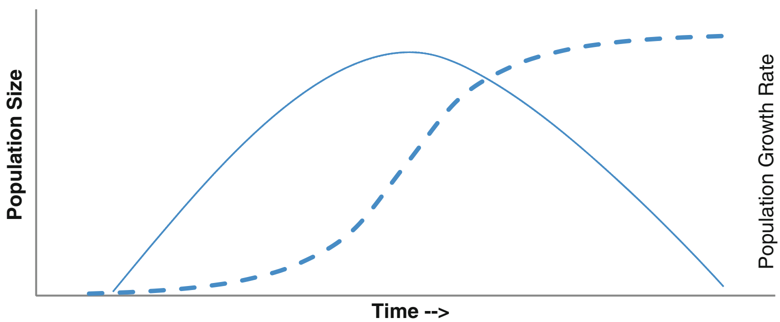

Fig. 16.2 A theoretical population growth curve (*dashed*) and its associated growth rate (*solid*)

twentieth century as a management paradigm upon which the fate of the world's fisheries would rest, and remains the core component of the majority of fisheries management institutions to this day. This positivist approach armed fisheries managers in developed countries with sophisticated population models, to calculate and supposedly catch at levels approximate to MSY. Unfortunately, simultaneous to the emergence and dominance of this paradigm for fisheries management, it became evident that fishery production did not always behave according to the models. Fish stocks did not exist in a vacuum, nor did humans always respect what was being recommended by the models. Thus, modeling stock yield alone has failed to enable sustainable management as stock after stock has collapsed either by faulty or simplistic production models or the failure of governance systems charged with their implementation (Beddington et al. 2009). Specifically, the MSY model on its own fails to incorporate the complexity of fisheries' ecology including the complexities of trophic relationships, oceanographic variation, land-based degradation, and coastal/marine habitat interactions. As "under-exploited" fisheries in developing countries are increasingly targeted, the potential downside of managing stocks according to these simplified rules of population biology without incorporating what we now know about ecosystems and human fishery exploitation is significant. Some of the complexities associated with fisheries management that are important to consider are described in the following sections of this chapter and, as the examples demonstrate, precautionary, ecosystem-based management approaches are imperative, especially when making fisheries management decisions in the absence of perfect information.

Trophic Cascades

Trophic cascades may result from poor management of a single fishery that causes a series of intertwined impacts across a range of other fisheries via ecological linkages. Due to their complexity, these cascades can be difficult to predict and even more difficult to reverse. One of the most infamous of trophic cascades resulted from the global collapse of the Atlantic Cod (*Gaddus morhua*),which plummeted from peak exploitation levels of 3.5 million tons in 1968 to less than a million today (Fig. 16.3). Though an early driver of the economy of the American colonies, in the 1960s and 1970s, US domestic catch of Atlantic cod produced only 5% of the global total. In the early 1980s, as the US domesticated all fisheries within a 200 nautical mile exclusive economic zone (EEZ), intense capitalization of the US cod fishery began a 10 year phase of heavy over-exploitation. This short-term boost to domestic production of cod was followed by a crash in the beginning of the 1990s that has led to current catch levels at less than 5% of their historic peak. The central blame for the crash of the cod fishery has been far too much fishing capacity, or fishing boats at sea, and quotas that had been set far too high to sustain the stock. The great mystery of the cod crash has been however, not its crash, given that catches and effort (fishing days) far outpaced what scientific advice suggested was "sustainable,"

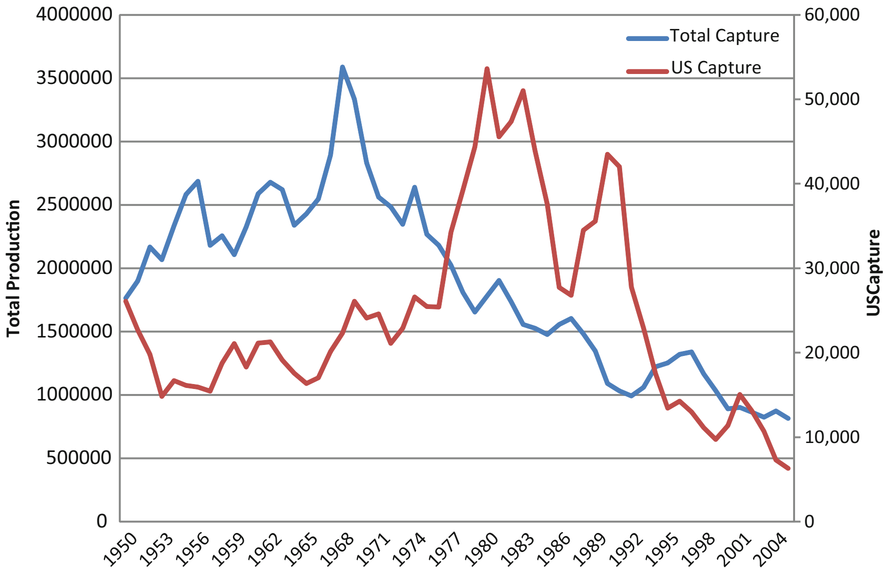

Fig. 16.3 Global and US cod production in metric tons (1950–2005) (Data source: FAO 2008)

but rather that with massive constraint in effort, the stock has yet to recover and has caused a series of other impacts. It has been demonstrated by Frank et al. (2005) that the loss of cod in the Northwest Atlantic Ocean has triggered a trophic cascade leading to a long-term regime shift of this large marine ecosystem. The loss of the cod has led to a boom in populations of crab, shrimp, and small pelagic fish, subsequently devastating zooplankton and causing a boom in the system's primary producers—phytoplankton. In addition to the trophic cascade resulting from the loss of cod, bottom habitat was permanently altered by trawlers dragging large-weighted nets across the ocean floor for cod. Bottom trawls contained high levels of bycatch (non-target species) and damage corals, sponges, and shellfish while leaving a large sedimentary plume in their wake, causing numerous secondary effects. Thus, while fishery managers in the US allowed increasing catches and capacity, the foundation for the large scale ecosystem was collapsing—and by the early 1990s, the cod fisheries began to disappear to the point where managers began to close off the fishery to allow for recovery. The collapse also had significant socio-economic impacts as the fishing communities that once depended on cod fisheries in the 1990s experienced extreme economic hardship as the cod fisheries were closed. Due to the collapse, the fishing communities of New England lost an estimated 14,000 jobs and $350 million were lost (Hennesy 2000). The same closures occurred at the same time in the Canadian waters to the North, as a moratorium on fishing the Grand Banks in 1992 caused the single largest sector layoff in Canadian history, and resulted in a fishing economy subsidized by unemployment insurance to this day (Shrank 2004).

Considering the implications of the previous example on a global scale, Myers and Worm (2003) estimate that in the aggregate populations of predatory fish today are only one tenth of the their historic population sizes, due to the targeted fishing in the twentieth century of tuna, swordfish, billfish, and cod, for example. The mass overfishing of sharks is increasingly being connected via trophic cascades to unintended consequences, documented empirically on the west coast, where the loss of sharks and subsequent rise of the cow-nosed ray has led to a crash of its food base—the economically important bay scallops (Myer's 2003). Research by the Wildlife Conservation Society at Glover's Reef in Belize is currently testing similar species interactions between reef sharks, southern stingrays, and the economically important lobster fishery. On coral reefs, outbreaks of crown-of-thorns starfish have been a phenomenon for millennia (Walbran et al. 1989), but it has been demonstrated that overfishing has led to significant increase in the occurrence of starfish outbreaks (Dulvy et al. 2004; Sweatman 2008). Crashing predator populations have forced fisheries to shift to the exploitation of forage fish, the historic prey of ocean predators. This phenomenon has been termed "fishing down the food web" (Pauly et al. 1998) and sets off a range of impacts that influence the recovery of oceanic predators, non-target species such as seabirds and marine mammals, as well as, a continued depletion of the ocean's living biomass, the impact of which we are still uncertain. Small-scale fishers may be particularly vulnerable to the impacts of trophic cascades as gleaners often target the lower end of the marine food chain, and as fisheries critical for food security become increasingly targeted by diversifying industrial or export-oriented exploitation methods.

Biophysical and Climatic Drivers

In addition to the scientific uncertainty inherent in managing a single stock based on its biological criteria alone, biophysical and climatic factors add another degree of uncertainty that can influence the system significantly. For example, it is now understood that one of the major factors leading to the decline of the California sardine fishery in the early half of the twentieth century was the compounding impact of over-exploitation and an ecosystem shift that resulted from changes in surface winds that warmed the typically cold nutrient rich California current that is characteristic of that coastline. Made famous by John Steinbek's *Cannery Row*, the boom of California's Sardine fisheries production led to a famous crash in the 1950s initially blamed on overfishing. However, it is now increasingly attributed to a changing ocean current regime off the California coast, which altered the speed of nutrient-rich coastal upwelling. Climatic change may have different impacts on different fisheries. For example, decreasing water temperatures may result in higher anchovy production while increasing water temperatures may result in higher sardine production (Rykaczewski and Checkley 2008). Catch data that is evidence of this mechanism can be witnessed in the multi-decadal trends of alternating catches of sardines and anchovies off the coast of Peru (Fig. 16.4). Considering the

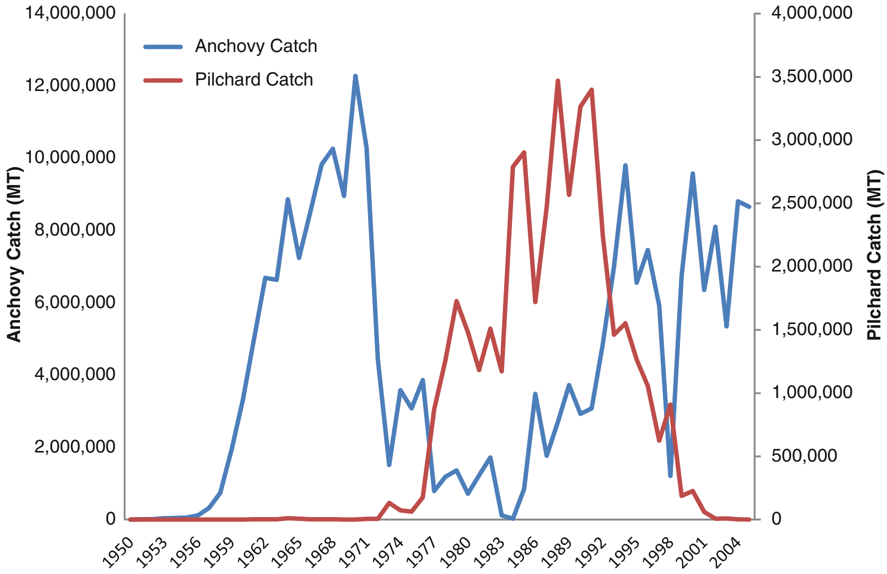

Fig. 16.4 Oceanographic impacts on Peruvian anchovy and pilchard fisheries (Data source: FAO 2008)

dynamism of fisheries that fluctuate temporally, it is clear that while computing allowable catch is necessary, fisheries managers must be willing to accept that stock outputs are dependent on many variables. To better model and predict these complex systems, managers must incorporate the impact of oceanographic changes on ecological structure in environmental modeling exercises. Predictive oceanographic models prove crucial to ensure that capacity and effort allotments in a given fishing season can efficiently be distributed so as to not to over-catch a particular stock, or waste capital on non-existent target species. For small-scale coastal economies, generating such models can be cost prohibitive, especially with the addition of increasingly unprecedented oceanographic conditions resulting from a changing global climate. Technical assistance is needed to provide essential understanding of these biophysical and climatic drivers, and equip communities and developing economies with the knowledge necessary to adaptively manage their resources through change.

Land-Based Impacts

Agriculture, industry, urbanization, freshwater extraction, and shoreline development can all have significant impacts on the health of the marine ecosystems that support fisheries production. One of the most glaring examples of this on a large scale is the seasonal dead zone that emerges out of the Mississippi delta in summer

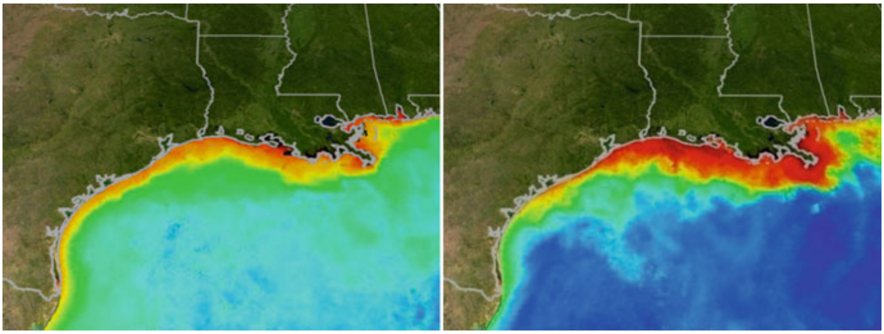

Fig. 16.5 The Gulf of Mexico "Dead Zone." Mississippi delta in the winter (*left*) and summer (*right*). Red represents algal blooms resulting from summer nutrient loading that effectively create anoxic conditions in the gulf decimating marine life (Data source: NASA/Goddard Space Flight Center Scientific Visualization Studio, 2004)

months when these waters experience high levels of eutrophication, a process whereby water bodies receive excess nutrients that stimulate excessive plant growth resulting in heightened levels of primary productivity (Fig. 16.5). The excessive plant growth is driven by nitrogen and phosphorus-laden agricultural run-off from the Mississippi River watershed, which includes some of the US's more agriculturally productive states. This initial boom in primary production is followed by massive decomposition of the algae by bacteria, which significantly lowers the level of available oxygen in waters and, thus, creates an oxygen depleted "dead" zone, where fish and crustaceans are unable to extract enough oxygen from the water to survive. The Mississippi delta is an extreme example of an area, which should be extremely rich and productive in fisheries, but has been significantly diminished in fisheries potential due to land-based management practices. This phenomenon is not limited to the Gulf of Mexico: a recent study has indicated that there are over 400 dead zones that are increasingly encroaching upon the ocean (Diaz and Rosenburg 2008). A combination of conceptually simple, ecological, and technological interventions, such as the establishment and maintenance of species-rich riparian or stream side forest buffers along streams and forests, improved efficiency, and thus, minimization of fertilizer utilization, and better timing of fertilizer applications are examples of potential technical solutions for stemming the world's increasing dead zones. For fisheries management downstream of productive agricultural systems, inter-jurisdiction and inter-agency cooperation are necessary to successfully manage near-shore marine ecosystems. Land-based problems for fisheries are not only limited to agricultural settings. Increased logging and land clearing have been demonstrated to significantly impact fisheries production of anadromous fish in riverine habitat (Cedarholm et al. 1997), lake environments (FAO 2008), and diversity in coral reef environments (Fabricius 2005). With deforestation rates significantly higher in tropical developing countries, the interaction between this land-use conversion and marine productivity is an increasing threat for coastal fisheries and food security—particularly in sensitive coral reef environments.

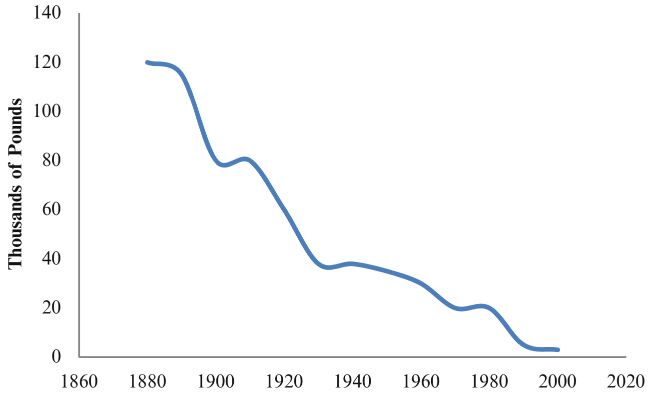

Fig. 16.6 Chesapeake bay oyster Production 1860–2000 (Data source: NOAA Chesapeake Bay Office)

Habitat Degradation

While management of upstream land use is important, it is imperative to protect and ensure the long-term functioning of critical fisheries supporting habitat. For example, in coral reef fisheries, many target fish depend on interconnectivity between sea-grass beds for larval stages, mangrove swamps as a juvenile, then an intact back-reef as a sub-adult prior to completing their lifecycle on the fore-reef (Mumby 2006). Furthermore, coral reefs and mangroves play critical storm protection roles and, thus, may provide a buffer for coastal communities against extreme weather events (see Rumbaitis del Rio; and Ingram and Khazai, in the Climate Change and Disaster section of this volume), in addition to providing many other ecosystem services. Worldwide, it is estimated that 35% of Mangroves were cleared in the 1980s and 1990s for near-shore construction, fuel wood, and the development of aquaculture (Valiela et al. 2001). In the Caribbean, due to the combined impacts of overfishing, land-cover change, hurricane damage, disease, and now climate change, reefs, which once were populated with 50% live coral, now only have 10% coverage (Gardner et al. 2003). Habitat degradation often occurs from targeted fishing methodologies such as dynamite fishing in Indonesia and the Philippines, which significantly reduces the diversity and abundance of future fishery potential (Fox et al. 2003). In temperate environments, bottom trawling is recognized as one of the most destructive forms of fishery extraction. Habitat altering bottom dredging significantly inhibits the ability of an ecosystem to produce both diverse and productive fisheries as demonstrated in the case of oysters (Lenihan and Peterson 1998; Kirby 2004) and scallops (Bradshaw et al. 2001). Historic overfishing of oysters on the east coast of the US fisheries such as the Chesapeake bay (Fig. 16.6) are so depleted that active habitat restoration programs are now necessary. In an extreme reaction, the loss of reefs on the eastern seaboard of the US has now prompted the state of Delaware to deposit 619 derelict New York City Subway cars along with

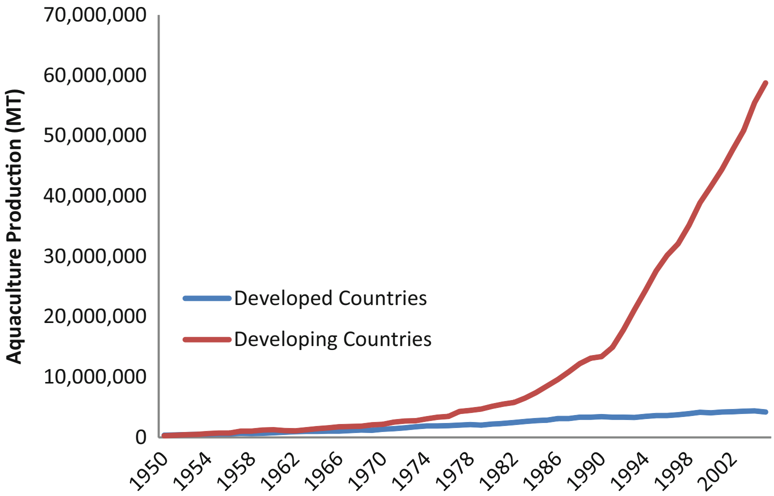

Fig. 16.7 Global aquaculture 1950–2006 (Data source: FAO 2008)

decommissioned ships to create Redbird Reef (after the "Redbird" brand Subway cars used) (State of Delaware 2009). The costs of attempting to provide artificial solutions to lost habitat are in many cases cost prohibitive, and have potential for unintended consequences as the State of Florida learned in its infamous attempt to restore fish habitat by dumping nearly two million waste tires on its reefs in the 1970s (Florida Department of Environment and Conservation 2009). The tires not only contained several toxins, but also proved inadequate at performing their intended task, artificial habitat, as few organisms settled on the "reef": what was meant to provide increases in marine biodiversity and productivity accomplished the opposite. In many developing countries, expensive restoration programs may be cost prohibitive, so, in these places, it is significantly more economically efficient to conserve habitat critical for fisheries in a precautionary manner rather than *post-facto* restoration programs.

The Rise of Aquaculture

To meet rising global seafood demand associated with both increasing population needs for subsistence as well as increasing demand by the wealthy, an exponential rise in fisheries production from aquaculture has occurred, almost entirely within the developing world (Fig. 16.7). Aquaculture production globally is now contributing well over 60 Million MT, and at the current rate of increase will shortly supersede wild capture fisheries as our primary source of food fish production. This transition presents another challenge as industrialized aquaculture can potentially take pressure off of wild stocks, but has caused significant challenges to the sustainability of marine ecosystems, including habitat conversion, disease, increased pressure on wild-caught fish as sources of fishmeal to feed aquaculture species, effluents, and "escapees" mixing with wild populations among the top problems. The impacts of aquaculture are well-known and some steps are being taken to improve their sustainability. For example, at the high end of shrimp production, highly efficient, closed cycle aquaculture ponds are nearly independent of the ocean systems, recycling waste water and severely limiting need to constantly discharge into the natural environment. Some populations of aquaculture production are now completely independent from their original wild stock, eliminating pressure on source populations. A critical issue is the dependence of some luxury aquaculture species such as salmon and shrimp on wild-sought fishmeal for food (Naylor et al. 2000). The impact of aquaculture on wild-caught fisheries is critical for poor coastal communities where the majority of production growth in aquaculture is predicted. Disease is also a major challenge to aquaculture production. In 2000, shrimp exports from aquaculture in Ecuador were valued at over $1 billion, while the following year, due to disease outbreaks resulting from mangrove clearing and unsustainable farming practices, the export value dropped 80% (McClennen 2006).

Management Solutions

New paradigms in developed and over-exploited fisheries are emerging to address the multiplicity of issues driving the history of fisheries management failures. However, as the examples presented in this chapter suggest, there are no one-size-fits-all solutions for fisheries management. While a great deal depends on fishery biology, even more depends on the linked ecological, social, political, and economic systems that are or are not established to extract these resources. Ecological and economic solutions to restore fisheries in historically over-exploited fisheries are proving increasingly successful (Worm et al. 2009). In developing countries, where primary production of fishery resources is critical for sustained poverty reduction of coastal populations, and in some cases, a driving force of national GDP, consideration of these emerging solutions is critical.

Ecosystem-Based Management

The fisheries management failures described within this chapter demonstrate that single species management paradigms are inadequate for the long-term sustainable extraction of the resource. In contrast, Ecosystem-Based Fisheries Management (EBM) has emerged to shift the dominant focus of fisheries management toward management of a more complicated and intricate system, where precautionary measures are a must and uncertainty rules (Pikitch et al. 2004). In the US Magnuson-Stevens Fishery Conservation and Management Act reauthorization of 1996, this philosophy was made into law, calling for an ecosystem-wide approach to managing US fisheries. The challenge of ecosystem-based management requires increasingly complex understandings, not only of single populations, but of entire ecosystem dynamics, including the cumulative impacts of human use. An ecosystem approach to fisheries management continues to utilize population models to calculate sustainable yield, but also considers the impact and need for habitat protection, no fishing reserves, species interactions, and environmental factors. The creation of Marine Protected Areas (MPAs) for habitat protection and fishery reserves is a critical component of most EBM regimes, though cannot alone provide all solutions for all fisheries. EBM is a new foundational approach upon which other management tools and policies should be based.

Assigning Resource Rights

A property rights approach to fisheries management has long been advocated by economists who have pushed for private ownership of resources (Gordon 1954). This approach responds to the well-known "Tragedy of the Commons," whereby unregulated utilization of an exhaustible public good provides high incentives to exploit it as much as possible in the present, as there is no guarantee of future production levels (Hardin 1968). However, this tragedy is not inevitable: in some cases, such as the Maine lobster fishery, the ability of fishers to successfully manage small-scale near-shore fisheries using a bottom-up approach has been demonstrated (Schlager and Ostrom 1992). One approach that has been popularized to deal with open access challenges is the use of "catch shares," which divide fishing rights into individual quotas or Individual Transferable Quotas (ITQs). This rights-based approach provides a certain percentage of the total allowable catch, guaranteed to each ITQ holder, and allows for market-based trading of these rights. By providing ownership for a given portion of a fishery, economists theorize that the value of future catches will provide a self-enforcing mechanism to ensure that present-day catch is sustainable and avoids a "Race to Fish" contest where too many fishers fight for too few fish. Costello et al. (2008) have demonstrated the success of this mechanism in recent years to limit the collapse of fisheries in developed country fisheries. A recent Pew report has challenged the universal applicability of catch shares, emphasizing the importance of enabling factors, and suggesting that catch shares are but one component of a successful fisheries management regime (2009). Assigning private rights to once open-access fishery resources is a contentious issue in many places—given historic flexibility in usage. Once trading of individual quotas are allowed, larger conglomerate fishers can start to buy out smaller rights owners, creating significant issues of equity unless the market is appropriately controlled. In addition, a rights-based approach, as described by Pew, requires powerful enabling conditions, such as strong scientific understanding of resource biology, strong legal and governance regimes, and an efficient market mechanism—many enabling conditions not present in emerging coastal economies and markets.

Supporting Local Governance of Small Scale Fisheries

In developing countries, an essential element of property rights is the right of artisanal and traditional fishers to exclusively extract the near-shore fishery. Competition between national or international industrial fishing fleets and subsistence or small-scale fisheries is a hotly contested issue. While the EEZ of most developing countries are typically licensed out to foreign fleets, the territorial seas typically extend from 3 to 12 nautical miles from shore and are often reserved for national or even artisanal usage. This is by no means universal: in a number of countries, the most valuable industrial fisheries, such as shrimp fisheries, are in the near-shore waters. As a result, in some places, there is no coastal buffer to reduce conflict between industrial trawlers and small-scale fishers. This is the case on the west coast of Madagascar, where a recent government decree has removed coastal fishing restrictions on shrimp trawlers, with potential tragic implications for existing subsistence coastal fishers. In an effort to support food security and conserve a range of ecosystem services critical for coastal populations in developing countries, it is increasingly important to consider the potential for an array of locally managed fishery regimes, each customized to local culture, economics, and ecological context (McClanahan et al. 2008), rather than focusing management and policies solely on the large-scale, industrial fisheries. Doing this effectively will require enabling localized solutions over top-down approaches and fisheries policies that empower coastal communities to manage their own near-shore fisheries. Examples of this include the *qoliqoli* system in Fiji (Teh et al. 2009) and the emerging beach management unit (BMU) in Kenya (Cinner et al. 2009), in which communities are given exclusive rights to autonomously manage their near-shore marine resources separately from national fisheries authorities. Both of these regimes reallocate the right to fish in a particular coastal area to individual local communities, providing for decentralized and increasingly adaptive management regimes, where output is reserved for those most proximate to the resource. In many places where well-defined usage rights exist between traditional and industrial fisheries, illegal incursion into the near-shore can devastate a resource and the economy of small-scale resource users. While the allocation of catch shares may be the property rights issue of the future in developed countries, near-shore usage rights for traditional and small-scale commercial fishers is a continuing struggle, even when recognized by the law.

Market-Based Approaches

Given the challenge of effective governance in the international fishing sector, pressure is being placed on fisheries management regimes to use market approaches to help drive sustainability. Increasingly, large corporations such as WalMart, McDonald's, Loblaw (Canada's largest food distributer) are committing to sustainability targets of 100% of fish food products served on their shelves or at their registers. Many of these commitments are grounded in a certification scheme provided by the Marine Stewardship Council (MSC), an entity established by Unilever and WWF in the mid-1990s, which certifies fisheries after a lengthy evaluation process. Many supermarket chains are endeavoring for similar goals, albeit with their own customized approach, including Publix, Wholefoods, and Stop & Shop. These market-based approaches to sustainability have a powerful effect, though remain at the whim of individual companies' commitment and bottom line. Voluntary industry standards have emerged, particularly in the shrimp farming industry, and may be one driving force in the increasing sustainability of this notoriously unsustainable sector. As yet, the impact of these initiatives has not yet been demonstrated to effectively change behavior in wild-caught fisheries management in developing countries—though a significant effort is currently underway by Darden Restaurants to do so in the Central America lobster fishery, in partnership with the US Agency for International Development in the Global FISH Alliance. Even more problematic is that early analysis indicates that only well-managed and regulated fisheries are seeking MSC certification, such that the costs associated with it are minimized, as are the environmental benefits (Kaiser and Edwards-Jones 2006). The power of purchasers to induce real transformation beyond brand image is as yet unrealized.

International Policies

While governments are doing a great deal to help drive the sustainability of fisheries, in many cases, they are also heavily subsidizing their decline. A recent World Bank publication, *The Sunken Billions: An Economic Justification for Fisheries Reform* (2008), demonstrates that over-capacity of our global fishing fleet is losing the global economy \$50 billion per year, and over \$2 trillion over the past three decades, due to wasted effort and resources in the global race of too many boats chasing too few fish. Subsidization while being highly economically inefficient has also driven numerous fisheries to over-exploitation. In a simplistic view, the fishery sector is considered just like other economic sectors; hence, it has been believed that increased capital investment will lead to increased production. Evidence of this is well ingrained in international development agencies, such as the reaction to the 2005 tsunami in Indonesia in which boat buying programs were largely driven by the assumption that this would provide quick fixes to the economic troubles of the devastated communities. In many cases, as demonstrated by the World Bank above, it is actually the reduced capitalization of fishing fleets, both small and large, and investment in rational management regimes that are most needed. On the path to sustainability, it is imperative that harmful subsidies that perpetuate overcapitalization of the fishing sector are removed.

Management Relevant Science

Critical to all interventions is an increased investment in the scientific understanding of the resource itself—especially those that are new to exploitation. Even if the process of serial depletions is remediated, pressure to find new fishery resources will continue to mount along with ever-rising global demand. A precautionary approach suggests we should exploit only so much as our science allows. A classic example of exploitation without appropriate scientific certainly, was the orange roughy fishery in the South Atlantic. Newly discovered on deep seamounts in the mid-1970s, the orange roughy produced exponentially increasing output for 15 years. Production started to drastically decline after a sharp peak in 1990 of over 90,000 metric tons as it emerged that roughy did not reach sexual maturity for 33 years (Clark 2001). Today, the fishery yields roughly 15% of its historic 1990 production level. In the Caribbean, the Nassau Grouper has become largely commercially and in many cases ecologically extinct, as commercial operators targeted spawning aggregations for export with little understanding of the critical nature of this phenomenon (Sala et al. 2001). New fishery exploitation and improved exploitation of existing fisheries will require continuing monitoring and growth of our understanding of marine ecosystems and their fisheries.

Conclusion

Over the past century, humanity has rendered some fisheries ecologically and commercially extinct. As discussed throughout this chapter, managing wild-caught fisheries has become increasingly complex, necessitating an approach that manages entire ecosystems rather than single stocks. For the rapidly growing fisheries and aquaculture in the developing world to continue apace, significant improvement of our understanding of the marine environment must be attained. Ironically, though the science of ecosystem-based management and innovative fishing effort allocation regimes such as ITQs are being piloted in the waters of well-resourced developed countries to rebuild collapsed stocks– solutions are needed most immediately in developing countries, where loss of fishery production, will not only mean GDP declines, but drastic increases in poverty and significant loss of food security. The regimes that provide fixes to fisheries in developed countries cannot be transferred without careful consideration and adaptation to local economics, social, political, and ecological conditions.

Growing global demand for seafood will continue to rely on continued significant growth in production from the waters of developing countries. Importantly, if the crashes experienced by northern fishers are to be avoided in the south, economically rational ecosystem-based management must be successfully implemented on a global scale. Foreign aid targeted at supporting the scientific understanding of tropical systems, technological improvements to aquaculture, and improved support for effective, equitable governance, and management could help stimulate this process. This will require a major shift in current aid funding. For example, the United States, while importing over 85% of its seafood allocates less than 1% of its agricultural foreign assistance to improving fisheries (OECD 2009). Globally, the OECD as a whole in 2007 failed to do much better with only 4% of agricultural assistance going to improve the management of fisheries. According to the OECD (2009), the US spends less than $10 million per year on international fisheries programs through foreign assistance, while the National Marine Fisheries Service, requested nearly $800 million in Fiscal Year 2008 to manage domestic fisheries (NOAA 2010), from which only 15% of our seafood comes. This trend may be shifting with increasing investments by USAID in the Coral Triangle and Central America for both fisheries and marine biodiversity. In the United States, demand side efforts to foster sustainability have been made in a variety of ways, from seafood awareness cards, to lobbies for import restrictions on unsustainable foreign fisheries in favor of sustainable domestic catches; however, the latter is an ecologically impractical solution with unpalatable results for poor coastal fishers in developing countries.

On the other hand, the expansion of aquaculture nearly entirely in developing countries to meet continually rising global demand should be a significant target for foreign assistance with considerable support put toward building capacity in sustainable aquaculture practice. Though some target species do rely on other marine protein for fishmeal, herbivorous fish such as catfish, tilapia, and filter-feeding shellfish are significant portions of farmed production. This said, even the least efficient and energy-intensive farm-raised fish provide a much better protein conversion ratio than pigs, chicken, and cattle. The growth of aquaculture should be promoted as a potentially sustainable source of seafood, both at commercial and small scales, that can both relieve pressure on wild stocks and do minimal harm to natural ecosystems.

In conclusion, the continued viability of the half billion people dependent upon fisheries as a viable economic sector, and the millions of people who are dependent on healthy marine ecosystems for their survival will demand a drastically different approach toward fishery management than has been witnessed in the latter half of the twentieth century. With global production increasingly coming from poorer developing countries and farm-raised versus wild caught harvests, effective management is needed to ensure the integrity of the ecological support systems upon which this growing global economic sector depends. Within the poorest countries of the world, it is recommended that the lessons of the north be studied, and ecosystem-scale management of marine systems be actualized as a means of ensuring food security, poverty prevention, and economic growth. Increasing production of wild-caught fisheries for the purposes of economic growth should be carefully considered, given the significant losses at a global scale being currently induced by over-capacity. Improved scientific understanding of newly exploited species and ecosystems is critical to supporting effective ecosystem-based management. A paradigm shift is also necessary for wild-caught fisheries, such that fishing less in the short term can yield more in the long term. Fisheries production targets need to be adjusted for stable wild-caught fisheries with ample room for management error, with the intent that marine resources provide not a growth, but a foundational sector of developing economies. A concomitant significant increase in resources should be devoted internationally to ensure the ecologically sound expansion of the global aquaculture industry, which managed correctly can provide for sustainable economic growth of coastal economies, but managed poorly can devastate coastal and ocean ecosystems, livelihoods, and biodiversity. Healthy marine ecosystems and the fisheries dependent upon them, if conserved, will provide a resilient resource for coastal peoples, as a long-term food supply and an array of other critical ecosystem services independent of global economic changes.

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Chapter 17
Participatory Zoning to Balance Conservation and Development in Protected Areas

Lisa Naughton-Treves

Land Use Zoning to Balance Conservation and Local Livelihoods

In the past 25 years, the area of land under legal protection has increased exponentially, particularly in biologically diverse areas of the tropics. Tropical parks and reserves are vital for biodiversity conservation, given that they often hold the last tracts of closed-canopy forest and endangered species amid landscapes dominated by agriculture (DeFries et al. 2005). But these protected areas are also home to some of the world's poorest citizens, who depend on tropical forests for income or as a "safety-net" during natural disasters or periods of social strife. In several cases, creating parks has undermined local incomes and security, particularly in Africa where parks are associated with exploitative colonial regimes (Adams and Hutton 2007). Urgent appeals to human rights concerns and equity have pushed a more people-centered approach to parks, as has the recognition that amidst desperate poverty, the long-term prospect for biodiversity conservation is poor (see Naughton-Treves et al. 2005; Adams and Hutton 2007).

Thus, by global mandates, protected areas (PAs) now are supposed to do far more than conserve biological diversity. These areas are charged with improving human well-being and providing economic benefits across multiple scales (WPC 2003, also see Chap. 18, this volume). Although expectations for protected areas have multiplied, it is not yet clear how to operationalize these plural objectives. As discussed in Chap. 13 (this volume), it is often very difficult to achieve multiple objectives with one policy instrument.

At many sites, managers and donor agencies have initiated participatory zoning projects to balance conservation and development around PAs. Participatory land-use

L. Naughton-Treves (✉)
Department of Geography, Land Tenure Center, Nelson Institute for Environmental Studies, University of Wisconsin-Madison, USA
e-mail: lnaughto@wisc.edu

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planning is an offshoot of zoning, which was invented in the early twentieth century as an urban land management device. Urban zoning is premised on managing incompatibility in land-use relationships to optimize social and economic ends for the broad public. In the late 1920s, rural zoning was invented to address the dual problems of abandoned land and inefficient government expenditures (Rowlands 1933). Zoning in any context ultimately involves negotiating the rights of the individual and the rights of society. Thus, zoning is an inherently political intervention that reveals underlying power struggles and social conflicts (Jacobs 1998).

Participatory zoning for environmental management purposes emerged in the zoning lexicon in the 1970s. PAs typically include substantial areas under customary and/or legal title, much of it pre-dating creation of the PA. It is financially and politically impractical to buy all these claimed lands for biodiversity protection. Moreover, imposing strict preservation or attempting resettlement may place undue hardship or risk on the local poor, who are often the least able to absorb such costs. Thus conservationists have increasingly turned to zoning as a tool. Although participatory zoning projects typically focus on *where* resources should be preserved or extracted, they also designate, explicitly or implicitly, *who* has authority and access to these areas. Zoning aims to promote broad societal benefits, but this may cause some claimants to lose access to certain rights, while others gain (or regain) access (Jacobs 1998). In the large and valuable areas that typify most PAs, many groups claim authority and access. Managing PAs effectively means bringing these often-competing groups together to negotiate rules transparently and democratically for managing use and avoiding conflicts.

Ideally, participatory zoning balances ecosystem conservation and economic development goals across large areas and among diverse stakeholders. In any one locale, tradeoffs between biodiversity conservation and economic development are likely, but at larger scales, side-by-side integration may be possible (Robinson and Redford 2004). Buffer or multiple-use zones can be established to soften the line between preserving biodiversity and extracting resources. Community mapping, aided by GPS/GIS technology, can bolster traditional resource use claims. Boundary demarcation also ought to highlight areas of special ecological importance and reflect dynamic ecological processes, including wildlife migrations and disturbance regimes.

Participatory zoning promises to link customary and scientific knowledge and build alliances among competing groups (Healy 1999). Yet, participatory zoning often is not truly participatory. Deeply political and often contentious governance decisions can be masked by bland planning terms such as "consensus" and "stakeholder." Newly formed management committees comprising local leaders and government authorities to oversee the zoning processes may not be able or willing to resolve conflicts over land. Despite inclusive rhetoric, participatory zoning may be a coercive exercise designed to contain local dissent, or it may be a political maneuver to postpone or prevent enforcing unpopular rules or confronting powerful commercial interests (Few 2001). In such cases, zoning may actually reduce the size of PAs and set a precedent for carving up the area. Ideally, parameters of authority and decision making are defined early in the zoning process.

Aside from these fundamental concerns, evidence from several PAs reveals that implementation and enforcement activities seldom match the complex zoning plans resting on office shelves. In the worst cases, "paper zones" have been drawn in "paper parks," leaving diverse ecosystems and poor residents' resources at risk to open access. To improve participatory zoning outcomes, it is critical to analyze global experiences. The following are case studies from three tropical PAs where participatory zoning attempted to link conservation with development, resolve conflict, and promote sustainability. All three cases engage the UNESCO Biosphere Reserve Model by attempting to demarcate a core protection area surrounded by zones allowing greater intensity of use (see Chap. 18, this volume). The three cases reveal that governance, funding commitments, ecological context, and the use of science and innovative mapping techniques are critical factors that can either stall or advance zoning outcomes.

Case Studies

Bolivia: Kaa-Iya del Gran Chaco National Park

In the early 1990s, the indigenous Isoseño-Guaraní people proposed the creation of the Kaa-Iya del Gran Chaco National Park (KINP) in a sparsely populated lowland region of eastern Bolivia, where the Isoseño-Guaraní have farmed and hunted for centuries. With industrial agriculture and petroleum extraction encroaching upon the region, the indigenous people saw a park as a buffer that might slow immigration, especially from the city of Santa Cruz. Promoting the creation of the park at the edge of their territory (Fig. 17.1) would be a way to protect indigenous land and traditions.

The park proposal was reviewed and approved in community meetings. The negotiating group (and now administrative arm) for the indigenous people was the Capitanía del Alto y Bajo Isoso (CABI), representing some 10,000 people in 23 communities. In 1995, the Bolivian government approved the proposal and established a 3.4 million hectare park to be co-managed with CABI. A park management committee was formed including members of the three indigenous groups, and authorities from the regional and three municipal governments (Noss and Castillo 2007).

The direct local economic impact of the park creation was initially low because the area was so remote, most of it located even beyond indigenous territoris (Noss and Castillo 2007, Winer, but see Lowrey in 2008). The original decree establishing the park also identified three integrated management areas where indigenous groups would be able to extract resources in the future. The park decree also defined a core zone where no extraction was allowed. In the subsequent participatory development of the KINP management plan, biodiversity and socioeconomic teams that included indigenous and other local technicians generated maps of the PA and integrated these into a new zoning plan that includes additional core protected areas, extensive areas for non-extractive and extractive use (e.g., livestock raising), special use areas for a gas pipeline, and recovery areas. This zoning was reviewed by local communities (see Lowrey 2008 for critique of review process) and approved by the national government.

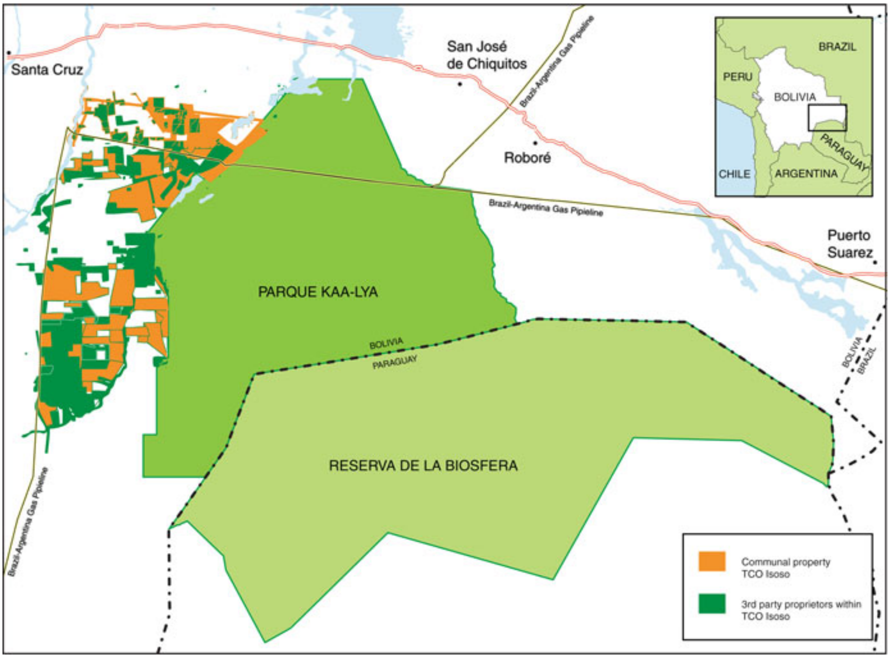

Fig. 17.1 Kaa-Iya National Park, neighboring indigenous territory and Biosphere Reserve, Bolivia (Source: O. Castillo, Wildlife Conservation Society, Santa Cruz, Bolivia. Revised with permission by UW-Madison Cartography Lab)

In 1996, under Bolivia's agrarian reform law, CABI requested a 1.9 million hectare indigenous territory adjacent to KINP, where indigenous groups would have unique authority over land and resource use. The government accepted, and an ongoing titling effort is expected to award roughly 1 million hectares of formerly untitled land to CABI, with the remaining 0.9 million hectares consolidated in private, non-indigenous ownership. At the time, there were also plans to run the Bolivia–Brazil gas pipeline through the area and hydrocarbon concessions were granted inside KINP. Exploration activities revealed that local oil and natural gas reserves were not economically viable, but the pipeline was constructed nonetheless for regional transport (Noss and Castillo 2007). Because of its standing as co-manager of the park, CABI was able to negotiate an agreement with the sponsors of the pipeline to establish a trust fund for managing the park, as well as additional funds for titling indigenous lands, strengthening local institutions and promoting development via the purchase of hand pumps to supply water to households and community livestock ranches.

As the representative of the Isoseño-Guaraní indigenous people, CABI owns the indigenous territory under a communal title, which does not allow for sub-division or sale by individuals or even communities. Private landholders, however, can sell their properties. Anyone can build roads and otherwise develop their land, although legally, development must accord with government-approved land management plans. CABI's successful request for an indigenous territory actually exceeded the areas of current use identified in the participatory maps, but CABI based its claim on historical occupation plus future space requirements (Noss and Castillo 2007).

KINP is a largely successful example of the devolution of land rights to local groups, including co-management by an indigenous organization. A critical element has been a supportive national policy framework, within which the Government of Bolivia adopted the approach known as "parks with people" since 1991. Also, critical to successes was collaboration between local and international organizations and agencies. The Wildlife Conservation Society (WCS) and the United States Agency for International Development (USAID) provided financial and technical assistance and helped open political space for negotiation (Winer 2003). The international pipeline sponsors (including the World Bank) had policies favoring just compensation and indigenous peoples (Winer 2003). CABI's authority as park co-manager provided a final key ingredient, justifying their space at the negotiating table.

Recent significant political changes threaten to undermine this collaboration. The co-administration agreement expired in 2006 and has yet to be renewed. Moreover, a local municipality is pushing for changes in park boundaries so as to expand its authority and allow road improvements through KINP (Noss and Castillo 2007). Mapping technology and social science also were and remain essential ingredients. In 1996, anthropologists worked with representatives from each community to map resource use in neighboring areas. GPS experts then helped community representatives transfer survey information onto topographical maps, which identified hunting and fishing areas, natural resource collection areas, and sacred sites (Noss 2007).

Zoning at KINP improved local communities' welfare and advanced conservation goals by creating a park as a buffer from colonization. Moreover, in parallel with the park, the legal titling of the neighboring indigenous territory allowed residents to better defend their land claim and revitalize their traditional production systems. Secure tenure over resources motivated the indigenous groups to manage the PA actively. Local leaders hope this will reduce degradation of ecologically sensitive areas and promote sustainable use of animals and plants. Field data suggest that important species are being conserved, for example tapir and white-lipped peccary, which are important game species disappearing from surrounding regions (Noss 2007).

Peru: Tambopata National Reserve and Bahuaja Sonene National Park

Like KINP, Tambopata is a vast, sparsely inhabited lowland region. Indigenous groups, miners, agriculturalists, tourism agencies, loggers, and oil companies all claim parts of this forested frontier. For 17 years, zoning negotiations have been ongoing, prompting four legal boundary changes to accommodate shifting socio-economic conditions. Such instability has made signaling and implementing zones more difficult.

Conservationists aimed large when they created the 1.5 million hectare Tambopata-Candamo Reserved Zone (TCRZ) in 1990. Although the founders did not initially consult many local residents, under Peruvian law a Reserved Zone is transitory, allowing time for subsequent negotiation and ecological assessments necessary for more permanent boundaries (Ricalde 1989). The TCRZ eventually resulted in (a) a National Park (the highest category of protection), (b) a National Reserve (a category that allows for limited use of natural resources), and (c) some areas remaining as private landholdings (Fig. 17.2).

Many local citizens initially opposed TCRZ for fear of losing access to land and resources. Peruvian NGOs sought to increase local support and negotiate public consensus for land use in TCRZ. With financial support from the MacArthur Foundation and USAID, these NGOs worked with local agriculturalists, indigenous groups, and state agencies to title land within and outside the Reserve. They also proposed the creation of a one million hectare national park, "Bahuaja-Sonene," at the uninhabited core of TCRZ. After a year of public meetings, local stakeholders approved the proposal to create a park in remote, largely uninhabited regions and agreed to participate in land-use zoning for the nearby multiple-use Reserve (Chicchón 2000).

When the national government legally established the park in 1996, it reduced the park's extent by approximately 60% to accommodate a one million hectare exploratory oil and natural gas concession for Mobil Oil straddling the proposed park and adjacent area. Despite a stipulation that the Natural Resource Institute (INRENA) would incorporate the areas relinquished by Mobil Oil into the national park, once the oil exploration was completed, local citizens felt deceived and protested violently. Many felt they had agreed to forego logging and agriculture in this remote area, only to have an oil company enter instead. Some local leaders walked away from the planning process, others struggled to convince their constituents to remain involved. Impetus for a renewed zoning effort came in 1999 when Mobil Oil released its concession due to inadequate reserves, and INRENA followed through with the plan to incorporate that land into the national park (Fig. 17.2). At the same time, Peru passed a new law promoting zoning in PAs.

Peru's 1999 Protected Areas Law (enacted in 2001) delineates categories of PAs according to levels of resource use and requires zoning within the master plan of each PA, guided by a Local Planning Committee formed of representatives of agriculturalist and indigenous federations, mining cooperatives, conservation NGOs, tourism companies, and staff from Peru's National Council for the Environment and INRENA. Thus, a new zoning effort was launched in Tambopata. According to participants, national agency representatives served as "catalytic agents" and technical advisors, but leadership and zoning decisions ultimately came from the Local Planning Committee as a whole. Simultaneous to this third phase of zoning at Tambopata, the Research Institute of Peru (IIAP) began zoning the broader region (Madre de Dios) following the Amazon Cooperation Treaty plan for "ecological-economic" zoning (Sombroek and Carvalho 2000). This larger process heightened public interest in land-use planning, and by some accounts made the Tambopata effort longer and more conflictive because it was now taken seriously.

After months of meetings, the Local Planning Committee reached consensus on the creation of a National Reserve in the area bordering Bahuaja-Sonene National Park. Any legally documented land claims within the Reserve could be excised if owners demanded to be "liberated" from the Reserve. The local plan was then sent to the national office of INRENA where it sat for 6 months. Local citizens subsequently

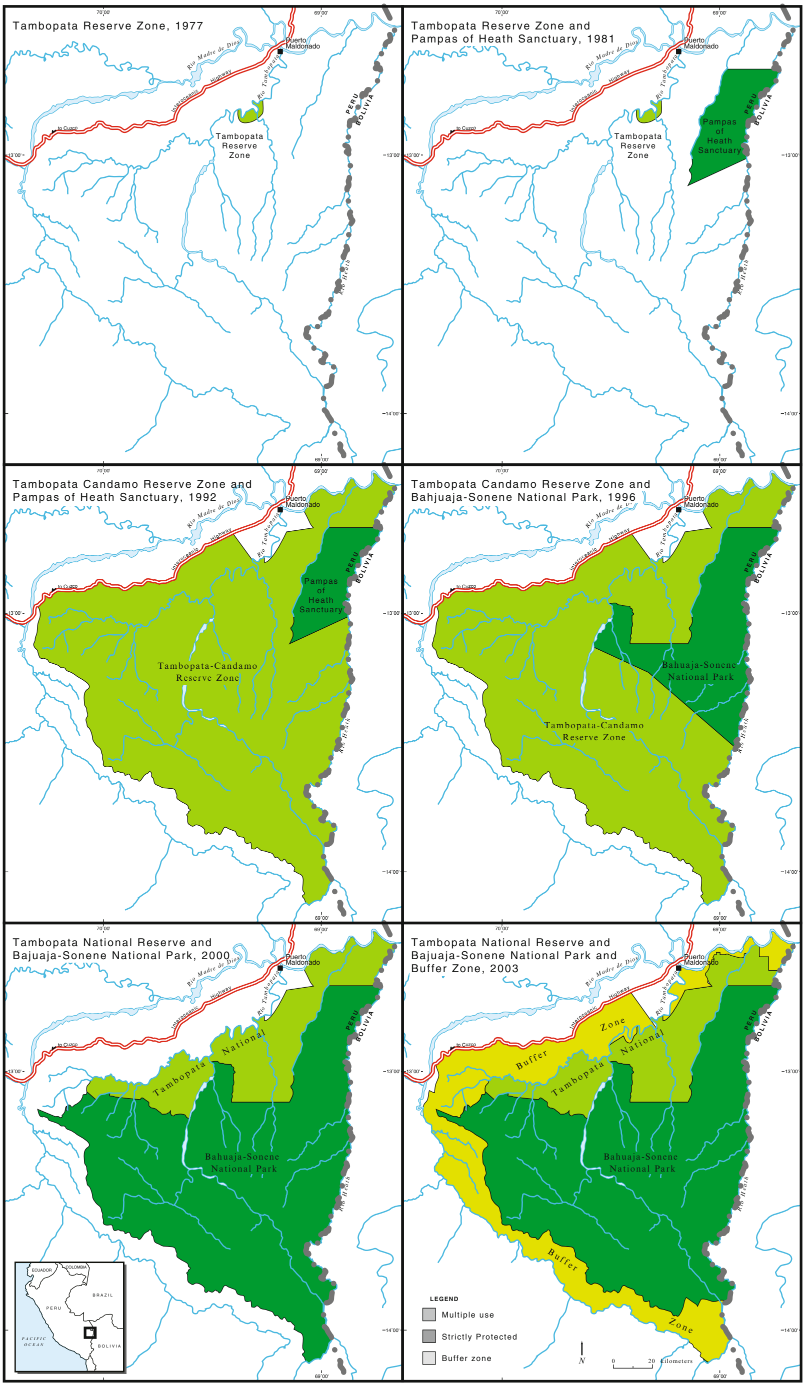

Fig. 17.2 Changing boundaries of protected areas in and around Tambopata Province, Peru, 1977–2003 (Source: CI-Peru. Revised with permission by UW-Madison Cartography Lab)

learned that INRENA had developed a separate zoning proposal for TCRZ, with seeming disregard for the local plan. This rebuff added to local protests regarding logging and mining restrictions. In 2000, the agriculturalists' federation mobilized the local population to demand significant reductions in the size of TCRZ. To draw attention to their campaign, protesters marched in the street and temporarily closed the regional airport.

Facing bureaucratic stalemate and local unrest, the Local Planning Committee urged the national office of INRENA to consider the locally developed zoning proposal. Later that year, INRENA announced the official zoning of TCRZ. The northern sector of TCRZ became the Tambopata National Reserve, and Bahuaja-Sonene National Park was doubled in size (Fig. 17.2). This final zoning plan largely followed that of the Local Planning Committee regarding land use outside the Reserve. Several communities were "liberated" from TCRZ per their wishes. Also, excised was the headwater region of a major tributary of the Tambopata River, an ecologically important area and home to an indigenous group in voluntary isolation. These headwaters became part of a buffer zone, an ambiguous category that failed to prevent the subsequent proliferation of illegal gold mining and logging. Other communities in the south were "discovered" to lie within the expanded park, apparently due to cartographic error. But given the lack of park enforcement in the south during this period, public protest was negligible.

The Local Planning Committee's 1999 proposal for *internal* zoning of the Tambopata National Reserve was largely ignored. Government officials explained that the local plan had not incorporated sufficient scientific and ecological considerations, and erroneously proposed illegal land uses. In 2001, the Local Planning Committee was renamed the "Management Committee" and a fourth phase of zoning began to sort out land use within Tambopata Reserve. Several zoning "veterans" participated for the fourth time. Workshops were held in communities within the Reserve in which citizens were asked once again to draw maps delineating their resource use areas.

This time, however, ecologists and foresters also demarcated ecologically sensitive areas for restoration or protection, and by local accounts, their voice carried special weight. In the final plan, use zones within Tambopata Reserve generally conformed to previous patterns of extraction, although one area of intact forest was zoned for "special use" due to the recent arrival of a group of colonists who were uniquely able to use political connections to lobby for rights to clear forest for subsistence agriculture. Another area of active mining was re-zoned for tourism and ecosystem restoration, but this has not been enforced due to budget constraints and the periodic threat of public demonstration and even violence against environmental authorities on part of the miners (Carlos Ponce, personal communication).

The outcomes of zoning efforts at Tambopata are mixed. On the positive side, information was generated and disseminated, which encouraged public dialogue. A large area was eventually legally designated for protection after Mobil Oil rejected it. However, some indigenous communities, such as the Ese'eja, believe they lost part of their territory in the process. Conservationists are meanwhile concerned that buffer zone rules are too ambiguous to protect forest and wildlife. Even some zones within the Reserve are neither publicly recognized nor enforced (e.g., miners work in the ecological restoration zone). Local PA managers blame budget shortfalls, pointing out that in some years donors spent more than $100,000 on participatory planning, while INRENA has less than $10,000 to implement such plans (Landeo 2006).

Philippines: Mt. Pulag National Park

Mt. Pulag encompasses 11,560 ha of mid-elevation forest and grassland in the Philippines, a remnant of biodiverse habitat elsewhere largely converted to agriculture. Park boundaries were originally set in 1987 without any ground survey work nor formal acknowledgement of indigenous people's ancestral claims to the land and resources. Zoning efforts began at Mt. Pulag as prescribed under the National Integrated Protected Areas Systems Act of 1992, with funding from the European Commission. This process included community consultations and ecological surveys. The Philippines is one of the few Asian countries to officially endorse indigenous peoples' presence and resource use within PAs. Specifically, the Indigenous Peoples Rights Act (1997) supports the transfer of title and management authority for ancestral domains within PAs to defined indigenous communities. But other national legislation (Local Government Codes) confers management authority to municipal governments.

Zoning decisions were complicated by the presence of four overlapping indigenous groups' territories within the park (Fig. 17.3). Competition for the mountain itself was intense, given its spiritual significance and tourism value. To resolve conflicting land claims, a Protected Area Management Board (PAMB) was created, comprised of indigenous leaders, municipal officials, and park staff. Yet, the PAMB had uncertain authority and, after approximately 10 years of deliberation, failed to reach consensus. As funding dwindled in 1999–2000, PAMB rushed to finalize a zoning plan. Two rounds of public meetings and hearings produced two conflicting resolutions. Two communities neighboring the park (the Kalanguya tribal organization and the Kabayan municipality) endorsed the park boundaries but claimed ancestral domain rights to the entire park area. Park officials rejected this proposal on the grounds that no such "ancestral park" category existed, and this would exclude other local municipalities. In a second proposal, two indigenous communities proposed excising certain areas from the park (Pinel 2007).

Rather than changing park boundaries, the Department of Environment and Natural Resources (DENR) delineated a core area and multiple-use zones. The PAMB approved the zoning plan, as did one municipality (Kabayan). However, several stakeholders filed objections with support from provincial political bodies. As a result, the DENR never recommended the plan to Congress. As public meetings

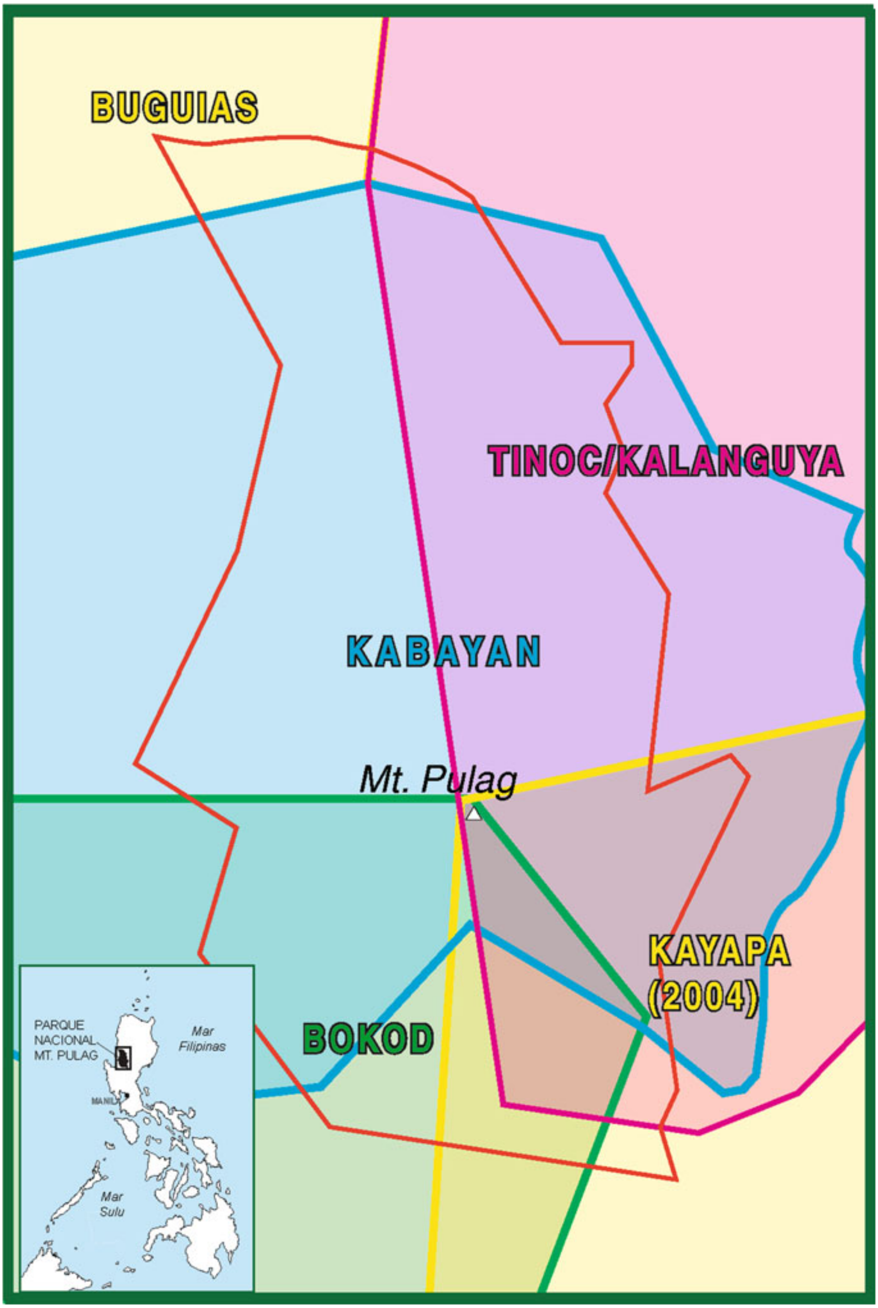

Fig. 17.3 Ancestral domains claimed by indigenous representatives in zoning initiative, Mt. Pulag National Park, Philippines, 1997 (For illustrative purposes only) (Source: DENR, Philippines, and Pinel 2007. Revised with permission by UW Madison Cartography Lab)

continued (and consumed nearly 80% of the park budget), two roads were built illegally in cloud forest habitat within the park.

These roads were sanctioned and funded by municipal authorities, including those serving on the PAMB, which as a body publicly rejected such actions. The roads signal the limited power of zoning in light of municipal politicians' drive to improve infrastructure and please voters.

Mt. Pulag's boundaries were never changed and a Congressional Act promoting indigenous management of park areas was never adopted. The DENR currently retains authority over the park. In the future, indigenous communities within the park will hold overlapping authority if the Indigenous Peoples Rights Act is enforced and title is transferred. Despite these uncertainties, the DENR has meanwhile attempted to implement the zones set by PAMB through the use of forest rangers and local officials, but enforcement is hindered by shortages of staff, funds, and authority. Conflicts persist in the multiple-use zones where there is steady encroachment. Although local communities appear to support the strict protection zone, they too have limited authority and negligible budgets. The PAMB recently requested funding to mark the physical borders of zones, hoping this would effectively limit agricultural expansion into the park.

Participatory zoning has had limited success from both a governance and ecological standpoint at Mt. Pulag. Despite considerable investment and years of deliberation, zoning has not resolved land-use conflicts. Contradictory and shifting national policies hindered collaborative planning, as have overlapping claims by indigenous groups and municipal governments.

Lessons

Participatory land-use planning may be slow and uncertain, yet it remains one of the few options for managing landscapes for both conservation and development. Therein lies its prominence in regional conservation efforts. In all three cases reviewed here, participatory zoning initiatives fostered dialogue and introduced a sense of landscape-level conservation among multiple stakeholders. The participatory process also partially redressed boundary-drawing "errors" from the time of the PAs' creation; for example, those boundaries that erased ancestral claims and/or excluded critical ecosystems. The literature is peppered with references to "erroneous" boundaries drawn at the creation of the parks, yet case studies show that there never will be perfect, conflict-free boundaries. In the best instances, customary and scientific knowledge were brought to bear on land allocation decisions and the resulting land-use zones achieve broad integration of poverty reduction and conservation, even if they also reflect local compromises on biodiversity and/or income generation. PAs face increasing management complexity, however, and the case studies underscore the need for better understanding of governance, science, and innovative mapping methods to improve planning exercises.

Governance

Legal Framework

State and national legal frameworks fundamentally shape the outcome of participatory zoning efforts. Like many other developing countries, Bolivia, Peru, and the Philippines all recently passed legislation promoting participatory zoning in PAs and the creation of park-level management committees or boards incorporating local representatives to guide the process and build public alliances. These reforms conform with international donors' call for participatory PA management, as well as national campaigns to decentralize environmental governance.

New legislation helped launch and legitimize zoning initiatives in PAs, but new laws can also create uncertainty and result in overlapping jurisdictions. In Mt. Pulag, competing and unclear national policies about indigenous territories and municipal-level authority within PAs confused local negotiations. Amid uncertainty and shifting rules, competing interest groups may be reluctant to negotiate or compromise. National governments also may undermine local collaborative planning more directly. In Tambopata, one government agency sponsored local deliberations about conserving a pristine area, while another issued an oil concession in the same area.

In KINP, planning worked more smoothly because of government recognition of the indigenous groups, through CABI, which proposed the park. The government ceded park administration to CABI, yet maintained the management of concessions. When the government imposed an oil pipeline through the park to meet national interests, this superseded local rights to resource use. Yet, as the administrator of the PA, CABI was able to negotiate with the pipeline sponsors to establish a trust fund to support park management and provide resources for communal land titling and development activities. The case of CABI reveals that participatory zoning can open possibilities for new alliances and/or new forms of negotiating resource access, with significant risks and/or benefits for conservation and local rights and livelihoods (Lowrey in 2008).

Collaboration

Not all stakeholders will be winners, yet building alliances and collaboration among multiple stakeholders can lead to more equitable and less costly management and monitoring for PAs. As environmental governance is decentralized, local participation becomes more important, though planners should not assume shared goals among various constituents. Local citizens may see PAs as an imposition on their land rights, and enforcing conservation can be a sensitive issue, given the exclusionary and abusive record of some park administrations. In turn, local demands can be politically charged and may not include biodiversity conservation as a goal.

In Machalilla National Park, Ecuador, zoning initiatives raised public expectations that land-use restrictions would be entirely lifted; in essence some citizens hoped the park would be "de-gazetted" (Alvarez 2006). Thus, some conservation agencies resist granting control of forests and wildlife to local groups. For example, park authorities at Tambopata were reluctant to accept the original zoning plan negotiated by NGOs and local communities for fear it would downgrade the protection of key ecosystems. Elsewhere, the responsibility for PA management has been transferred to local institutions, but without economic support, implementation has stalled (Larson and Ribot 2004).

People will participate in a meaningful way if they think it is in their best interest. At KINP, CABI worked with conservationists to establish a park and, thus, gained their support for titling indigenous land in the adjacent area. By contrast, as the Mt. Pulag case illustrates, buy-in to the process may be lost if the rules change midstream, or if the PA management committee has uncertain authority. Participants may then merely go through the motions of participation and negotiation, or may actively subvert the process. Leaders of participatory processes ought to publicly acknowledge the uncertainties involved and elicit people's involvement in participatory adaptive management—including evaluation and adaptation of the process.

The merits of zoning as a conflict-management strategy are uncertain. In KINP, zoning helped secure claims and reduce conflict between indigenous communities and other stakeholders. Yet, efforts to draw boundaries among indigenous groups heightened competition at Mt. Pulag (Pinel 2007). Zoning can destabilize communities' traditional management practices in common areas and lead to an acceleration of ecosystem degradation if communities do not understand the rationale of zoning or were not involved in its design and implementation. Judging meaningful "community understanding" is a fraught process, observers may reach different conclusions about the same exercise (e.g. community participation in case of Kaa-Iya, re: Lowrey 2008, Castillo date). Zoning efforts are most likely to be effective if they are scaled to managerial capacity and are viewed as legitimate by local citizens and key stakeholder groups.

Financial and Institutional Support

Participatory planning is costly and slow. Defining and identifying property rights can be a contentious and lengthy process. In all three case studies, participatory zoning lasted well over a decade and exhausted scarce financial resources. For example, 80% of the annual budget for Mt. Pulag was spent on planning meetings. One community neighboring Mt. Pulag held 28 meetings over a 10-year period to discuss a boundary location. They never reached consensus. Critics argue that rapid deforestation and biodiversity loss leaves no time to wait for public consensus. The case studies reveal a serious problem in following plans through to implementation: far more is spent on planning and public meetings than on implementation. Furthermore, zoning plans may come to nothing if managing institutions are powerless. In such cases, public deliberation on zoning may not be an appropriate intervention and alternative, smaller-scale strategies (conservation concessions or conservation easements, as discussed by Chap. 12, this volume, for instance) may prove more effective.

Planning for Change

In principle, parks are permanent, and, thus, promise protection of biodiversity and critical ecosystem services in face of future economic demands. Yet conservationists and development practitioners must recognize, respond to, and manage change over time. At all three sites, zoning negotiations took place amidst shifting resource use and political alliances. In Tambopata, earlier zoning plans focused on balancing agriculture with forest conservation. Later efforts had to contend with booms in mining, tourism, and logging. In the 1990s, Tambopata's citizens voted in a mayor who declared the region "the biodiversity capitol of the world." They later elected a governor who promised to degazette the regions' parks and reserves for local benefit. Some communities of subsistence farmers who had originally lobbied to be "liberated" from Tambopata Reserve later regretted being excluded when they faced subsequent colonization of their lands. Thus, although zoning may place new restrictions on land use, it may also offer security with respect to land claims, especially for the rural poor.

Experts stress that zoning rules are not intended to be permanent (Jacobs 2007). For example, in the Brazilian Amazon, ecological-economic zoning projects assume a 5–25 year planning horizon (Sombroek and Carvalho 2000). Zoning, thus, offers flexibility, but ever-changing boundaries are difficult to administer and leave biodiversity vulnerable to economic and political instability. Thus, a balance must be found between adaptability and consistency for enforcement.

Another critical change factor is population growth within PAs. The sparsely populated landscape of the Bolivian Chaco apparently favored zoning efforts while claims to the more densely settled, fertile lands of Mt. Pulag were seriously contested. In Latin America, indigenous communities in PAs are growing faster than populations in surrounding areas. This accelerated population growth may undermine sustainable use, or it could sustain biodiversity if such growth translates into political strength and a pro-conservation stance (see discussion by McSweeney 2005).

Role of Science

Zoning is meant to separate incompatible land uses within PAs (for instance, mining and ecotourism). A key role of science is to define what uses are indeed incompatible and set area parameters for sustainability. Many scientists involved in zoning favor assigning land uses based on land aptitude, priority of use, ecological functions, or ease of protection. Some conservation biologists advocate spatially explicit modeling that combines the abundance of species with the cost of protection for the maintenance of regionally important biodiversity. However, such approaches demand a balance of scientific rigor and political acceptability, a compromise that may not satisfy any of the participants. Scientists may assume a similar approach for delineating agricultural areas, hunting territories, or logging tracts. Decades of zoning experience in the North American context reveal the limits of top-down, science-based land-use planning.

Managers likely may want more precisely defined categories than what participatory processes yield. After all, measuring management effectiveness is difficult when categories are vague. Yet efforts to reach public consensus often lead to the creation of ambiguous categories. For example, a guard at Tambopata explained that he was unsure how to enforce a large zone designated for "economic development harmonious with biodiversity conservation." Such vague designations reflect pragmatic ambiguity or the political advantages of avoiding difficult decisions about priority land uses; in other words, building public consensus on zoning plans can be easier (though not ultimately effective) if the management objectives for contested zones remain vague.

In a participatory process, scientists usually do not make the decisions, or, if their decisions defy local priorities, their decisions may not be implemented. Scientists involved in ecological zoning are more likely to be effective if they are transparent in their work, incorporate local ecological knowledge, and clearly communicate the benefits of protective zones for neighboring communities. Recent advances in GIS technology and participatory mapping open the way to better communication and collaboration. In all three case studies, public dialogue was aided significantly by participatory mapping.

Innovative Mapping Methods

Methods that emphasize community participation in the mapping and zoning process to capture the cultural and socio-economic importance of land, as well as geographic characteristics, include participatory three-dimensional modeling, and community integrated geographic information technology. In the former, projects build 3-D maps of zoned areas with input from the communities. In the latter, projects use technologies such as GIS to capture community knowledge and perceptions of place. Innovative mapping approaches ought also to include information on resource control and governance.

Satellite imagery allows practitioners to compare large areas of land and to differentiate land types. Zones are delineated according to the analysis and weighting of land attributes such as type of soil, topography, hydrology, and prevalent agricultural use. Public construction of maps helps participants visualize pressure on protected areas and understand overlapping resource claims. In short, participatory GIS mapping can aid in the analysis of complex spatial data and also facilitate public dialogue. Yet, caution is warranted. Practitioners keen to utilize new mapping technologies ought not to intimidate local stakeholders or confer inappropriate formality or legitimacy to proposed zoning boundaries (Harris and Hazen 2006).

Lessons from the Three Case Studies

- Participatory land-use zoning is a slow and uncertain endeavor yet remains one of the few options for integrating conservation with development at the landscape level.
- Ecological context matters. Highly imperiled habitats may require more agile strategies (e.g., conservation concessions).
- Within parks, areas of fertile soil or high tourism potential often attract multiple and conflicting claims.
- Institutions strongly shape zoning outcomes. New national policies promoting participatory planning open opportunities for negotiation but may also create confusion. Amid shifting and/or ambiguous policies, local stakeholders will be reluctant to compromise in land-use planning exercises.
- Amid decentralization, municipal governments have more power and this needs to be considered carefully in zoning negotiations.
- Newly created local park management committees have uncertain authority. Their role deserves critical attention.
- Support for implementation is often grossly neglected. Zoning is most likely effective if scaled to managerial capacity and viewed as legitimate by local citizens.
- Field research and monitoring is essential for lasting conservation. Science needs to be presented in a transparent fashion. Opportunities to adapt and rezone for biodiversity purposes need to be built into plans, just as such opportunities are offered for future economic development.
- Innovative mapping promises to link customary and scientific knowledge and facilitate negotiation. Yet donors should not spend money on elaborate maps if that means that there is no funding left for implementation.

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Chapter 18
The Role of Protected Areas for Conserving Biodiversity and Reducing Poverty

Margaret Buck Holland

Introduction

The practice of setting lands aside for the preservation of natural and cultural heritage, sometimes in the face of a perceived threat to that site or resource, is hardly a modern concept. For centuries, even millennia, human societies have established protected areas, often for the safeguarding of sacred sites, forests, or hunting grounds (Chape et al. 2005). Most mark the beginning of our present-day network of government-established protected areas with the declaration of the first national park, Yellowstone, by the U.S. federal government in 1872. The early protected areas of the U.S. system embodied the romantic concept of wilderness, one captured in the writings and actions of Thoreau, Muir, and others as the sublime: natural landscapes where one might best sense the presence of God (Cronon 1995). This wilderness concept propelled a preservationist mode of protected area creation and management, one that focused on protected areas as natural monuments, places that should be kept free of human activity. By the second half of the twentieth century, the primary motivations behind conservation and protected area establishment shifted from the wilderness and preservationist ethic to one focused on conservation of rapidly disappearing habitats, often in landscapes where the spheres of human societies and nature were interwoven (McNeely 2005). Today, protected areas (PAs) are often cited as the cornerstones of biodiversity conservation, the most effective tool for in-situ conservation, as well as essential safeguards of ecosystem services (CBD 2006; MEA 2005).

In this chapter, I explore the change in both the extent and mission of the global protected area network. I highlight several key points at which ecologists have helped to shape that change, as well as the global agenda and targets established for protected areas, with respect to biodiversity conservation and poverty reduction.

M.B. Holland (✉)
Geography & Environmental Systems, University of Maryland, Baltimore Country, Baltimore, Maryland, USA
e-mail: mholland@umbc.edu

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The chapter concludes with an exploration of current research on protected areas and human welfare, identifying key research gaps and management challenges facing protected area networks today, particularly in light of new policy programs focused on climate change adaptation and mitigation.

Change in Extent and Context of the Global Network of Protected Areas

In the post World War II era of massive consumption and growth by human societies, the world's ecosystems have experienced rapid and widespread change, to a greater extent than any other time in the course of human history (MEA 2005). In turn, these human-induced ecosystem changes have triggered a potentially irreversible loss of biological diversity. Current extinction rates for species are estimated to be 100 times greater than background rates, (often referred to as the sixth mass extinction), and predictions are that they will continue to increase tenfold in the near future (Ricketts et al. 2005). The recently developed Living Planet Index of global biodiversity points to a 30% decline over the past 35 years, while human consumption of natural resources, or "humanity's demand on the planet," has more than doubled over that same timeframe (Hails et al. 2008). Given this grim picture, it is no surprise that the field of conservation biology has often been referred to as a "crisis" or "mission-driven" discipline (Soulé and Wilcox 1980; Redford and Sanjayan 2003; Meine et al. 2006).

When ecologists Robert MacArthur and E.O. Wilson published the theory of island biogeography in the 1960s, it did not take long before colleagues noticed the theory's applicability to protected area design, as many observed that parks were isolated features in fragmented landscapes of land use, and posited that the risk of species loss from these areas was inversely related to its size, as predicted by the species–area relationship (Lovejoy 2006; Diamond 1975; Terborgh 1975; Wilson and Willis 1975). By 1980, when the terms biodiversity and conservation biology were first coined, protected areas were recognized as critical tools for stemming the tide of species losses (Soulé and Wilcox 1980).

The now widely accepted definition of a protected area, as released by the International Union for the Conservation of Nature (IUCN) in 1994 is:

> An area of land and/or sea especially dedicated to the protection and maintenance of biological diversity, and of natural and associated cultural resources, and managed through legal or other effective means (Chape et al. 2003).

Over the past several decades, the number and extent of PAs have grown dramatically: between the first and the fifth World Parks congresses (1962–2003), the total number of documented areas increased tenfold, extending over 16.3 million km², or 11.5% of the Earth's land surface (Figs. 18.1 and 18.2) (Chape et al. 2003). This number does not include 1.84 million km² of marine protected areas that have also been placed under some form of conservation management (Chape et al. 2003). The most recent release of the World Database on Protected Areas (WDPA) includes

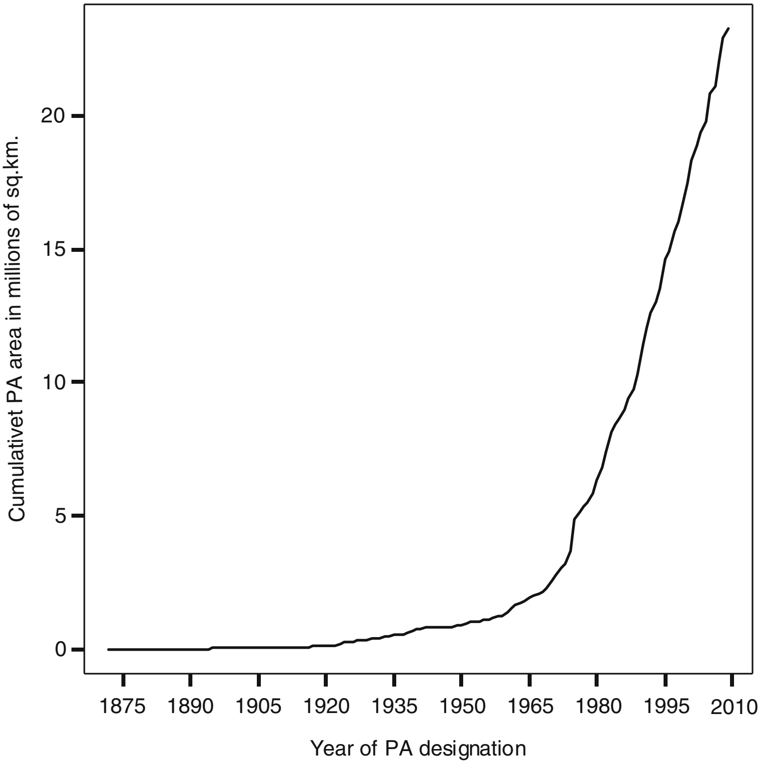

Fig. 18.1 Cumulative growth in global PA extent, 1872–2009. Graph includes data for only those classified as designated or inscribed, with a known year of designation, and a documented area (WDPA 2009)

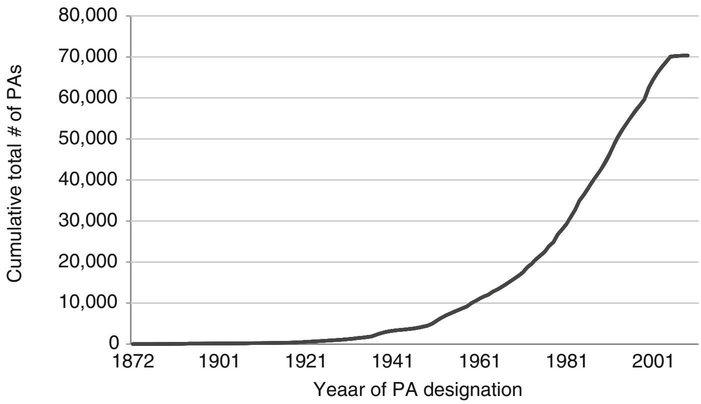

Fig. 18.2 Growth in cumulative total # of PAs, 1872–2009. Graph only includes those PAs that are: designated/inscribed, with a documented year of designation and area extent. Note: It was not until 1921 that the number of global PAs surpassed 500, as registered by the IUCN

mapped information for 122,750 PAs, which have been formally designated or inscribed (UNEP-WCMC 2009).

Ecologists stress that PAs are *cornerstones* of biodiversity conservation: that they are crucial pieces of a larger structure, and cannot be successful mechanisms for slowing biodiversity loss if they exist in isolation (Hansen and DeFries 2007). Yet, PAs are becoming islands. A recent analysis of the change in tropical forest cover both within a global set of parks and related buffer areas revealed this dilemma: protected areas are increasingly isolated in a fragmented landscape of varied human uses (DeFries et al. 2005).

Concerns over the rate of biodiversity loss and the increasing isolation of PAs are further complicated by the fact that not all protected areas are created equal, in terms of their potential for conservation effectiveness. Part of this is due to the fact that not all protected areas are established with biodiversity conservation as the main objective, and are thus assigned management that can range widely in terms of permitted human use. In an attempt to synthesize the more than 1,000 different terms used by countries in classifying their PA networks, the IUCN has developed six categories to describe varying management regimes, ranging in level of protection and permitted use. According to the UN List of Protected areas in 2003, approximately 70% of the total number and 80% of the area of PAs worldwide had been classified using an IUCN category (Chape et al. 2003). These six include, according to degree of protection, from strictest to those with the highest level of permitted use: strict nature reserves and wilderness areas (Ia and Ib), national parks (II), natural monuments (III), habitat management areas (IV), protected landscapes/seascapes (V), and managed resource protected areas (VI).

According to the IUCN, categories I–III are primarily concerned with conservation where "direct human intervention and modification of the environment has been limited," and categories IV–VI involve PAs where "significantly greater intervention and modification will be found" (IUCN 1994). Categories V and VI are the most recent additions to the group, and much of the recent growth in PAs over the past 15 years can be mainly credited to the recognition of these two categories, as well as the inclusion in the WDPA and UN List of those areas which are as of yet unclassified by the IUCN (Fig. 18.3). Zimmerer et al. (2004) identify this as the "second wave" of global conservation, and during the late 1980s and 1990s, the growth of PAs designated for "sustainable utilization" outpaced those under strictly protective management (Zimmerer et al. 2004).

Proponents of restricted use protected areas, such as national parks and biological reserves (Types Ia, Ib, and II), have decried the addition of categories V and VI, arguing that the mission of these areas is to promote sustainable development, rather than maintain a focus on biodiversity conservation (Locke and Dearden 2005). In fact, while the most recent World Parks Congress, held in Durban in 2003, heralded the achievement of surpassing the 10% global target for protected area coverage of terrestrial lands (at 11.5%), the Millennium Ecosystem Assessment estimated for the same year that coverage only ranged between 4.0% and 9.5% for each terrestrial biome. This discrepancy appeared because the MEA based its estimates on protected areas classified as IUCN I–IV only, suggesting that ecologists are not convinced of the conservation effectiveness of PAs in categories V–VI (see MEA, Chap. 4, p. 83).

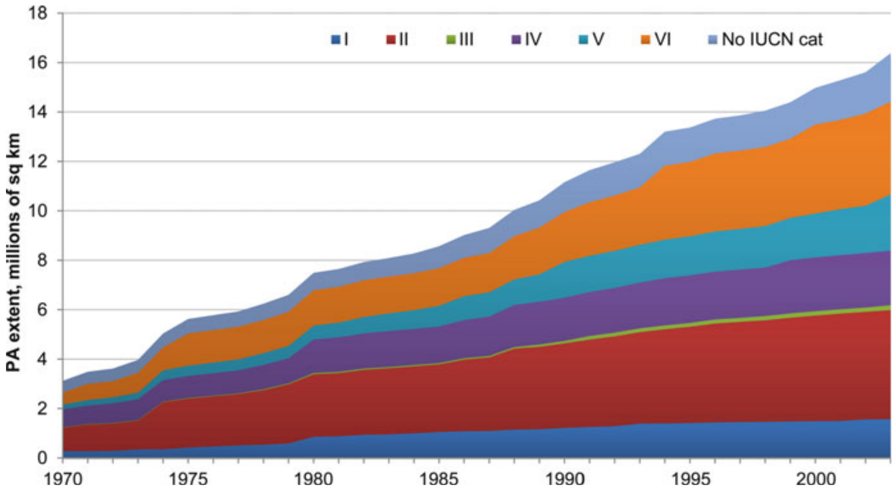

Fig. 18.3 Growth in terrestrial area covered by global network of protected areas, 1970–2003 (Source: Chape et al. 2003; UNEP-WCMC 2009)

In fact, a recent review of literature suggests that there is a dearth of empirical evidence overall on the ecological performance of protected areas, with research dominated by small numbers of scattered case studies (Gaston et al. 2008). While Gaston et al. (2008) note there is little doubt that, *globally*, protected areas are successful at safeguarding a significant portion of biodiversity from external threats, several key ecological knowledge gaps remain, focused mainly on the performance of individual PAs, or even small portfolios of protected areas. One main knowledge gap, as outlined by the authors, is the ecological performance of PAs under varying types and levels of management (Gaston et al. 2008).

Closely related to performance, effectiveness is a term that surfaces often in the discourse on protected areas, and has emerged as a major focus of PA-related research over the past decade. The emphasis on effectiveness can be roughly divided into two categories: (a) the ability of protected areas to safeguard resources and ecosystem services from external influences or pressures and (b) evaluations of the management of individual or small networks of PAs. The former group of PA effectiveness research centers its methods on the use of GIS and satellite remote sensing technologies to analyze land-use/cover change, (primarily in the form of forest cover), in and around protected areas. There is much about a protected area's ability to conserve biodiversity and maintain ecosystem services that is not captured through the analysis of land-use/cover change. Nevertheless, intact forest (or other habitat) is a useful gauge for understanding how well a PA can mitigate land-use conversion, even when adequate management, political, or financial support might be lacking (Naughton-Treves et al. 2005).

Improved management effectiveness of PAs, the latter of the two groups, is a primary goal of the Convention on Biological Diversity's Programme of Work on Protected Areas (PoWPA), with specific targets set for 2010 (terrestrial PAs) and 2012 (marine

PAs). An extensive review by Leverington et al. (2008) collected reports on management effectiveness of approximately 6,300 PAs in 100 countries, close to 6% of the total number of PAs listed as designated in the WDPA (2009). This represents a surge in the evaluation of PA management effectiveness over the past several years. Nevertheless, it is still a far cry from the target of 30% of each country's PA network by 2010 (for CBD Party countries) (Chape et al. 2005). Furthermore, while the authors note that two methodologies dominate the field (Rapid Assessment and Prioritization of Protected Areas Management (RAPPAM) and the World Bank/World Wildlife Fund (WWF) Tracking Tool), they also identified 40 different methods used in their collection of reports (Leverington et al. 2008).

A common thread in these reviews of research on the performance and effectiveness of protected areas is that a plethora of methodologies and case studies exist, without much overlap or standardization among them. In addition, rarely is there an effort to evaluate PAs across these varying measures of performance and effectiveness. Recently, however, researchers have issued a call for pluralism in the approach to protected area effectiveness (Caro et al. 2009). The challenge is that many of these measures are rooted in disparate fundamental ideas about the purpose of PAs, particularly in terms of their scope of impact and engagement with local communities. This has added complexity to the measures of effectiveness, as concerns about the role of PAs extend to impacts on park residents, or communities who directly rely upon the ecosystem services that protected areas harbor.

A Broader Mandate for Protected Areas

This emphasis on the role of PAs as influencing "benefits beyond boundaries," (phrase borrowed from the World Parks Congress in 2003), is part of a broader globalizing conservation movement, a process that began in the late 1980s and early 1990s, aligning the goals of conservation with those of sustainability, or sustainable development. Zimmerer (2006) referred to this as the "third wave of conservation," suggesting that the phase of rapid increase in the establishment of protected areas worldwide generated an unprecedented convergence of conservation with community livelihoods, agricultural, and other landscapes previously dedicated to resource use (Zimmerer 2006). In fact, many of the same international conventions that have set conservation targets for protected area coverage, have also committed to integrating PA planning and management with the global development agenda, of which poverty reducation is now of primary concern (Fig. 18.4 presents dual timeline of these global agreements and targets). With regards to recent conservation targets for protected areas, the figure of 10% dominates, (as shown in the left-hand column of Fig. 18.4).

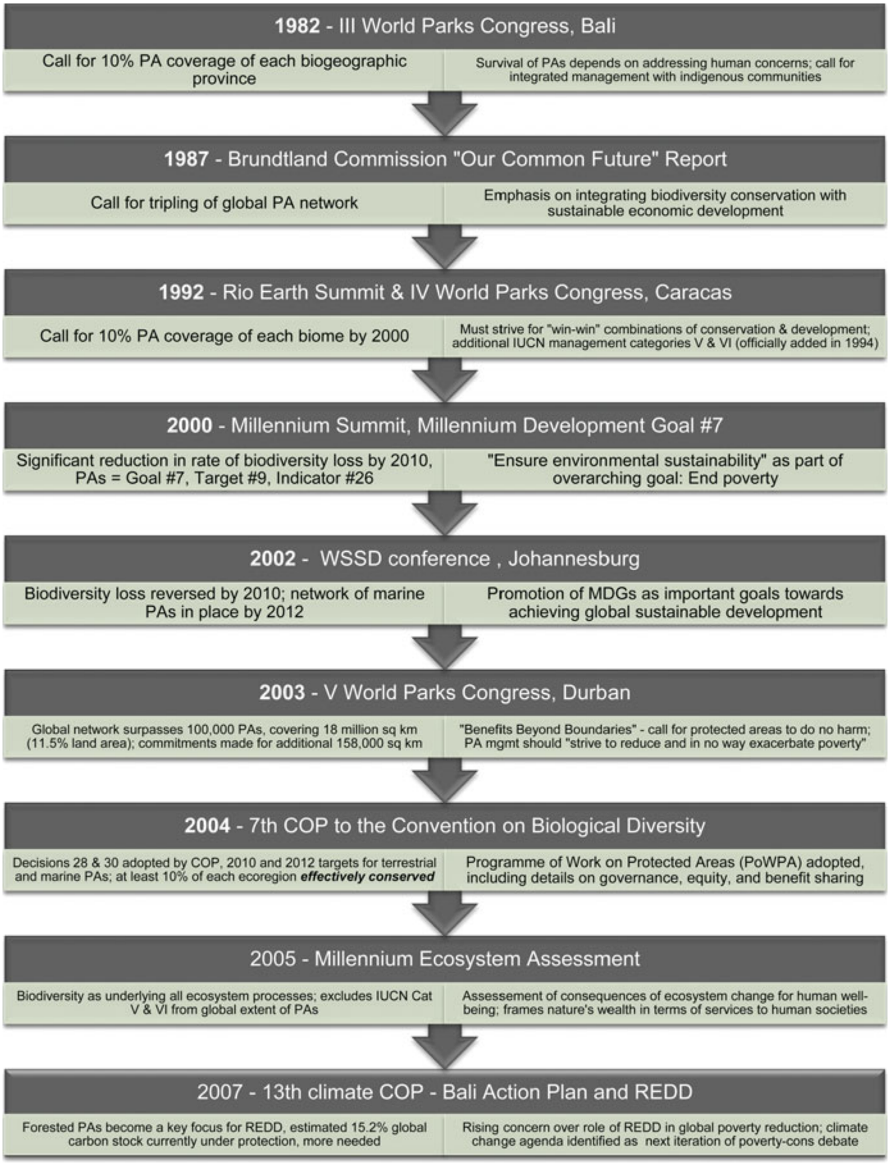

Fig. 18.4 Timeline of recent international conventions and actions, separated according to dual goals for protected areas: biodiversity conservation (*left*) and sustainable development (*right*) (Sources: BrundtlandCommission 1987; McNeely 2005; Naughton-Treves et al. 2005; Roe 2008; UNEP-WCMC 2008)

Global Conservation Targets: The Magic 10%

The first international convention to call for a 10% target for protected area coverage was the 1982 World Parks Congress, which took place in Bali, Indonesia. This was a specific call for 10% coverage of each biogeographic province (Udvary 1975). At the same time, participants in the Congress emphasized that protected area networks in the lesser developed countries would survive only if they also addressed human concerns. Five years later, the Brundtland commission released the report "Our Common Future," which was the first document to bring the term sustainable development onto center stage in the global development dialogue. The report also stated that protected area extent worldwide would need to be tripled in order to reverse the loss of critical ecosystems (BrundtlandCommission 1987). Given the extent of the global PA network at that time, this was essentially another 10% target.

The ubiquity of the figure of 10% is more evident than ever with the concurrent goals established by both the CBD and the Millennium Development Goals: 10% of Earth's land surface (and 12% of marine area), 10% of each country's land and marine areas, 10% of each biome (as delineated by the World Wildlife Fund), 10% of each terrestrial ecoregion (WWF), and, most recently, 10% of the world's forest types. Many scientists worry that global targets for conservation have created a false sense of security and success in the crusade to stem the loss of biodiversity. A review of conservation targets set by policy versus those based on scientific evidence found that the average targets recommended by evidence-based studies were three times higher than those put forth by policy-driven processes (Svancara et al. 2005). In considering the 10% target and year 2000 deadline set forth at the IV World Parks Congress, Soulé and Sanjayan expressed concern that such deadlines were unrealistic and that achieving 10% coverage of each biome would still result in at least a 50% reduction in global species richness (Soulé and Sanjayan 1998). It is apparent from the literature that 10% has been the magic number chosen for the sake of political feasibility, rather than based on evidence of the rates and locations of species extinctions or habitat loss.

In stark contrast with the concrete conservation targets established for protected areas, those agreements connecting PAs with sustainable development and human welfare have been focused primarily on overall improvement, with little guidance on how achievements might be measured. Prominent among this set of international agreements are the Millennium Development Goals (MDGs), established in 2000 with the ultimate goal of reducing extreme poverty by half by 2015 (Fig. 18.3). In particular, the MDG Goal seven of "Ensuring environmental sustainability," sets the reduction of biodiversity loss as one of its primary targets, (Target 7b), with significant reduction by 2010 (UNDP 2009). Protected areas are included as an indicator towards achieving this target, emphasizing that parks have the potential to "yield large social and economic, as well as ecological dividends" (UNDP 2009). While the majority of the 21 targets for the 8 MDGs are quantifiable measures, the target for reducing biodiversity loss is *directional*, perhaps in recognition of the challenge in measuring changes in the rate of biodiversity loss, particularly over such a short time period. Despite this, the inclusion of protected areas in the MDGs can be seen as an indication of just how extensive the mandate for protected areas has become: to now hold a place on international development and poverty reduction agendas (Naughton-Treves et al. 2005). This convergence of agendas for conservation and poverty reduction efforts has resulted in conflict and considerable debate as to whether or not “win-win” solutions are truly achievable.

Poverty Reduction Versus Biodiversity Conservation: A Collision of Goods?

Win-win scenarios were precisely what community-based natural resource management (CBNRM) and integrated conservation and development projects (ICDPs) were trying to achieve when they first began in the late 1980s. And yet, by the early 1990s, the proliferation of such projects was met with criticism, with several analysts arguing that the ambitious goal of achieving successful conservation and development was resulting in failure on both ends (Brandon and Wells 1992; Wells and McShane 2004). Similar concerns were raised that this was a classic problem of poor design and unrealistic assumptions, resulting in projects that were “Jack of all trades; master of none” (Robinson and Redford 2004). Even though the intention of these critiques might not have been to condemn the ICDP or CBNRM projects, the result in discourse and practice was the creation of a new divide, between those still believing in the potential of ICDPs, and those arguing that conservationists should re-focus efforts and limited funds on a prime directive: to slow the tide of biodiversity loss and establish more protected areas. Many viewed this latter group as symbolic of a return to the strict protectionism of the past, connected with both conservation policy and PA management, labeling the approach as *fortress conservation*, or “back to the barriers” (Hutton et al. 2005).

The turn of the millennium brought with it a new shift in this debate, with concerns focused on human rights issues and poverty reduction. Protected areas were directly implicated in these concerns, specifically as related to interactions and engagement with indigenous and traditional peoples (Chapin 2004), as well as involuntary displacement or resettlement of resident populations from PAs declared under strict management regimes (Geisler 2003; Cernea and Schmidt-Soltau 2006). The message emerging from World Parks Congress in 2003 was that the establishment and management of protected areas were to *do no harm* to local communities (WPC 2003). Implicit in this message is the idea that parks should not further impoverish people.

Poverty Reduction Versus Biodiversity Conservation

What has been more recently coined as the poverty and biodiversity conservation debate is really the latest iteration in a more extensive debate. The definition of the complex relationships that exist between humans and the environment has long been an issue of conflict, despite the ultimate irony that any connection we define between our "selves" and nature is ultimately a human construct and therefore inherently misconstrued. Naeem (Chap. 19, this volume), discusses in depth how our vision of ourselves in relation to nature influences the way in which we approach "sustainable development." The recent focus on biodiversity conservation (and by extension, protected areas) and poverty reduction is rich in rhetoric and poor in empirical evidence, on either side (Redford et al. 2006). The history of the debate, the modern version of which could be dated to the Stockholm conference in 1972, could be likened to a pendulum, a spiral, or a shifting balance beam: never quite achieving equilibrium. It has most recently been described as fiery, polarized, and stagnant – even a "dialogue of the deaf" (Redford et al. 2006). What is evident is that the struggles over a limited pool of financial resources for conservation and development as well as a sense of urgency for action drive the engine of this pendulum. A recent review by Dilys Roe highlights the three main concerns emerging from the debate:

1. The accountability and practices of big conservation NGOs (BINGOs), with respect to local communities, in particular;
2. An increasingly "back to the barriers" approach (Hutton et al. 2005) to conservation (with emphasis on forced resettlement from PAs);
3. The relative absence of biodiversity conservation from the development agenda, where poverty reduction is central (Roe 2008).

In an earlier piece, Adams et al. (2004) offer a typology for variations on the philosophical and practical interpretations on poverty reduction and biodiversity conservation. Their balanced discussion of this typology illustrates how individuals and institutions, operating at varying scales, might fall within specific categories, or viewpoints. They conclude that efforts must be made to recognize other institutional viewpoints and work between them; otherwise, failure is inevitable (Adams et al. 2004).

West and Brechin compare this debate to the modern form of tragedy, as defined by the German philosopher Hegel, when they noted: "the greatest and most troubling conflicts are not between good and evil, but between good and good" (Brechin et al. 2003). In fact, Hegel defined tragedy as a situation in which two rights or values are in fatal conflict – a "collision of goods."

And yet, as is noted by Roche (2006), tragedy inevitably involves some sort of resolution:

> For Hegel tragic fate is rational: reason does not allow individuals to hold on to one-sided positions. Because each stance is constituted through its relation to the other, the elimination of one stance leads to the destruction of the other. The human result is death, but the absolute end is the reestablishment of ethical substance. (Roche 2006, p. 17).

In many locations, there is evidence of a new reality emerging in the form of payments for ecosystem service (PES) initiatives or small corridor projects that are moving beyond missteps of the past. This change is also reflected in the broader discourse on conservation and development, where many argue that it is premature to give up on efforts to achieve both goods (Adams et al. 2004), even if the ultimate solution is to minimize tradeoffs (DeFries et al. 2007).

Identifying Research Gaps on Protected Areas and Poverty

When the discussion focuses specifically on protected areas, few acknowledge the substantial variation in ecosystems and social systems that fundamentally shape park–people relations. PAs are not randomly scattered across a landscape. They are created in areas of significance for biodiversity, ecosystem service provision, critical wildlife habitat, and, as is often the case of older PAs, areas of natural beauty or cultural/spiritual significance. Some are also located for strategic or political purposes, such as the group of peace parks established in Central America in the 1990s. Protected areas are not actors, even though this is how they are often referred to in discourse. Rather, they are societal constructs and forms of management for terrestrial or marine areas. While tied physically to a geographic location, protected areas are shaped by interests and actors that range from the very local to the global scale.

Our ability to generalize on the overall impact of protected areas on local communities is obscured by muddy concepts of "local," "poverty," and even "protected area." Moreover, substantial variation in ecological and social systems and park histories profoundly shape PA–people interactions. Undoubtedly, such variation prevents ultimate conclusions about the relationship between protected areas and poverty, but a short overview of PA impacts on local populations and human welfare at different scales helps to illuminate both the relationships we have identified and the knowledge gaps where further research and method development are needed.

Review of Recent Studies on Protected Areas and Poverty

Empirical studies on the spatial and temporal dimensions of poverty, biodiversity trends, protected areas, and ecosystem dynamics are critical vehicles for moving beyond the debate over biodiversity conservation and poverty. GIS and related technologies have opened up frontiers of policy development and practice by enabling the relatively quick integration of complex datasets, and permitting analysis at varying levels of spatial scale. Whereas earlier research on protected areas and poverty might have been limited to field-level case studies, in recent years, several researchers are trying to model this interaction at much broader spatial scales. In the following sections, I review the relevant literature published on studies from both the local and regional/global scales, and note the main gap in research that bridges site-level understanding with regional trends.

Individual PA Case Studies

Research at the level of individual protected areas and groups of households or communities tends to center on two main themes: identifying the costs and benefits of protected areas for local communities/households and sifting through the heterogeneity of relationships that exist between poor households and their proximate natural resource base, most often that located within a protected area.

The field of environmental economics has contributed greatly to this broad set of literature in the form of cost–benefit analysis of individual protected area establishment. In an analysis of costs borne by local populations in the creation of Ranomafana National Park in Madagascar, Ferraro estimates the aggregate present value of the opportunity costs to be approximately $3.37 million, (in 1991 USD), with the present value cost per household ranging from $353 to $1,316, an amount that the author notes is substantial compared with other households in the region (Ferraro 2002). While Ferraro notes that the economic burden of PA establishment, in this case, is impacting the livelihood opportunities of local communities, (with disproportionate burden on the poor), he does suggest that there are significant national and global benefits of the park creation, primarily through its value for biodiversity and ecosystem service provision. Ferraro concludes that opportunities for increased funding for conservation in the area could help offset the costs of PA establishment, if targeted appropriately (Ferraro 2002).

Most of these analyses, however, focus on the costs of PAs imposed on local communities overall, with little empirical attention given to the costs incurred by the different groups of poor within these communities, nor differentiated benefits of protected areas extending to the communities.

In her longitudinal analysis of forest use by local populations living near Kibale National Park in Uganda, Naughton-Treves found correlation between proximity to the park boundary and poverty (Naughton-Treves 2007). She cautioned against establishing causality between distance to PA and poverty, however, noting that the poorest households located further from the park in the earliest time period of analysis resorted to selling off their land and relocating to urban areas, or forest frontiers; whereas the poorest households close to the park remained on the land, albeit poor.

Studies by McSweeney (2002, 2005) and Takasaki et al. (2004) explore ways in which households not only rely upon forested PAs for income and food, but as "natural insurance" or as a coping strategy in the wake of shocks, such as natural disasters, economic downturns, or even political instability. In their research on river-dwelling peasant households living in and around Pacaya-Samiria National Reserve in the Peruvian Amazon, Takasaki et al. (2004) examine the responsive strategies employed by households when faced with natural shocks (i.e. flooding) and health shocks (i.e. human illness). They found that flooding, in particular, resulted in coping strategies among households that involved intensified use of forest resources, (more so than shocks related to human health issues), and also found that households that were asset poor were more likely to turn to intensified forest use, (through non-timber forest product (NTFP) collection), rather than those with even minimal assets, such as fishing nets (Takasaki et al. 2004).

Kendra McSweeney's multi-year survey of forest product sale by the Tawahka Sumu population of Honduras, (in/around the Tawahka Asangni Biosphere Reserve) revealed the heterogeneity that can exist within villages, and even at the household level, in the degree to which forest dwellers can be classified as *forest dependent* (McSweeney 2002). She also found that the households with the highest levels of forest product earnings, which otherwise might be labeled as those that are most forest dependent, were actually among the less poor within the communities – those who could rely on multiple sources of income. In a subsequent study of the Tawahka, McSweeney examined the coping strategies of households following the enormous shock of Hurricane Mitch in 1998, and analyzed the degree to which forest land and resources acted as a form of *natural insurance* during the after-shock period (McSweeney 2005). In this situation, she concluded that land wealth strongly conditioned the degree to which households could be said to have utilized the forest as a form of natural insurance, or as a safety net, during this time of shock. In the absence of landholdings, households were less able to cope, and restrictions (i.e. bans on commercial sales of forest products) acted as a form of poverty trap for certain land-poor groups (McSweeney 2005).

It is difficult, however, to compile quantitative assessments from these local-scale studies and examine trends across ecoregions, countries, or biomes, for example. Studies that make use of baseline data are among the most desirable, (ex-ante in terms of PA establishment), but such "natural experiments" often prove too challenging in terms of planning and funding of research. One potential longitudinal study is currently underway and focuses on change in community-level welfare pre- and post-park establishment in Gabon, directed by David Wilkie and partners. Since 2006, the research team has been conducting community-level and household panel surveys on a sample of 1,000 households that have traditionally used the resources within four new protected areas, and comparing those results over a 5-year period with those of an equal sample of control households (Wilkie et al. 2006).

Regional and Global-Scale Research: Early Insights and Data Challenges

Analysis of the spatial relationship between protected areas and poverty at the global or regional scale is necessarily simple, given the coarseness of scale and the challenge of assembling uniform datasets for analysis.

Very few common measures of poverty exist that have been standardized to a regional or global scale. Many researchers turn to datasets provided by the Center for International Earth Science Information Network (CIESIN) at Columbia University, which use such variables as infant mortality and child malnutrition as a sole proxy measure, or correlate, for poverty. Others incorporate more traditional measures, such as the Human Development Index (HDI), a composite using health, education, and income-based parameters, or the single income-based measure of population living on less than \$1/day. The HDI is published by the United Nations Development Program (UNDP), and national statistics on population living on less than \$1/day (as well as less than \$2/day) are published and updated by the World Bank. All four of the measures mentioned above are incorporated into the MDGs, as a means of tracking progress towards certain development targets (UNDP 2009).

Similar to correlates of poverty, the data for the protected area side of these analyses is limited in definition, with the WDPA offering the most comprehensive and standardized database available for analysis of PAs at these scales. Most often, protected area presence is modeled using the number and extent of PAs in the area of interest. A slightly more complex measure might include grouping the PAs according to their condition or IUCN management category, recognizing that varying degrees of permitted access and use may translate into different relationships with poverty in the region. The analysis of resource use across parks in Ecuador and Peru by Naughton-Treves et al. (2006) found dramatic differences in the intensity and extent of use within PAs of similar management categories (Naughton-Treves et al. 2006). This observation cautions researchers to further examine whether these categorizations, such as IUCN management category, are realities in the on-the-ground context of their study region.

In a recent special issue of the journal *Oryx*, Upton et al. (2008) examine the relationship between protected area networks (as measured by number, extent, and management category) and national-level poverty indicators, measured using the HDI, GNI per capita, and % of population living on less than $1/day (Upton et al. 2008). Using Spearman's rank correlations between the poverty and protected areas datasets, they observe that individual PAs in the poorest countries are larger and categorized in the more restricted use IUCN management categories I–III. In wealthier countries, the results show that the overall total area protected is greater, yet with a tendency towards greater numbers of small PAs. They conclude that protected area networks have no discernible correlation at a national scale on the incidence of poverty (Upton et al. 2008). The results of this study contrast with the analysis by Geisler (2003) on the growth of protected area networks in 38 African countries, and the observed tendency of greater PA growth in countries with the highest incidence of poverty. Geisler's observations were based on comparing protected area growth from 1985 to 1997 with the bottom half of countries ranked according to the Human Development Index (HDI) (Geisler 2003).

In the same special issue, research by de Sherbinin (2008) assesses whether poverty near large "strictly" protected areas, (defined as IUCN categories I–III), is higher than national rates, using the subnational infant mortality dataset produced by CIESIN (de Sherbinin 2008). While infant mortality rates were somewhat higher around large protected areas, the author does not attribute causality because any apparent influence of protected areas is muted by other possible factors, such as spatial poverty traps existing in the area (de Sherbinin 2008). Here, de Sherbinin uses the term "spatial poverty trap" to refer to the unique effect that geographic location alone can have on keeping households from lifting out of poverty, as identified by Jalan and Ravallion (2002) in their study of rural households in China (Jalan and Ravallion 2002).

Finally, a recent article in *Science* has challenged the assumption that parks act as negative buffers to development (Wittemyer et al. 2008). In an analysis of human population change around 306 PAs in 45 countries in Latin America and Africa, the authors conclude that PAs act as an attracting force for human migration. They also find no difference in the occurrence of child malnutrition, (an indicator often used for poverty), for populations living within and proximate to versus distant from PAs.

In a direct challenge to Wittemyer et al.'s results, Joppa et al. (2009) replicate the study and find that population growth near protected areas is no different than growth in other rural parts of the same country. Furthermore, they posit that any detected disproportionate increase in population around PAs is due to growth in nearby population centers (Joppa et al. 2009). In further discussion of both articles, Nelson et al. (2009) note that the primary data for the analysis of population growth in these studies was used erroneously and that it is not possible to derive rural population growth estimates from the dataset, as assembled by Nelson et al. for the UN Environment Programme (Nelson et al. 2009). In the end, Nelson et al. (2009) concur with the overall analysis by Joppa et al. (2009) and conclude that there is no observable difference between total growth rates immediately around parks and total growth rates in the broader landscape across parks in Africa and Latin America (Nelson et al. 2009). Overall, the back-and-forth discussion over the initial analysis by Wittemyer et al. (2008) sheds light on the need to exercise caution when working with regional and global scale datasets, as these have usually been assembled from national-level statistics and require a firm understanding of the methods and assumptions involved in their compilation.

Overall, the results from this set of articles suggest that, while poverty and protected areas might spatially coincide, there is no clear impact of PAs on poverty, either negative or positive. Furthermore, none of the authors has been able to attribute causality at this coarse scale, due to data limitations and the inability to control for the suite of factors that can influence poverty. Nonetheless, such observations help us to identify the scales at which this relationship is most active and relevant for policy and management related to environmental governance and human development. Despite the limitations of these primary data inputs to the analyses, studies at the regional-global level are an important step in addressing the issue of scale in defining the complex relationship between poverty and protected areas.

This review has hovered around two opposite ends of the spatial spectrum, when discussing the relationship between PAs and poverty. There is an obvious gap in research efforts to model these relationships at a scale that can bridge this spectrum, and provide insight at the scale of nations, landscapes, and sub-regions, where management efforts and policy recommendations are most relevant.

The current focus of the debate on integrating conservation and poverty reduction is rich in rhetoric and poor in empirical evidence, on either side (Agrawal and Redford 2006). Often each side posits win-win solutions to the dual-challenges of achieving conservation objectives and improving human well-being, or otherwise eludes the concerns of opposing views. Only in the past few years have efforts begun that extend across the chasm and engage both sides in more productive discourse, (e.g. the Poverty Conservation Learning Group (PCLG): www.povertyandconservation.info). Others have even begun identifying big win–small loss solutions (DeFries et al. 2007), or exploring trade-offs, (e.g. the Advancing Conservation in a Social Context: Working in a World of Tradeoffs project, www.tradeoffs.org) (Sunderland et al. 2007). As new lessons emerge on the interaction between conservation action and poverty reduction initiatives, we can use new understanding to appraise the risks and realize the potential benefits of alternative policies and programs such as payments for ecosystem services (PES) and Reduced Emissions from Deforestation and Degradation (REDD) (MEA 2005; Coad et al. 2008a), as discussed in the PES section of this volume. The challenge will lie in maintaining the optimism of achieving win-win solutions as well as a sense of urgency given poverty and biodiversity loss, all the while defining and adjusting to the trade-offs and big win–small loss scenarios that will inevitably surface.

Conclusion: New Direction

With increasing understanding of the interactions between ecosystem function, biological diversity, and climate change/destabilization, ecologists recognize that protected areas represent only one mechanism, one keystone in a landscape approach to conservation. In a recent review of the literature, however, Gaston et al. (2008) concluded that there is a dearth of empirical knowledge on the ecological performance of protected areas at all scales of observation (individual, PA portfolio, and networks of PAs). Ecologists are needed not only to help fill this knowledge gap, but to help translate and disseminate the adaptive management techniques that will be necessary to adequately conserve biodiversity through protected areas and other landscape conservation mechanisms. With the broadening of the definition (and mandate) of "protected area" to include co-managed and community-managed areas as well as private reserves, indigenous reserves, and lands or marine areas with easements or conservation incentives, such as payments for ecosystem services, ecologists must learn how to communicate their knowledge to a much larger group of actors than the traditional national park manager. There will be increasing demands for the scientific community in general to offer numbers and concrete objectives derived from objective science relating to the *what*, *where,* and *how much* for biodiversity conservation and protected area establishment. As these objectives are matched alongside those of poverty reduction, ecologists can inform the development of common sets of metrics for analysis and modeling across scales. Additionally, as more is understood about *what* and *where*, ecologists can help move the policy discussions beyond that of "minima" and into an improved understanding of *how* (Redford and Sanjayan 2003). This could often involve a dose of realism. DeFries and Hansen suggest that, in much of the world where it is unrealistic to find true "win-win" scenarios for balancing protected areas and human needs, the question becomes: "Are there key locations within the greater ecosystem that, if managed appropriately, would maintain ecological function while minimizing restrictions on human use?" (DeFries et al. 2007). In other words, they ask: Are there big win–minimal loss situations?

The more recent focus of landscape or ecosystem-level planning for effective protected area management and biodiversity conservation has led some to worry that conservation scientists are shifting back to a model of "fortress conservation" and channeling funding for conservation away from the need to address human welfare concerns (Roe 2008). The perception is that by effectively “zooming out” and focusing on landscape-level processes, particularly with respect to climate change, concerns regarding the effects on local communities have become muted. While planning for protected areas and their role within a given ecosystem might best occur at a coarser scale, it is not possible to remove the very local interactions that take place between the acts of creating and managing those areas, and the relative livelihood impacts experienced by those living in their realm of influence. Perhaps, the most pertinent message the ecologist can convey in the midst of this polarized debate is that we must stop thinking of protected areas as individual entities, even actors, in our landscapes. Failure to do so only results in parks becoming less effective, both in terms of addressing human welfare concerns and in conserving biological diversity.

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Chapter 19
Looking Forward: The Future and Evolving Role of Ecology in Society

Shahid Naeem

If the misery of our poor be caused not by the laws of nature, but by our institutions, great is our sin.

(Charles Darwin, quoted in Gleick 1998)

Ecology and Poverty: An Unorthodox Perspective

Environmental Sustainability and Sustainable Development

If a development strategy is ecologically sound, meaning that it is founded on ecological principles and is environmentally sustainable, then it qualifies as sustainable development, but qualifying as such makes no guarantees about whether it will or will not promote poverty reduction. There are, of course, different definitions of sustainable development, but a universal requirement for any development program to be sustainable is that its activities that are designed to meet the needs of the present generation will not jeopardize the ability of future generations to meet their needs. This requirement is akin to long-term (i.e., multigenerational) ecological stability where the water, nutrient, and energy needs of millions of species, on a global scale, are met generation after generation for tens to thousands of years. Because ecological systems appear globally to exhibit slow dynamics (Fig. 19.1), it makes sense that ecology is a science to which we might turn for understanding how to achieve environmental sustainability. To put it into an ecological perspective, consider Fig. 19.1, which

S. Naeem (✉)
Department of Ecology, Evolution, and Environmental Biology & Center for Environmental Research and Conservation, Columbia University, New York, NY, USA
e-mail: sn2121@columbia.edu

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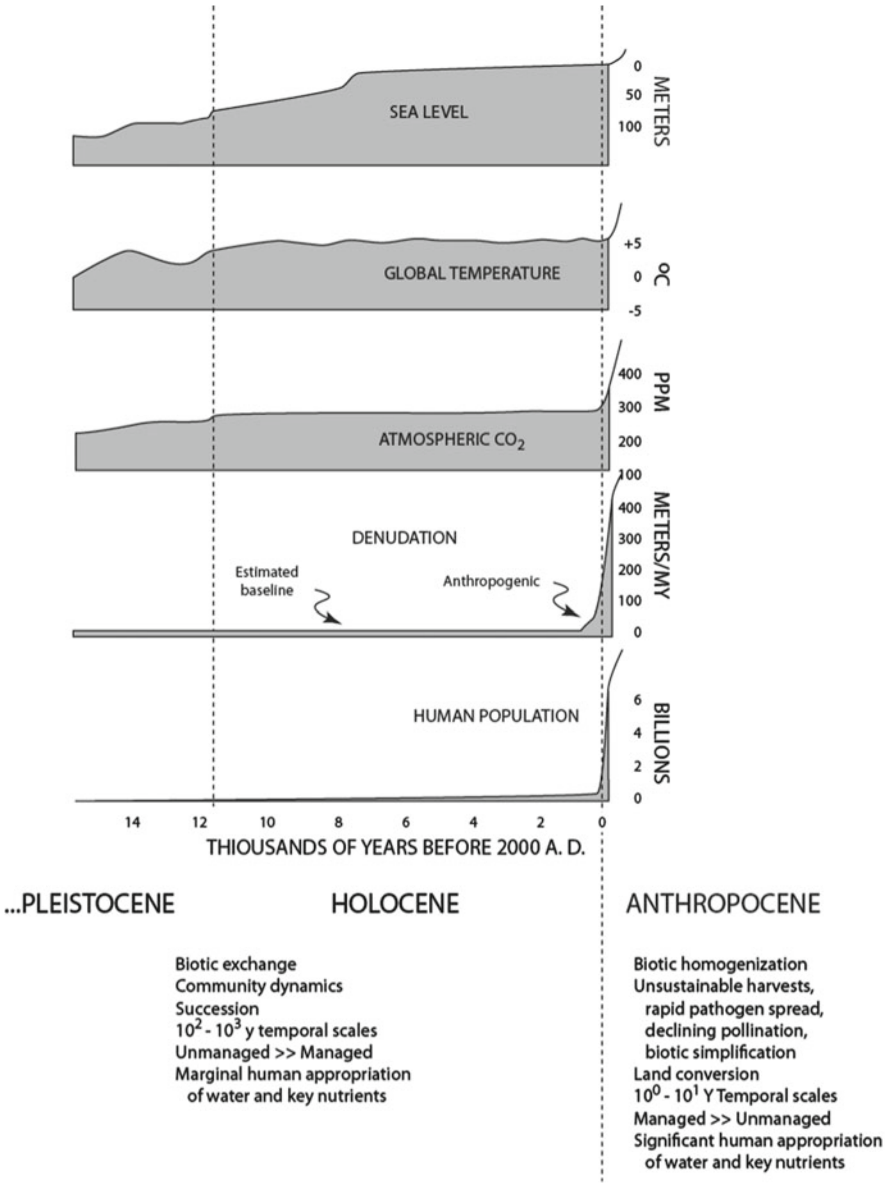

Fig. 19.1 Achieving environmental sustainability in the Anthropocene. During the Holocene, the period in which humanity flourished, the environment was steady over long periods in terms of human generations. It stands to reason, then, if in the Holocene, environmental sustainability for roughly 30 million species lasted for thousands of years, these sciences may inform our desire to achieve environmental sustainability in the Anthropocene. The text at the bottom considers basic differences between the Anthropocene and Holocene. See text for further discussion (Adapted from Zalasiewcz et al. 2008)

compares the Holocene, the last 12×10^3 years, to what some call the Anthropocene (Crutzen 2002; Zalasiewcz et al. 2008; Steffen et al. 2009), which covers roughly (although, the starting date is debated) the last three centuries (more than that if the starting point is with the commencement of agriculture 10,000 years ago). In the Holocene, the dominant ecological processes that shaped the environment were biotic exchange (the rare occurrence of species moving long distances between ecosystems), community dynamics (the population dynamics of species that interact with one another through competition, facilitation, and consumption), and succession (the predictable return of disturbed ecosystems to their original states); these processes operated over temporal scales of 10^2–10^3 years. During this epoch, unmanaged habitats were much more abundant than managed habitats, and human appropriations of water, energy, and key nutrients were marginal. In sharp contrast, the Anthropocene is exhibiting exponential changes in our environment (Fig. 19.1), characterized by biotic homogenization (widespread occurrences of domestic and exotic species), unsustainable harvests (e.g., collapsing fisheries, land degradation through poor agricultural or forestry practice), rapid pathogen spread, declining pollination, biotic simplification (once complex communities made up of hundreds to thousands of species are becoming domesticated or weed-dominated systems of low complexity), and land conversion of unmanaged habitats to managed habitats (e.g., farms, plantations, grazing lands, urban, peri-urban and suburban habitats); these processes are operating on time scales of 10^0–10^1 years, yet resulting in significant environmental change and significant human appropriation of water, energy, and key nutrients. Clearly, the ecology of the Anthropocene is very different from the ecology of the Holocene, and because humanity desires the environmental steadiness (i.e., sustainability) of the Holocene environment, but not necessarily its specific conditions, such as relatively few humans, it stands to reason that ecology could shed some light into how one achieves environmental sustainability in the modern world.

If a key aspect of the definition of sustainable development is that nations will continue to develop and poverty reduction is one of the goals of such development, it is important to ask whether poverty reduction is really compatible with achieving environmental sustainability and whether ecology, which explains why the steady Holocene is so different from the rapidly changing Anthropocene, can inform practices and policies aimed at promoting sustainable development. This may seem a strange observation following a collection of chapters that argue that sustainable development is critical to alleviating hunger, insuring access to water, improving health, conserving biodiversity, and addressing climate change, all of which are key ingredients to development strategies for tackling poverty. The chapters in these volumes, however, concern the idea that sustainable development requires the integration of the social (e.g., economics, anthropology, sociology) and natural (e.g., chemistry, physics, biology) sciences to achieve sustainable development and alleviate poverty, especially in rural areas. I am less sanguine, however, about the popular idea that the science of ecology, that is, the science of the study of the relationship between organisms and their environment, should be integral to poverty reduction. In this chapter, I will develop what may seem to some as an unorthodox perspective; the science of ecology may partly explain why poverty exists, but ecological processes actually promote rather than alleviate poverty.

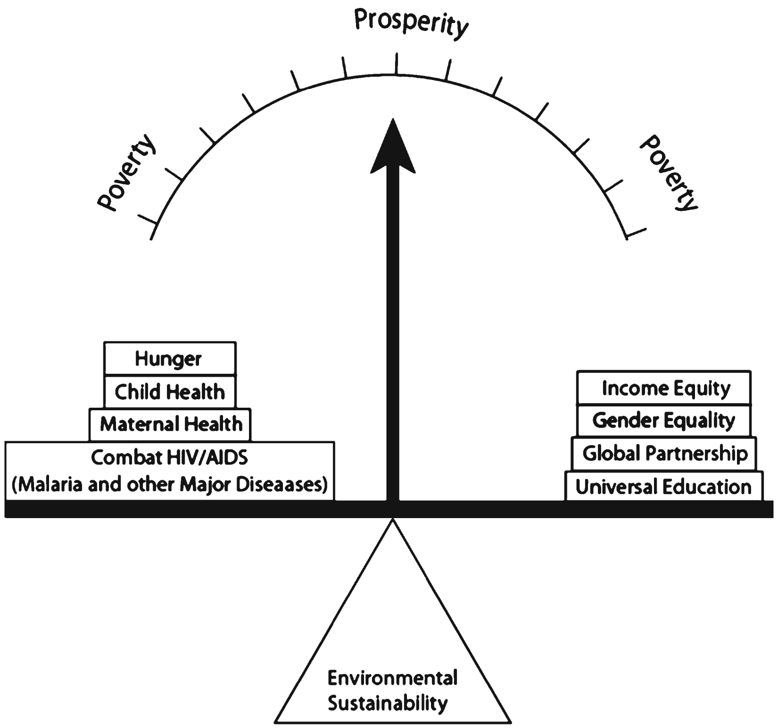

Fig. 19.2 Environmental sustainability as a foundation for sustainable development and poverty reduction. This framework is derived from the Millennium Development Goals, a series of benchmarks for poverty reduction and sustainable development that anchor social endeavors to insuring environmental sustainability (shown here as a fulcrum). I have placed biological goals, such as health and hunger on the right, with more social endeavors on the right, such as achieving universal education and a global partnership. Prosperity occurs when social programs attain their goals without jeopardizing environmental sustainability, otherwise, the system tilts towards increasing poverty among people

To explore how natural and social sciences are integrated into strategies for poverty reduction, we can begin by examining the United Nations Millennium Development Goals (MDGs) (http://www.un.org/millenniumgoals/) as a primary example. The MDGs are founded on the idea that sustainable development is a pathway to achieving poverty reduction. They are, however, primarily benchmarks for improving human well-being by 2015, not a framework for sustainable development. In Fig. 19.2, I construct a framework for sustainable development and poverty reduction consistent with the MDG benchmarks. This MDG Framework (Fig. 19.2) consists of a mix of natural resource management strategies and social programs that are balanced on a foundation of environmental sustainability. The MDG benchmarks concern improving human well-being, such as improving child health, maternal health, and combating diseases, and balancing these with social goals such as striving for greater gender equality, universal education, and building a global partnership. The first MDG is to end poverty and hunger, and is the benchmark by which the success of the overall program can be judged. The MDG Framework therefore suggests that there is some optimum for human well-being (which I have labeled *prosperity*) that can be achieved, but when resource management and social programs are unbalanced, the outcome is suboptimal (which I have labeled *poverty*). That is, when natural resource management interferes with social programs or vice versa, and if any activity interferes with environmental sustainability, the balance is tipped and poverty ensues (Fig. 19.2).

Note that it is only the goal of environmental sustainability (MDG 7) which insures *sustainable* economic development, meaning that development does not jeopardize the welfare of future generations (i.e., avoids intergenerational negative externalities, as economists would say). The other MDGs are not about sustainable development; they are merely laudable goals for any development plan, sustainable or otherwise. Countries meeting these goals are not compelled to prove that they did so following a pathway of sustainable development, but the assumption is that if environmental sustainability was achieved, then their pathway was one of sustainable development.

Like the MDGs, these volumes provide guidance for sustainable economic development, but their focus is on the scientific foundations. The volumes consider hunger, water, health, and energy (natural science-based issues), and climate change (as a driver or stress), followed by the roles of education, gender, population, and economics (social science-based issues) in achieving sustainable development and poverty reduction. The editors' overarching thesis for the volumes is that the growth of sustainable development requires some grounding in ecology and they point to the EcoAgriculture Partnerships, the EcoHealth symposium at the Ecological Society of America meetings in 2006, and EcoNutrition proposed by Deckelbaum et al. (2006), as evidence of a nascent movement to integrate ecological science into sustainable development and poverty reduction.

What do these volumes tell us about the role of ecology in achieving sustainable development and poverty reduction? Here, I will first consider why ecology is perceived as a critical science in sustainable development and poverty reduction. I will argue that this perception stems from the conflation of ecology with environmentalism and the idea that nature is our friend, which has led to the perception that ecological processes inherently improve human well-being. To test the idea that ecology should be a more prominent science in sustainable development and poverty reduction, I examine three core concepts in ecology to see what they tell us about the issues. Finally, I will suggest that what emerges from these volumes is a different framework for sustainable development and poverty reduction and a different theory of poverty – one that involves delineation between what I will call *ecological poverty* and *social poverty* and the interaction between the two.

Is Nature Friend or Foe?

Ecology is the science most closely identified with Nature, but humanity has had a complex relationship with Nature and this clouds our understanding of ecology and its role in sustainable development and poverty reduction. The question, Is nature our friend or foe?

reflects a dialectic that has long shaped much of environmental debate and shapes the issues that these volumes address. For example, de Sherbinin (Chap. 5, Vol. 2) on population, references the undercurrent of environmental rhetoric in which some perceive humanity as a sort of planetary pox and a threat to Nature. Similarly, Smukler et al. (Chap. 3, Vol. 1) note that some see Nature as an impediment to producing the food needed to feed a billion people suffering from hunger, while others see it as a guide for developing sustainable agro-ecosystems. Adamo and Curran (Chap. 7, this volume) note that human migration, and the plights that accompany it, are (in part) struggles to escape environmental change or the vagaries of Nature. The issues are, of course, much more complex, but throughout these volumes and in most environmental discourse, there is a wide range of opinion as to whether humans have rights to reproduction, water, food, and security and whether poverty arises because of Nature's laws or because we ignore the laws of Nature.

Whatever one's perspective about Nature and its laws (or ecology and its principles), humanity is at a crossroads where we face three choices: (1) return to a point where humanity's needs can be met by natural processes as they once were, (2) move forward along the same path of traditional development until natural resources are exhausted, but substitutes are found along the way, or (3) move forward along a new path of restoration and conservation that improves the efficiency of our use of natural resources without impairing economic development. All three pathways, one backward and two forward, aim to improve human well-being, but reflect two different environmental philosophies: Arcadian environmentalism, whose subscribers feel we must learn to live in harmony with Nature as it is versus Promethian environmentalism, whose subscribers feel we should manipulate Nature to improve our well-being (Lewis 1992). Thus, the nature-is-our-friend-or-foe dialectic shapes the modern economic development debate as it has shaped environmental debates throughout history. Because many of the chapters in these volumes focus somewhat on the third pathway, they understandably see Nature as our friend, though they recognize limitations to this perspective, and because ecology is the principle science of Nature, they see ecology as a potentially valuable science for achieving sustainable development.

Core Ecological Constructs and Poverty

Environmental Sustainability and Sustainable Development

The consensus represented by the Brundtland report (WCED 1987) was a call for sustainable development, but while we understood what was desired (development without intergenerational negative externalities), the means to get there were less clear. Numerous calls for science-based sustainable development, specifically synthesizing the social and natural sciences, led to the proposed new field of *sustainability science* (e.g., Lubchenco et al. 1991; National Research Council 2000; Kates and Parris 2003; Sachs 2004; Watson and Zakhri 2005; Clark 2007; Holdren 2008). These volumes are among the first to contribute to this effort.

All sciences (natural, social, applied) are part of sustainability science, but because environmental issues such as climate change, biodiversity loss, emerging diseases, collapsing fisheries, invasive species, and many others are what motivate the call for sustainable development, ecology takes center stage in sustainability science. I have selected three core ecological constructs here to see what they might tell us about sustainable development and poverty. The first is the concept of the *niche* which provides insights into the basic relationship between an organism and its environment. The second is the concept of the *lognormal(ish) distribution* of abundance which provides insights into the distribution of natural resources among species. I use the parenthetical suffix *ish* to qualify that, in fact, there are many distributions that could serve equally well, but as I will explain below, phenomenologically, they are generally similar to the lognormal distribution. The third is the concept of *biodiversity and ecosystem functioning* which concerns how the diversity of life influences the magnitude and stability of ecosystem processes, such as production and decomposition.

Ecological Concept 1: The Niche

The concept of the *niche* is a venerable construct in ecology concerning how the distribution of species is shaped by the availability of habitat and natural resources (Colwell and Fuentes 1975). The concept continually evolves, beginning with Grinell (1917) who saw the niche as the habitat within which a species could persist, something that could be quantified in terms of climate, resource availability (e.g., water, nutrients, energy, such as light), and other habitat factors. Elton (1927), in contrast, argued that a species' niche is the role it plays in the community, much the way a butcher, baker, and candle stick maker each play unique and important roles in their community. Hutchinson (1978) provided possibly the most widely used concept of the niche, dividing a species' niche into its *fundamental niche*, which consists of the set of environmental conditions under which a species can persist in the absence of other species, and its *realized niche*, which is the smaller set of conditions that a species actually occupies because competitors, predators, disease, and other species limit its ability to grow and persist even where abiotic conditions are favorable. In general, it is assumed that a realized niche is smaller than a fundamental niche, but this may not be true when facilitation or mutualism is involved. For example, the realized niche of plants that have symbiotic associations with mycorrhizal fungi may be much larger than the fundamental niche because the presence of fungal associates helps the plants persist in places they could not otherwise inhabit.

More recent advances in the concept of the niche include species' impacts on their environment, sometimes referred to as *niche construction* (Liebold 1995;

Laland and Sterelny 2006; Marco 2008). Ecosystem engineers (e.g., Jones et al. 1994) are probably the most well- known examples of species that construct their niche, such as beavers (*Castor canadensis*) that build dams to create the ponds where they live. Other examples include legumes that influence soil fertility, corals that build reefs, vegetation that stabilizes shorelines, grazers that hinder the incursion of woody species into grasslands, or social insects such as termites that can alter the structure and biogeochemistry of entire landscapes.

What is the human niche? Humans, as a biological species, have a fundamental niche like any other large-bodied, omnivorous, social mammal, and this human fundamental niche is undoubtedly larger than what is realized when competition, predation, and disease take their toll, producing the human-realized niche. The human constructed niche, however, is staggering in its extent, larger than any species has ever had in all 3.5 billion years of life's history on Earth. Our constructed niche is virtually the entire surface of Earth. Because we can modify biodiversity (eliminate threats and favor those species we like) and habitat (set fires, build roads, canals, reservoirs, and shelters), there are few places on terrestrial Earth that we do not live and few places on Earth, including marine and freshwater systems, that we do not come in contact with or influence directly (e.g., trawling ocean floor, as McClennen discusses, Chap. 16, Vol. 2) or indirectly (e.g., acidification of the ocean through increases in atmospheric carbon dioxide through fossil fuel burning) (Kareiva et al. 2007). Thus, while most species on Earth are constrained by interactions with other species, climate, and resource limits, the human niche is constrained only by biophysical limits (Fischer et al. 2007; Rockstrom et al. 2009).

Ecological Concept 2: The Lognormal(ish) Distribution

The natural world is lognormal, not normal or Gaussian, meaning that many phenomena are better represented by skewed or unequal distributions rather than symmetrical distributions (see Limpert et al. 2001, for a review). The common perception is that most things are distributed normally, which means that most things have average values while things with high and low values are equally rare. In reality, however, habitat and natural resources (i.e., water, energy, and nutrients) are highly unequally distributed among species. In general, few species are common while the vast majority of species are rare.

The prominence of the lognormal distribution in ecology was first discussed in the 1960s by Preston (1962), but long before Preston's foundational work, the fact that most species are rare and only a few common was well recognized in ecology in the form of the species–area relationship. The species–area relation was described in the 1920s by Arrhenius (1921) who recognized that when one looks for species in a landscape, at first it is easy to find new species, but it takes increasingly more effort to uncover further new species. The two observations, the lognormal distribution of species abundances (few dominant and many rare) and diminishing returns for one's efforts in finding new species in a landscape, are related (May 1975).

The question of why most species are rare and only a few are common has dominated ecological research for decades and remains one of the most active fields in the discipline (May 1975; Sugihara 1980; Plotkin et al. 2000; He and Legendre 2002; Hubbell 2005; Harte et al. 2009). The reason for the lognormal(ish) distribution was first famously (and incorrectly) explained by MacArthur (1960) as the result of species dividing resources, a theme later explored by Sugihara (Sugihara 1980). The pattern has also been correlated with energy (Wright 1983; Wylie and Currie 1993a, b; Storch et al. 2005), geographic range (Hanski and Gyllenberg 1997), endemism (Harte and Kinzig 1997), and a combination of speciation, emigration, and other processes (Hubbell 2001; Alonso et al. 2006).

The fact that most species command little of the world's biological assets suggests that the vast majority of species are what one might consider *ecologically poor*. Because research on the distribution of species is dominated by correlative studies, in fact we cannot be sure what causes most species to be rare, but whatever the cause, the fact is that habitat and natural resources are highly unequally distributed with a relative handful of species commanding most of the world's space and resources. Preston (1950), who noted that species abundances fit the lognormal distribution, also noticed that human income distribution, as described by Pareto, was similar to the commonness and rarity of species. Preston also observed that the distribution of energy states among molecules in an ideal gas described by the Maxwell and Boltzman laws was also similar to the Pareto law. Indeed, Limpert et al. (2001) note that the distribution of latency periods of infectious diseases, mineral resources, the length of words in the English language, the age of marriage in Danish women, the age of onset of Alzheimer's disease, and the income of Swiss households, along with many other phenomena are also well-fitted by the log-normal distribution. Preston suggested that perhaps human wealth (and poverty) is the outcome of the same kinds of processes that govern other patterns in nature, such as the lognormal distribution of species. Figure 19.3 illustrates the similarities by showing the frequency of species found at particular abundances for trees, butterflies, and birds and the frequency of household monthly incomes. All four plots show that most species are rare (low abundance) while a small number of species have enormous numbers and command much of the ecological assets. Likewise, most human households are poor in comparison to an extremely small number of extraordinarily wealthy households.

To explore the relationship between ecological poverty and human poverty, we can see if patterns in income distribution are influenced by national wealth – a way to relate total economic resources at the disposal of a country with how those economic resources are divided among its citizens. Using the per capita Gross Domestic Product (GDP) adjusted for Purchasing Power Parity (PPP) as a metric of a nation's wealth and comparing it to income inequality as indexed by the Gini Coefficient, there is a strong, negative correlation between the two (Fig. 19.4). Thus, as wealth increases in many nations, the percent of resources that are captured by the wealthy

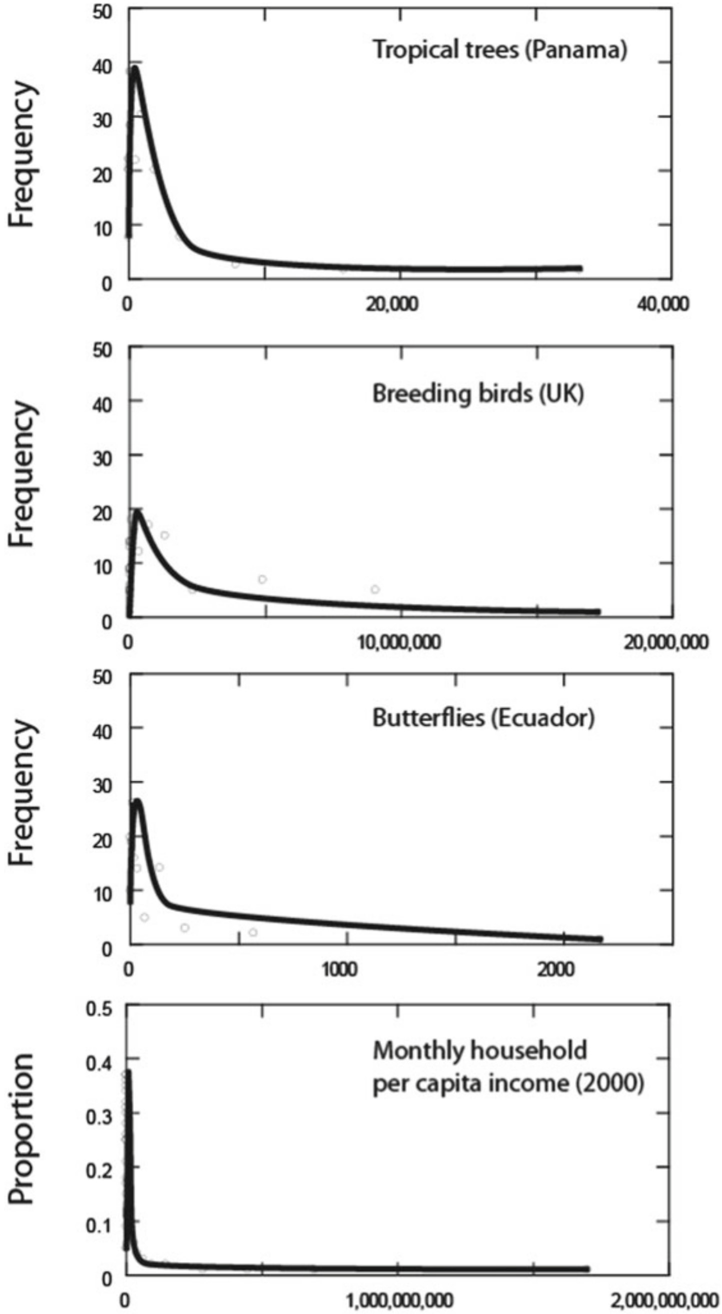

Fig. 19.3 Lognormal(ish) distributions of species and wealth. Curves are modeled after figures shown for trees, birds, and butterflies in Williamson and Gaston (2005). *Frequency* refers to species out of total found in all samples. Curve for global monthly household incomes is based on a figure in Bussolo et al. (2009) where income was reported in 1993 US dollars adjusted for Purchasing Power Parity. *Proportion* indicates proportion of households. Curves are not statistically derived and drawn only for illustrative purposes

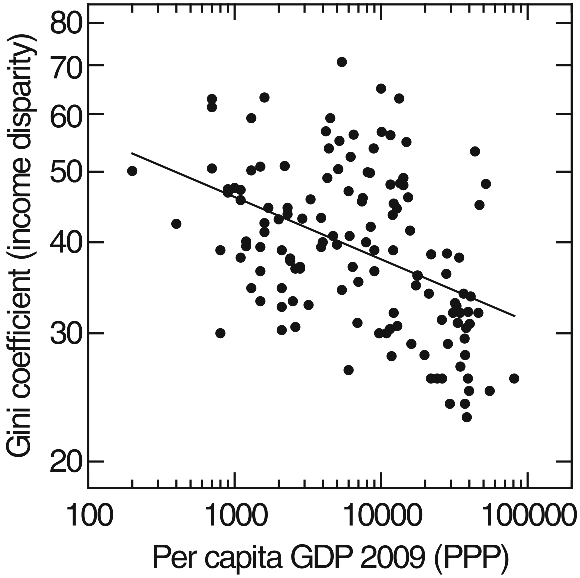

Fig. 19.4 Wealth and income inequity. Data derived from United States Central Intelligence Agency World Facts Book. Each point represents values for a single nation

may increase more than the percent captured by the poor, which is quite similar to what happens among species in ecological systems where a few species command most of the resources and the majority of others get very little. (I say "may" to acknowledge that these indexes, the data used to calculate them, and their interpretations are often controversial).

Note that this species-level perspective does not examine resource allocation within species, which is an important issue, but not easily resolved. The assumption is generally that resources within a population are distributed as normal distribution – most individuals obtain similar amounts of resources, but a few are on the margin and a few do very well. That is, fitness is normally distributed. That income, as a proxy for resource allocation, is so hugely skewed in humans represents potentially an additional issue of relevance to poverty reduction. That is, not only has human success impoverished the rest of our biota, but within our species, many individuals have been impoverished by others.

Ecological Concept 3: Biodiversity and Ecosystem Functioning

One of the most familiar yet misunderstood terms in ecology is *biodiversity*. Created in the 1980s specifically to describe the diversity of life on Earth well beyond the minor aspect of diversity that concerns taxonomy, the term has unfortunately devolved in its common usage to being synonymous with taxonomic diversity. Biodiversity actually refers to the sum of all molecular, genetic, taxonomic, physiological, morphological, behavioral, evolutionary, and functional diversity as it is distributed over space and time. Equating biodiversity solely with species richness creates the unfortunate perception that only the number of species matters when, in fact, what matters is the relative abundance (commonness and rarity) and the influence biodiversity has on ecosystem functions and ecosystem services (MEA 2005b; Naeem et al. 2009b).

Few things in ecology have been as well documented as the influence of biodiversity on the functioning of ecosystems and the services they provide (Schulze and Mooney 1993; Kinzig et al. 2002; Loreau et al. 2002; Naeem et al. 2009a). The first experimental demonstration of biodiversity's importance was published over 15 years ago (Naeem et al. 1994). Like most new disciplines, however, biodiversity and ecosystem function (BEF) was surrounded by much controversy until scientific consensus was reached (Hooper et al. 2005). In spite of the initial controversy, the idea that biodiversity was the foundation of ecological systems and its conservation was instrumental in improving human well-being became the central architecture of the Millennium Assessment's conceptual framework that was used by over 1,300 natural and social scientists from around the world (MEA 2003). Since 1994, over 900 studies on BEF have been published and recent meta-analyses have identified the most robust of its findings for both terrestrial (Cardinale et al. 2006) and marine ecosystems (Worm et al. 2006). The basic conclusion is that diversity influences both the magnitude and stability of ecosystem functioning and the delivery of ecosystem services (Naeem et al. 2009b).

The ecological basis for biodiversity conservation being instrumental to human well-being and poverty reduction is well known and is the foundation for the United Nations Convention on Biological Diversity (CBD), which was established in 1992. Although tangible biodiversity benchmarks were largely left out of the MDGs, its importance in economic development has continued to rise (Adams et al. 2004; Mooney and Mace 2009; Sachs et al. 2009; Walpole et al. 2009). Indeed, 2010, was both the International Year of Biodiversity and the date for nations to meet CBD targets for significantly reducing the loss of biodiversity (Sachs et al. 2009; Walpole et al. 2009).

The reasons that biodiversity conservation is a tool for poverty reduction are relatively straightforward: greater biodiversity means greater stability and efficiency of ecosystem service provisioning, whether the ecosystems are managed (e.g., agroecosystems) or unmanaged (e.g., grasslands and wilderness areas). Because rural poverty is often associated with a population's demand for ecosystem services exceeding what ecosystems can locally provide, using biodiversity to improve the efficiency and stability of ecosystem service provisioning are key arguments for preserving biodiversity.

Biodiversity might appear to contrast with the concepts of the niche and lognormal(ish) distribution in that it improves economic growth rather than constrains it, but retaining biodiversity requires allocating increasing amounts of water, energy, and nutrients for every additional species conserved, which means less for humans. Paradoxically, our vibrant, resilient natural world is one in which there are millions of species, but virtually all of them are rare. Because per-species gains in ecosystem function diminish with each species conserved, it creates the mistaken impression, even among scientists, that conserving biodiversity for its role in ecosystem function should not be a major motivation for biodiversity conservation (e.g., Schwartz et al. 2000; Srivastava and Vellend 2005).

Ecological Poverty and Social Poverty

Collectively, as these three central concepts in ecology show, the very same ecological processes that yield the rich, vibrant, and resilient natural world we know also promote a kind of ecological poverty or extreme resource inequality among species – most species control little in the way of habitat and natural resources. It may seem to us a terrible state of affairs that perhaps 90% of the world's species exist in a state of ecological poverty, but over decadal or centennial time frames, the flow of nutrients, water, and energy throughout ecosystems occurs with greater efficiency and resiliency when many species are involved. Thus, somewhat paradoxically, widespread *ecological* poverty is the reason the world is vibrant and resilient.

Extreme resource inequality among species is due to natural processes, but extreme resource inequality among humans is due to both natural and social processes. This suggests we divide human poverty into two components: *ecological poverty* and *social poverty*. If we define human poverty as extreme deprivation in one or more of the constituents of human well-being, the constituents would include social needs such as security, good social relations, health, freedoms, and choices, which require good social institutions and good governance; and basic ecological needs such as water, nutrients, and energy. Based on this premise, human poverty is clearly a mix of social and natural components. Health, for example, is as much an ecological factor (e.g., naturally occurring mosquitoes carrying *Plasmodium* protists that cause malaria) as it is a social factor (the availability of bed nets and antimalarial drugs). We can describe human poverty by the simple formula,

$$\begin{aligned} \textit{Human poverty} &= \textit{Ecological poverty} + \textit{Social poverty} \\ &\quad + \left(\textit{Ecological poverty} \times \textit{Social poverty}\right), \end{aligned}$$

where ecological poverty is the extent to which the environment does not allow an ecological population to persist (replace itself exactly each generation, as in the concept of the niche described above) and social poverty is where security, social relations, health, freedoms, and choices are below what people desire.

Ecological poverty would be the natural state of humans as large-bodied, long-lived, omnivorous, social mammals. We would be, as we were prior to significant economic development, merely one species among the 10–100 million other species whose realized niche would be constrained to climatically equitable regions with sufficient food and water to sustain us at a level where our population, over the long term, exactly replaced itself in the face of competition, predation, and disease. By today's Promethian standards, we would most likely be considered extremely poor, though Arcadians might argue we were richer by being part of Nature rather than being removed from it.

Ecologically, niche construction is what allowed us to expand beyond anything our basic biology would predict. Our population is enormous, we are distributed just about everywhere on Earth, live longer than most species, consume more natural resources per unit mass than other species, prey upon species just about anywhere they exist, including the ocean floor, and are much more in control of our well-being than other species because we can regulate ecosystem function and ecosystem services including our internal ecosystems such as through the use of antibiotics. Like other species that construct their niche, we have done so by manipulating the biological, physical, and chemical aspects of our habitats, but our technology has allowed us to go much further and become the dominant species on Earth. This unprecedented success, however, is tempered by the enormous environmental peril in which we have placed ourselves (and the entire living world) and by the enormity of human poverty.

The third term in the formulation above, the interaction between the ecological poverty and human poverty, is the focus of much of these volumes. The interaction can be negative, such as when social poverty leads to degradation of habitat and exacerbates ecological poverty, or when ecological poverty, such as reduced watershed outflow, leads to the degradation of social conditions and exacerbates social poverty, precipitating forced migration. The interaction can also be positive, where social poverty improves ecological conditions or ecological poverty improves social conditions.

This clarification of human poverty encompassing both ecological and social poverty suggests a different framework for sustainable development. In the next section, I consider what this framework might look like.

An Ecological Framework for Sustainable Development and Poverty Reduction

In Fig. 19.2, I presented an MDG framework for achieving sustainable development that was consistent with the MDG benchmarks. The skeletal framework, meaning just the main components of the framework and how they are linked, may be written as:

$$\textit{Environmental sustainability} \rightarrow \textit{Social programs} \rightarrow \textit{Human well-being}.$$

This framework suggests that economic development without intergenerational negative externalities (sustainable development) is achieved if social programs to improve human well-being are founded on the principles of not degrading natural capital or lowering the stability of ecosystems (environmental sustainability). However, the role of ecology is ambiguous in this framework, especially because the framework leaves out biodiversity, ecosystem functioning, and ecosystem services.

The Millennium Assessment (MA) provided a different framework that linked biodiversity with ecosystem functioning, which was, in turn, linked to the provisioning of ecosystem services (the benefits we derive from ecosystems) as the foundation for human well-being. We can write the Millennium Assessment (MA) skeletal framework as,

$$\textit{Biodiversity} \rightarrow \textit{Ecosystem function} \rightarrow \textit{Ecosystem services} \rightarrow \textit{Human well-being}.$$

In this framework, the natural sciences are on the left side while the social sciences are on the right side of the scale. The fleshed out version is much more complex (MEA 2005a), containing interactions among these components, but this skeletal framework facilitates comparison with the MDG and the framework based on these volumes that I propose below. In either the skeletal or more complex versions, however, there is a sense that natural scientists would continue their work (e.g., link biodiversity to ecosystem functioning and assume these were translatable to ecosystem services) and then, like a relay race, pass the baton to the social scientists (e.g., assume ecosystem function was translatable to ecosystem services and link ecosystem services to human well-being). In fact, ecosystems are as directly influenced by social factors as human well-being is directly influenced by natural processes.

The chapters in these volumes, in combination with the idea that human poverty is the sum of ecological poverty, social poverty, and the interaction between the two, requires a different framework from the MDG and MA frameworks. The skeletal version I propose that fits these volumes is,

$$\textit{Natural Resources} \rightarrow \textit{Habitat} \rightarrow \textit{Ecosystem goods and services} \\ \rightarrow \textit{Human well-being}.$$

In this framework, which I will call the Ecological Sustainable Development (ESD) framework, biodiversity does not appear explicitly because it consists of the diversity of plants, animals, and microorganisms found within the habitat that govern the flow of natural resources (e.g., energy, nutrients, and water) in the ecosystem. This framework is more accurate than the MA framework because ecosystems do not exist independent of biodiversity nor does biodiversity exist independent of ecosystems. There is, after all, no such thing as biodiversity outside the context of an ecosystem or any such thing as an ecosystem that has no biodiversity. In the ESD framework, ecosystem services are derived from habitats and the magnitude and stability of the delivery of those services are dependent on how we engineer natural resources (e.g., through irrigation or fertilizer addition) or manipulate biodiversity (e.g., replacing native species with domestic species and eradicating pests). The end result is to improve human well-being (and alleviate poverty).

Figure 19.5 provides a fleshed out version of this skeletal ESD framework. Natural resources (shown simply as energy, nutrients, and water) are added (e.g., irrigation, fertilizer, fossil fuels) or extracted as goods and services from habitats (e.g., diverting watershed outflow for drinking, removing vegetation for nutrients as food, and removing dung as biofuels). Goods and services also conventionally

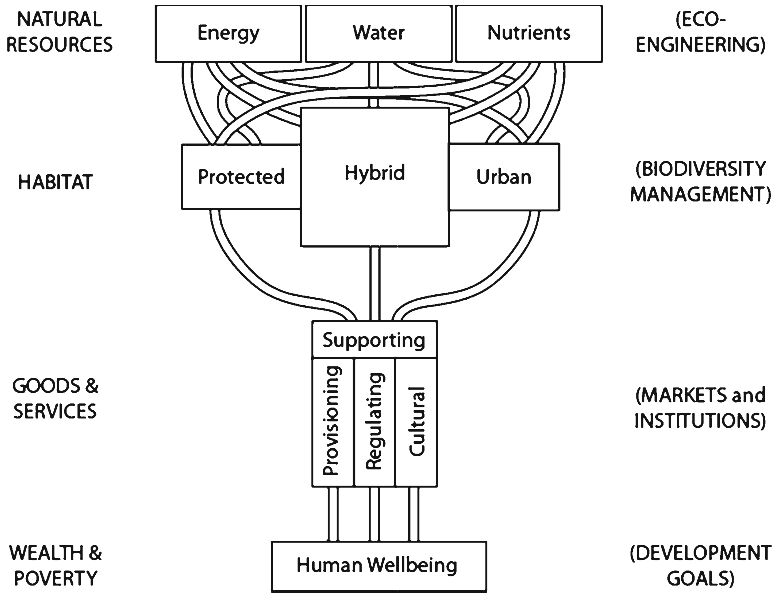

Fig. 19.5 A framework for sustainable development. For the author, the framework that emerges from these volumes links energy, water, and nutrients (the components of natural resources) to human well-being (which scales from being wealthy to being poor) through habitat (ranging from protected to highly managed, such as urban ecosystems, with the majority of habitats, labeled *Hybrid* in the diagram, falling somewhere between the two), and goods and services (following the MA typology of supporting, provisioning, regulating, and cultural services). Interventions are shown on the right. Eco-engineering refers to fertilizer addition, irrigation, or other physical and chemical changes to energy input, hydrology, and biogeochemical processes. Habitat primarily provides opportunities for managing species through conservation, planting, extracting, extirpating, and other changes to the plants, animals, and microorganisms found in the habitat. Depending on how biodiversity is managed, supporting services will secure provisioning, regulating, and cultural services. Markets can be used to manage ecosystem goods and services for which markets exist (e.g., food, timber, recreation) and institutions can provide for management of ecosystem goods and services for which there is no market (e.g., nutrient cycling, spiritual values of biodiversity, naturally occurring pest control). Human well-being is shaped by development goals, which include reducing hunger, improving water, education, security, and health, and building good social relations. See text for further explanation

include biotic services, such as pollination by bees, dung removal by dung beetles, litter decomposition by soil invertebrates, and biocontrol by bats (e.g., Hattenschwiler and Gasser 2005; Larsen et al. 2005; Kremen et al. 2007; Vandermeer et al. 2008; Williams-Guillen et al. 2008; Kasina et al. 2009). Human well-being is the result of all these goods and services being delivered to people; natural resources and biotic services. Disruptions, such as pollution via excess fertilizer use or salinization from irrigation with high mineral content water in low drainage regions or loss of biodiversity, lower the quantity and reliability of the delivery of ecosystem goods and services.

The value of this ESD framework over the MDG and the MA frameworks is that it shows how eco-engineering and management of biodiversity influence ecosystem goods and services. It also shows how social mechanisms, such as markets (e.g., see the chapters on Payments for Ecosystem Services, this volume) can regulate ecosystem services, and how development goals based on human well-being shape the overall strategy.

Ecological Sustainable Development

Energy, water, and nutrients represent natural resources, which exist as abiotic, biotic, or fossil resources. These are manipulated by eco-engineering, which constitutes activities such as mining, irrigation, drilling wells, building dams, or harvesting, burning, planting, breeding, genetically engineering, transporting, or extracting biological resources. The resources themselves may be in the form of biomass found in habitats, such as energy in the form of burnable biofuels such as wood or dung, or as nutrients in the form of edible plants, livestock, bushmeat, and seafood. Below, I will consider the topics covered in these volumes in light of the ESD framework.

Water

The chapters on water by Brown, Randhir and Hawes, and FitzHugh et al. (Chapters 6–9, Volume 1) address sustainable development and water as a natural resource. From these chapters, it is clear that water is a looming environmental crisis that presents enormous challenges in achieving sustainable development and alleviating poverty. Freshwater is only 2.5% of the global water supply and only 0.77% is held in freshwater ecosystems and life on Earth (Postel et al. 1996). Gleick (1998) suggests that basic daily requirements for water by humans are 5 l for drinking, 20 l for sanitation, 15 l for bathing, and 10 l for food preparation, which tallies conveniently to 50 l per person per day. This does not even include the productive uses of water such as for irrigated agriculture and livestock raising. Gleick estimated that 62 countries were not meeting basic water needs for their citizens and over two billion people are currently below the minimum needs. Jackson et al. (2001) estimates were slightly more pessimistic and indicated that already half of terrestrial water is appropriated by people, yet one billion people continue to lack access to sufficient fresh water for drinking and three billion people lack access to sufficient water for sanitation. When a developed nation meets its citizens' water needs, such as the United States, up to 90% of its naturally flowing water may be impounded, which diverts water away from other species.

All three chapters draw attention to the stress that existing demands place on watersheds, how population growth will generate greater demand, and how climate change will generate greater variability, thus increasing the risk of water shortages or floods. Randhir and Dawes focus on the watershed as the appropriate scale and consider the importance of ecological and climatic processes in watersheds that govern outflow volume, quality, and resilience, as well as, the complex social issues that concern watershed management. They reflect upon the need to address the lack of natural capital markets, restore watersheds, and develop holistic and participatory management strategies. Brown reviews social mechanisms, such as water markets, conservation incentives, and investment in hydrologic management for managing watersheds to meet future needs and alleviate poverty. Underlying environmental issues surrounding water are covered in greater detail by FitzHugh et al.'s chapter, yet the ecology of human water needs in comparison to the water needs of habitats and their biodiversity is addressed only in the context of being careful not to draw so much water that we leave too little for ecosystems to perform.

These reflections about water are not strictly ecological, but a mix of social and environmental concerns. From an ecological perspective, what is of interest is the enormous per-capita and population needs for water that reflect the human-constructed niche. The level of water consumption by our species is vastly more than other mammals because other species do not use water for sanitation, bathing, food preparation, agriculture, recreation, and industry. To illustrate, a medium-sized elephant of 4,000 kg (100 times larger than a medium-sized human of 40 kg) drinks about 225 l a day, a ratio of 0.055 l water per kg of biomass. An average human of 40 kg needs approximately 5 l of water a day for drinking, which yields a ratio of 0.125 l water per kg of biomass, a 2.27-fold greater need than that of an elephant. If we consider all human water needs, the ratio is 1.25 l of water per kg of biomass, or 23 times what an elephant needs. Being smaller than elephants, we have higher metabolic rates and would naturally need more water per kilogram, so perhaps we should compare ourselves to smaller mammals. Going to the other extreme, a 30 g mouse's consumption ratio is about 1.0 l of water per kg of biomass, ten times that of human needs for drinking. The higher metabolic rate of a mouse clearly calls for more water per unit biomass. Compared to total human water needs, however, even at an enormously high metabolic rate, a mouse needs 20% less than a human since, again, it does not need water for activities such as bathing, sanitation, and food preparation.

Humans, compared to other terrestrial species, no matter what their size, consume a lot of water. If we include what we use for agriculture, given our food primarily comes from agriculture, our consumption is even greater. Postel et al. (1996) estimated that in addition to humans appropriating about half of terrestrial rainfall that runs off into streams, rivers, and lakes for irrigation, industry, and drinking, we appropriate about 26% of terrestrial rain that evaporates either directly into the atmosphere or through plants as they respire (evapotranspiration), which means less and less water is available to meet human needs.

Given that biologically we greatly exceed water use in proportion to other species, what would an ecologically sustainable development strategy for water use be? The idea that human poverty is associated with the interaction between ecological poverty and social poverty is captured by Chap. 8 in Vol. 1 by Fitzhugh et al. who state "Improving the balance between human and ecosystem needs for water is critical because of the degree to which valuable ecosystem services may be damaged by water development that doesn't take into account ecosystem needs." However, it has proved challenging to fully integrate this perspective into practice and policies. For example, Richter et al. (2003) propose an "ecologically sustainable water management" framework for providing urban needs for water while sustaining the other goods and service deliveries provided by the ecosystems that generate water supplies (see Chap. 8 in Vol. 1 for a discussion of this framework). It consists of six steps that involve assessments of human water needs, water needs of associated ecosystems to ensure delivery of goods and services, identifying "incompatibilities" between these two needs, and then building an adaptive management plan based on solutions that resolve the incompatibilities. This sustainable management framework for water, however, is one that begins with a focus on social processes by equating the problem with the way human activities alter watershed outflow, searching for solutions by fostering collaborative dialogue among actors and conducting management experiments, then devising adaptive management plans based on iterations of these processes.

In contrast, an ESD strategy would begin with natural resources (top of Fig. 19.5) and assess how much water consumption humans would appropriate as a biological species that is interacting with other species and a functioning part of their ecosystem (realized water niche) and how much they would appropriate to meet minimum social standards (constructed water niche). The difference between the two approaches is the target for minimal development. Water consumption within the limits of the realized water niche would support each person replacing themselves at each generation, though they would have higher reproductive output when water was in excess and lower when water was in short supply. Further, water appropriations would be tied to nutrient and energy appropriations, while ensuring that other species are not adversely affected so that the magnitude and stability of ecosystem function is retained. This means ensuring biodiversity is conserved and retaining the lognormal(ish) distribution of species.

An ESD strategy is one that would ensure that biodiversity, species distributions, and the magnitude and stability of ecosystem functions are either not impacted by human appropriations of water when they exceed their realized niche, or that humans compensate for lost ecosystem functioning. The human-constructed water niche must necessarily and substantially exceed the realized water niche since socially acceptable minimal per-capita water needs are well above basic biological needs. Eco-engineering, biodiversity management, and social mechanisms and institutions, such as markets, education, gender equality, and other activities shaped by social benchmarks for human well-being, are designed to achieve this balance of water appropriation with the delivery of other ecosystem goods and services.

While based on ecological principles to parse the problem and devise solutions, the ESD framework has seeming disadvantages in that it requires much information on biodiversity and ecosystem processes, and is likely to require rather complex ecological and ecohydrological modeling. An ESD framework is also more difficult to explain to actors in the system; it may seem abstract, academic, and counter to popular conceptions of "Nature as our friend." The ESD framework has built into it the idea that even the minimal human needs typical of extreme poverty may be in excess of what is ecologically sound, which runs the risk of making it seem as if an ESD strategy pits the needs of humans against the needs of Nature. ESD obviously does not pit people against nature; however, it simply seeks ecological baselines to build a science-based strategy to achieve environmental sustainability and alleviate poverty, rather than the other way round.

Nutrients and Energy

Other chapters in these volumes similarly capture the importance of the interaction between ecological poverty and social poverty but are not primarily ecological. Smukler et al., (Chap. 3, Vol. 1) for example, note that the goal of sustainable development should be to maximize ecosystem services not just for subsistence, but for rising out of poverty, and they provide examples of where necessary inputs for food production should not erode other services. To fit within the ESD framework, however, one would have to again ask what human nutrient, energy, and water use would be if humans were simply a biological species integrated into a functioning ecosystem and what population density could be sustained in the absence of interventions (which are forms of niche construction). Once this is assessed, one can determine what levels of nutrient, energy, and water appropriation for the existing or future population will achieve those desires without impairing ecosystem functioning. Smukler et al. do highlight enhancing ecosystem service provisioning via habitat modifications which is congruent with the ESD.

The topic of energy is treated similarly. Doll (Chap. 16, Vol. 1) provides a framework for sustainable development of energy resources that contributes to poverty reduction, recognizing that energy use is something that allows humans to expand beyond their biophysical boundaries (niche construction), which is clearly an ecological perspective. Current strategies for sustainable energy use, however, remain dominated by social (e.g., markets), health (e.g., the respiratory diseases associated with smoke inhalation), and engineering considerations (e.g., storage and cook stove design), rather than ecological processes. Ganz et al. (Chap. 17, Vol. 1) on energy similarly recognize that what is needed is primarily a social "energy-poverty reduction" approach. This approach would consider the entire biomass-fuel (i.e., the burning of wood, charcoal, dung, agricultural residues, and other biological materials) supply chain and develop strategies that benefit the poor as well as biodiversity (more in the context of protecting species rather than securing ecosystem function). The first part of their three-part framework calls for assessing community dependencies on ecosystem services and what options exist for poverty reduction through improved provisioning of biomass-fuels while the remainder of the framework is largely social (e.g., markets, transportation, finance, and politics).

In slight contrast, an ESD approach to energy would begin by considering human consumption of biomass for light, heat, cooking, and other uses, as solely part of our constructed niche. Basic energy needs are largely governed by social issues – are the people living in a habitat where heating is essential, are they preferentially eating food that requires considerable cooking as opposed to directly consumable foods, are they lighting homes for work, education, entertainment, or other social activities? Like water for bathing, food preparation, recreation and sanitation, using biomass for heating, light, and cooking are non-ecological activities and are part of our constructed niche.

Health, Finance, Gender, Education, Conservation, and Other Social Dimensions

Other chapters call attention to additional dimensions, challenges, and solutions to sustainable development and poverty reduction. The science of ecology is not the starting point in these chapters by design, to illustrate the complexity of these issues. For example, Milder et al. (Chap. 5, Vol. 1) note that one must work at the landscape level where tradeoffs among multiple households and multiple development plans occur, as their examples of Rio Copan in Honduras and Kijabe in Kenya demonstrate. Jenkins (Chap. 10, Vol. 2) discuss finance mechanisms such as payment for ecosystem services (PES), but note that the current challenge is that a large fraction of the world's poor live in ecosystems where biological diversity is high, but are moving to peri-urban or urban regions, leaving behind landscapes that are being converted to production which provide a limited number of ecosystem services. Fisher (Chap. 13, Vol. 2), using the example of the Eastern Arc Mountains of Tanzania to point out that the consumption of one beneficiary of an ecosystem service at the local scale can impair the livelihood of others at higher scales. Thus, PES schemes need to consider land tenure, land ownership, tradeoffs among different livelihoods, scale, and benefit sharing, all of which are important social issues, but not primarily ecological.

The ecology of PES concerns primarily tradeoffs, where, for example, water extraction may impact wildlife, conversion to agriculture may lead to the loss of medicinal plants, or the use of pesticides harms pollinators, natural biological control, and wildlife. Market mechanisms should enhance synergies and minimize or eliminate harmful tradeoffs. An ESD approach would begin, again, with understanding the ecological boundaries and constraints of human integration into ecosystem processes, how far outside those constraints people wish to live to achieve their goals for well-being, and what market mechanisms would support the necessary niche construction to attain their goals for well-being.

Concerning gender, Gutierrez-Montes et al. (Chap. 4, Vol. 2) provide further detail by emphasizing differences in decision making, learning, knowledge transfer, and other gender differences that influence development – one of the better known examples being that aid to women improves child health better than aid delivered to men. These differences vary from place to place, but again, the focus is not on ecological principles so much as how the outcome is governed by complex linkages among social and ecological factors. An ESD approach would include gender differences in understanding how shifts in the allocation of water, energy, and nutrients affect the well-being of both genders. Schroeder (1993), for example, illustrated how the establishment and irrigation of community gardens during a drought in Gambia, West Africa, benefitted women, but market incentives for orchard products led men to plant orchards in these gardens, which shaded the fruits and vegetables leading to a reversal in gains for women. An ecological perspective would have identified the incompatibility of orchards and gardens due to one shading the other. Rocheleau (1991), concluded from her studies of local responses to a drought in Kenya, East Africa, that, "If research results in documentation and discussion of gendered ethno-science at community level, then rural women may make more informed choices about which species, which skills, and which visions of nature and society to carry into the shaping of their emerging ecological and economic futures." The same can be said about ESD, which should promote a gendered ecological science when documenting the allocation of water, energy, and nutrients among people and their biota within their ecosystems.

Ecological science could play a prominent role in education, but what is needed, as the chapter by Sears and Steward (Chap.3, Vol.2) illustrates, is a hybrid approach in which synergies between local ecological knowledge and the ecological science are promoted. They suggest that education in rural communities should strongly emphasize ecological knowledge as this is what will empower local people, foster better self-governance of, and access to resources. Ecological knowledge is experience, such as how, what, where, and when to grow and harvest plants for food, fuel, medicine, and food, or how to use the natural history of species to inform hunting, fishing, and the management of species that serve as natural resources. But experiences with Nature go both ways, sometimes suggesting that decoupling from Nature can free us from its vagaries (Nature as foe) or conversely that coupling, such as through PES programs, can improve well-being (Nature as ally). Sears and Steward see little role for formal ecological science in education, arguing that, "The stringent rules of reductionist scientific inquiry, isolating causal factors and mechanisms, render much of the research information too narrow to be effectively applied in complex ecological systems. The utility of the scientific and technical information is only as good as its relevance to local people and conditions." Because both local ecological knowledge and the science of ecology can be too broad or too narrow to be of utility, finding their common ground is important. Finding where each makes unique contributions and resolving incompatibilities are also important endeavors. I would also argue that utility is not the sole basis for local ecological knowledge or ecological science given that what is useful knowledge today may not be useful tomorrow, and what is not useful today may be useful tomorrow.

The role of ecological science is clearly integrated into the health chapters. Hess and Myers (Chap. 12, Vol. 1) define the issue of climate change and health from a clearly ecological perspective; "As humans evolved, we were actors on the global ecological stage no different from any other organisms with whom we shared the planet. However, over the last 2–300 years, we have increasingly monopolized the stage, becoming the playwright as well as the dominant actor." Likewise, Keesing and Ostfeld (Chap. 13, Vol. 1) point to the necessity of including disease ecology in preemptive medicine. Levy et al. (Chap. 14, Vol. 1) clearly anchor ecosystem services to the ecosystems and the diversity of organisms that generate them. Myers (Chap. 11) also consider how emerging, re-emerging, and changing patterns in disease are attributable to human alterations of basic ecological factors such as biophysical properties of ecosystems and landscapes, genetics of pathogens, life histories of disease and vector organisms, and biodiversity.

Finally, the idea that biodiversity conservation should be instrumental to all sustainable development strategies, presents itself most often as an argument for protecting species for their cultural and genetic resource values. While the importance of biodiversity in ecosystem functioning is addressed throughout these volumes, a strong emphasis on conservation of taxonomic diversity remains pervasive. The former ideology is most helpful for promoting the value of biodiversity in all its dimensions and as a means for managing the magnitude and stability of ecosystem services. This common confusion about biodiversity is picked up by developing nations as evidenced in their Poverty Reduction Strategy Papers (PRSPs), filed by 65 developing countries with the International Monetary Fund. Biodiversity is only mentioned once in the correct ecological context in these PRSPs. That is, out of a total of 12,366 pages of reports from 65 countries, biodiversity is only mentioned on 158 pages in 51 reports (14 made no mention of it at all), and of the nations that included the term biodiversity in their reports (which was rare even for those that did, with 45% using the term only once), the concern was reducing the loss of taxonomic diversity. Only Lesotho's PRSP captured biodiversity's value and it is worth quoting from their report:

> A powerful indicator of severe environmental decline is loss of bio-diversity. In Lesotho this manifests itself through changes in flora and fauna, and by loss or decline of habitats such as grasslands, marshes, bogs and reed meadows. The direct consequences of loss of biodiversity on the rural poor are multi-faceted. One is the impact on fuel woods, referred to above. Equally worrying is the decline in plants of medicinal value, which are used extensively by poor households unable to afford modern health services. For those whose livelihoods depend on livestock, the replacement of grasses by invading unpalatable shrubs is a chief concern. As rangelands lose their diversity their quality declines, as does the productivity of the livestock and, hence, the income of their owners.

For ecology to be fully embedded in poverty reduction efforts, biodiversity must be framed within the proper ecological context including a clear understanding of biodiversity's functional role in the provisioning of ecosystem services.

Conclusion: Toward an Ecological Sustainable Development

Jenkins (Chap. 10, Vol. 2) (Chap. 10) conclude, and most would agree, that sustainable development remains elusive in spite of the first Earth Summit being held in 1992 and the idea of sustainable development emerging in 1987 with the publication of the Brundtland report (WCED 1987). One reason for this is the lack of a clear framework based on a well-developed scientific foundation. These volumes

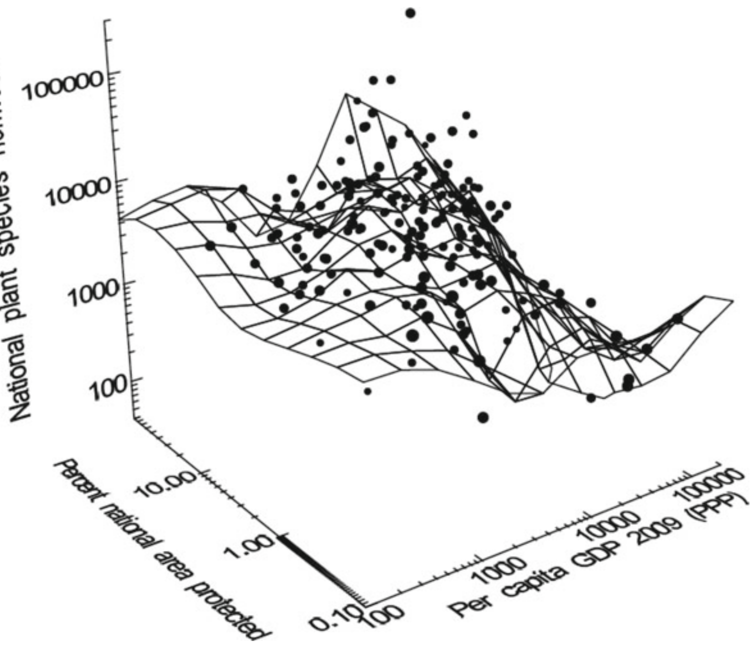

Fig. 19.6 Biodiversity, conservation, and wealth. Data derived from the World Resources Institute (plant species richness and percent national area protected) and the United States Central Intelligence Agency World Fact Book. *Points* represent values for single nations. The *grid* represents a Lowess smoother with tension set at 0.5

have risen to the challenge. My purpose was to address the role of ecology, both in general and in specific reference to the topics in these volumes.

In spite of international commitments to pursue sustainable development that would link development with biodiversity conservation, there is little evidence that they have been coupled. If we plot wealth as a proxy for development, area set aside for protection as a proxy for understanding the importance of ecology in one's nation, and plant species richness as a proxy for biological diversity (even though species richness is not the best measure of biodiversity), we find no significant associations (i.e., no significant Pearson correlations among log-transformed variables) (Fig. 19.6). We do see that countries of intermediate wealth (not the poorest) have the highest diversity, but that the amount of protected area is independent of either wealth or diversity. Now, in the third decade since global the commitment to sustainable development, and in the second decade since we agreed to combat climate change (UNFCC) and biodiversity loss (the CBD), it is time to work towards linking wealth with environment through sustainable development.

This lack of progress is partly attributable to a lack of a well-developed scientific basis for sustainable development, but building this scientific basis is not easy. Ecology as a science is a powerful, predictive science about the relationship between organisms and their environment, thus, if it is to be integrated into sustainable development, it should be used to analyze how humans, as organisms, relate to their environment. The thesis I have presented here suggests that ecological sustainable development (ESD) would be based on understanding how humans as biological entities are a species like any other, and contribute to the magnitudes and stability of ecosystem functions, which concern the flow of nutrients, energy, and water.

Humans, like all biological species, have fundamental, realized, and constructed niches, but our success is primarily through niche construction. Development concerns our constructed niche and how much it improves human well-being beyond what ecological processes would provide. Clearly, by niche construction, the potential for improving human well-being is ultimately limited by biophysical boundaries. The closer we come to those boundaries, however, the greater income inequality seems to get and the worse our environment gets, putting not only poor and vulnerable people at greater risk, but threatening the survival of humanity as a whole if crossing those boundaries destabilizes Earth systems (Walker et al. 2009).

If we divide human poverty into the sum of ecological poverty and social poverty, as defined above, it becomes evident that current challenges of alleviating poverty arise from a history of development that did not take into account the interaction between ecological poverty and social poverty, as supported by the chapters on hunger, water resources, health, energy, and climate change. The chapters on societal paradigms, population, financing, and ecosystem-based management outline solutions that recognize the interaction between ecological poverty and social poverty. Recognizing the interaction between ecological poverty and social poverty is valuable because progress in achieving the MDG's and alleviating poverty remains elusive. The MDGs are the most visible manifestation of the continuing primacy of social endeavors over ecological ones, in spite of the fact that its partner, the MA, pointed squarely to the need to rethink the way we work to better human well-being.

The Millennium Assessment, the Millennium Development Goals, and the Millennium Declaration will stand as tributes to humanity's goal at the dawn of this century to change its course and secure its survival against the threats the modern world posed. Poverty is only one of many looming global-scale environmental challenges that we will fail to meet without the proper interactive institutions (Walker et al. 2009). I, like the authors in this book, am optimistic that these goals will be achieved, and part of that optimism comes from the solutions that we can see once science is applied to the problems of sustainable development.

In our endeavors, however, we should be clear about the natural world and its processes and what ecology tells us. Ecology is neither savior nor demon. As Darwin noted (cited in Gleick 1998), the great sin of poverty is if it is the result of our institutions, not if it is an outcome of the laws of nature.

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Chapter 20
Conclusion: Integrating Ecology and Poverty Reduction

Jane Carter Ingram, Fabrice DeClerck, and Cristina Rumbaitis del Rio

Conclusion

As discussed throughout the chapters of the two volumes comprising this series on *Integrating Ecology and Poverty Reduction*, in recent years an increasing amount of global attention has focused on the role of the natural environment in contributing to poverty reduction (McNeely and Scherr 2003; Ash and Jenkins 2007; World Bank 2007; Tekelenburg et al. 2009; Chivian and Bernstein 2008; Galizzi and Herklotz 2008). These volumes complement and build upon this growing body of work, but look specifically at the ecological dimensions of multiple development challenges related to rural poverty and the ways in which ecological science can be applied to address some of these challenges. The majority of the chapters comprising the two volumes have focused on these issues in poor, rural areas, where approximately 70% of the developing world's 1.4 billion extremely poor people live (IFAD 2011). In these places, direct dependence on nature for subsistence is often high, and access to social services, markets, and economic options is often limited. However, several chapters in these volumes, such as the chapter on water supply planning by Fitzhugh et al. (Chap. 8, Vol. 1) and population by Marcotullio et al. (Chap. 8, Vol. 2) suggest that experiences gained from the successes and failures of natural resource management in developed countries and nuanced understandings of how urbanization influences poverty and the environment may be useful for informing decisions and polices in rural areas of developing countries.

Several recurrent messages have surfaced from this body of work that illustrate the importance of ecological science for understanding challenges related to poverty reduction and the enabling conditions that influence the effective application of ecological science and tools for addressing those challenges. Broadly, these messages

J.C. Ingram (✉)
Wildlife Conservation Society, 2300 Southern Boulevard, Bronx, NY 10460, USA
e-mail: cingram@wcs.org

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can be encapsulated by three overarching themes: the challenges of preventing and managing complex trade-offs; the importance of social and economic contexts for determining the application and utility of ecological science; and the paradigm shifts that will be required to effectively integrate ecology into development practice and planning. These themes are certainly not new ones within development or environmental fields; however, the collective chapters in these volumes focus on these themes through the lens of ecology, as it relates to multiple development challenges and potential solutions.

Understanding and Managing Complex Trade-Offs

These chapters encourage a careful consideration of potential ecological and social trade-offs that may result from projects aimed at poverty reduction and conservation. The importance of addressing trade-offs lies in the risks of losing biodiversity and critical ecological functions; crossing thresholds, beyond which it may be difficult to restore ecosystem services; and causing unintended consequences on poor communities, who may be left more vulnerable as a result of even the most well-intended actions. Others have also recognized the importance of these issues and, consequently, efforts to identify and model trade-offs to inform decision making have grown in recent years (Tallis and Polasky 2009). Despite this increasing attention on trade-offs and the proliferation of new tools to address them, as these chapters reveal, it remains a challenge to identify and manage them, especially, when they occur across interconnected social and ecological systems and across a range of spatial and temporal scales. These chapters demonstrate these challenges in the context of several types of trade-offs: trade-offs between and within development goals; spatial and temporal tradeoffs; economic trade-offs; trade-offs among natural resource user groups; and trade-offs between technology and nature, as a provider of ecosystem services that contribute to poverty reduction.

Many of the chapters in Vol.1., *Integrating Ecology and Poverty Reduction: Ecological Dimensions*, which focus on hunger, water, energy, health, climate change, and disasters, explore the difficulty of navigating the temporal and spatial trade-offs associated with managing farms, watersheds, landscapes, and coastal zones for the multiple needs of the poor, such as food, nutrition, water, fuel, health, and physical security. Historically, development efforts have often focused on maximizing one ecosystem service, such as food production, at the expense of other services that are critical for the livelihoods of rural populations in the long term, as discussed by Smuckler et al. with respect to food production, Ganz et al. with respect to energy production, and Randhir and Hawes with respect to watershed management (Chaps. 3, 7, and 17, Vol. 1). Even within single development sectors, trade-offs have occurred: for example, in some cases, total food production has been enhanced at the expense of overall nutrition (Chap. 4, Vol. 1). Collectively, these chapters emphasize that projects designed to reduce rural poverty should consider the many ecosystem services that are important to rural households and how interventions aimed at improving one service may impact other key components of people's livelihoods and well-being. Such a holistic way of thinking may be facilitated by a social-ecological systems approach that includes an appreciation of the connectedness of natural, social, and economic systems across multiple spatial and temporal scales. Ecologists are equipped to think this way and can contribute to such approaches by fostering understanding on the ecological relationships among ecosystem services at multiple spatial and temporal scales (Carpenter et al. 2009; Bennett et al. 2009). Tools, models, and frameworks that can elucidate the social, economic, and ecological trade-offs occurring across different spatial and temporal scales as a result of natural resource management practices and policies are critical for informing decision making. Much work in this area is currently under way by groups such as the Natural Capital Project (NatCap 2007).

The chapters in this volume, *Integrating Ecology and Poverty Reduction: The Application of Ecology in Development Solutions*, address how trade-offs related to balancing conservation and poverty reduction can be managed, negotiated, and, in some cases, avoided through education, gender equality, demography, innovative financing, and ecosystem governance. In some cases, mechanisms such as payments for ecosystem services (PES) can address potential economic trade-offs associated with implementing environmentally sustainable practices by paying people, in cash or in kind, to protect or enhance ecosystem services, as described by Jenkins (Chap. 10, this volume) and as demonstrated by Sachedina and Nelson (Chap. 12, this volume) in Tanzania. While payments through PES (or PES-like mechanisms) primarily aim to incentivize conservation and enhancement of ecosystem services, they may also provide an important source of income or other resources to poor, rural communities, where few other markets or income-generating opportunities exist. However, as Fisher (Chap. 13, this volume) discusses, it can be difficult to achieve multiple policy goals, such as conservation and poverty reduction, through a single instrument such as PES, without compromising success in one or both of the goals. For example, the distribution, stocks, and flows of ecosystem services may vary across a landscape, as Estrada and DeClerck describe (Chap. 14, this volume), and do not coincide necessarily with areas on a landscape that are the highest priorities for poverty reduction. Thus, some trade-offs in the level of ecosystem services conserved or rural income generated, for example, may be inevitable if a tool like PES is being applied to achieve conservation and contribute to development goals. While tools like PES can be useful for addressing economic trade-offs associated with conservation, some ecosystem services currently do not have value in traditional markets (deGroot et al. 2010) such as, the health benefits provided by ecosystems as described in the chapters by Meyers; Keesing and Ostfeld; and Levi et al. (Chaps. 11, 13 and 14, Vol. 1) and protection from extreme events, as explored in chapters on climate change and disasters (Chaps. 20–23, Vol. 1). Thus, it is important to quantify and articulate potential trade-offs involving a range of ecosystem services, some of which may be very localized and non-monetary in value, in ways that are clear and meaningful to affected stakeholders. To do this, ecologists will need to work in combination with professionals from other disciplines, policy makers and local communities to identify the suite of ecosystem services that are important to different user groups; to generate knowledge on how services are provided ecologically; and to develop guidance on how to manage and monitor the condition of key services important at local, national and international scales.

Trade-offs may be inevitable in landscapes or seascapes where multiple stakeholders have competing and conflicting uses for the ecosystem services available. In many cases, it will be necessary to negotiate compromises that require some or all stakeholders to yield their ideal patterns of ecosystem service “consumption” to ensure that a wide range of a landscape or seascape’s biodiversity and ecological functions are conserved in the long term. These negotiations must be grounded in the reality that the diverse benefits provided by nature may accrue at very different spatial or temporal scales to different groups of people. In cases where nature cannot provide services in the quantity or at the scale desired, production or consumption of one service may be increased through management practices and/or technology. However, this could result in a change in the provisioning of other ecosystem services, which must be explored carefully and communicated clearly with all affected people. As the chapters on the governance of ecosystems (Chaps. 16–18, Vol. 2) discuss, balancing competing claims for natural resources can be aided significantly by credible science to support decision makers in developing natural resource management plans and identifying science-based targets to monitor progress toward those plans.

One common decision that often involves trade-offs is the choice between using engineered or technological providers of services rather than relying on ecosystems to supply those services. For example, a seawall may be able to provide highly predictable, measurable protection against coastal storms, but may degrade coastal ecosystems that provide a range of critical services for local communities. Thus, in some cases holistic vulnerability of poor, coastal communities may be reduced more effectively in the long-term through conserving coastal ecosystems, such as mangroves, that provide storm protection services, in addition to other important services, such as food production from fisheries, fuelwood, construction materials, climate regulation through carbon storage, and biodiversity for tourism and cultural purposes (see Chap. 22, Vol. 1). Shifting toward more holistic approaches and away from static engineered approaches may be feasible and preferable in certain contexts, but will require research to quantify the social, economic, and ecological advantages of nature-based versus traditional, built approaches (Chaps. 9 and 22, Vol. 1). In cases where ecological thresholds have already been crossed and the ecosystem may no longer be able to provide key services, technological substitutes for services may be the only option. As Smuckler et al. discuss (Chap. 3, Vol. 1), inorganic fertilizers may be needed in some cases to “jump-start” extremely degraded soils and rebuild the productivity of the system again, even if they are not desirable as a singular approach in the long term. However, the authors note that financial and ecological trade-offs may still occur in such dire situations, even when few other alternatives are available. While ecological science and management can help avoid trade-offs and their negative consequences in some cases, in other cases, trade-offs may be inevitable and ecological science can merely contribute to identifying what they might be so that people can plan accordingly.

The Importance of Social and Economic Context for Applying Ecological Science and Tools

Many of the chapters in these volumes emphasize that prevailing social and economic conditions must be considered in order to understand how humans utilize and value ecosystems, so that ecological science and tools can be more effectively applied and used. For example, multiple, interacting scales of governance influence the use of natural resources, such as local rules, national policies, and international treaties. To be effective, political processes and institutions of influence should be compatible with the scale of the ecological processes and services they are intending to conserve, although this does not always happen, as McClennen points out with respect to fisheries management and as Holland demonstrates with respect to protected areas (Chaps. 16 and 18, this volume). Furthermore, relationships that may influence decisions about natural resource management change across spatial and temporal scales. For example, Bremner et al. (Chap. 6, this volume) state that relationships between population, poverty, and environmental degradation that exist at a national scale may be difficult to identify at local scales. Thus, an ongoing challenge for ecologists is to translate observations and findings related to important ecological processes and patterns into information that is meaningful to stakeholders and resonates at scales of relevance for decision making. To do this, information must be obtained, collected, and applied within the context of how prevailing environmental, social, cultural, and economic conditions influence choices about natural resource use or the results may be irrelevant for addressing problems, as discussed in Chaps. 16, 17 and 18 with respect to energy challenges in Vol. 1 and Chap. 11 with respect to carbon projects in this volume. A failure to do this could lead to perverse policies that exacerbate poverty and environmental degradation in the long-term rather than improving them. Collectively, these chapters emphasize that for ecological science and tools to be useful, it is crucial for ecologists to engage with local people, natural resource users and groups who are addressing social, governance, and economic challenges in the places where they work to better understand the broader framework within which environmental problems are situated, and the spatial and temporal scales at which information will be useful to support decision making over natural resources.

Fostering a New Paradigm

Throughout these volumes, authors have called for a new way of thinking about the relationships between people and nature if ecological science and tools are to be more regularly and seamlessly integrated into development practice and policies. In Naeem's language (Chap. 19, this volume), paradigms such as "nature is our friend" or "nature is our foe" have been critical for shaping economic and environmental debates throughout history and continue to dominant many discussions regarding conservation, sustainable development, and poverty reduction. However, such simplistic views of the way in which people interact with their environment have contributed little to our ability to conserve ecosystems and reduce poverty of the world's poorest people. In contrast, Naeem (Chap.19, this volume) uses several ecological concepts to illustrate the nature of poverty and, based on this understanding, demonstrates how human poverty consists of the interactions between ecological poverty and social poverty. Other chapters in these volumes encourage new thinking about the impact of human population on natural resources (Chaps. 6 and 7, this volume), the environmental benefits of urbanization (Chap. 8, this volume), the role of women in natural resource management (Chap. 4, this volume) and the importance of rural education for addressing poverty and ecosystem degradation (Chap. 3, this volume). Many of these chapters have promoted holistic, interdisciplinary frameworks for illustrating the relationships between poverty and the natural environment that require a nuanced understanding of social and ecological interactions (for example, Chaps. 7 and 17, Vol. 1; and Chap. 6, this volume). Naeem presents an example of this with the Ecological Sustainable Development framework (Chap. 19, this Volume). Such guiding frameworks and principles that promote consilience across disciplines will be helpful for revisiting deeply held ideas regarding the ways in which humans use and are embedded in ecological systems and will be required to design and implement novel, innovative, and lasting solutions to environmental degradation and poverty reduction in the coming decades of the twenty-first century. As Naeem states in this volume, *ecologically* sustainable development should be our goal and, as such, must be founded on a better understanding of how humans interact with and affect an ecosystem like any other species, by contributing to ecosystem functions that influence the flow of nutrients, energy, and water, for example.

Summary

Ecological systems are a source of the many diverse services that make life on earth possible. In rural areas of many developing countries, the major geographical focus of these volumes, it can be difficult to access affordable and/or appropriate substitutes for many of the services provided by nature such as food production; pest control; soil fertility; water for drinking, bathing, and cooking; energy for cooking, heating, and electricity; shelter; disaster protection; and medicine. If multiple development challenges are to be solved simultaneously and sustainably, solutions for poverty reduction must not undermine the persistence of species and ecological functions that generate the many ecosystem services that rural communities currently depend upon for their livelihoods and well-being and that will be critical for future generations. These volumes have attempted to address the ecological nature of some of the major challenges related to poverty reduction and the ways in which ecological science can be more effectively leveraged within, political, economic, and cultural processes mechanisms, and institutions to address some of these problems. We hope that the chapters included within these volumes have catalyzed discussions and ideas that will ultimately foster a world in which extreme poverty is a concept of the past and ecological sustainability is a guiding principle of the future.

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