ECOAGRICULTURE AND BIODIVERSITY CONSERVATION: CONCEPTS, APPROACHES, AND EVIDENCE FROM NORTHERN LATIN AMERICA

A Dissertation

Presented to the Faculty of the Graduate School

of Cornell University

in Partial Fulfillment of the Requirements for the Degree of

Doctor of Philosophy

by

Jeffrey C. Milder

February 2010

ECOAGRICULTURE AND BIODIVERSITY CONSERVATION: CONCEPTS, APPROACHES, AND EVIDENCE FROM NORTHERN LATIN AMERICA

Jeffrey C. Milder, PhD
Cornell University 2010

Ecoagriculture is an approach to landscape management that simultaneously advances agricultural production, conservation of biodiversity and ecosystem services, and sustainable rural livelihoods. In this dissertation, I address gaps in the science and application of ecoagriculture by reviewing key antecedents of this approach, enumerating a framework for ecoagriculture practice, and evaluating ecoagriculture outcomes—particularly biodiversity conservation—in pasture-dominated landscapes in northern Latin America.

Ecoagriculture builds upon multiple disciplines and traditions of practice including landscape and ecosystem approaches, adaptive management, and integrated natural resource management. These antecedents inform the Landscape Measures approach, a structure for guiding multi-stakeholder planning, decision-making, and monitoring of ecoagriculture objectives at a landscape scale. Pilot applications of this approach in Honduras and Kenya suggest its value for promoting landscape multifunctionality by documenting landscape dynamics, fostering constructive dialogue among diverse stakeholders, and improving empirical bases for collective decision-making.

Previous research has indicated that a substantial proportion of native species may persist in Neotropical agricultural mosaics. However, the consistency of these findings—and thus their applicability to unstudied locales—remains unknown. I conducted a coordinated multi-site study to evaluate the consistency of relationships between bird and butterfly assemblage characteristics and land use practices in four landscapes spanning four countries in northern Latin America: Colombia, Costa Rica, Honduras, and Nicaragua. Bird and butterfly species richness and abundance were significantly related to land use across all four landscapes, and were generally positively correlated with degree of tree cover. In the Honduras landscape, I also investigated the relative influence of land use, vegetation, and landscape composition and structure on bird and butterfly assemblages. These assemblages were more strongly related to land use and plot-scale habitat features than to gradients of landscape composition and structure. Across all landscapes, habitat heterogeneity fostered high levels of beta diversity and high overall species richness, but only a modest presence of forest-dependent and high conservation value species.

The findings underscore the importance of considering multiple conservation values when evaluating conservation outcomes in ecoagriculture landscapes. These values include protecting species of intermediate conservation concern and species that support rural livelihoods—not only globally threatened biodiversity.

BIOGRAPHICAL SKETCH

Jeffrey C. Milder is an ecologist and land use planner with diverse experience in research and professional practice. Prior to coming to Cornell, from 1997-2003, he founded and managed the community planning practice at Daylor Consulting Group, a Massachusetts-based design firm. While at Cornell, he has focused his research and teaching on the impacts and opportunities of land use models that integrate conservation and human use, including agricultural mosaics in the Neotropics and conservation-oriented real estate projects in the United States. In conjunction with his dissertation research, Mr. Milder held the position of Research Associate at Ecoagriculture Partners (a Washington D.C.-based nongovernmental organization), where he applied his research approach to key ecoagriculture issues including payment for ecosystem services and biofuel production. He is co-author, with Dan L. Perlman, of *Practical Ecology for Planners, Developers, and Citizens* (Island Press, 2005). Mr. Milder holds a Bachelor of Arts in Earth and Planetary Sciences, *magna cum laude*, from Harvard University and a Master of Science in Natural Resources from Cornell University.

To my parents, Gail and Robert:

you helped me discover a passion for nature, love for learning, and instinct for helping

that inspired these pages.

ACKNOWLEDGMENTS

This project could not have succeeded without the generous support of many people. First and foremost, I thank my family and friends, and especially my wife, Nina, for enduring and cheering me throughout the long process of my PhD.

I thank my committee members James Lassoie, Louise Buck, Stephen DeGloria, and Thomas Gavin for their insight, guidance, and encouragement throughout the project. I also thank Fabrice DeClerck of the Centro Agronómico Tropical de Investigación y Enseñanza (CATIE) in Costa Rica for providing intellectual and logistical support, and for creating a welcoming environment for me to conduct research at CATIE and its field sites. Numerous faculty, students, and conservation professionals graciously offered input and suggestions throughout the project. I thank William Philpot, Charles Smith, and the Cornell Office of Statistical Consulting for input on my research methods and data analysis, and Fabrice DeClerck for helpful comments on draft portions of this dissertation.

I am also grateful to my two research assistants, Andre Sanfiorenzo and Cliff Cordy, who were exceptionally dedicated and conscientious in carrying out their complex and often demanding assignments related to GIS analysis and bird monitoring, respectively. Diego Tobar conducted butterfly monitoring and served as a helpful liaison within CATIE. Dalia Sanchez conducted field reconnaissance and vegetation monitoring in Copán and supported my visits to that site. I thank Robert Gallardo for providing Cliff and me with information on Copán’s avifauna. Numerous scientists and field staff at CATIE and its partner organizations conducted and supervised the collection of much of the field data that I used in my research, and I thank them and project coordinator Muhammad Ibrahim for allowing me to work with these datasets.

Sara Scherr and other colleagues at Ecoagriculture Partners supported my research from the beginning and helped me to situate this project within the fast-moving field of ecoagriculture. I especially appreciate the encouragement of Louise Buck, who first invited me to work with Ecoagriculture Partners in 2005 and has been a valued advisor and colleague throughout the project.

Funding to support my PhD was provided by a National Science Foundation Graduate Research Fellowship (2005-06 and 2007-09), an EPA STAR fellowship (2006-07), and a Cornell University Teaching Assistantship (fall 2009). Ecoagriculture Partners and a Guani Family Fellowship in Conservation Biology provided funding for research and travel expenses. Additional funds to conduct field work in Honduras, Nicaragua, and Colombia were provided to partner organization CATIE by the Alcoa Foundation, Rufford Foundation, and World Bank-Bank Netherlands Partnership Program. CATIE also provided in-kind resources including staff time, office space, and field coordination. I thank all of these organizations for supporting my research.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

LIST OF BOXES

CHAPTER 1

INTRODUCTION

1.1 PROBLEM STATEMENT

As the third millennium begins, society faces unprecedented challenges related to its use and stewardship of Earth's natural resources. Chief among these are widespread poverty, rapidly rising demands for food and fiber, and extensive ecosystem degradation and biodiversity loss. Worldwide, more than a billion people survive on less than US$1 per day and 2.7 billion people live on less than US$2 per day, nearly three quarters of them in rural areas (IFAD 2009a). Disease and malnutrition continue to cause untold suffering even though their sources are often simple and seemingly preventable. As difficult as it has proven to meet the needs of Earth's current inhabitants, future population growth and economic trends are projected to increase global food demand by 70% by 2050 (FAO 2009). The challenge is not only poverty, but also affluence: rising demand for biofuels and meat, if not restricted, could displace billions of hectares of natural forest, including the world's largest terrestrial carbon stocks (FAO 2006; Gurgel et al. 2007). Our planet is on the cusp of its sixth mass extinction, in which it stands to lose up to half its species in the next century, many of which are yet unknown to science (Wilson 2002). At the same time, many of Earth's critical life support services—including water supplies, natural disaster mitigation, and soil fertility—are in steep decline (MA 2005). Further degradation of the ecosystems that furnish these services will make it even more difficult to achieve food production and poverty reduction goals, and could catastrophically increase humanity's vulnerability to climate change (IFAD 2009b).

Each of these challenges is now being addressed by global initiatives of significant scope and ambition. However, addressing each challenge individually will not be sufficient; in fact, it could be quite futile, creating competition among objectives that must all be achieved if many billion people are to attain a comfortable and sustainable existence on a finite Earth. Yet such competition is occurring at all levels, from local to global. Agribusiness and government planners claim the same areas for cropland expansion that conservationists have identified as a high priority for ecosystem conservation (McNeely and Scherr 2003). Meanwhile, rural development projects focused on anti-poverty improvements related to health, nutrition, or water all too often neglect the ecological root causes that contribute to problems in these areas and that may determine whether improvements are fleeting or lasting.

As the limitations of single-objective projects and single-objective thinking to address sustainability challenges becomes more apparent, rural landscapes and their stewards are increasingly being called upon to provide society with a broader set of goods and services. In Europe, for example, the notion of multifunctional landscapes has been widely promoted as a way to meld economic, environmental, aesthetic, and cultural objectives on rural lands. At the same time, conservationists and researchers have advanced the idea of wildlife-friendly farming as a way to improve the habitat value of human-dominated rural landscapes. In a similar vein, the concept of ecoagriculture is an approach to landscape management that simultaneously pursues the goals of agricultural production, sustainable rural livelihoods, and conservation of biodiversity and ecosystem services (Scherr and McNeely 2008). Ecoagriculture recognizes that the only way to meet all three of these critical objectives at a regional or global scale is through land use systems that advance multiple goals in the same geographic space.

Ecoagriculture is already being practiced in thousands of locations worldwide, with promising results for regions where food production, poverty alleviation, and biodiversity conservation are all high priorities. However, much remains to be learned. Although many of the component ideas and disciplines are not new, methodologies for evaluating landscape multifunctionality—and processes for achieving it—are both rapidly evolving areas of inquiry. The work presented in this dissertation is intended to advance the science and practice of ecoagriculture by addressing several key challenges and gaps, which are explored sequentially in the next three chapters.

1.2 DISSERTATION OVERVIEW

The first aspect that I address, in Chapter 2, is to situate ecoagriculture intellectually and historically, as a modern concept that builds upon several disciplines and traditions of practice related to rural development and ecosystem management. When the idea of ecoagriculture was first being formulated, in the early 2000s, practitioners and researchers viewed the lack of credible tools for planning and monitoring multifunctional rural landscapes as one of the greatest impediments to improving and mainstreaming ecoagriculture (Rhodes and Scherr 2004; Buck et al. 2006). Yet such tools are of little value without people and institutions to apply them and integrate them into decision-making processes. Accordingly, Chapter 2 defines both the technical and the social dimensions of ecoagriculture and contextualizes this paradigm in relation to other landscape approaches to sustainable land management that offer valuable lessons in both domains. Through literature review, synthesis, and case studies, I describe and document a set of processes and tools by which ecoagriculture can be achieved in practice.

In addition to exploring the concept of ecoagriculture, Chapter 2 focuses specifically on how this paradigm can contribute to global efforts to achieve the Millennium Development Goals (MDGs), which the United Nations adopted in 2000 as a framework for ameliorating poverty worldwide by 2015. Many early efforts to implement the MDGs were faulted for promoting an overly sectoral approach to poverty alleviation that paid insufficient attention to the environmentally-based root causes of poverty (Sanderson 2005; WRI 2005). By contrast, multi-objective approaches to rural landscape management that include food production, livelihood, and conservation goals are more likely to foster sustainable gains in human development, especially where livelihoods are heavily dependent on the natural resource base. To this end, Chapter 2 introduces the Landscape Measures framework as an ecoagriculture approach to advancing the MDGs.

Unlike the conceptual approach to inquiry used in Chapter 2, Chapters 3 and 4 draw upon empirically based research designs to explore one particular aspect of landscape multifunctionality: the potential to conserve biodiversity in agricultural landscapes. Specifically, I evaluate the conservation value of pasture-dominated landscapes in northern Latin America by examining the relationships between site- and landscape-scale management practices and assemblages of birds and butterflies—two taxa that are frequently used as indicators of habitat quality and that respond to different habitat features at different scales. Pasture and pasture-dominated land mosaics occupy 27% of Central America's land base, three times the area occupied by all other agricultural production systems combined (FAOSTAT 2004). Thus, it is a high priority to identify specific combinations of management practices that are economically viable yet improve the conservation value of pasture-dominated landscapes for plant and animal habitat, buffer zones, or functional corridors.

In addition to addressing a set of research questions related to biodiversity conservation in agricultural landscapes, my goal in Chapters 3 and 4 is to apply several novel or under-utilized methodologies that I hypothesized could help advance research of this sort. One such approach, which I present in Chapter 3, is to use both continuous and categorical habitat descriptors as potential predictors of faunal assemblages. Continuous habitat variables quantify gradients in the landscape—such as quantity of trees, percent forest cover, or level of structural connectivity—which have the potential advantage of being more ecologically relevant to native species than human-defined categories such as land use. A second approach is to quantify habitat variables at multiple scales, from the plot to the landscape, to determine the relative importance to conservation outcomes of individual and collective management decisions at these different scales. In Chapter 3, I use both of these research approaches to evaluate relationships among land use, agricultural management, and bird and butterfly conservation in an agricultural landscape in the Río Copán watershed in western Honduras.

Chapter 4 expands the scope of this inquiry by evaluating bird and butterfly conservation in four pasture-dominated landscapes across northern Latin America. These include the Copán landscape assessed in Chapter 3 as well as landscapes in Matiguás, Nicaragua; Esparza, Costa Rica; and Quindío, Colombia. The study sites are broadly similar in that they are all mosaics of pasture, forest, and annual and perennial cropped areas; however, they differ to some degree in their geographic and physiographic context. Thus, they are an excellent laboratory for testing the consistency of habitat-wildlife relationships within the widespread Neotropical land use system of pasture-dominated agricultural mosaics.

Identifying generalized habitat-wildlife relationships is critical for informing policies, incentives, and management practices to improve the conservation value of agricultural landscapes. However, the type of coordinated cross-site research that is necessary to reveal such general patterns has historically been uncommon because of the immense amount of data required. To conduct such research, I collaborated with scientists at the Centro Agronómico Tropical de Investigación y Enseñanza (CATIE) in Costa Rica their partners to obtain bird data and satellite imagery from prior research conducted in Matiguás, Esparza, and Quindío. I combined this information with data from original field studies (including the Copán data presented in Chapter 3 plus butterfly studies in Matiguás and Quindío) and landscape analysis (for all four landscapes) that I conducted or coordinated from 2007-2009. I then analyzed these datasets to control for confounding variables to allow for cross-site analysis.

An additional methodological challenge that I address in Chapter 4 is the characterization of species assemblages for the purpose of evaluating the conservation value of human-modified landscapes. Traditionally, measures such as species richness, abundance, and similarity have been used most commonly; however, such indices do not distinguish species according to their relative contribution to specific conservation goals, such as minimizing extinction risk or stabilizing populations of vulnerable species. To address this limitation, I develop and apply the Biodiversity Conservation Value (BCV) metric, a tool for quantifying the conservation value of species assemblages according to multiple traits of their component species. I then use BCV and other metrics to evaluate the consistency of bird and butterfly responses to land use patterns across all four study locations.

Chapter 5 synthesizes conclusions from Chapters 2-4 and discusses their implications for conservation and sustainable livelihoods in agricultural landscapes. Appendices A through C provide additional detail on aspects of the research methodology. Taken together, the five chapters present conceptual and empirical perspectives on ecoagriculture, exploring both the opportunities for and the limitations to achieving multiple management objectives in human-dominated rural landscapes.

LITERATURE CITED

Buck, L.E., J.C. Milder, T.A. Gavin, and I. Mukherjee. 2006. Understanding ecoagriculture: a framework for measuring landscape performance. Ecoagriculture Discussion Paper #2. Ecoagriculture Partners, Washington, D.C.

FAO [Food and Agriculture Organization of the United Nations]. 2006. Livestock's long shadow. FAO, Rome.

FAO [Food and Agriculture Organization of the United Nations]. 2009. Global agriculture towards 2050. FAO, Rome.

FAOSTAT. 2004. Database of the Food and Agricultural Organization of the United Nations. Online: faostat.fao.org.

Gurgel, A., J.M. Reilly, and S. Paltsev. 2007. Potential land use implications of a global biofuels industry. Journal of Agricultural & Food Industrial Organization 5(2): Article 9.

IFAD [International Fund for Agricultural Development]. 2009a. Rural poverty portal. Online: www.ruralpovertyportal.org.

IFAD [International Fund for Agricultural Development]. 2009b. Climate change: building the resilience of poor rural communities. IFAD, Rome.

MA [Millennium Ecosystem Assessment]. 2005. ecosystems and human well-being: synthesis. Island Press, Washington, D.C.

McNeely, J.A., and S.J. Scherr. 2003. Ecoagriculture. Island Press, Washington, D.C.

Rhodes, C., and S.J. Scherr, editors. 2004. Developing ecoagriculture to improve livelihoods, biodiversity conservation and sustainable production at a landscape scale. Conference proceedings from the First International Ecoagriculture Conference and Practitioners' Fair, Sept. 25 - Oct. 1, 2004.

Sanderson, S. 2005. Poverty and conservation: the new century's "peasant question?" World Development 33: 323-332.

Scherr, S.J., and J. McNeely. 2008. Biodiversity conservation and agricultural sustainability: towards a new paradigm of 'ecoagriculture' landscapes. Philosophical Transactions of the Royal Society B 363: 477-494.

Wilson, E.O. 2002. The Future of Life. Knopf, New York.

WRI [World Resources Institute in collaboration with United Nations Development Programme, United Nations Environment Programme, and World Bank]. 2005. World resources 2005: the wealth of the poor—managing ecosystems to fight poverty. World Resources Institute, Washington, D.C.

CHAPTER 2

LANDSCAPE APPROACHES TO ACHIEVING FOOD PRODUCTION, NATURAL RESOURCE CONSERVATION, AND THE MILLENNIUM DEVELOPMENT GOALS[1]

2.1 INTRODUCTION

The Río Copán watershed in western Honduras is not unlike many agricultural landscapes throughout the developing world. A journey through this 600 square kilometer watershed reveals a mixture of small and mid-sized farms producing cattle, coffee, and subsistence crops. Residents here face many challenges: recent population growth has led to deforestation and water pollution, while agricultural productivity is generally low and poverty levels remain high, especially among the indigenous Mayan population.

Environmental degradation is both a cause and a consequence of these problems. Poverty has driven many local people to cut wood in the vanishing native pine-oak forests or to cultivate or graze hillsides that are too steep for these purposes. Such practices, in turn, contribute to silted rivers unsuitable for human or livestock consumption and to landslides that routinely close roads and isolate villages from needed goods and services for weeks or months at a time. To meet the Millennium Development Goals (MDGs) in the Río Copán watershed will require not just schools, health centers, and high-yielding crop varieties; it will require a suite of coordinated

---

[1] This chapter is adapted from a manuscript of the same title by Jeffrey C. Milder, Louise E. Buck, Fabrice A.J. DeClerck, and Sara J. Scherr. My original contribution to this work included conducting the literature review, writing the entire manuscript except for the two case study descriptions, preparing all figures, and editing the entire manuscript.

activities, many of them focused on environmental restoration and natural resource management.

Fortunately, unlike many rural communities that address poverty issues piecemeal at the household or village level, the communities in the watershed have recognized that these challenges grow from—and, in turn, influence—key dynamics and ecosystem processes operating at the scale of the entire watershed, and sometimes beyond. For local leaders, the wake-up call that spurred this landscape-level thinking arrived suddenly, drenching them, quite literally, like a bucket of cold water from above. In 1998, Hurricane Mitch tore through the region, wreaking havoc not just on de-vegetated hillsides but on the farms, villages, waterways, and infrastructure below.

After taking stock of the extensive damage, the four municipalities in the watershed decided to band together to form a regional coalition aimed at preventing such devastation in the future, and at finding solutions to shared problems such as erosion, water pollution, and poor human health. They created a vision and plan for the watershed's future and, for the past several years, have been using this plan to target and guide externally-funded rural development activities. The problems and challenges in the watershed are not solved, but their root causes and interactions are now better understood. This knowledge encourages leaders to find solutions that do not trade off one landowner's wellbeing for another's, or one development objective for another, but that seek to maintain and restore the landscape's natural and human capital for the benefit of all.

2.2 INTEGRATING RURAL DEVELOPMENT AND NATURAL RESOURCE MANAGEMENT

Leaders in the Río Copán watershed have learned through experience what many scientists have documented over the past two decades: ecosystem services are critical to human wellbeing, especially in rural landscapes in developing countries. The Earth's natural capital of clean water, soils, fish, wildlife, and other resources provides about two-thirds of household income for the rural poor (MA 2005) and 26% of all wealth in low-income countries (World Bank 2006). Environmental causes are responsible for nearly one-fourth of the global disease burden, and more than four million children die each year from illnesses such as diarrhea, malaria, and respiratory infections that could be significantly mitigated by improved environmental management (Prüss-Üstün and Corvalán 2006). In light of the fundamental role of natural capital in supporting human wellbeing, it is especially worrisome that 15 of the 24 key ecosystem services upon which humans depend are being degraded or used unsustainably (MA 2005).

The eight Millennium Development Goals (MDGs) provide a global framework for ameliorating extreme poverty worldwide by 2015 (see Table 2-1 for a summary of the MDGs). World leaders and major development funding agencies have acknowledged that environmental factors are either at the root of, or closely linked to, MDGs 1 through 6—those relating to food security, human health, education, and gender equality (DFID et al. 2002; DFID 2006; Sachs and Reid 2006). This connection means that much of the recent progress toward meeting the MDGs (UN 2008) is likely to be fleeting if the natural capital that underlies these improvements continues to decline (WRI 2005). Yet, despite these well-documented linkages, the treatment of the environment in the MDGs "…harkens back to old, outmoded ways of thinking" (WRI

2005:154). Rather than being framed as a cross-cutting theme that underlies the long-term achievement of other poverty alleviation goals, the environment is addressed only in MDG 7. And although the revised MDG targets and indicators issued in 2008 provide more specific measures of success for MDG7, these measures still fail to address many of the aspects of environmental management that are most relevant for sustaining the ecosystem services that are critical for poverty alleviation (WRI 2005; DFID 2006).

Unfortunately, this inattention to natural capital as a foundation of human wellbeing has been reflected in global funding priorities and implementation frameworks for poverty alleviation. For example, Poverty Reduction Strategy Papers (PRSPs)—the vehicle by which national governments formulate objectives for meeting the MDGs and establish their priorities for international aid—have often paid insufficient attention to the environment (Bojö et al. 2004; WRI 2005). This undervaluing of environmental factors is likely a result both of the stated priorities of the aid agencies themselves (World Bank and IMF 2005) and of the apparent tendency of some governments preparing PRSPs to favor more fundable infrastructure projects over environment and agriculture projects identified as priorities by local communities and district-level agencies (Swallow 2005). The general result, at the field level, has been an overly sectoral approach to rural development that neither integrates environmental and livelihood objectives nor adequately addresses the environmental drivers underlying development goals (Sanderson 2005).

Table 2-1. The Millennium Development Goals and targets. Source: UN 2008.

Goal	Targets
MDG 1: Eradicate extreme poverty and hunger	Target 1: Halve, between 1990 and 2015, the proportion of people whose income is less than $1 a day
	Target 2: Achieve full and productive employment and decent work for all, including women and young people
	Target 3: Halve, between 1990 and 2015, the proportion of people who suffer from hunger
MDG 2: Achieve universal primary education	Target 1: Ensure that, by 2015, children everywhere, boys and girls alike, will be able to complete a full course of primary schooling
MDG 3: Promote gender equality and empower women	Target 1: Eliminate gender disparity in primary and secondary education, preferably by 2005, and in all levels of education no later than 2015
MDG 4: Reduce child mortality	Target 1: Reduce by two thirds, between 1990 and 2015, the under-five mortality rate
MDG 5: Improve maternal health	Target 1: Reduce by three quarters the maternal mortality ratio
	Target 2: Achieve universal access to reproductive health
MDG 6: Combat HIV/AIDS, malaria, and other diseases	Target 1: Have halted by 2015 and begun to reverse the spread of HIV/AIDS
	Target 2: Achieve, by 2010, universal access to treatment for HIV/AIDS for all those who need it
	Target 3: Have halted by 2015 and begun to reverse the incidence of malaria and other major diseases
MDG 7: Ensure environmental sustainability	Target 1: Integrate the principles of sustainable development into country policies and programs and reverse the loss of environmental resources
	Target 2: Reduce biodiversity loss, achieving, by 2010, a significant reduction in the rate of loss
	Target 3: Halve, by 2015, the proportion of the population without sustainable access to safe drinking water and basic sanitation
	Target 4: By 2020, to have achieved a significant improvement in the lives of at least 100 million slum dwellers
MDG 8: Develop a global partnership for development	Target 1: Address the special needs of least developed countries, landlocked countries and small island developing states
	Target 2: Develop further an open, rule-based, predictable, non-discriminatory trading and financial system
	Target 3: Deal comprehensively with developing countries' debt
	Target 4: In cooperation with pharmaceutical companies, provide access to affordable essential drugs in developing countries
	Target 5: In cooperation with the private sector, make available benefits of new technologies, especially information and communications

In light of these shortcomings, many have argued that the rural development agenda must be re-formulated to integrate environmental sustainability at all scales, from international funding priorities to on-the-ground projects. This chapter suggests that such integration needs to include a strong focus on the landscape scale—the level at which many ecosystem processes operate and at which interactions among environment and development objectives are often mediated (O'Neill et al. 1997) (Box 2-1). For example, in many landscapes, conservationists and rural development advocates have both targeted the same land or water resources for advancing their respective objectives—often with little communication or recognition of the conflicts between these aspirations (Wood et al. 2000; McNeely and Scherr 2003). In such situations, landscape-scale assessment, negotiation, planning, and monitoring can help identify actions and policies that increase synergies while decreasing tradeoffs (Palm et al. 2005). On the other hand, if tradeoffs are not explicitly acknowledged and addressed through negotiated solutions, then sectoral programs and investments will move forward in isolation, leading to composite outcomes that are likely to be far sub-optimal, especially for less powerful stakeholders.

The purpose of this chapter is to explore the theory and practice of landscape approaches to sustainable rural development and to illustrate the ways in which this paradigm can be applied to address the MDGs. The chapter begins by introducing and reviewing existing landscape approaches. Next, I present the Landscape Measures framework, a landscape approach that was developed specifically for use in 'ecoagriculture' landscapes where food production is a key objective. I then introduce some tools for implementing the Landscape Measure approach, focusing on those that apply ecological knowledge and methods. I illustrate the use of such tools by elaborating on the Copán case study introduced above as well as a recent project in

Kenya. Finally, I conclude by identifying important actions for mainstreaming landscape approaches to help achieve the MDGs.

Box 2-1. Why use a landscape perspective to address food security and rural poverty?

The reasons for working at a landscape scale stem not only from the biophysical realities of how natural resource-dependent systems function, but also from the growing interdependence and interconnectedness of rural regions. Motivations include:

1. **Scale of key ecological functions and processes.** Recent scientific research has demonstrated that flows of water, nutrients, sediment, plants, animals, and disease organisms in agricultural regions often operate beyond the farm or village level to encompass the entire landscape (Forman 1995). Many of these flows are critical to human wellbeing, providing ecosystem services such as clean water for human consumption, irrigation water, and natural pest control. Major threats, such as insect-borne diseases, crop and livestock predation, and various natural disasters, are also mediated at the landscape scale.
2. **Scale of key institutional frameworks.** In many developing nations, government authority and social programs have been devolved to smaller units of government operating at the district level (Molnar et al. 2007). At the same time, villages, communities, and non-governmental organizations are increasingly forming partnerships, networks, and alliances to addressed shared objectives (Pretty and Ward 2001). Both trends create opportunities to analyze and address challenges at a landscape scale. Conversely, inaction or ineffective policies at the landscape or sub-regional levels can keep rural households mired in “poverty traps” even when effective action is taken at the farm or village scale (Barrett and Swallow 2006). Thus meso-scale institutional arrangements are especially important in determining whether rural communities can spring out of self-reinforcing poverty traps.
3. **Changing face of the rural agricultural economy.** Throughout the world, the role of subsistence farming is in decline, while market-linked agriculture is expanding, even among small farmers. This trend is being reinforced by development and aid agencies, many of whom emphasize market access and rural enterprise development in their programs (WRI 2008). As rural communities become more tied to one other, more dependent on physical infrastructure and regional markets, and more influenced by global economic forces, it is necessary to widen the lens through which rural livelihoods are understood and advanced.

Box 2-1 (continued)

4. **New market opportunities.** Markets are beginning to place value on rural land uses that protect or enhance ecological values. Eco-certification allows producers to receive price premiums for ecologically friendly production practices, while payments for ecosystem services compensate land stewards for protecting carbon stocks, biodiversity, or watershed functions. These new market opportunities will shift incentives for rural land managers and motivate a greater focus on management at the landscape or watershed scale, where many ecosystem services are mediated.
5. **Climate change.** Resulting largely from anthropogenic forcing mechanisms, climate change is occurring faster and more dramatically than at any time in recent history. Without greater emphasis on resilience, adaptation, and regional cooperation to accommodate shifting patterns of agricultural suitability, water availability, and habitat quality, these rapid climate shifts could easily undermine local development or conservation successes (Fairhead 2004).
6. **Increased emphasis on resilience and adaptation.** The reality of climate change combined with ecologists' recognition of ecosystems as dynamic, non-equilibrium systems has led to an increased interest in resilience and adaptation as important objectives for rural landscapes (Sayer and Campbell 2004). As population growth and ecosystem degradation combine to create increasingly thin margins of error for human wellbeing in many landscapes, the ability to re-evaluate circumstances and adapt management solutions based on new information will be critical for human wellbeing (Diamond 2004). Doing so requires the continual development and use of knowledge at appropriate scales within an adaptive management framework (Röling and Wagemakers 1998; Plummer and Armitage 2007)

2.3 AN INTRODUCTION TO LANDSCAPE APPROACHES

Notwithstanding the limitations of current mainstream rural development priorities, many rural land stewards, non-governmental organizations (NGOs), researchers, and supporters have come to embrace the complexity of rural landscapes and have developed evidence-based management approaches that address the spatial, thematic, and human scope of the challenges themselves (Lal et al. 2001). I refer to these as landscape approaches and suggest that they have five defining characteristics: 1) a landscape-scale focus, 2) treatment of landscapes as complex systems, 3) management for multiple objectives, 4) adaptive management, and 5) management through participatory processes of social learning and multi-stakeholder negotiation. Each of these characteristics is discussed below.

First and most obviously, landscape approaches seek to address livelihood needs and environmental challenges at a landscape scale. There are many possible ways to define landscapes, but for management purposes it is helpful to define them functionally according to the objectives at hand and the physical extent of the features and processes that mediate these objectives (Buck et al. 2006). Precise boundaries are often ambiguous because the various biophysical gradients, socio-cultural attributes, and political jurisdictions found on the land operate at multiple scales and rarely coincide with one another. Thus, landscape approaches incorporate multi-scale linkages, helping to coordinate small-scale management efforts while considering relevant aspects of the landscape's regional and global context.

Second, landscapes are analyzed as complex systems—that is, assemblies of interconnected components that are expected to fulfill a specific set of purposes (Collins et al. 2007). Recent research on coupled human and natural systems has solidified the analytical foundations for understanding the reciprocal influences between humans and their environment at multiple scales (Liu et al. 2007). This field proposes increased emphasis on indirect linkages, feedbacks, and multi-temporal analysis when investigating or managing properties of interest such as the resilience and vulnerability of agroecosystems, which, by definition, encompass human goals, human behavior, and ecosystem dynamics. A range of methods for aiding in such analysis already exists, including system dynamics modeling, agent-based modeling, and various GIS-based tools. For example, Parker et al. (2003) illustrate how multi-agent system models of land use/land cover change can elucidate feedbacks between land stewards and the environment in the (very common) circumstance where landscape change is largely a composite outcome of numerous household-level decisions. In practice, coupled systems thinking can help policy makers anticipate future trends, manage interactions among landscape components, and expose “blind spots” that can emerge from unanticipated feedbacks (Maarleveld and Dangbegnon 1999).

Third, landscape approaches manage for multiple objectives, among which there are likely to be both synergies and tradeoffs. Multi-objective management is essential when landscapes are expected to provide more than one type of product or service—as indeed most landscapes are—and when stakeholders disagree on the goals of management and their relative importance. Furthermore, indicators for the various management goals are likely to be non-commensurable (‘apples and oranges’) such that it is difficult to define any aggregate measure of landscape success even if the relative importance of each goal can be ascertained (López-Ridaura et al. 2005; Munda 2005). For this reason, multi-objective management is rarely amenable to the type of optimization algorithms that have transformed the management of single-objective initiatives such as maximizing corporate profitability or designing the most cost-effective system of nature reserves (Röling 2002). Instead, multi-objective initiatives are likely to be understood and reported using a combination of quantitative and qualitative metrics that track whether the landscape is progressing toward the sustainable provision of the desired environmental and socioeconomic outcomes (Buck et al. 2006).

Fourth, landscape approaches are predicated on adaptive management: "…a formal, systematic, and rigorous approach to learning from the outcomes of management actions, accommodating change and improving management" (Nyberg 1999). Adaptive management is essentially the scientific method applied to real-world challenges. Resource managers begin by hypothesizing models of cause and effect, then test these models through specific interventions and policies, monitor the outcomes of these interventions, and use the resulting information to refine the causal models and improve the interventions. Over time, managers become more knowledgeable about the system and better able to respond to changing conditions, thereby increasing the resilience of ecosystems and communities in the face of natural and anthropogenic dynamics (Folke et al. 2002). Adaptive management has its intellectual roots and early experience in ecosystem management (Holling 1978) and is now widely viewed as the preferred approach for addressing complex natural resource management challenges amid incomplete information (Lee 1993; Salafsky et al. 2001). More recent formulations of this paradigm recognize that resource management is not simply a technical puzzle to be solved through better information, analysis, and planning. It is a social dilemma in which the perceptions, priorities, capabilities, and negotiation capacity of land stewards and institutions determine sustainability at least as much as the management practices themselves (Röling 2002; Ison et al. 2007). These ideas underlie the practice of adaptive collaborative management, which positions 'experts' and their technical tools in the role of facilitators or technical advisors to assist a process that is guided by stakeholders themselves (Buck et al. 2001; Colfer 2005).

This leads to the fifth and final characteristic of landscape approaches: an ongoing, participatory process of 'social learning' through which stakeholders iteratively discover and generate relevant knowledge, negotiate goals and objectives, implement management plans, and evaluate outcomes (Leeuwis and Pyburn 2002; Steyaert et al. 2007). In the context of adaptive management, social learning encourages stakeholders to articulate and discuss their understanding of reality and mental models of cause and effect when formulating goals, objectives, and plans (van Noordwijk et al. 2001). These understandings are refined over time based on evidence from project monitoring as well as external sources. Because it provides a built-in mechanism for incorporating new information and responding to novel circumstances, social learning is essential for ensuring the sustainability and resilience of human and natural systems (Röling and Wagemakers 1998; Olsson et al. 2004).

2.4 CONTEMPORARY USES OF LANDSCAPE APPROACHES

I conducted a literature review to identify the ways in which landscape approaches have been used to address rural poverty and natural resource conservation challenges. This section provides a brief history of the development of landscape approaches and some leading examples of recent practice.

The roots of landscape approaches can be traced to the emergence of the sustainable development concept in the late 1980s (WCED 1987; Lele 1991). This framework ushered in a wave of Integrated Conservation and Development Projects (ICDPs) that included both rural development and environmental (particularly biodiversity protection) objectives. However, the outcomes of ICDPs proved generally to be disappointing. In many projects, the nexus between the development activities and conservation objectives was poorly conceived or fallacious: win-win solutions were assumed rather than acknowledging and addressing tradeoffs. Furthermore, local participation was often token, resulting in mis-directed efforts yielding transient benefits that evaporated when project funding ended (McShane and Wells 2004). Some observers blamed these failures on fundamental flaws in the integrated project model itself (Terbough 1999) while others argued that the basic ideas were sound but had not been fully embraced in most first-generation ICDPs (Brechin et al. 2003). In retrospect, it can be said that these projects aspired to multi-objective rural land management but typically lacked most of the other attributes of landscape approaches, such as adaptive management in a social learning context. These omissions were often important causes of the projects' shortcomings.

The disappointing results of early ICDPs coincided with a growing awareness of ecosystem services and their role in sustaining society (Daily 1997; Costanza et al. 1998). This theme was echoed in the 1998 systemwide review of the Consultative Group on International Agricultural Research (CGIAR), which urged the 16 CGIAR centers to move beyond crop research to advance the field of natural resource management to support global food production (CGIAR 1998). Building on earlier formative work by the World Agroforestry Center and Center for International Forestry Research, the centers responded by adopting a program on Integrated Natural Resource Management (INRM), which they defined as a research and management approach that "…aims at improving livelihoods, agroecosystem resilience, agricultural productivity and environmental services [by] augment[ing] social, physical, human, natural and financial capital" (ICARDA 2005). While INRM is not specifically a landscape approach, it envisions management and analysis at multiple nested scales including that of the landscape (Campbell et al. 2001; Izac and Sanchez 2001). Recent INRM initiatives by several of the CGIAR centers have included a strong landscape emphasis and illustrate how 'action research' can facilitate stakeholder dialogue, planning, and management for conservation, food production, and livelihood objectives (Gottret and White 2001; Frost et al. 2006; Pfund et al. 2008).

Despite these promising initiatives, landscape-level planning and analysis does not yet play a significant role in mainstream agricultural investment, management, or policy. Nevertheless, there is some tradition of spatial thinking in agriculture, and this is gradually expanding to encompass larger scales and broader disciplinary foci. For instance, agricultural investment decisions are commonly made using spatially-sensitive methods such as agroecological suitability classification (based on factors such as altitude, rainfall, and soil type) and market analysis (based on transportation costs, access to inputs, value chain mapping, and distance to storage or processing facilities). Spatial zoning for agriculture is now becoming more nuanced, with certain agricultural uses contingent on the adoption of conservation management practices. Farmers are increasingly choosing to coordinate across sites to address challenges such as pest control, salinization, and limited availability of irrigation water. Such efforts are being supported by new scientific tools such as spatial modeling of nutrient flows, and by new policy instruments such as nutrient trading systems. The concept of foodsheds has encouraged more systematic spatial analysis of food supplies and value chains around major population centers (Kloppenburg et al. 1996). All of these approaches are beginning to increase the scale at which agricultural management is considered as well as the level of integration among production, conservation, and livelihood dimensions.

Concurrently, conservationists have begun to implement landscape approaches such as biological corridors, landscape-scale conservation planning, and green infrastructure planning to address the challenges of habitat fragmentation and ecosystem degradation in populous regions (Rosenberg et al. 1997; Benedict and McMahon 2006). Many such projects seek to address livelihood needs in concert with biodiversity conservation by engaging private and communal land stewards in transitioning to more conservation-friendly agriculture and livelihood strategies (e.g., Miller et al. 2001). A new generation of multi-objective landscape-scale projects by groups such as WWF and the Wildlife Conservation Society can be seen as a maturation of the ICDP concept to embrace genuine local participation and a broader set of spatial and temporal scales to address the poverty-biodiversity nexus (USFS 2006; Redford and Fearn 2007; COMACO 2009). For instance, the IUCN/WWF Forest Landscape Restoration initiative aims to restore ecosystem goods and services by increasing tree cover in degraded landscapes while engaging stakeholders to address institutional barriers at multiple scales (Barrow et al. 2002; Sayer and Buck 2008). A complementary process for landscape monitoring and adaptive management has also been developed, which uses the Capital Assets Framework (Carney 1998) to track multiple landscape variables and to use this information to aid in participatory decision-making (Sayer et al. 2007).

The preceding examples were of landscape approaches initiated by international NGOs and research centers. However, much of the impetus for landscape-level planning and management emerges from local and regional initiatives. For example, the practice of participatory watershed management arose as an alternative to ineffective top-down watershed planning. In this approach, priorities are negotiated at the watershed scale but implemented at the community level through micro-watershed plans focused on practices such as re-vegetation, soil management, and erosion control (Hinchcliffe et al. 1999; Kerr 2002). More generally, the concept of community-based natural resource management has been widely applied to forest, water, wildlife, rangeland and other common property or state-owned resources to secure tenure rights and support collective management and shared benefits (Leach et al. 1999; Borrini-Feyerabend et al. 2000). At a larger scale, the concept of territorial management has been used to assert local control over rural development processes, including land and resource use. This approach is best developed in Latin America, where it has been applied in the context of indigenous reserves as well as mainstream planning for rural areas (Sepúlveda et al. 2003).

Overall, this analysis revealed many instances of both community-led and externally driven initiatives that met three or four of the characteristics of landscape approaches described above, but relatively few that met all five. Of those cases that exhibited all five characteristics, most were being carried out in forested landscapes where the objective was to reconcile biodiversity conservation and poverty alleviation. To my knowledge, landscape approaches have rarely been applied to areas where cropland or rangeland is a major land use and where food production for a large local population is a central goal.

2.5 ECOAGRICULTURE AND THE LANDSCAPE MEASURES APPROACH

The lack of methods and tools for landscape-scale management and monitoring of agroecosystems was a frequent theme at the first Ecoagriculture Conference and Practitioners' Fair in Nairobi in 2004. Many of the researchers, government and NGO representatives, community leaders, donors, and farmers at the meeting were involved in implementing or promoting ecoagriculture—that is, efforts to simultaneously achieve food production, conservation, and rural livelihood goals at a landscape level

(McNeely and Scherr 2003; Scherr and McNeely 2008). Conference participants could point to many examples where ecoagriculture principles had been implemented successfully. Yet, their ability to sustain, document, and scale up these successes was limited by the dearth of existing frameworks or processes for planning and monitoring ecoagriculture landscapes. What was needed was a landscape approach that spoke to the particular issues and challenges of ecoagriculture contexts where food production (cropping, livestock, agroforestry, or fisheries) comprises a significant portion of the land base and the local economy.

The Landscape Measures approach (LM), which is described and illustrated in the remainder of this chapter, was developed to address this need. Formulated as part of Ecoagriculture Partners' Landscape Measures Initiative (LMI), the LM consists of a set of processes and tools for negotiating, planning, implementing, and evaluating ecoagriculture practices and innovations (Buck et al. 2006). Like other landscape approaches, the LM is predicated on stakeholder-driven adaptive management embedded in a social learning process (Figure 2-1). However, the LM is designed around the four major goals of ecoagriculture: 1) conserving biodiversity and ecosystem services, 2) producing food, 3) improving rural livelihoods, and 4) building effective institutions for cross-sector planning, analysis, and action. As such, the LM includes tools and methods specifically oriented toward these goals and toward measuring and negotiating the interactions among them.

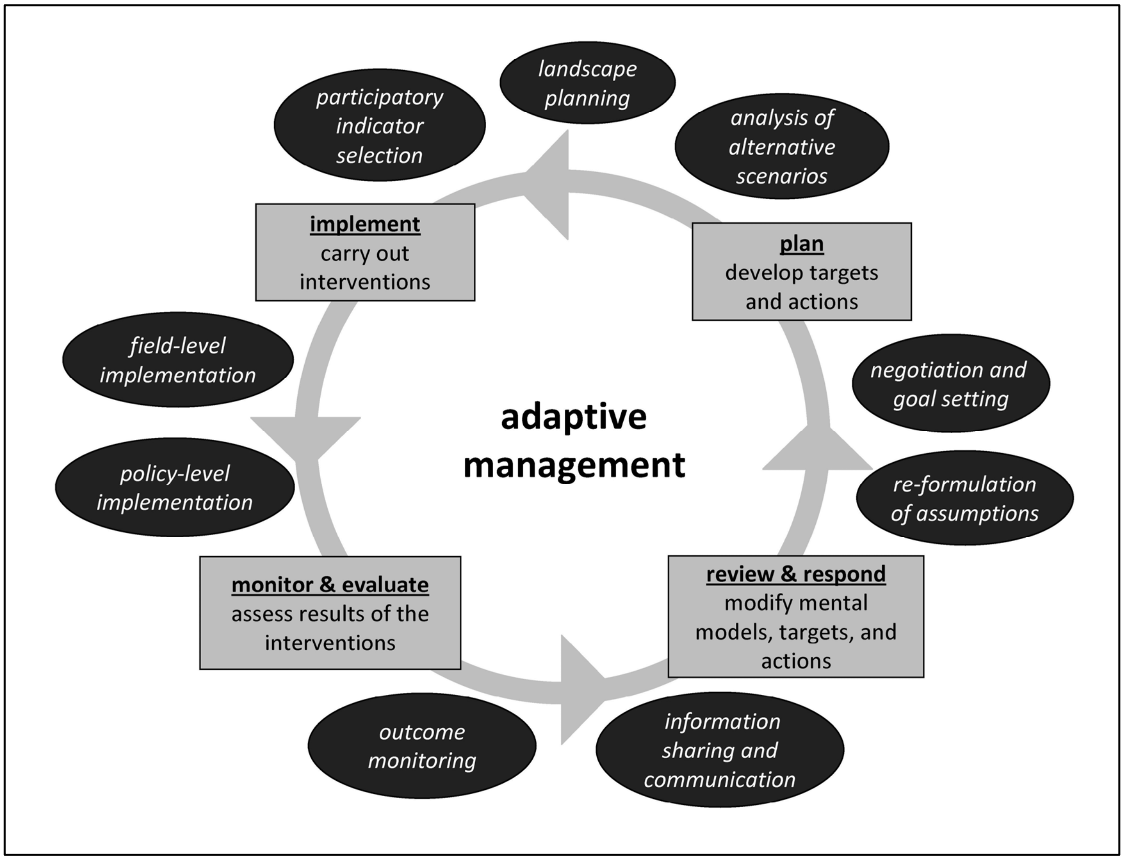


Figure 2-1. Key roles of the Landscape Measures approach (LM) for guiding adaptive management for food production, conservation, livelihoods, and institutional capacity in rural landscapes. The standard adaptive management cycle is depicted in gray, while key LM processes and tools for each phase of the cycle are shown as black ovals.

2.5.1 The Landscape Measures framework

One of the salient challenges of working at a landscape scale is to incorporate the important goals, processes, and dynamics into adaptive management without getting mired in excessive detail and layers of complexity (Lynam et al. 2007). To address this challenge, the LMI conducted a year-long consultative process that engaged scientists and practitioners from diverse disciplines and sectors in conversations about how to track change across multiple dimensions at a landscape scale (Buck et al. 2006). One outcome of these conversations was a set of “20 Questions” about landscape performance that represented the key variables that are likely to be important in ecoagriculture landscapes worldwide (Buck et al. 2006; Box 2-2). The 20 Questions offer tangible criteria for assessing progress toward the four broad goals of ecoagriculture. In turn, stakeholders can answer the questions by selecting and evaluating context-appropriate indicators and means of measure (Table 2-2). Because many of the 20 Questions focus explicitly on the interactions among conservation, food production, rural livelihoods, and supporting institutions, they can help spur cross-sector dialogue and encourage stakeholders to negotiate tradeoffs among competing interests rather than avoiding such important conversations.

Box 2-2. Twenty questions for assessing the performance of ecoagriculture landscapes (adapted from Buck et al. 2006)
Conservation Goal: The landscape conserves, maintains, and restores wild biodiversity and ecosystem services.
Criterion C1: Does the landscape contain an adequate quantity and suitable configuration of natural and semi-natural habitat to protect native biodiversity?
Criterion C2: Do natural and semi-natural habitats in the landscape approximate the composition and structure of the habitats historically found in the landscape?
Criterion C3: Are important species within the landscape biologically viable?
Criterion C4: Does the landscape provide locally, regionally, and globally important ecosystem services?
Criterion C5: Are natural areas and aquatic resources degraded by productive areas and activities?
Production Goal: The landscape provides for the sustainable production of crops, livestock, fish, forests, and wild edible resources.
Criterion P1: Do production systems satisfy demand for agricultural products (crops, livestock, fish, wood) by consumers inside and outside the landscape?
Criterion P2: Are production systems financially viable and can they adapt to changes in input and output markets?
Criterion P3: Are production systems resilient to disturbances, both natural and human?

Box 2-2 (continued)
Criterion P4: Do production systems have a neutral or positive impact on wild biodiversity and ecosystem services in the landscape?
Criterion P5: Are species and varietal diversity of crops, livestock, fisheries and forests adequate and maintained?
Livelihoods Goal: The landscape sustains or enhances the livelihoods and wellbeing of all social groups who reside there.
Criterion L1: Are households and communities able to meet their basic needs while sustaining natural resources?
Criterion L2: Is the value of household and community income and assets increasing?
Criterion L3: Do households and communities have sustainable and equitable access to critical natural resource stocks and flows?
Criterion L4: Are local economies and livelihoods resilient to change in human and non-human population dynamics?
Criterion L5: Are households and communities resilient to external shocks such as flooding, drought, changes in commodity prices, and disease epidemics?
Institutions Goal: The landscape hosts institutions that support the planning, negotiation, implementation, resource mobilization, and capacity-building needed to integrate conservation, production and livelihood functions.
Criterion I1: Are mechanisms in place and functioning for cross-sectoral interaction at landscape scale?
Criterion I2: Do producers and other community members have adequate capacity to learn and innovate about practices that will lead to integrated landscapes?
Criterion I3: Does public policy support integrated landscapes?
Criterion I4: Are market incentives conducive to integrated landscapes?
Criterion I5: Do knowledge, norms, and values support integrated landscapes?

Table 2-2. Hierarchical framework of the Landscape Measures approach (LM) for identifying and tracking progress toward landscape objectives. Similar to other recent methods for landscape evaluation (e.g., CIFOR 1999; LAC-Net 2006), this hierarchical approach helps ensure that all major system components are considered while leaving room to interpret these components in relation to the landscape's specific biophysical and socio-cultural context.

Hierarchical level	Selection process	Description
Goals	Universal; part of the LM framework	Comprises the four broad goals of ecoagriculture: sustainable food production, viable rural livelihoods, conservation of biodiversity and ecosystem services, and effective supporting institutions.
Criteria	Universal; part of the LM framework	The 20 Questions, which enumerate five specific sub-goals for each of the four ecoagriculture goals.
Indicators	Place-specific; selected by stakeholders	Tangible factors or characteristics in the landscape that are measured to reveal how well each criterion is being fulfilled. Stakeholders select indicators that are relevant to the landscape context and to their specific objectives.
Means of measure	Place-specific; selected by stakeholders	Specific methods or techniques for evaluating indicators, such as household interviews or land cover analysis. Stakeholders select means of measure that are appropriate to the desired level of precision and availability of monitoring resources.

The 20 Questions provide a useful complement to the Millennium Development Goals, targets, and indicators for monitoring the performance and sustainability of rural landscapes. Whereas the targets for MDGs 1 through 6 are focused on specific human wellbeing outcomes, the 20 Questions help elucidate some of the ecological drivers that undergird long-term human wellbeing in rural landscapes. In addition, the 20 Questions offer a more detailed framework for monitoring MDG 7 (environmental sustainability) by focusing on local and landscape-scale ecosystem structure and function. This focus can help address recent calls for improved monitoring of ecosystem services in assessing progress toward the MDGs—for example, by tracking

soil fertility, hydrological function, and the maintenance of biodiversity, as well as the ways in which local people value, utilize, and sustain such ecosystem services (WRI 2005).

The LM is designed to be used in all phases of the adaptive management cycle, including goal setting, planning, and monitoring (see Figure 2-1):

Goal setting and stakeholder negotiation. The framework and 20 Questions provide a roadmap to landscape multi-functionality, identifying those functions that local and external stakeholders typically expect a landscape to fulfill. In recent experience applying the framework, nearly all of these 20 factors have proven relevant in landscapes across a diverse range of contexts. By providing a broad view of what would constitute successful landscape management, the framework can also help ensure that goals are not skewed too far toward or away from any single interest group. Under-represented stakeholders are given greater legitimacy in negotiations while all participants are encouraged to consider landscape processes or objectives that may be outside their ordinary purview.

Landscape planning. In rural landscapes in developing countries, there is a significant history of spatial planning for single objectives or projects (e.g., plantation forestry, large-scale agriculture, and conservation networks), but much less experience with multi-functional landscape planning (Selman 2002). Such planning can identify and promote synergies among disparate landscape objectives to a much greater degree than sectoral plans that optimize for a single outcome. Essentially, multi-functional landscape planning for ecoagriculture is the process of making the 20 Questions spatially explicit by establishing land and resource use parameters that can be implemented locally. The resulting spatial plans will often have a high proportion of multi-use zones (such as agroforestry or rotational grazing), substantial integration of activities on the landscape, and a relatively fine spatial resolution, reflecting the knowledge-intensive, ecosystem-based management that is proposed (Scherr et al. 2009). Integrated planning can also help ensure that sectoral plans are consistent with broader goals and will register positively against multiple criteria in the LM framework. Although landscape planning requires technical expertise, the process need not be controlled by outside experts; indeed, facilitated multi-objective planning processes can be an effective vehicle for engaging diverse stakeholders to influence management and policy outcomes (Wollenberg et al. 2000).

Landscape monitoring. One constraint to the use of ecosystem-based approaches to poverty alleviation is the inadequacy of environmental monitoring systems in many parts of the developing world (WRI 2005:161). Tracking landscape change requires going beyond project-based evaluation monitoring that focuses on a small set of landscape variables that the project expects to influence. Instead, monitoring should track all key system components so that it can reveal unexpected results of interventions as well as complex interactions of policy or management changes with other landscape dynamics. The LM helps define the scope of landscape monitoring by identifying a series of objectives for which stakeholders can select context-appropriate indicators for measuring progress over time. Data on these indicators then feeds back into the social learning process, expanding the base of information upon which future plans and decisions are made (Sayer and Campbell 2004).

2.5.2 Implementation process

As with other landscape approaches, the LM is implemented through a process of social learning and negotiation among landscape stakeholders to adaptively manage land, natural resources, capital assets, and market and policy structures. Consistent with the multi-scaled nature of landscapes, adaptive management must engage participants at many levels. Local participation and leadership are essential, but external stakeholders and higher-level agencies must also be represented to the extent that they have a legitimate interest in the landscape. Processes that fail to engage external actors who have the will and power to exert significant influence (such as agri-business or international NGOs) are unlikely to be successful. Instead, conflict and trade-offs between local and external interests must be acknowledged and clarified so that negotiation can take place.

Implementation of the LM usually requires a 'landscape facilitator'—individual(s) or organization(s) who work on a systematic and sustained basis to convene stakeholders, guide negotiation, manage information, and promote collective action (Laumonier et al. 2008; Buck and Scherr 2009). Steyaert and Jiggins (2007) define facilitation as "…a combination of skills, activities and tools used to support and guide learning processes among multiple interdependent stakeholders [to] bring about systemic change in complex situations…." Ideally, the landscape facilitator should be a neutral party that is dedicated only to the social learning process itself, as guided by the 20 Questions—not to any specific outcomes. Truly disinterested parties are rarely available as they have little incentive to participate; instead, facilitators are often drawn from the ranks of NGOs and research organizations, which often have a disciplinary or normative bias, if not a deliberate agenda. In these cases, facilitators must be scrupulous in acknowledging their biases and working to subordinate them to the larger process.

One key role of the landscape facilitator is to integrate stakeholders' disparate knowledge systems, data needs, and ways of communicating and using information. Past experience indicates that for scientific information to support sustainable development, greater efforts are needed to bridge the realms of knowledge generation and decision-making by ensuring that information is credible, salient, and legitimate to decision makers (Cash et al. 2003; Dietz et al. 2003). Yet, farmers, government agencies, and international donors each have very different conceptions of credibility, salience, and legitimacy. Furthermore, knowledge of rural landscapes can be rooted in many different epistemologies. Landscape level innovation systems integrate experiential or 'tacit' knowledge—gained by people who live in the landscape and are intimately familiar with aspects of its workings over time—with evidence of phenomena that are revealed through scientific inquiry and likely to be less visible to local people. Combining these approaches can provide a richer understanding of the landscape, and one that is credible to local and external stakeholders alike (Bell and Morse 2001).

Although the LM is predicated on significant coordination among sectors and scales in rural landscapes, the goal is not to establish a centralized landscape 'secretariat' but rather a web of activity nodes that are knit together by shared purpose, shared information, and dedication to evidence-based decision making. These nodes come together from time to time to negotiate and establish broad-level goals, formulate plans, identify needed collaborations, and share monitoring results to understand the interactive effects of different projects and programs on the landscape. Actual management and policy interventions are carried out at a range of scales—from the household to the region or beyond—but these interventions occur within the context of the landscape planning and monitoring process (Figure 2-2).

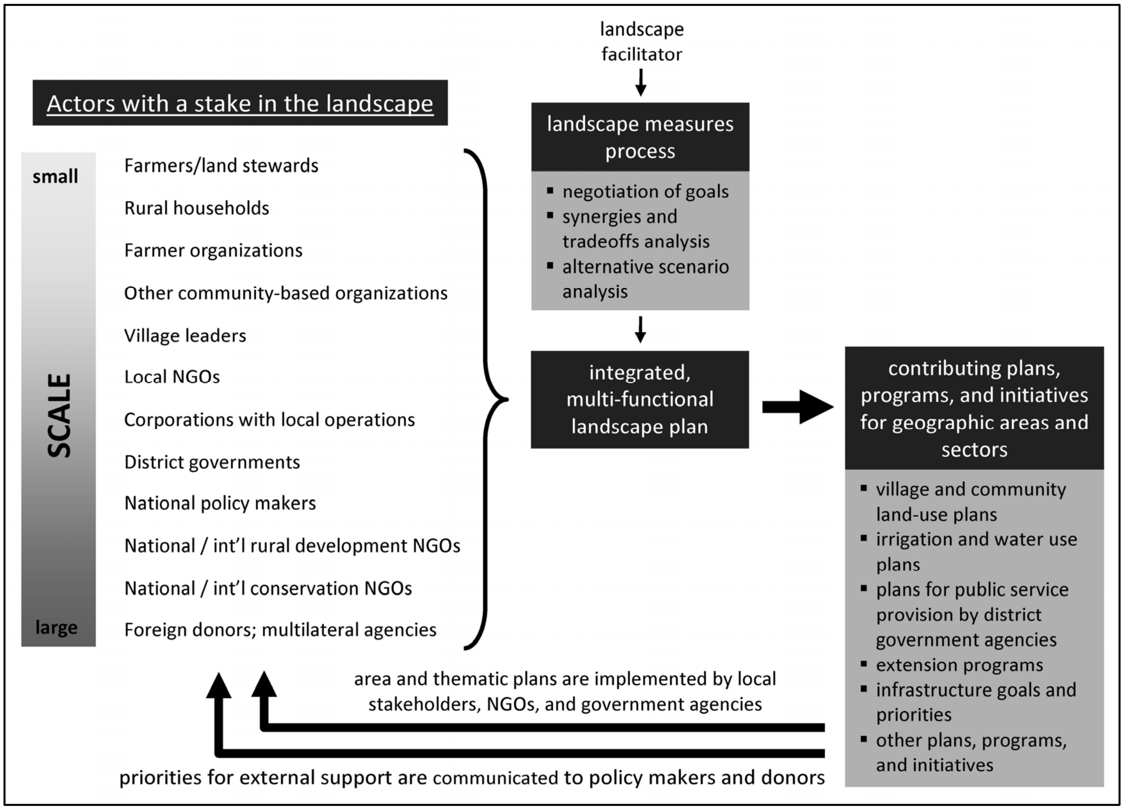


Figure 2-2. Idealized representation of the interactions among stakeholder groups in the Landscape Measures approach (LM). Moving from left to right in the diagram: 1) A wide range of actors—operating at multiple scales—have a stake in rural landscapes. Many of these groups are already linked to each other through social networks, joint projects, and so forth, and the LM can strengthen or augment such linkages. 2) These diverse actors come together to participate in the LM under the auspices of a landscape facilitator. Negotiation and social learning supported by technical analysis lead to the formulation of an integrated, multi-functional landscape plan. 3) Landscape actors then incorporate information, insights, and agreed-upon goals and objectives from the broader LM process into their geographically- and sectorally-focused activities, programs, and plans. These activities are implemented on the ground and communicated to stakeholders operating at other scales (especially donors and policy makers). Over time, the relationships depicted here are sustained and strengthened in an iterative process, while the resulting plans and activities are frequently revisited in light of new circumstances, new priorities, and new landscape monitoring data.

2.5.3 Ecologically-based tools for implementation

This sub-section highlights some of the more promising tools that have recently been developed to aid in the implementation of the LM. These and other tools are also presented in an online portal for practitioners known as the Landscape Measures Resource Center (LMRC) (LMI 2009).

As discussed above, a key challenge of multi-stakeholder adaptive management is to bridge different types and uses of knowledge by different landscape actors. One way to do so is through landscape monitoring programs that incorporate both scientific and community knowledge (Place and Were 2005). For example, several of the methods in the LMRC combine social learning with scientifically rigorous sampling and analysis methods to add external credibility to community-generated datasets while bringing local relevance to monitoring data demanded by outside donors and program evaluators. A second challenge is to generate sufficient knowledge about landscape dynamics, even when the funding and personnel resources available for the task are quite limited. This need highlights the importance of approaches that derive additional value from existing monitoring efforts, employ participatory monitoring, and take advantage of newly available low-cost data collection and analysis tools.

One such method—repeat ground-based photo-monitoring—can be a cost-effective way to track changes in vegetation and land use when aerial imagery is unavailable or unaffordable (Lassoie et al. 2006). In this method, scientists use stratified sampling to establish points throughout the landscape from which digital photographs are taken in all directions. The photos are analyzed according to a protocol that yields standardized quantitative and qualitative descriptors, which are entered into a database. As the photo points are re-visited over the course of months and years, the data begin to reveal trends in land use, agricultural management, vegetation condition, and other factors. The digital photographs themselves can be taken by local people, providing a credible and easily interpretable data source for household- and village-level adaptive management while generating high quality data through systematic aggregation across the network of photo points. More generally, participatory monitoring and evaluation can often yield data that are widely credible if it follows a scientifically designed protocol (Bonney et al. 2009).

A second method in the LMRC toolkit achieves the opposite type of knowledge transfer, taking data that are collected for external evaluators and making them relevant to local land stewards to use in adaptive management. On eco-certified farms throughout the world, large amounts of data are collected annually to meet the auditing requirements of various certification systems. Yet much of this information is filed away, never to be used by land stewards in the service of improved management. For these data to be useful to landscape stakeholders, they must be entered into appropriate information systems, aggregated, analyzed, and communicated effectively. For example, monitoring data on agrochemical usage, cover cropping, or soil erosion potential could be spatially plotted in a geographic information system (GIS) to visualize trends across space and time. This information could then be combined with downstream water quality monitoring data to track the relationship between on-farm practices and watershed-level ecosystem services. Again, approaches from the field of ecology can be used to help establish appropriate sampling protocols, aggregation methods, and analysis techniques.

Central to the LM is the use of integrative indicators that provide answers to several of the 20 Questions at once. An important integrative indicator in almost every landscape is the composition and configuration of land use and land cover. Basic land cover maps can be created by interpreting aerial imagery or compiling data from field surveys or repeat ground-based photography. Maps can then be analyzed quantitatively to derive key measures of composition (e.g., area under native forest) and structure (e.g., degree of interspersion of complementary or conflicting land uses). Often, these measures can be further extrapolated to estimate outcomes related to food production, species viability, hydrological functions, and other key landscape parameters.

Given the great interpretive power of such composition and structure measures, landscape design principles have been proposed as heuristics for maintaining ecological integrity in the context of endeavors such as regional planning (Forman 1995; Dramstad et al. 1996; Lindenmeyer et al. 2008) and agroecosystem management (Fischer et al. 2006; Harvey 2008). Similar principles and proxies could be developed for other objectives of landscape multi-functionality, including increased agricultural production, decreased disease burden attributable to environmental factors, and other goals related to the long-term fulfillment of the MDGs. Recent work on ecosystem service mapping has begun to relate landscape composition, ecological integrity, livelihood potential, and economic value in a spatially explicit manner (e.g., Troy and Wilson 2006; Egoh et al. 2008). These efforts suggest how GIS-based analyses can be used to track many of the 20 Questions with relatively fine spatial and temporal resolution.

The final tool presented here is the use of systems dynamics modeling: computer applications that allow a user to simulate complex systems by tracking numerous interacting variables over time (Sterman 2000). Although system dynamics modeling is based on a mechanistic view of systems, its great advantage is that it can account for much higher levels of complexity than is possible through human intuition and *ad hoc* methods, making it valuable for landscape approaches. Key applications include understanding causal relationships in the landscape, identifying high-leverage 'pressure points' for landscape change, determining thresholds at which dramatic changes may occur, exploring alternative scenarios through participatory modeling, and measuring the success of interventions by comparing actual landscape outcomes to simulated outcomes under alternative management programs (Campbell et al. 2001; Sandker et al. 2007).

2.6 CASE STUDY 1: APPLYING THE LM IN COPÁN, HONDURAS

The Copán case study illustrates the application of the LM to conduct a broad-reaching baseline evaluation of landscape conditions and to elucidate and prioritize community needs. Honduras currently has the highest poverty rate in Central America (70%) and ranks 115 out of 170 countries globally in the index of human development (Programa Estado de la Nación 2008). The Copán region is somewhat insulated from the worst poverty due to tourism revenue associated with the local Mayan ruins. Ironically, however, the landscape's most impoverished residents are the Chorti Maya, whose ancestors built these temples. As such, the landscape contains a diverse mix of stakeholders, ranging from wealthier landowners concentrated around the colonial town of Copán Ruinas—whose income is principally drawn from ecological and cultural tourism—to coffee and cattle farmers and the *campesinos* they hire to work their lands, to the Chorti Maya, who are largely segregated from the Mestizo majority and work as farm laborers or depend on subsistence agriculture.

As discussed earlier, there is already some institutional capacity in the Río Copán watershed for carrying out landscape approaches to natural resource management and community development. A regional governing body known as the MANCOSARIC represents the watershed's four municipalities and works to improve basic human services while facilitating adaptive co-management with an emphasis on improving flows of ecosystem services and reducing risks from natural hazards such as flooding and landslides. The MANCOSARIC also helps empower local governments to take responsibility for natural resource stewardship through integrated watershed management.

In 2007, the MANCOSARIC and its partners decided to implement the LM and the 20 Questions to provide a baseline evaluation of the watershed that would help them understand the current status of the landscape, identify priorities, and refine current landscape management plans. The landscape was particularly suitable for such evaluation because of the existence of the MANCOSARIC governing body, which was well positioned to utilize the information generated. The evaluation also promised to offer a wider perspective on the region and a staring point for initiating critical discussion on stakeholder priorities.

The baseline evaluation conducted by Bejarano (2009) was designed to synthesize useful information from pre-existing studies while generating strategic new data to answer some of the 20 Questions deemed most critical by local stakeholders. Many of the landscape performance measures included in the assessment were derived or extrapolated from land use patterns and dynamics. In this regard, the MANCOSARIC was fortunate to have a 1 m resolution IKONOS satellite image of the landscape acquired in 2007 that was classified into land uses at the plot scale (Sanfiorenzo 2008). This land use map provided a foundation for much of the landscape evaluation, allowing stakeholders to analyze information on agricultural production, conservation, and livelihood indicators in a spatially explicit manner to understand where improvements were most needed.

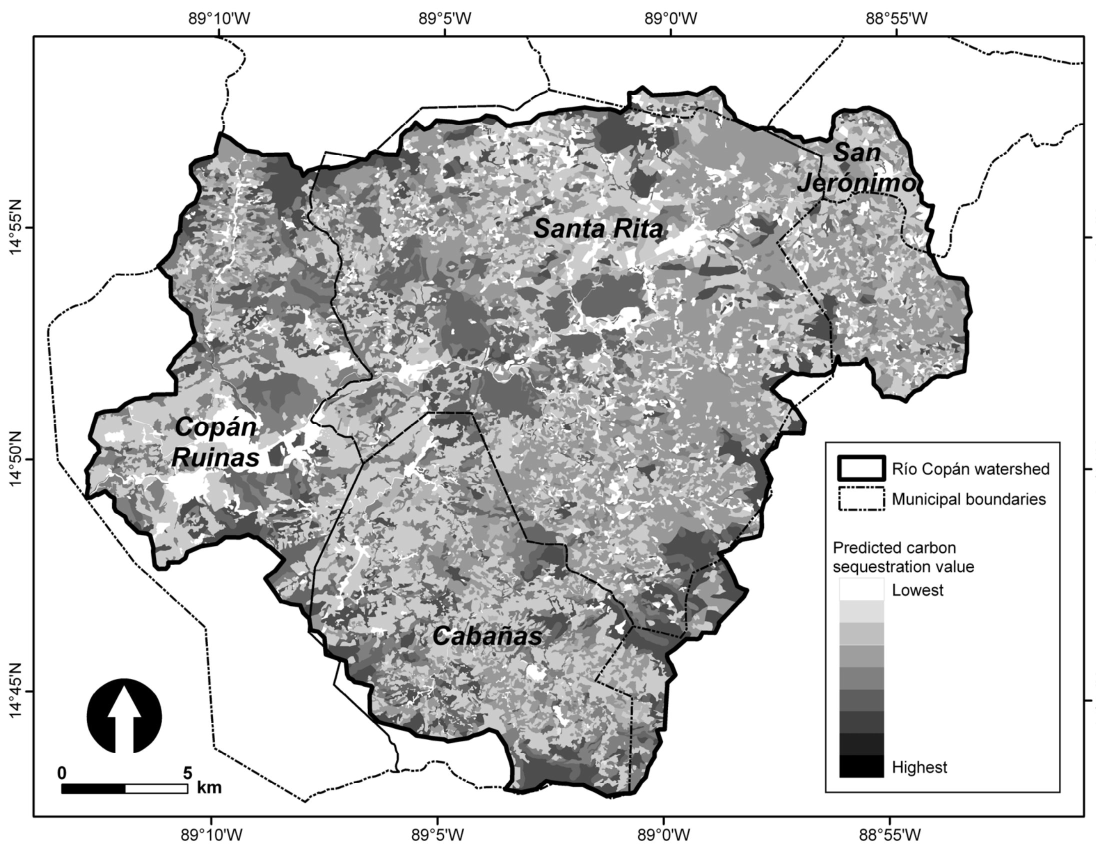


Figure 2-3a. Predicted distribution of carbon sequestration services provided in the Río Copán watershed based on estimates of the capacity of each land use to store carbon (Murgueitio et al. 2004).

One application, for example, was the interpretation of land use patterns to estimate the provision of ecosystem services throughout the watershed (Figures 2-3a and 2-3b).

While land use is not a precise proxy for such services, prior study has yielded enough information on the relationships between land use, biodiversity conservation, and carbon storage to help identify hotspots where ecosystem services have been eroded and where restoration efforts could address both conservation and livelihood goals. The spatially explicit nature of these maps facilitates negotiation by identifying specific property owners and municipalities that could benefit from interventions.

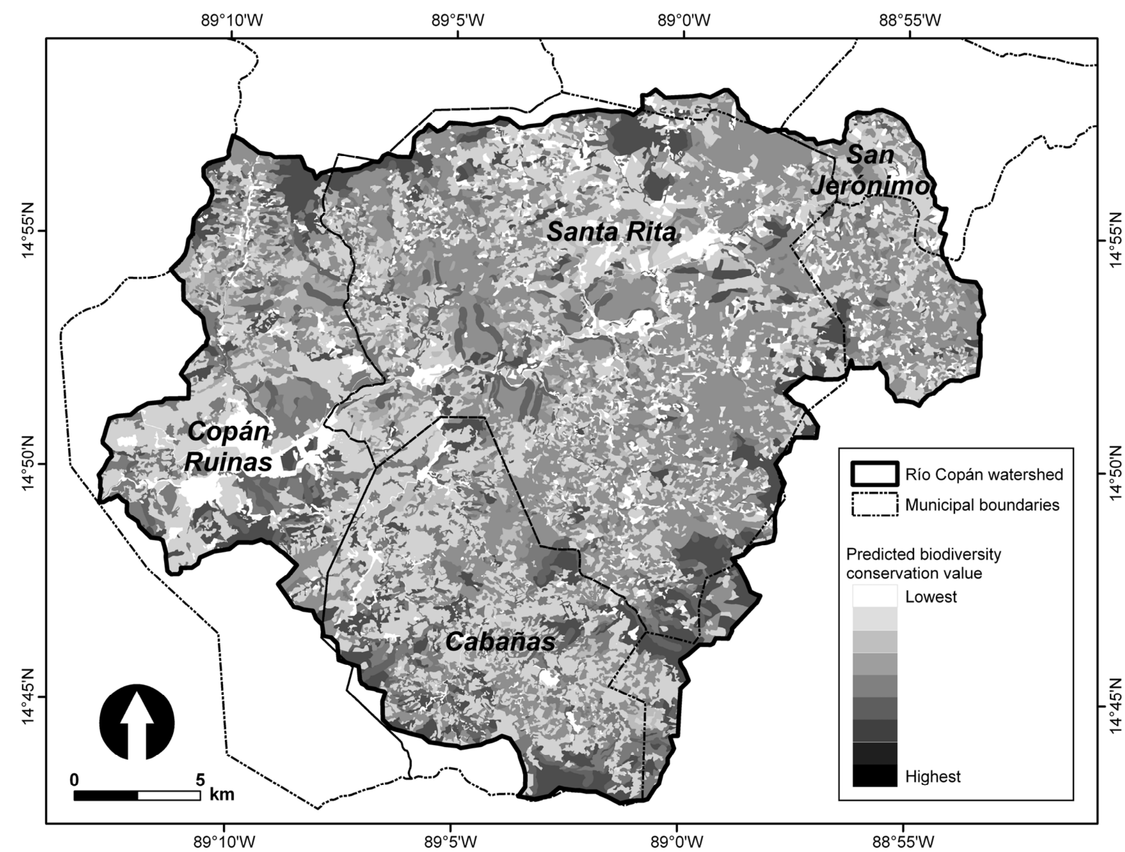


Figure 2-3b. Predicted biodiversity conservation value within the Río Copán watershed based on estimates of the capacity of each land use to sustain native species (Murgueitio et al. 2004).

While landscape composition and structure metrics were an important part of the landscape evaluation, it was critical to supplement these measures with household interviews and plot-level field studies to answer many of the 20 Questions. For example, one of the surrogate measures for conservation criteria 1 and 3 (see Box 2-2) was to ask farmers when they had last seen a wild deer. Representative patches of each forest type in each community were also surveyed to evaluate vegetation structure and evidence of degradation from grazing, timber or fuelwood extraction, and other human activities. This study indicated that forests are more degraded in Cabañas—where the economy is heavily based on natural resources—than in Copán Ruinas, a larger town with a more diversified economy.

The evaluation of livelihood indictors was primarily based on household interviews (45 per municipality), but these were spatially stratified and located with GPS coordinates to allow spatially explicit analysis of the relationships among multiple goals. Interviews revealed household members' education levels, production activities, agricultural yields, farm income, total income, and other factors. Results were integrated with those from earlier household surveys focusing on farm-level conservation practices and access to water and energy resources. Both sets of interviews also assessed the degree to which local social service and resource management entities were providing households with services, extension, or training—or even the degree to which farmers were aware of relevant projects. These data helped define the effectiveness and sphere of influence of local institutions relative to their mission and objectives. The data also revealed spatial patterns of wealth and poverty—including both current income and capacity to improve and adapt household livelihood strategies. Again, the evaluation documented greater levels of poverty and need in the more resource-dependent communities outside of the tourism nexus (and MANCOSARIC headquarters) in Copán Ruinas.

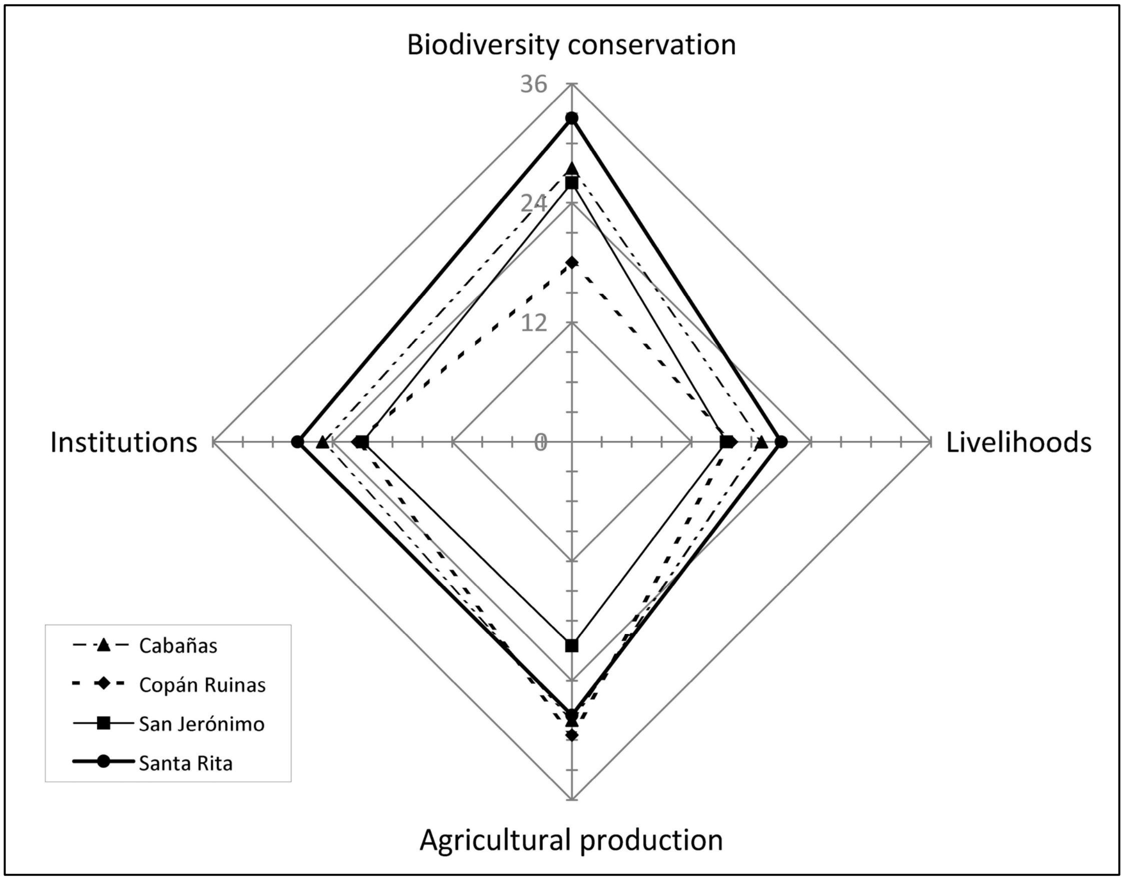


Figure 2-4. Spider diagram indicating current performance of each of the four municipalities in the Río Copán watershed with respect to each of the four axes of ecoagriculture: agricultural production, conservation, livelihoods and institutional support. Scores for each axis are reported in dimensionless units based on summing the scores for the five criteria under each ecoagriculture goal. These data are derived from mixed methods including household interviews, vegetation sampling, and land use analyses, as described in the text. The diagram provides a simplified performance metric to help assess existing conditions, set priorities, and establish a baseline against which to measure future progress. This figure is re-drawn from Bejarano 2009.

The landscape evaluation reported answers to each of the 20 Questions individually but also amalgamated outcomes into the four 'axes' of ecoagriculture to help frame stakeholder discussion about landscape priorities (Figure 2-4). This type of synthesis is rife with challenges and value judgments. (How do you weigh each indicator? Can landscape outcomes be traded off against each other, or must some or all objectives be met at a basic level?) But rather than forming an insurmountable barrier, such value questions can provide a starting point for dialogue about synergies and tradeoffs among disparate objectives.

In addition to providing a baseline assessment of landscape performance, the evaluation also explored various policy alternatives for improving outcomes to several of the 20 Questions. Framing policy analysis in terms of the 20 Questions is an alternative to sectoral analyses that predict the direct results of interventions but not their indirect or feedback effects. For example, Sanfiorenzo (2008) conducted landscape modeling to evaluate the effects on biodiversity of proposed policies for reducing erosion, landslides, and water pollution in the landscape, which hinder progress toward several of the MDGs. A baseline analysis evaluated forest patch size, fragmentation, and functional connectivity of the existing landscape from the perspective of the genus *trogon*—forest-dependent birds that are also highly sought after by ecotourists. Forest cover comprised 25% of the 598 square kilometer watershed, but this habitat was fragmented into 145 isolated patches. Sanfiorenzo (2008) then evaluated the effects of three potential policies: 1) enforcing the Honduran law to protect 10 m forested buffers alongside all rivers and streams, 2) converting steep slopes (14-40%) to agroforestry systems such as shaded coffee or pasture with high tree density, and 3) revegetating all very steep slopes (>40%) to natural forest or timber plantations. The models revealed that riparian buffers would decrease the number of isolated forest fragments from 145 to less than 40, while the three policies in combination would increase suitable trogon habitat from 22% to 38% of the landscape. The analysis not only sheds light on several of the 20 Questions (e.g., C1, C4, P4, L3, and L5; see Box 2-2); it also identifies the most promising target areas for restoration.

Reflecting on the LM evaluation in Copán, the approach at first glance seems similar to standard assessment methods, such as Rapid Rural Appraisal, that combine interviews and other forms of baseline data collection to identify needs and priorities. However, on closer examination, several key differences emerge. One is the use of an integrative framework to steer communities and field technicians to consider the possible importance or feedback effects of issues that have been neglected locally. Second is an emphasis on land use and landscape patterns as durable—though manageable—underlying drivers of many of the socioeconomic themes that are often the focus of rural appraisals. Third is a focus on quantitative indicators that can be cost-effectively measured on a regular basis to track the direct and indirect effects of landscape interventions, as well as the feedbacks between these interventions and exogenous policy and market forces.

Based on the cost of the initial assessment, repeating LM evaluations every 2-3 years as part of a landscape planning and adaptive management program would cost approximately \$50,000 to \$70,000. As the MANCOSARIC has learned, however, such up-front investment can pay for itself many times over by helping to attract and target foreign assistance to communities that have a clear vision for the future and understand which projects and interventions will help them achieve this vision.

2.7 CASE STUDY 2: APPLYING THE LM IN KIJABE, KENYA

The second case study documents the use of the 20 Questions and two participatory evaluation tools in the Landscape Measures Resource Center as a basis for initiating dialogue about landscape dynamics and priorities. The case takes place in the Kijabe landscape on the eastern slopes of the Aberdare Mountains, just northwest of Nairobi, Kenya. Here lies the Kikuyu Escarpment Forest, a hotspot for plant and bird diversity that is also the watershed supplying water to more than a million of Nairobi's inhabitants. The landscape is a mosaic of ancient forests, tree plantations, and diverse agricultural plots, supporting a mixed agricultural economy and extensive tea production. However, recent population growth had led to increased pressure on the forest: cattle and sheep were killing seedlings, residents were cutting wood for charcoal production, and illegal loggers were exploiting the forest.

Recognizing the dependence of local livelihoods on the health of the forest, local leaders, with financial support from BirdLife International, established the Kijabe Environmental Volunteers (KENVO) to educate, train, and support local residents in forest conservation and restoration efforts. KENVO began with a seedling initiative that organized landscape residents to plant and protect native trees to restore the ailing forest. By raising and selling the trees to KENVO, women and youth groups were able to earn income while supplying their farms with useful agroforestry trees. Meanwhile, a growing contingent of innovative farmers was building on KENVO's ideas by diversifying and intensifying their production systems to integrate small animals, bees, and fish farming and by utilizing organic wastes to enhance soil fertility. As these farmers increased their incomes and were able to realize prized education and health benefits for their families, others took notice and the ideas began to spread.

By 2007, KENVO had enjoyed significant success, ridding the area of illegal loggers and spawning numerous community-led forest restoration groups where none had existed before. KENVO's founder, David Kuria, remarked on residents' deep pride in these achievements but emphasized that "for conservation in this area to succeed, communities must continue to benefit."

2.7.1 Participatory landscape evaluation

In this context, KENVO was interested in using the LM to re-assess its strategic direction and provide local stakeholders a forum in which to express their needs and priorities. For its part, Ecoagriculture Partners' LM team sought to apply and evaluate the Landscape Performance Scorecard and Institutional Performance Scorecard tools, which it had recently designed for a Ugandan landscape with similar land use and livelihood dynamics. Both scorecards are based on the 20 Questions and offer a format for discussion and participatory evaluation of these questions to initiate dialogue on landscape dynamics.

KENVO convened a group of 22 stakeholders for a five-hour workshop at its strategically-located office and meeting space in the landscape. About two-thirds of the participants were farmers, while others represented public agencies of forestry and natural resources, agriculture and livestock, and social services as well as leaders of church groups and other local organizations. KENVO's multi-lingual professional staff issued the invitations, arranged for teas and lunch to be provided, and co-led the workshop with the LM team. The LM team prepared color-coded copies of scorecards, data capture forms, and written instructions for the exercise.

The group began by translating each of the 20 Questions (see Box 2-2) into terms that made sense in the Kijabe landscape, a process that involved discussing various local examples that were meaningful to participants. Next, each participant filled out a copy of the Landscape Performance Scorecard, which required evaluating each question on a five-point scale for the Kijabe landscape. The group then prepared for the institutional scoring exercise by brainstorming to identify all public, private, civic, or hybrid organizations that they considered to have an effect on the landscape's current status and future direction. Using a similar scorecard format, participants scored each institution based on its fulfillment of its mission and its contribution to the objectives articulated in the 20 Questions. The meeting facilitators entered all scorecard data into a Microsoft Excel data capture form, computed summary results, and generated illustrative spider diagrams of the results, all of which were projected for the group to view. Discussion ensued about the results and what they implied about the landscape's current balance among conservation, food production, and livelihood performance. Following the meeting, a group of Kenyan participants met with the LM team to review the workshop process, assess the relevance and usability of the scoring tools, and determine whether the landscape perspective was helpful or viewed by participants as abstract or irrelevant.

2.7.2 Outcomes of the landscape evaluation

The landscape evaluation process exceeded the expectations of KENVO and the LM team in three respects. First, the level of engagement and application of participants' knowledge to the tasks at hand were impressive and inspiring. Participants devoted much more time and effort to the institutional scoring than had been anticipated, producing an institutional map of the landscape that KENVO and its members have used subsequently in publications, presentations, and discussions with collaborators.

Second, the exercise stimulated creative thinking and discussion about strategic new directions for KENVO's activities. For example, the landscape scorecard made evident the fact that Kijabe was performing better with respect to conservation goals than livelihood goals. Reflecting on this result, participants realized that external investment in the landscape had been driven for some time by the agendas of conservation groups whose aims were to restore forest habitat for wildlife. While participants were proud of their conservation achievements, they articulated a need to pursue parallel improvements in food production and livelihood security. This discussion generated a list of concrete steps toward which the group agreed to organize, including improving farmers' access to markets for specialty products and securing credit for new enterprises. Results of the institutional scoring exercise stimulated participants to target private sector organizations—particularly companies dealing in agricultural products—for recruitment into KENVO's activities. They also used the newly-created institutional map to explore the potential of linking organizations to create agri-eco-tourism enterprises that would benefit entrepreneurs and the community by taking advantage of the landscape's strategic location and dramatic views into the rift valley.

A third outcome of the exercise was KENVO's decision to invest in the development of additional tools and analyses for assessing landscape performance and promoting 'landscape literacy' among residents and stakeholders. This decision stemmed partly from a growing realization—supported by the landscape scoring process—that important conservation benefits and other ecosystem services were being provided in the agricultural mosaic itself, not just in the Kikuyu forest. With encouragement and a modest seed grant from Ecoagriculture Partners, KENVO's leaders generated sufficient resources to commission the National Museums of Kenya to conduct a biodiversity inventory in the agricultural portions of the landscape to complement the previous inventory of the forest. KENVO also commissioned a socioeconomic study of farming households to increase their understanding of local livelihood strategies and generate baseline information against which change could be measured over time. In addition, KENVO worked with the Ecoagriculture Working Group at Cornell University to create a land use/land cover map that they could used to communicate with residents about land use dynamics and opportunities for forest restoration to provide conservation and livelihood benefits.

2.7.3 Conclusion

The post-workshop evaluation revealed that the landscape and institutional scoring tools—and the process by which they were implemented—were relevant and worthwhile. Participants were visibly engaged throughout the workshop and contributed impressive knowledge and insight from their individual perspectives. The discussion and use of the scorecards ran smoothly, with no apparent confusion, and the resulting baseline evaluations were judged to be credible by the people and organizations who participated. At the same time, however, the landscape evaluation did not merely reiterate what participants already knew. New information was brought forward through the multi-stakeholder forum and, more importantly, participants were able to organize and understand existing knowledge in new ways that made the trajectory, opportunities, and threats in the Kijabe landscape more apparent. This new understanding helped generate ideas about KENVO's future priorities for landscape level planning and management while solidifying KENVO's commitment to continuing to invest in strategic landscape information to support such planning and management. A further measure of impact, to be assessed later, would be KENVO's repeat use of the scorecard tools to evaluate changes in landscape performance attributable to its programs and to other factors.

2.8 TOWARD MAINSTREAMING OF LANDSCAPE APPROACHES

The case studies from Honduras and Kenya illustrate the ways in which landscape-scale negotiation, planning, and monitoring will be crucial for meeting the MDGs on a sustained basis in rural landscapes. As documented in this chapter, landscape approaches have begun to be used in recent years, but further work is needed to continue to develop the science and practice of multi-stakeholder, multi-objective adaptive management at the landscape scale. Mainstreaming landscape approaches will also require the adoption of favorable policy, market, and institutional frameworks at the national and international levels. Many of these changes will entail substantial re-allocations of power, authority, and resources, and could take years or decades to achieve. Key actions needed to support landscape approaches include:

1) Shift power over land and resource management to landscape-level institutions that have (or can develop) the capacity to carry out such management. Continued devolution of government authority will be an important part of this process in many countries.
2) Legitimize and provide sustained support for multi-stakeholder processes in landscapes. Re-orient government line agencies toward a service role in which they provide technical resources and facilitation for these processes and subsequently incorporate landscape-level goals and plans into agency priorities and programs. Recognize roles for business, NGOs, farmers' organizations and citizen groups in implementing action and tracking progress based on these plans.
3) Expand opportunities for training and knowledge sharing around landscape-scale analysis, planning and monitoring, moving beyond fixed-curriculum extension to include demand-driven programs and peer-to-peer networks, with learning across sectors. Support action learning through partnerships between practitioners and researchers.

4) Clarify and adjust land and resource tenure arrangements so that households and communities are motivated and able to implement concepts or plans that emerge from landscape-level adaptive management processes.
5) Create more equitable approaches to the governance of natural resources so that corporate and government interests are required to participate in multi-stakeholder planning processes rather than shortcutting such negotiations through inside channels. This applies to both common-pool resources such as forests and oceans and privately-owned resources whose management affects public goods like water supply and biodiversity.
6) Eliminate market-distorting policies and subsidies that hinder evidence-based management of water, soil, crops, and land. Establish markets for ecosystem services to internalize externalities associated with the management of rural landscapes, and encourage public and private procurement of agricultural products from farmers using ecoagriculture practices.
7) Re-align the priorities of government agencies, donors and NGOs to incorporate environmental sustainability and ecosystem management into agricultural and rural development programs, and to track human welfare in a way that accounts for the stocks and flows of natural capital that support rural livelihoods.

Historically, the link between environmental sustainability and the wealth of rural communities has been widely ignored or neglected, especially in the fertile, productive landscapes that supply much of the world's food. Technological innovations, inexpensive farm inputs, large subsidies from nature, and the relief valve of the agricultural frontier have all held crisis at bay in many rural landscapes. Going forward, however, this picture is likely to change. As population pressures mount, suitable vacant land diminishes, and productivity gains from technological innovation plateau in post-Green Revolution areas, healthy ecosystems will become increasingly fundamental to human wellbeing. As the margin of error for meeting livelihood needs in rural landscapes shrinks, the demand for effective landscape approaches will grow. Acting now to develop the science, the tools, and the institutional support mechanisms for landscape-scale adaptive management will ensure that such processes are fully functional at the time they are most needed.

LITERATURE CITED

Barrett, C.B., and B.M. Swallow. 2006. Fractal poverty traps. World Development 34: 1-15.

Barrow, E., D. Timmer, S. White, and S. Maginnis. 2002. Forest landscape restoration: building assets for people and nature, experience from East Africa. IUCN, Gland, Switzerland and Cambridge, UK.

Bell, S., and S. Morse. 2001. Breaking through the glass ceiling: who cares about sustainability indicators? Local Environment 6: 291-309.

Bejarano, L.F. 2009. Evaluación metodológica del enfoque de Ecoagricultura para medir el desempeño de un paisaje con matriz agropecuaria en la subcuenca del Río Copán, Honduras. Master's thesis. Centro Agronómico Tropical de Investigación y Enseñanza, Turrialba, Costa Rica.

Benedict, M.A., and E.T. McMahon. 2006. Green infrastructure: linking landscapes and communities. Island Press, Washington, D.C.

Bojö, J., K. Green, S. Kishore, S. Pilapitiya, and R. Reddy. 2004. Environment in poverty reduction strategies and poverty reduction support credits. World Bank Environment Department Paper No. 102. World Bank, Washington, D.C.

Bonney, R., C.B. Cooper, J. Dickinson, S. Kelling, T. Phillips, K.V. Rosenberg, and J. Shirk. 2009. Citizen science: a developing tool for expanding science knowledge and scientific literacy. BioScience 59: 977-984.

Borrini-Feyerabend, G., M.T. Farvar, J.C. Nguinguiri, and V.A. Ndangang. 2000. Co-management of natural resources: organising, negotiating and learning-by-doing. GTZ and IUCN, Kasparek Verlag, Heidelberg, Germany.

Brechin, S.R., P.R. Wilsusen, C.L. Fortwangler, and P.C. West, editors. 2003. Contested nature: promoting international biodiversity with social justice in the 21st century. SUNY Press, Albany, New York.

Buck, L.E., C.C. Geisler, J. Schelhas, and E. Wollenberg, editors. 2001. Biological diversity: balancing interests through adaptive collaborative management. CRC Press, Boca Raton, Florida.

Buck, L.E., J.C. Milder, T.A. Gavin, and I. Mukherjee. 2006. Understanding ecoagriculture: a framework for measuring landscape performance. Ecoagriculture Discussion Paper #2. Ecoagriculture Partners, Washington, D.C.

Buck, L.E., and S.J. Scherr. 2009. Building innovation systems for managing complex landscapes. Pages 164-186 in K.M. Moore, editor. The sciences and art of adaptive management: innovating for sustainable agriculture and natural resource management. Soil and Water Conservation Society, Ankeny, Iowa.

Campbell, B., J.A. Sayer, P. Frost, S. Vermeulen, M.R. Pérez, A. Cunningham, and R. Prabhu. 2001. Assessing the performance of natural resource systems. Conservation Ecology 5(2): Article 22.

Carney, D. editor. 1998. Sustainable rural livelihoods: what contribution can we make? Department for International Development, London.

Cash, D.W., W.C. Clark, F. Alcock, N.M. Dickson, N. Eckley, D.H. Guston, J. Jäger, and R.B. Mitchell. 2003. Knowledge systems for sustainable development. Proceeding of the National Academy of Science 100: 8086-8091.

CGIAR [Consultative Group on International Agricultural Research System Review Secretariat]. 1998. Third system review of the Consultative Group on International Agricultural Research (CGIAR). CGIAR, Washington, D.C.

CIFOR [International Center for Forestry Research]. 1999. Guidelines for developing, testing and selecting criteria and indicators for sustainable forest management. CIFOR Toolbox, Part 1. CIFOR, Bogor, Indonesia.

Colfer, C.J.P. 2005. The complex forest: communities, uncertainty, and adaptive collaborative management. Resources for the Future and CIFOR, Washington, D.C.

Collins, K., C. Blackmore, D. Morris, and D. Watson. 2007. A systematic approach to managing multiple perspectives and stakeholding in water catchments: some findings from three UK case studies. Environmental Science & Policy 10: 564-574.

COMACO [Community Markets for Conservation]. 2009. COMACO website. Online: www.itswild.org.

Costanza, R., R. d'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R.V. O'Neill, J. Paruelo, R.G. Raskin, P. Sutton and M. van den Belt. 1998. The value of the world's ecosystem services and natural capital. Ecological Economics 25: 3-15.

Daily, G.C. 1997. Nature's services. Island Press, Washington, D.C.

DFID [Department for International Development], European Commission, UN Development Programme, and World Bank. 2002. Linking poverty reduction and environmental management. World Bank, Washington, D.C.

DFID [Department for International Development]. 2006. DFID's approach to the environment. DFID, London.

Diamond, J. 2004. Collapse: how societies choose to fail or succeed. Viking, New York.

Dietz, T., E. Ostrom, and P.C. Stern. 2003. The struggle to govern the commons. Science 302: 1907-1912.

Dramstad, W.E., J.D. Olson, and R.T.T. Forman. 1996. Landscape ecology principles in landscape architecture and land-use planning. Harvard University Graduate School of Design, Cambridge, Massachusetts.

Egoh, B., B. Reyers, M. Rouget, D.M. Richardson, D.C. Le Maitre, and A.S. van Jaarsveld. 2008. Mapping ecosystem services for planning and management. Agriculture, Ecosystems and Environment 127: 135-140.

Fairhead, J. 2004. Achieving sustainability in Africa. Pages 292-306 in R. Black and H. White, editors. Targeting development: critical perspectives on the Millennium Development Goals. Routledge, London.

Fischer, J., D.B. Lindenmayer, and A.D. Manning. 2006. Biodiversity, ecosystem function, and resilience: ten guiding principles for commodity production landscapes. Frontiers in Ecology and the Environment 4: 80-86.

Folke, C., S. Carpenter, T. Elmqvist, L. Gunderson, C.S. Holling, and B. Walker. 2002. Resilience and sustainable development: building adaptive capacity in a world of transformations. Ambio 31: 437-440.

Forman, R. T. T. 1995. Land mosaics: the ecology of landscapes and regions. Cambridge University Press, Cambridge, UK.

Frost, P., B. Campbell, G. Medina, and L. Usongo. 2006. Landscape-scale approaches for integrated natural resource management in tropical forest landscapes. Ecology and Society 11(2): Article 30.

Gottret, M.V., and D. White. 2001. Assessing the impact of integrated natural resource management: challenges and experiences. Conservation Ecology 5(2): Article 17.

Harvey, C.A. 2008. Designing agricultural landscapes for biodiversity conservation. Pages 146-165 in S.J. Scherr and J.A. McNeely. Farming with nature. Island Press, Washington, D.C.

Hinchcliffe, F., J. Thompson, J.N. Pretty, I. Guijt, and P. Shah. 1999. Fertile ground: the impacts of participatory watershed management. Intermediate Technology Publications, London.

Holling, C.S. 1978. Adaptive environmental assessment and management. Wiley, New York.

ICARDA [International Center for Agricultural Research in the Dry Areas]. 2005. Current definition of INRM. Online: www.icarda.cgiar.org/INRMsite/index.htm.

Ison, R., N. Röling, and D. Watson. 2007. Challenges to science and society in the sustainable management and use of water: investigating the role of social learning. Environmental Science & Policy 10: 499-511.

Izac, A.N., and P.A. Sanchez. 2001. Towards a natural resource management paradigm for international agriculture: the example of agroforestry research. Agricultural Systems 69: 5-25.

Kerr, J. with G. Pangare and V.L. Pangare. 2002. Watershed development projects in India: an evaluation. IFPRI Research Report 127. International Food Policy Research Institute, Washington, D.C.

Kloppenburg, J., J. Hendrickson, and G.W. Stevenson. 1996. Coming in to the foodshed. Agriculture and Human Values 13: 33-42.

LAC-Net [Regional Model Forest Network for Latin America and the Caribbean]. 2006. Estándar de principios, criterios e indicadores de la Red Regional de Bosques Modelo para América Latina y el Caribe. Online: www.bosquesmodelo.net/foro/documentos/Estandares_principios_criterios.doc.

Lal, P., H. Lim-Applegate, and M. Scoccimarro. 2001. The adaptive decision-making process as a tool for integrated natural resource management: focus, attitudes, and approach. Conservation Ecology 5(2): Article 11.

Lassoie, J.P., R.K. Moseley, and K. E. Goldman. 2006. Ground-based photomonitoring of ecoregional ecological changes in northwestern Yunnan, China. Pages 140-151 in C. Aguirre-Bravo, P.J. Pellicane, D.P. Burns, and S. Draggan, editors. Monitoring science and technology symposium: unifying knowledge for sustainability in the Western Hemisphere. Proceedings RMRS-P-42CD. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado.

Laumonier, Y., R. Bourgeois, and J. Pfund. 2008. Accounting for the ecological dimension in participatory research and development: lessons learned from Indonesia and Madagascar. Ecology and Society 13(1): Article 15.

Leach, M., R. Mearns, and I. Scoones. 1999. Environmental entitlements: dynamics and institutions in community-based natural resource management. World Development 27: 225-247.

Lee, K.N. 1993. Compass and gyroscope: integrating science and politics for the environment. Island Press, Washington, D.C.

Leeuwis, C., and R. Pyburn. 2002. Social learning in rural resource management. Pages 11-22 in C. Leeuwis and R. Pyburn, editors. Wheelbarrows full of frogs: social learning in rural resource management. Koninklijke Van Gorcum, Assen, The Netherlands.

Lele, S.M. 1991. Sustainable development: a critical review. World Development 19: 607–621.

Lindenmayer, D., R.J. Hobbs, R. Montague-Drake, J. Alexandra, A. Bennett, M. Burgman, P. Cale, A. Calhoun, V. Cramer, P. Cullen, D. Driscoll, L. Fahrig, J. Fischer, J. Franklin, Y. Haila, M. Hunter, P. Gibbons, S. Lake, G. Luck, C. MacGregor, S. McIntyre, R. Mac Nally, A. Manning, J. Miller, H. Mooney, R. Noss, H. Possingham, D. Saunders, F. Schmiegelow, M. Scott, D. Simberloff, T.

Sisk, G. Tabor, B. Walker, J. Wiens, J. Woinarski, and E. Zavaleta. 2008. A checklist for ecological management of landscapes for conservation. Ecology Letters 11: 78-91.

Liu, J., T. Dietz, S.R. Carpenter, C. Folke, M. Alberti, C.L. Redman, S.H. Schneider, E. Ostrom, A.N. Pell, J. Lubchenco, W.W. Taylor, Z. Ouyang, P. Deadman, T. Kratz, and W. Provencher. 2007. Coupled human and natural systems. Ambio 36: 639-649.

LMI [Landscape Measures Initiative]. 2009. The Landscape Measures Resource Center. Ecoagriculture Partners, Washington, D.C., and Cornell Ecoagriculture Working Group, Ithaca, New York. Online: www.landscapemeasures.org.

López-Ridaura, S., H. van Keulen, M.K. van Ittersum, and P.A. Leffelaar. 2005. Multiscale methodological framework to derive criteria and indicators for sustainability evaluation of peasant natural resource management systems. Environment, Development and Sustainability 7: 51-69.

Lynam, T., W. de Jong, D. Sheil, T. Kusumanto, and K. Evans. 2007. A review of tools for incorporating community knowledge, preferences, and values into decision making in natural resources management. Conservation Ecology 12(1): Article 5.

MA [Millennium Ecosystem Assessment]. 2005. Ecosystems and human well-being: synthesis. Island Press, Washington, D.C.

Maarleveld, M., and C. Dangbegnon. 1999. Managing natural resources: a social learning perspective. Agriculture and Human Values 16: 267-280.

McNeely, J.A., and S.J. Scherr. 2003. Ecoagriculture. Island Press, Washington D.C.

McShane, T.O., and M.P. Wells, editors. 2004. Getting biodiversity projects to work: towards better conservation and development. Columbia University Press, New York.

Miller, K., E. Chang, and N. Johnson. 2001. Defining common ground for the Mesoamerican Biological Corridor. World Resources Institute, Washington, D.C.

Molnar, A., S.J. Scherr and A. Khare, 2007. Community stewardship of biodiversity. Pages 268-285 in S.J. Scherr and J. McNeely, editors. Farming with nature: the science and practice of ecoagriculture. Island Press, Washington, D.C.

Munda, G. 2005. "Measuring sustainability": a multi-criterion framework. Environment, Development, and Sustainability 7: 117-134.

Murgueitio, E., M. Ibrahim, E. Ramirez, A. Zapata, C.E. Mejia, and F. Casasola. 2004. Land use on cattle farms: guide for the payment of environmental services. CIPAV, Cali, Colombia.

Nyberg, B. 1999. An introductory guide to adaptive management for project leaders and participants. British Columbia Forest Service, Victoria, Canada.

Olsson, P., C. Folke, and F. Berkes. 2004. Adaptive comanagement for building resilience in social-ecological systems. Environmental Management 34: 75-90.

O'Neill, R.V., C.T. Hunsaker, K.B. Jones, K.H. Riitters, J.D. Wickham, P.M. Schwartz, I.A. Goodman, B.L. Jackson, and W.S. Baillargeon. 1997. Monitoring environmental quality at the landscape scale. BioScience 47: 513-519.

Palm, C.A., S.A. Vosti, P.A. Sanchez, and P.J. Ericksen, editors. 2005. Slash-and-burn agriculture. Columbia University Press, New York.

Parker, D.C., S.M. Manson, M.A. Janssen, M.J. Hoffmann, and P. Deadman. 2003. Multi-agent systems for the simulation of land-use and land-cover change: a review. Annals of the Association of American Geographers 93: 314–337.

Pfund, J.-L., P. Koponen, T. O'Connor, J. Boffa, M. van Noordwijk, and J. Sorg. 2008. Biodiversity conservation and sustainable livelihoods in tropical forest landscapes. Pages 297-322 in R. Lafortezza, J. Chen, G. Sanesi, and T.R. Crow, editors. Patterns and processes in forest landscapes. Springer, New York.

Place, F., and E. Were, editors. 2005. Proceedings of 5th Workshop of the Integrated Natural Resource Management (INRM) Stakeholder Group, October 20-21, 2003. World Agroforestry Centre, Nairobi.

Plummer, R., and D. Armitage. 2007. A resilience-based framework for evaluating adaptive co-management: linking ecology, economics and society in a complex world. Ecological Economics 61: 62-74.

Pretty, J., and H. Ward. 2001. Social capital and the environment. World Development 29: 209-229.

Programa Estado de la Nación. 2008. Estado de la región en desarrollo humano sostenible: un informe desde Centroamérica y para Centroamérica. Programa Estado de la Nación, San José, Costa Rica.

Prüss-Üstün, A., and C. Corvalán. 2006. Preventing disease through healthy environments. Towards an estimate of the environmental burden of disease. World Health Organization, Geneva.

Redford, K.H., and E. Fearn, editors. 2007. Protected areas and human livelihoods. WCS Working Paper No. 32. Wildlife Conservation Society, New York.

Röling, N. 2002. Beyond the aggregation of individual preferences. Pages 25-48 in C. Leeuwis and R. Pyburn, editors. Wheelbarrows full of frogs: social learning in rural resource management. Koninklijke Van Gorcum, Assen, The Netherlands.

Röling, N.G and M.A.E. Wagemakers. 1998. A new practice: facilitating sustainable agriculture. Pages 3-22 in N.G. Röling and M.A.E. Wagemakers, editors. Facilitating sustainable agriculture. Cambridge University Press, Cambridge, UK.

Rosenberg, D.K, B.R. Noon, and E.C. Meslow. 1997. Biological corridors: form, function and efficacy. BioScience 47: 677-687.

Sachs, J.D., and W.V. Reid. 2006. Investments toward sustainable development. Science 312: 1002.

Salafsky, N., R. Margoluis, and K. Redford. 2001. Adaptive management: a tool for conservation practitioners. Biodiversity Support Program, Washington D.C.

Sanderson, S. 2005. Poverty and conservation: the new century's "peasant question?" World Development 33: 323-332.

Sandker, M., A. Suwarno, and B.M. Campbell. 2007. Will forests remain in the face of oil palm expansion? Simulating change in Malinau, Indonesia. Ecology and Society 12(2): Article 37.

Sanfiorenzo, A.R. 2008. Contribución de diferentes arreglos silvopastoriles a la conservación de la biodiversidad, mediante la provisión de hábitat y conectividad en el paisaje de la sub-cuenca del Río Copán, Honduras. Master's thesis. Centro Agronómico Tropical de Investigación y Enseñanza, Turrialba, Costa Rica.

Sayer, J., and B. Campbell. 2004. The science of sustainable development: local livelihoods and the global environment. Cambridge University Press, Cambridge, UK.

Sayer, J., B. Campbell, L. Petheram, M. Aldrich, M. Ruiz Perez, D. Endamana, Z.N. Dongmo, L. Defo, S. Mariki, N. Doggart, and N. Burgess. 2007. Assessing environment and development outcomes in conservation landscapes. Biodiversity and Conservation 16: 2677-2694.

Sayer, J., and L. Buck. 2008. Learning from landscapes. Arborvitae Special. IUCN Forest Conservation Program, Gland, Switzerland.

Scherr, S.J., and J. McNeely. 2008. Biodiversity conservation and agricultural sustainability: towards a new paradigm of 'ecoagriculture' landscapes. Philosophical Transactions of the Royal Society B 363: 477-494.

Scherr, S., J.A. McNeely, and S. Shames. 2009. Ecoagriculture: agriculture, environmental conservation, and poverty reduction at a landscape scale. Pages 64-

86 in N. Gillis and P. Galizzi editors. The role of the environment in poverty alleviation. Fordham University Press, New York.

Selman, P.H. 2002. Multi-function landscape plans: a missing link in sustainability planning? Local Environment 7: 283-294.

Sepúlveda, S., A. Rodríguez, R. Echeverri, and M. Portilla. 2003. El enfoque territorial del desarrollo rural. Instituto Interamericano de Cooperación para la Agricultura, San José, Costa Rica.

Sterman, J.D. 2000. Business dynamics: systems thinking and modeling for a complex world. McGraw-Hill, New York.

Steyaert, P., M. Barzman, J. Billaud, H. Brives, B. Hubert, G. Ollivier, and B. Roche. 2007. The role of knowledge and research in facilitating social learning among stakeholders in natural resources management in the French Atlantic coastal wetlands. Environmental Science & Policy 10: 537-550.

Steyaert, P., and J. Jiggins. 2007. Governance of complex environmental situations through social learning: a synthesis of SLIM’s lessons for research, policy and practice. Environmental Science & Policy 10: 575-586.

Swallow, B. 2005. Potential for poverty reduction strategies to address community priorities: case study of Kenya. World Development 33: 301-321.

Terborgh, J. 1999. Requiem for nature. Island Press, Washington, D.C.

Troy, A., and M.A. Wilson. 2006. Mapping ecosystem services: practical challenges and opportunities in linking GIS and value transfer. Ecological Economics 60: 435-449.

UN [United Nations]. 2008. The Millennium Development Goals report. UN, New York.

USFS [United States Forest Service]. 2006. US Forest Service guide to integrated landscape land use planning in Central Africa. Unpublished paper.

van Noordwijk, M., T.P. Tomich, and B. Verbist. 2001. Negotiation support models for integrated natural resource management in tropical forest margins. Conservation Ecology 5(2): Article 21.

WCED [World Commission on Environment and Development]. 1987. Our common future. Oxford University Press, Oxford, UK.

Wollenberg, E., D. Edmunds, and L. Buck. 2000. Using scenarios to make decisions about the future: anticipatory learning for the adaptive co-management of community forests. Landscape and Urban Planning 47: 65-77.

Wood, S., K. Sebastian, and S.J. Scherr. 2000. Pilot analysis of global ecosystems: agroecosystems. International Food Policy Research Institute and World Resources Institute, Washington, D.C.

World Bank and IMF [International Monetary Fund]. 2005. Global monitoring report 2005: Millennium Development Goals: from consensus to momentum. World Bank, Washington, D.C.

World Bank. 2006. Where is the wealth of nations? Measuring capital for the 21st century. World Bank, Washington, D.C.

WRI [World Resources Institute in collaboration with United Nations Development Programme, United Nations Environment Programme, and World Bank]. 2005. World resources 2005: the wealth of the poor—managing ecosystems to fight poverty. World Resources Institute, Washington, D.C.

WRI [World Resources Institute in collaboration with United Nations Development Programme, United Nations Environment Programme, and World Bank]. 2008. World resources 2008: roots of resilience—growing the wealth of the poor. World Resources Institute, Washington, D.C.

CHAPTER 3

EFFECTS OF TREE COVER, LAND USE, AND LANDSCAPE CONTEXT ON BIRD AND BUTTERFLY CONSERVATION IN AN AGRICULTURAL LANDSCAPE IN WESTERN HONDURAS[2]

3.1 ABSTRACT

Previous research has indicated that a substantial portion of native flora and fauna may persist in agricultural mosaics in the Neotropics. The broader applicability of these findings to unstudied locales, however, remains unknown. This study expands the geographic breadth of research on biodiversity conservation in the agricultural matrix by evaluating bird assemblages in eight land uses and butterfly assemblages in six land uses in an agricultural landscape in the Río Copán watershed of western Honduras. The study also investigates the relative influence of land use, plot-scale vegetation characteristics, landscape composition, and landscape structure on these assemblages. Bird species richness and abundance were highest in successional vegetation; intermediate in broadleaf forest, riparian forest, shade-grown coffee, and live fences; and lowest in pasture and pine forest. Butterflies exhibited a contrasting pattern, reaching their greatest species richness in live fences and riparian forest, and their greatest abundance in live fences, riparian forest, and pasture. Continuous habitat descriptors related to vegetation and landscape context were generally less effective

[2] This chapter is adapted from a manuscript of the same title by Jeffrey C. Milder, Dalia Sánchez, Andre Sanfiorenzo, Diego E. Tobar, and Fabrice A.J. DeClerck. My original contributions to this work included designing and overseeing the entire study, including the research protocol, field studies, and the landscape analysis; conducting the data analysis; and preparing the entire manuscript including all figures and tables. D.S. conducted the vegetation field work. D.E.T. and two field assistants conducted the butterfly field work. My field assistant Cliff Cordy conducted the bird field work. A.S. assisted me in conducting the landscape analysis. D.S. and F.A.J.D. provided comments on a draft version of the manuscript that are incorporated into this chapter.

than categorical land use descriptors at explaining faunal assemblage patterns. Likewise, landscape context exhibited an equivocal and generally weak effect on faunal assemblages, and this effect was mostly limited to a 100 m zone of influence. Overall, bird and butterfly distribution in the study landscape is primarily a function of plot-scale habitat features; however, the distinctiveness of these local habitats fosters a high degree of beta diversity, resulting in significant species richness at the landscape scale. The observed bird assemblages were heavily skewed toward common and non-forest-dependent species, indicating that the more heavily managed central portion of the Río Copán watershed examined in this study contributes only modestly to global bird conservation objectives. However, butterfly assemblages contained a substantial proportion of forest-dependent species, which were observed not only in forest plots but also in and around tree cover in pastures and live fences.

3.2 INTRODUCTION

In recent years, conservationists have placed increasing emphasis on the importance of conserving biodiversity and ecosystem services outside of protected areas, including in agricultural landscapes (Harvey et al. 2008a; Gardner et al. 2009). Such efforts are recognized as critical for buffering and connecting nature reserves, maintaining populations of native species, and increasing the resilience of rural regions to climate change and other disturbances (Bennett 2003; Fischer and Lindenmayer 2007; WRI 2008). Considerable evidence also suggests that conservation in agricultural regions can help sustain economically important ecosystem services, including crop pollination, pest control, and water purification (Perfecto et al. 2004; Ricketts et al. 2004, 2008; Zhang et al. 2007).

Advocates of wildlife-friendly farming and ecoagriculture suggest that agriculture conducted at moderate intensity, with deliberate efforts to manage wild biodiversity and agrobiodiversity for multiple benefits, can play a substantial role in global conservations efforts (Vandermeer and Perfecto 2007; Scherr and McNeely 2008). Such strategies may be particularly appropriate on hilly or marginal lands not suitable for large-scale monoculture agriculture, and in areas where substantial rural populations depend on small- and medium-scale farming for their livelihoods (McNeely and Scherr 2003).

The potential for wildlife-friendly farming may be especially great in Central America (Harvey et al. 2008a). This region experienced heavy deforestation from 1950-1990, driven in part by policies and market forces that encouraged conversion of forest to pasture for beef production (Kaimowitz 1996). Although deforestation has now slowed in most parts of Central America, its legacy remains: pastures occupy 27% of the region's land base, more than three times the area occupied by all other agricultural production systems combined (FAOSTAT 2004, cited in Harvey et al. 2005a). Decades later, these pasture-dominated landscapes are in various states of degradation or regeneration, with many of them consisting of fine-grained mosaics of pasture, annual and perennial crops, fallows, and patches of second-growth forest. These landscapes also support rural indigenous and Mestizo populations, many of them living in poverty on small plots of land. In this context, wildlife-friendly approaches to livestock production, such as silvopastoralism, have been advocated as win-win strategies that could simultaneously increase farmer incomes and improve the conservation value of the agricultural matrix (Dagang and Nair 2003; Pagiola et al. 2004). Production of coffee and other perennial crops in structurally diverse agroforestry systems has also been promoted as a way to conserve biodiversity while enabling small farmers to produce cash crops for global markets (Mas and Dietsch 2004).

A substantial body of research has examined the potential for wildlife-friendly farming in Central America by quantifying plant and animal distributions within various farming systems and relating these distributions to variables such as agricultural management, land use, and landscape context. Most of these studies, however, have taken place in only a few fairly well-studied landscapes. For example, more than a decade of research in a rural landscape in Coto Brus, Costa Rica has documented the area's substantial—albeit modified—assemblage of native biodiversity (Daily and Ehrlich 1995; Daily et al. 2001, 2003; Hughes et al. 2002; Horner-Devine et al. 2003; Luck and Daily 2003; Mayfield and Daily 2005; Lindell et al. 2006; Ranganathan et al. 2007; Sekercioglu et al. 2007). Other research hotspots have included coffee agroecosystems in Chiapas, Mexico (Perfecto et al. 2003); mosaics of lowland rainforest and various cropping systems in Los Tuxtlas, Veracruz, Mexico (Estrada et al. 1993, 1994, 1997, 1998, 2000; Estrada and Coates-Estrada 2001, 2002); cacao agroforests and competing land uses in Talamanca, Costa Rica (Reitsma et al. 2001; Suatunce et al. 2003; Harvey et al. 2006a); and several pasture-dominated landscapes in Costa Rica and Nicaragua (Harvey et al. 2005b; Harvey et al. 2006b; Pérez et al. 2006; Medina et al. 2007). This collective body of research has provided important evidence on the contributions and limitations of wildlife-friendly farming as a conservation strategy in Central America. However, the limited geographic range of the research to date—particularly the strong focus on Costa Rica—hinders the ability of scientists and policy makers to derive valid understandings of broader patterns or to transfer knowledge to unstudied locales (Gardner et al. 2009).

This chapter expands the geographic breadth of research on the conservation potential of Central American agricultural mosaics by evaluating the relationships between biodiversity, land use, landscape context, and agroecosystem management in an agricultural landscape in western Honduras. In many ways Honduras epitomizes the challenge of reconciling conservation and rural livelihoods in Central America. With a deforestation rate of 3.1% per year, Honduras has, by far, the highest proportional and absolute rates of forest loss of any Central American nation (FAO 2006). Honduras also has the region's highest proportion of rural people living in poverty (75%) and extreme poverty (63%)—three times the rates in Costa Rica (IFAD 2009). Recent catastrophic destruction caused by Hurricane Mitch and other storms has highlighted the vulnerability of many current land management approaches and the importance of ecosystem-based management to improve the resistance and resilience of rural landscapes and livelihoods to such events.

To explore the conservation potential of agricultural mosaics in western Honduras, the study reported in this chapter evaluated site- and landscape-scale habitat conditions and their influences on bird and butterfly diversity throughout the study landscape. The general hypothesis underlying this approach is that bird and butterfly distributions are a function of habitat conditions created by human management decisions, which in turn influence the fitness of individuals and the viability of populations. As such, I hypothesized that patterns of bird and butterfly distribution would each respond to habitat conditions in consistent ways. Drawing on findings from previous studies in nearby countries (cited above), I further hypothesized that higher levels of tree cover and tree diversity at the plot scale as well as greater forest and tree cover in the surrounding landscape would correspond to higher bird and butterfly abundance and species richness. Finally, based on first principles from landscape ecology, I hypothesized that continuous descriptors related to tree cover, landscape composition, and landscape structure would be more effective than categorical land use descriptors at explaining bird and butterfly distributions (Ferrier 2002; McGarigal and Cushman 2005).

3.3 METHODS

This study relates a set of independent site and landscape management variables to a set of dependent biodiversity conservation variables that characterize bird and butterfly abundance and species richness. As such, the study is a "natural experiment" in which existing patterns of variability in the study landscape substitute for experimentally manipulated samples and controls (Hargrove and Pickering 1992; Oksanen 2001).

3.3.1 Study site

The study was conducted in the Río Copán watershed in the Copán Department of western Honduras (14°47' to 14°54' N, 89°2' to 89°10' W) (Figure 3-1). The watershed comprises an area of 598 square kilometers with elevations ranging from 600 to 1,700 m above sea level. To avoid the confounding effects of elevation, however, this study was confined to the lower, central part of the watershed between 600 and 1,200 m elevation. This zone includes many of the more heavily impacted agricultural portions of the watershed. Local rainfall averages 1,600 mm per year, with a distinct dry season from December to April and a rainy season from May through November. Average daily temperature in the study area is approximately 24°C.

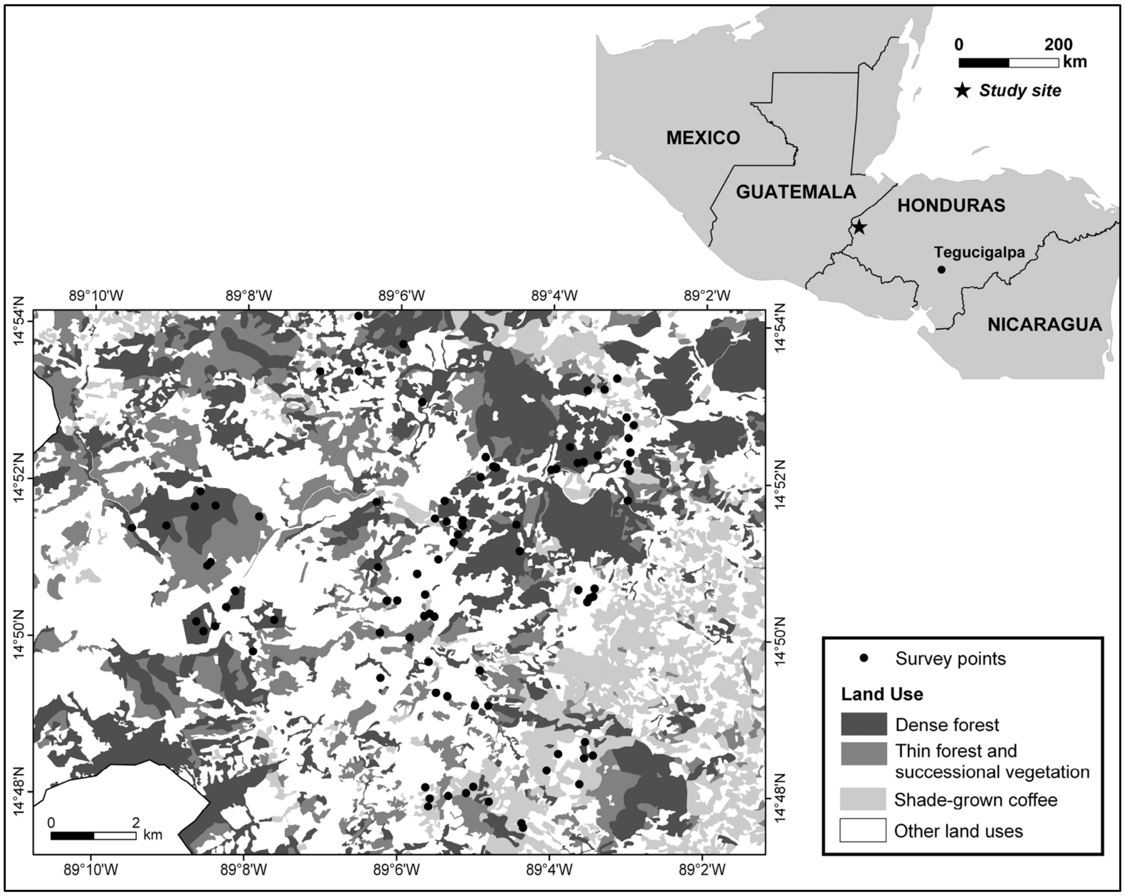


Figure 3-1. Map of the study landscape and its geographic context.

Biogeographically, the landscape is noteworthy in that it occupies a zone of convergence between the Atlantic and Pacific slopes, and contains diverse native ecosystems including humid broadleaf forest, pine/oak forest, and pre-montane moist forest (Sanfiorenzo 2008). The resulting vegetational diversity supports diverse animal assemblages, including at least 382 bird species recorded in Río Copán watershed (Gallardo, unpublished manuscript). In addition, Río Copán is a tributary of the Motagua River, which is considered a global conservation priority in its own right and because the quality of its water strongly affects the health of the Mesoamerican Reef where it empties into the Caribbean Sea in eastern Guatemala (TNC 2007).

The Río Copán watershed consists of a fine-grained mosaic of pasture, small agricultural plots, and natural and semi-natural forest habitats. The watershed contains 5,842 distinct patches (land units defined by different adjacent land uses), for a mean patch size of about 10.2 ha (Sanfiorenzo 2008). The majority of the land is occupied by various types of agriculture, including pasture (40%), shade-grown coffee (24%), and annual crops (1%). About one-fourth of the watershed is forested, but many of the forests are heavily impacted by cattle grazing, firewood harvesting, and timber extraction. The forests themselves are highly variable, and include pine/oak vegetation (7% of the watershed), broadleaf riparian vegetation (6%), broadleaf upland vegetation (8%), and mixed vegetation (3%). Eight percent of the watershed is in forest fallow (successional vegetation), while 3% consists of urban and other land uses (Sanfiorenzo 2008). The watershed contains no nationally or internationally recognized protected areas. The largest patches of broadleaf forest (several patches of approximately 500 ha each) are in the upper reaches of the watershed, outside of the study area. In the central portion of the watershed evaluated in this study, the largest patch of broadleaf forest is 110 ha, while the largest patches of pine forest are 200, 290, and 360 ha.

The Copán Department, which encompasses the Río Copán watershed, has an estimated population density of 101 persons per square kilometer, one of the densest in rural Honduras (Gallardo, unpublished manuscript). Land ownership in the watershed is moderately fragmented, with a large number of small (<1-20 ha) and medium (20-60 ha) farms and a few large farms and cattle and coffee estates. Rural inhabitants generally score low with respect to human development variables: for example, 72% of a representative sample of 91 farmers in the watershed had no more than a primary education (Sánchez 2006). Socioeconomic conditions are significantly worse among the indigenous Mayan population in the watershed than among the Mestizo majority (Bejarano 2009).

Consistent with other regions of Honduras (Barrance et al. 2003) and Central America (Harvey and Haber 1999; Harvey et al. 2008b), farmers in the Río Copán watershed maintain a significant quantity and diversity of trees on their farms. Farmers report that these trees serve a variety of practical functions, including firewood, fence posts, and lumber. Despite these reported livelihood advantages, however, small and medium farms in the watershed were found to have significantly less forest and significantly fewer trees than large farms (Sánchez 2006). These results suggest cultural and/or economic barriers to the wider adoption of potentially wildlife-friendly farming practices in the watershed.

3.3.2 Biodiversity sampling

Field sampling was conducted at 79 points stratified across eight land uses (Table 3-1). Within each land use, half the sample points were located in zones of low tree cover in the surrounding landscape (as defined by total tree cover within 3 km of the sample point) while the other half were located in zones of high surrounding tree cover. Except in a few instances where logistical constraints dictated otherwise, points of any given land use were separated by at least 400 m, and all sample points were separated by at least 200 m. In addition, to minimize the confounding influence of edge effect, all sample points were located at least 40 m from land use boundaries. All 79 points were sampled for birds and vegetation, and a subset of these points were sampled for butterflies. All field sampling protocols—as well as the landscape context analyses described in the next section—were spatially oriented around the 79 central sample points (Figure 3-2).

Table 3-1. Stratification of the 79 sample points across eight land uses. Note that only nine suitable sample points could be identified for the live fence land use.

Land Use	Description	# of Points Sampled
Broadleaf forest	Second-growth broadleaf forest with a closed canopy of mature trees	10
Pine forest	Pine or pine/oak forest with a closed canopy of mature trees	10
Riparian forest	Forest bordering perennial rivers or streams	10
Successional vegetation	Abandoned or fallowed agricultural land containing shrubby or woody vegetation approximately 3-8 years old	10
Shade-grown coffee	Coffee plantation with a tree overstory, typically containing multiple species	10
Pasture with high tree density	Pasture with 15-30% tree canopy cover	10
Pasture with low tree density	Pasture with 5-15% tree canopy cover	10
Live fence	Continuous rows of trees separating pastures or farms, and composed of multiple tree species and strata	9

At each point, a 0.1-ha vegetation quadrat was established, within which the species name and diameter at breast height (dbh: 1.4 m above ground) of all trees ≥5 cm dbh were recorded. Quadrats for most of the land uses consisted of a 20 x 50 m plot situated around the central sampling point. These standard dimensions were modified for the riparian forest and live fence land uses because of their linear form. Riparian forest plots consisted of a 10 x 100 m quadrat of riparian vegetation along one side of the stream. Live fence plots consisted of a 5 x 200 m fence segment. Vegetation sampling was conducted in July 2008 by Dalia Sánchez, an experienced botanist.

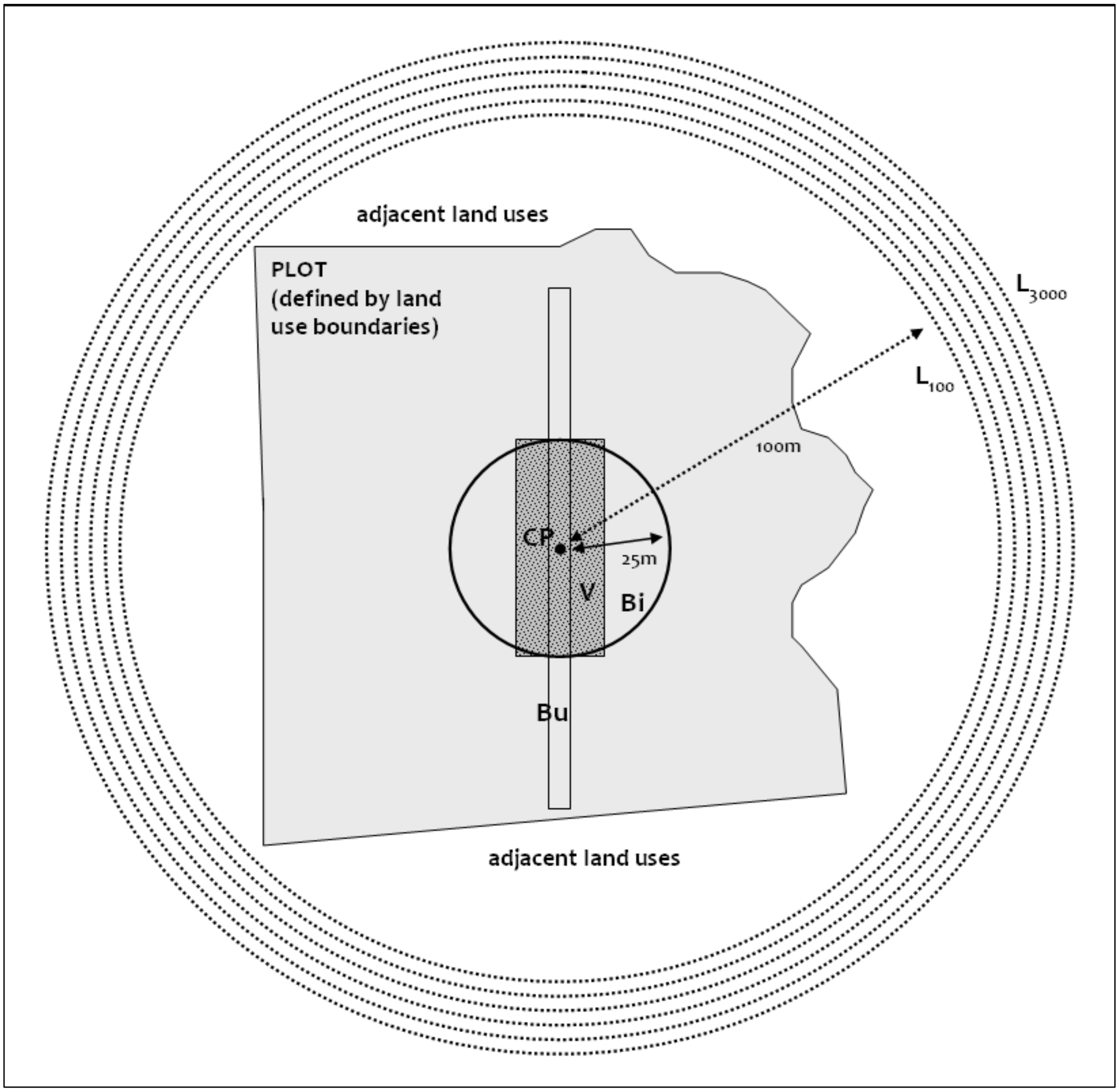


Figure 3-2. Schematic diagram of the sampling protocol. The central point (CP) is the point of orientation for all sampling, including a 20 x 50 m vegetation quadrat (V; shaded dark grey), a 25 m radius bird point count (Bi), and a 120 x 5 m butterfly transect (Bu). For each point, all these samples are contained within a single land use plot. Surrounding the central point, metrics of landscape composition and structure were evaluated for six concentric radii ranging from 100 to 3000 m (L_{100} through L_{3000}; depicted by dotted lines and not drawn to scale).

Bird sampling was conducted by ornithologist Cliff Cordy using a standard point count method in which all birds heard or seen within 25 m of the central sampling point during a 10-minute sampling period were recorded (Ralph et al. 1995). Point counts were conducted from 0600 to 1000 hours only in good weather conditions without heavy rain or strong winds. Upon arriving at each sampling location, the field technician waited five minutes prior to the start of the counting period to minimize the effect of any initial flushing of birds. Point counts were repeated six times per point from November 2008 through February 2009 for a total sampling effort of one hour per location, 10 hours per land use, and 79 hours for the entire study. A field guide (Howell and Webb 1995) and recorded bird songs (R. Gallardo, Enchanted Wings Nature Center, Copán, Honduras) were used to identify unfamiliar birds. Birds observed flying high above the plot were excluded from the analysis when they were judged to be unassociated with the habitat from which they were observed.

Butterflies were sampled at 36 of the 79 bird and vegetation points. These points included six sampling locations within each of the following six land uses: broadleaf forest, riparian forest, successional vegetation, pasture with high tree density, pasture with low tree density, and live fences. Butterflies were sampled by lepidopterist Diego Tobar and two field assistants, who established a 120 m-long transect through the central sampling point at each location (Figure 3-2). Each transect was surveyed by walking its length at a slow, constant pace for 45 minutes and recording all adult butterflies observed within 2.5 m to either side of the transect. Transects were visited between 0800 and 1600 hours on days with good weather. Each transect was surveyed six times between September 2008 and March 2009 for a total sampling effort of 4.5 hours per location, 27 hours per land use, and 162 hours for the entire study. To avoid time-of-day biases, both morning and afternoon visits were conducted at each sampling location. Individuals that could not be identified by sight were captured by net for later identification. Field guides, keys, and illustrations were used to identify unfamiliar species (DeVries 1987; DeVries 1997; Tobar et al. 2007).

3.3.3 Landscape analysis

To evaluate the influence of ecological context on animal assemblages, I quantified patterns of landscape composition and structure at six scales (concentric radii of 100, 200, 400, 800, 1500 and 3000 m) surrounding each sample point (Figure 3-2). I selected a set of six landscape metrics according to following criteria: 1) the metrics follow directly from the hypotheses posed earlier, in the introduction to this chapter; 2) they have a strong ecological relation to habitat quality for Neotropical birds and butterflies; and 3) they have performed well in prior studies of this type (Table 3-2). By circumscribing the set of landscape metrics and tying them to specific hypotheses, I was able to reduce the degree of multicollinearity often found among sets of landscape metrics and minimize the risk of drawing spurious conclusions from metrics with a weak theoretical or empirical basis (Hargis et al. 1998; Tischendorf 2001; Li and Wu 2004).

Three of the six landscape metrics were based on a recently created land use map of the watershed (Sanfiorenzo 2008). This map classified the watershed into 18 discrete land uses according to visual interpretation and systematic field verification of a 1 m resolution IKONOS satellite image acquired during February 2007. Based on this map, I used the software program FRAGSTATS (version 3.3; McGarigal et al. 2002) to calculate the percentage of the land within each radius surrounding each sample point that consisted of forested land uses and tree-covered land uses (see Table 3-2 for further explanation of these categories). In addition, I calculated contrast-weighted edge density to quantify graininess and habitat diversity in the area surrounding each sample point.

Table 3-2. Metrics of landscape composition and structure calculated for each of six concentric radii (from 100 to 3000 m) surrounding each sample point.

Metric	Type	Data source	Description
%_FOR	Composition	Land use map	Percentage of land in the surrounding radius consisting of closed-canopy broadleaf, pine/oak, or riparian forest.
%_TC	Composition	Land use map	Percentage of land in the surrounding radius with at least 15% tree cover (including all forest types, shade-grown coffee, successional vegetation, pasture with high tree density, and live fences).
CWED	Structure	Land use map	Contrast-weighted edge density (McGarigal et al. 2002). This metric quantifies the graininess of the landscape and the degree of contrast between adjacent patches.
NDVI	Composition	Multi-spectral imagery	Mean NDVI value within the radius of analysis.
TCB	Composition	Multi-spectral imagery	Mean tasseled-cap brightness value within the radius of analysis.
TCW	Composition	Multi-spectral imagery	Mean tasseled-cap wetness value within the radius of analysis.

I also calculated the Normalized Difference Vegetation Index (NDVI), tasseled-cap wetness, and tasseled-cap brightness to quantify landscape composition based on a 15 m resolution multi-spectral ASTER image acquired during February 2007. NDVI uses the red and near-infrared spectral bands to evaluate the level of green vegetation in an area, and has been used as a proxy for vegetation characteristics such as leaf area index (Liang 2004; Pettorelli et al. 2005). The Kauth-Thomas transformation, known more commonly as the tasseled-cap transformation, scales down the visible, near-infrared, and short-wave infrared bands of multi-spectral imagery (a total of nine bands in the case of ASTER imagery) to three axes of variation corresponding to the ground properties of brightness, wetness, and greenness (Kauth and Thomas 1976; Crist and Kauth 1986; Yarbrough et al. 2005). The tasseled-cap transformation has been used successfully to monitor vegetation properties and, in one instance, as a proxy for Neotropical bird assemblages (Ranganathan et al. 2007). Since the tasseled-cap greenness metric should be closely related to NDVI, only brightness and wetness were used in this study. See Appendix A for a detailed description of the method for processing the ASTER imagery to calculate these indices.

3.3.4 Data analysis

I analyzed the raw field data to provide descriptive statistics of plant and animal assemblages at each sampling point. For vegetation, I calculated tree species richness, tree density, and total basal area (as derived from dbh measurements). For birds and butterflies, I calculated species richness and abundance. I also partitioned the bird and butterfly species observed in each land use according to their degree of forest dependence, as classified by previous studies (for birds: Stiles 1985 supplemented by natural history information from and Stiles and Skutch 1989 and Howell and Webb 1995; for butterflies: DeVries 1987, 1997). To attain normally distributed data and avoid skewed results due to large groups of flocking birds observed at some points, I natural log (ln)-transformed all abundance data.

I used one-way ANOVA to evaluate differences in bird and butterfly species richness and ln-adjusted abundance among different land uses (all of these variables met the assumptions for ANOVA). Tukey's honestly significantly different (hsd) test was used to evaluate the statistical significance of these differences. I also generated similarity matrices to quantify the percent overlap in species composition among land uses.

I used simple linear regression to evaluate the relationships between faunal assemblage characteristics and each of the continuous vegetation and landscape context variables. Because many of the landscape context variables were significantly correlated with plot-scale land use, I used ANCOVA to evaluate the significance of each landscape variable at each scale of analysis once the effect of land use was considered.

Finally, I used multiple regression to identify the most important combinations of habitat descriptors for each faunal assemblage response variable. This process began by using stepwise regression to identify the best multivariate models at each scale, which were defined as those models that most fully yet parsimoniously explained the variability in the response variable. Three criteria were used to select the best overall model: 1) the model had the lowest, or nearly the lowest, Akaike information criterion (AIC) value (Burnham and Anderson 2002); 2) individual variables in the model were significant at $\alpha = 0.10$ or better; and 3) variables in the model had little multicollinearity (Pearson's $r < 0.40$). In cases where all three criteria could not be met, I selected the model that provided the best overall compromise for satisfying these considerations. Finally, I compared the predictive power of the categorical (land use) and continuous habitat descriptors for explaining patterns of animal distribution. All statistical analyses were conducted in JMP (version 8.0; SAS Institute 2008).

3.4 RESULTS

3.4.1 Tree, bird, and butterfly abundance and species richness

The study sampled a total of 2,211 trees of 145 species. The five most common species were *Pinus oocarpa*, *Gliricidia sepium*, *Quercus sapotifolia*, *Guazuma ulmifolia*, and *Bursera simaruba*. Patterns of tree density, species richness, and basal area were significantly different among the eight land uses evaluated (ANOVA, $p <$

0.0001 for all three variables) but these three factors did not co-vary consistently (Table 3-3). For instance, all three variables were comparatively high in riparian forests and comparatively low in both pasture land uses. However, shade coffee plantations had relatively high tree density, but low species richness and basal area. Conversely, pine forests had high basal area but moderate tree density and low diversity. These findings emphasize the importance of evaluating multiple attributes of the vegetation assemblages in the study landscape. Across all land uses, mean tree species richness per plot was only a small fraction of total richness per land use, indicating a high level of beta diversity across the landscape.

The study sampled a total of 3,939 birds of 139 species, and 5,287 butterflies of 119 species (Table 3-3). The five most common bird species were *Dives dives*, *Dendroica virens*, *Psarocolius wagleri*, *Dendroica magnolia*, and *Saltator atriceps*. The five most common butterfly species were *Hermeuptychia hermes*, *Anartia Fatima*, *Eurema daira*, *Eurema nise*, and *Mechanitis polymnia*. Bird species richness was strongly correlated with bird abundance at the plot level ($r = 0.86$). Butterfly species richness and abundance were also well correlated ($r = 0.67$), although this relationship did not hold true in pasture with low tree density, which had among the highest butterfly abundances but the lowest species richness. Measures of bird richness and abundance were uncorrelated with measures of butterfly richness and abundance ($r = 0.07$ and 0.01, $p = 0.70$ and 0.94 for species richness and abundance, respectively).

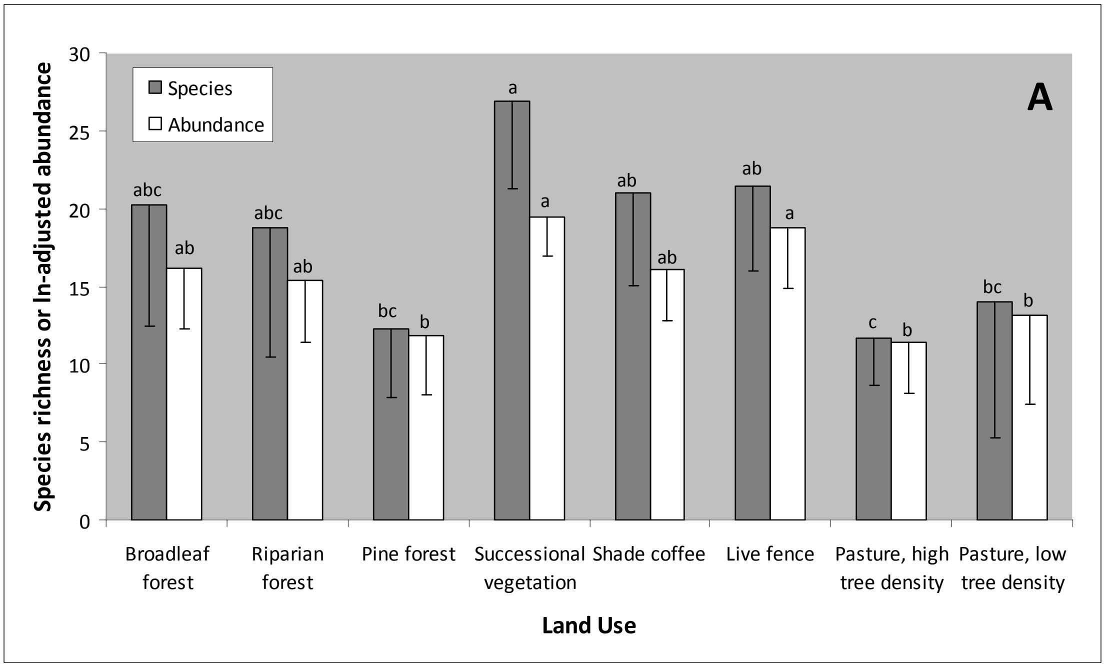
Table 3-3. Total abundance and species richness of trees, birds, and butterflies in each of the land uses surveyed. For butterflies, “ne” indicates a land use that was not evaluated.

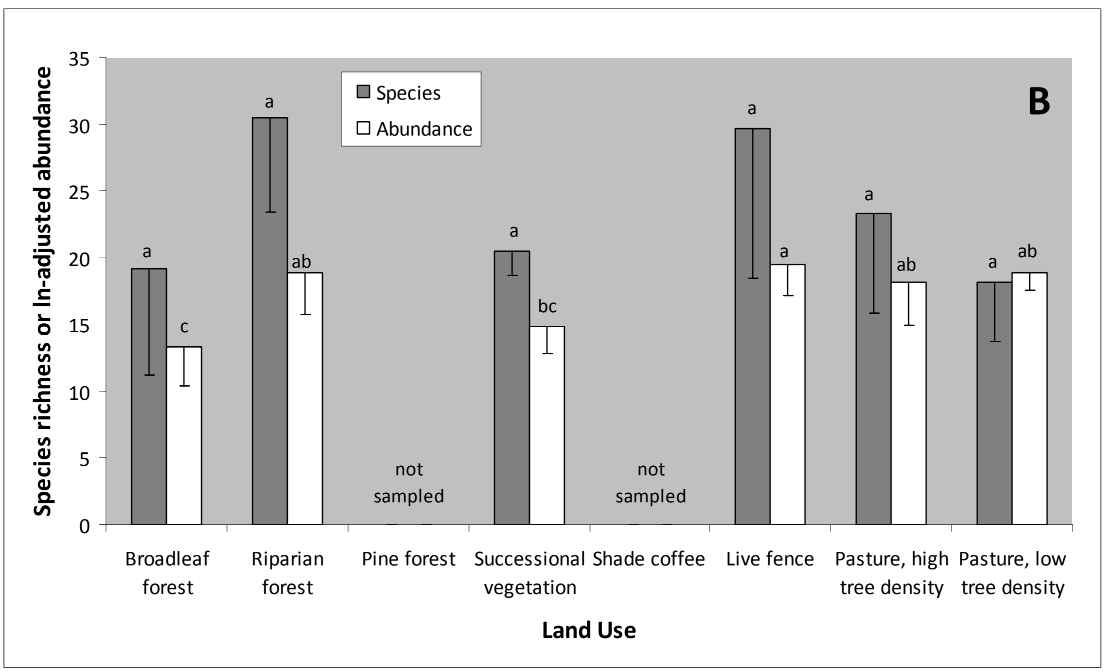
Biodiversity characteristic	**Land Use**								**Total**
	Broadleaf forest	Pine forest	Riparian forest	Successional vegetation	Shade coffee	Pasture, high tree density	Pasture, low tree density	Live fence	
Vegetation									
Total trees	362	219	412	348	251	160	70	389	2211
Total species richness	72	5	75	53	29	17	17	57	145
Mean species per plot	13.6	1.6	17.1	11.1	5.3	3.6	2.6	13.3	8.5
Mean basal area per plot (m^2)	1.72	2.52	2.75	0.58	0.85	0.74	0.23	2.08	1.42
Birds									
Total individuals	484	350	422	674	510	375	372	752	3939
Total species richness	64	41	68	72	62	48	66	72	139
Butterflies									
Total individuals	527	ne	1079	572	ne	944	1064	1101	5287
Total species richness	58	ne	77	56	ne	59	43	70	119

3.4.2 Patterns of faunal diversity associated with different land uses

Land use was a significant predictor of abundance and species richness for both birds and butterflies (Figures 3-3a and 3-3b). Bird mean abundance and mean species richness were highest in successional vegetation and somewhat lower in broadleaf forest, riparian forest, shade coffee, and live fences. Pine forest and pasture supported more depauperate bird assemblages. Surprisingly, pasture with low tree density had more bird individuals and species per plot, on average, than pasture with high tree density, although these differences were not statistically significant (Figure 3-3a).

Butterfly mean species richness and mean abundance were highest in the two linear land uses: riparian forest and live fences. Slightly lower mean abundances were found in the pasture land uses, while successional vegetation and broadleaf forest had substantially lower mean abundances. Mean species richness was intermediate in broadleaf forest, successional vegetation, and pasture with high tree density, and slightly lower in pasture with low tree density (Figure 3-3b). Because of the high degree of inter-plot variation in several of the land cover types, none of the pairwise comparisons of butterfly species richness among land uses were statistically significant (Tukey hsd test, $\alpha = 0.05$), although the model as a whole was significant.

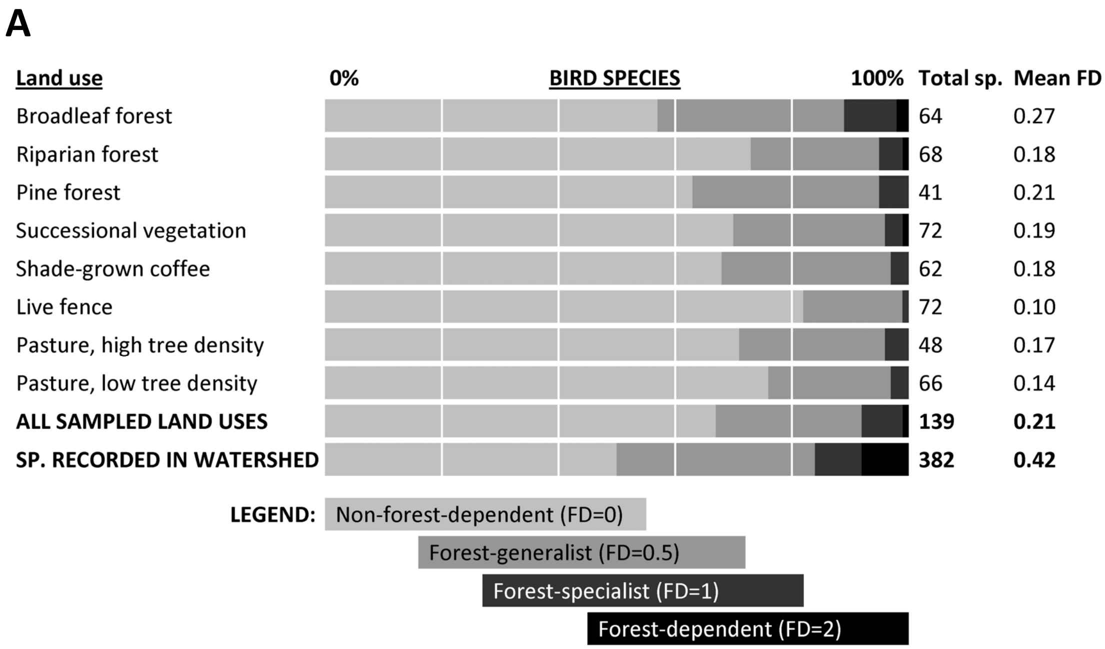


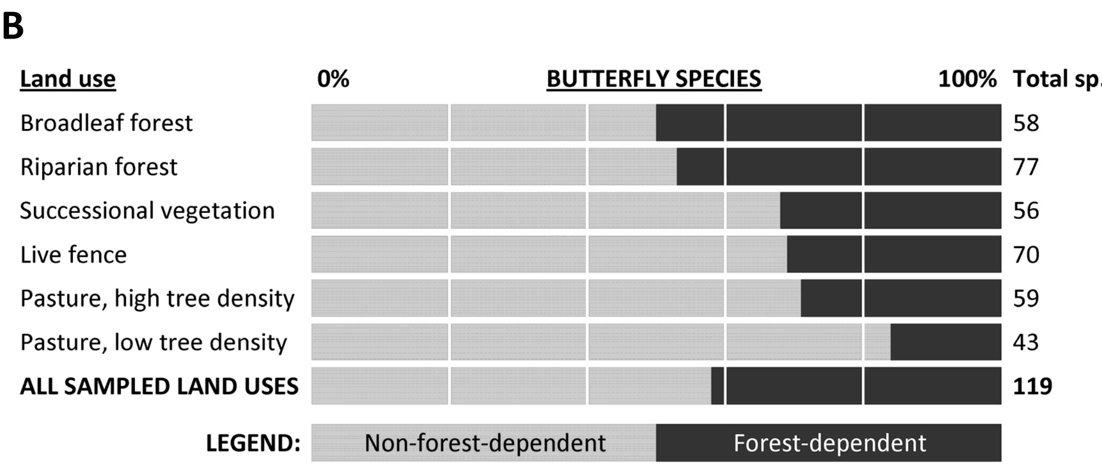


Figures 3-3a and 3-3b. ANOVA of bird (a) and butterfly (b) mean species richness and mean ln-adjusted abundance per plot. ANOVA results for bird species richness: $F_{(7,71)} = 6.57$, $p < 0.0001$; bird abundance: $F_{(7,71)} = 5.78$, $p < 0.0001$; butterfly species richness: $F_{(5,30)} = 3.24$, $p = 0.02$; butterfly abundance: $F_{(5,30)} = 5.90$, $p = 0.0007$. Error bars indicate one standard deviation. Different letters above each data series denote significant differences between pairs of land uses ($p < 0.05$) based on Tukey's hsd test.

Very few forest-dependent bird species were observed at the sample plots, even though many such species have been recorded previously in the Río Copán watershed. Not surprisingly, broadleaf forest had the highest proportions of forest-dependent and forest-specialist bird species, followed by pine forest, riparian forest, successional vegetation, and shade-grown coffee (Figure 3-4a). Live fences had the lowest proportion of forest-oriented bird species, although they were among the most species rich land use. In contrast to the low proportion of forest-oriented bird species observed, 42% of the 119 observed butterfly species are considered to be forest-dependent (Figure 3-4b). The proportion of forest-dependent butterfly species was highest in the most densely forested and tree-covered land uses, as expected. Contrary to the pattern observed for birds, live fences supported an intermediate number and percentage of forest-dependent butterfly species.

Species composition analysis revealed substantial species turnover among land uses for birds and butterflies alike (Table 3-4). Each land use contained unique bird and butterfly species, and overall 28% of butterfly species and 30% of bird species were observed in only a single land use. For birds, species composition was most unique in pine forest, which contained no more than 42% of the species found in any other single land use. For all other land uses, species overlap among pairs of land uses was approximately 45-75%, with slightly greater similarity among broadleaf forest, riparian forest, successional vegetation, and shade-grown coffee. For butterflies, there were few clear patterns except that the most species-rich land uses—riparian forest and live fences—tended to include a high percentage of species present in all the other land uses.





Figures 3-4a and 3-4b. Analysis of the bird (a) and butterfly (b) species observed in each land use according to their degree of forest dependence. For birds, the category 'non-forest-dependent' corresponds to Stiles (1985) forest dependency classifications 2-3 and 3; 'forest-generalist' corresponds to Stiles (1985) category 2; 'forest-specialist' corresponds to Stiles (1985) classification 1-2; and 'forest-dependent' corresponds to Stiles (1985) category 1. The Mean FD for each land use is the mean of the forest dependence scores (indicated in the legend) for all bird species observed in that land use. The analysis of all bird species recorded in the watershed is based on the observations of Gallardo (unpublished manuscript).

Table 3-4. Comparison of species composition among land uses for birds (a) and butterflies (b). Percentages indicate the proportion of species in the land use indicated in the heading row that were also observed in the land use indicated in the left-most column. Values >70% are indicated in **bold** while overlaps of <50% are *italicized*. Abbreviations are as follows: BF = broadleaf forest; PF = pine forest; RF = riparian forest; SV = successional vegetation; SC = shade-grown coffee; PH = pasture with high tree density; PL = pasture with low tree density; LF = live fence. The penultimate row of each table indicates the number of species that were observed only in that particular land use. The butterfly table excludes the pine forest and shade-grown coffee land uses, for which butterfly sampling was not conducted.

(a)

	BF	**PF**	**RF**	**SV**	**SC**	**PH**	**PL**	**LF**
BF	--	*38%*	65%	65%	69%	60%	*49%*	54%
PF	*24%*	--	*29%*	*29%*	*33%*	*42%*	*40%*	*29%*
RF	70%	50%	--	68%	67%	67%	62%	60%
SV	**75%**	53%	**72%**	--	**77%**	67%	65%	63%
SC	67%	50%	60%	65%	--	**71%**	57%	54%
PH	*46%*	50%	*47%*	*44%*	56%	--	52%	*44%*
PL	51%	65%	59%	58%	61%	**71%**	--	58%
LF	62%	53%	63%	63%	64%	67%	65%	--
Unique species	9	6	6	5	3	1	4	8
Total species	64	41	68	72	62	48	66	72

(b)

	BF	**RF**	**SV**	**PH**	**PL**	**LF**
BF	--	59%	60%	*44%*	*49%*	51%
RF	**77%**	--	**81%**	62%	70%	**75%**
SV	60%	61%	--	67%	**74%**	65%
PH	*47%*	51%	**72%**	--	**84%**	62%
PL	*37%*	*40%*	56%	59%	--	*48%*
LF	63%	**71%**	**81%**	**72%**	**79%**	--
Unique species	9	8	1	8	1	7
Total species	58	77	56	59	43	70

3.4.3 Effects of vegetation and landscape context on bird and butterfly assemblages

Bird abundance, bird species richness, and butterfly species richness were all significantly positively correlated with at least one vegetation metric (Table 3-5). However, none of the relationships between butterfly abundance and vegetation parameters were statistically significant. Tree species richness was the vegetation characteristic that most strongly explained patterns of bird and butterfly distribution, while tree density also helped explain patterns of bird distribution. Basal area offered little explanatory power: it was uncorrelated with measures of the bird assemblages and, while significantly correlated with butterfly species richness, provided no additional explanatory power beyond that already provided by tree species richness ($r^2 = 0.23$ for both models).

Table 3-5. Relationships between plot-level vegetation and plot-level bird and butterfly assemblage characteristics. The table reports pairwise correlations (Pearson's *r*) with significance indicated by asterisks: * $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$.

Vegetation characteristic	**Faunal assemblage characteristic**			
	Bird abundance	Bird species richness	Butterfly abundance	Butterfly species richness
Tree density	0.33*	0.28*	-0.10	0.20
Total basal area	0.02	0.01	0.00	0.35*
Tree species richness	0.40***	0.38***	0.06	0.47**

Within each land use type, sample points situated in a landscape context of low tree cover did not have significantly different bird or butterfly assemblage characteristics than those situated in a context of high tree cover (two-way ANOVA, $p > 0.10$ for landscape context, across all faunal response variables). Similarly, once the effect of plot-scale land use was considered, none of the six continuous landscape metrics strongly or consistently explained patterns in faunal response variables at any scale (ANCOVA, $\alpha = 0.05$). The only exception to this pattern was for tasseled-cap wetness, which was positively correlated with butterfly abundance for radii of 100, 200, and 400 m ($p < 0.05$).

3.4.4 Combinations of factors explaining patterns of bird and butterfly diversity

While few of the individual landscape metrics were significant predictors of bird or butterfly abundance or species richness, taken together these metrics were significantly related to bird and butterfly community characteristics (Table 3-6, row 4). However, for the most part, bird and butterfly responses were better explained by land use than by landscape context, vegetation characteristics, or a combination of these (Table 3-6, comparison of row 1 to rows 2, 4, and 5). The sole exception was for patterns of butterfly species richness, which were more fully explained by a combination of continuous vegetation and landscape context variables than by the categorical land use variable.

Overall, the most complete and parsimonious explanations for patterns of bird abundance, bird species richness, and butterfly abundance were provided by combinations of continuous and categorical habitat descriptors. The best explanation of butterfly species richness was provided by the continuous variables alone (Table 3-6, rows 5 and 6). The best models explained between 41% and 64% of the variability in the bird and butterfly response variables. The best model for each response variable included a different set of predictor variables, including some variables that were not significant in the univariate models (Table 3-7). Three of the candidate predictor variables (tree density, contrast-weighted edge density, and NDVI) did not appear in any of the models. Tree species richness at the plot scale and tasseled-cap wetness within the surrounding 100 m were significant predictor variables in the best-fit models for butterflies. Tree basal area at the plot scale and percent forest cover in the surrounding 100 m were also significant predictors of butterfly species richness. Bird assemblages were best explained by land use, percent tree-covered land in the surrounding 100 m, and tasseled-cap brightness and wetness for bird abundance and species richness, respectively.

To the extent that landscape context influenced faunal assemblages, this effect was localized. Landscape context exerted a moderate influence on butterfly abundance at a radius of 100 m (r^2 = 0.40 for the most parsimonious set of landscape context variables; Table 3-6), but this influence decreased somewhat at 200 and 400 m, and sharply at 800 m and beyond. Landscape context effects on bird abundance, bird species richness, and butterfly species richness were weak at 100 m, and remained roughly constant or declined further at coarser spatial scales.

Table 3-6. Comparison of different combinations of categorical and continuous habitat descriptors for explaining patterns of bird and butterfly distribution. For each category of model, the best individual model was selected according to the criteria described in the Methods section. For each response variable, the model with the lowest Akaike information criterion (AIC) value is denoted with **boldface**. "L.C. scale of analysis" indicates the scale over which the landscape context variables yielded the best model. Vegetation variables include tree density, tree basal area, and tree species richness. Landscape context variables include the six metrics described in Table 3-2.

Model results	Faunal assemblage characteristic			
	Bird abundance	Bird species richness	Butterfly abundance	Butterfly species richness
1. Land use only				
AIC	451	531	182	257
r^2	0.36	0.39	0.50	0.35
p	<0.0001	<0.0001	0.001	0.02
2. Vegetation only				
AIC	459	544	195	251
r^2	0.20	0.17	0	0.23
p	0.0006	0.0009	N/A	0.004
3. Plot-scale factors only (land use + vegetation)				
AIC	same as	same as	180	256
r^2	model 1	model 1	0.56	0.42
p			0.0003	0.01
4. Landscape context only				
L.C. scale of analysis	100 m	100 m	100 m	100 m
AIC	457	544	182	257
r^2	0.20	0.19	0.40	0.17
p	0.0002	0.001	0.0008	0.05
5. Continuous habitat descriptors only (vegetation + landscape context)				
L.C. scale of analysis	100 m	800 m	100 m	**100 m**
AIC	450	534	181	**250**
r^2	0.29	0.27	0.43	**0.41**
p	<0.0001	<0.0001	0.0004	**0.002**
6. All variables				
L.C. scale of analysis	**100 m**	**100 m**	**100 m**	100 m
AIC	**448**	**529**	**177**	258
Model r^2	**0.43**	**0.44**	**0.64**	0.45
p	**<0.0001**	**<0.0001**	**<0.0001**	0.01

Table 3-7. Multivariate models of habitat effects on bird and butterfly assemblage characteristics. For each response variable, the best model was selected according to the criteria described in the Methods section. Check marks indicate predictor variables that are included in each model, while the bracketed sign to the right of the check mark indicates whether the partial slope for each continuous variable is positive or negative. The second row indicates the radius of analysis for the landscape context variables in each best-fit model. The bottom three rows report the overall model fit and analysis of variance. See Table 3-2 for descriptions of the candidate predictor variables.

Model attributes		**Faunal assemblage characteristic**			
		Bird abundance	Bird species richness	Butterfly abundance	Butterfly species richness
Scale		100 m	100 m	100 m	100 m
Candidate Predictor Variables	Land use (categorical)	✓	✓	✓	
	Tree density				
	Tree total basal area				✓[+]
	Tree species richness			✓[+]	✓[+]
	%_FOR				✓[-]
	%_TC	✓[-]	✓[-]		
	CWED				
	NDVI				
	TCB	✓[+]			
	TCW		✓[-]	✓[+]	✓[+]
Model r^2		0.43	0.44	0.64	0.41
F		5.70	6.14	7.18	5.39
p		<0.0001	<0.0001	<0.0001	0.002

3.5 DISCUSSION

3.5.1 Bird and butterfly assemblages in the Río Copán watershed

As in other studies in farming landscapes in Central America, this study found that the agricultural mosaic of the Río Copán watershed is supporting a high diversity of birds and butterflies (139 and 119 species, respectively). However, the vast majority of the observed bird species were of low conservation concern. For example, none of the sampled bird species are on the IUCN Red List (Baillie et al. 2004), and less than 10% are considered to be forest-dependent (i.e., forest dependence classifications 1 or 1-2 as denoted by Stiles [1985] or inferred from natural history information in field guides). Furthermore, the observed bird assemblage represents only 36% of the 382 documented bird species in the Río Copán watershed (Gallardo, unpublished manuscript). Although increased sampling effort would likely have revealed some additional species, the difference in species richness—especially richness of forest-dependent species—between the study plots and the watershed overall is far too large to be explained by unobserved species alone. Instead, the results suggest that a high proportion of the watershed's bird species are absent from the heavily-managed central portion of the watershed evaluated here. Some such species have an affinity for montane forests, which are located at the periphery of the watershed and were not sampled in this study. Others may require larger or higher quality forest patches than those generally found in the central portion of the watershed, most of which have diminished floristic and structural diversity associated with wood cutting, cattle incursions, or past disturbances.

The observed butterfly assemblages were much more evenly balanced between forest-dependent and non-forest-dependent species than were the bird assemblages. Likewise, significant numbers of both nectar-feeding and fruit-feeding butterfly species were recorded at the sample plots. Since there is no butterfly species list for the Río Copán watershed as a whole, it is not possible to evaluate the degree to which the central portion of the watershed represents the butterfly fauna of the entire watershed. Nevertheless, based on species composition, this zone does appear to provide a more complete set of suitable habitat niches and resources for butterflies than for birds.

Consistent with prior studies in human-dominated tropical landscapes, the species richness and abundance of birds were poorly correlated with those of butterflies throughout the study landscape. This finding reinforces earlier evidence that any single taxon of tropical animals is unlikely to provide a reliable surrogate for other taxa, either for biodiversity monitoring or for conservation management (Lawton et al. 1998; Perfecto et al. 2003; Harvey et al. 2006b; but see Schulze et al. 2004 and Barlow et al. 2007).

3.5.2 Relationships between land use and faunal assemblages

Patterns of bird and butterfly distribution responded to land use in fairly consistent ways. However, not all of these relationships were as expected. In general, birds were more closely affiliated with dense vegetative cover from trees, shrubs, and vines while butterflies had a high affinity for semi-open habitats and linear land uses. Contrary to expectations, there was no significant difference between pasture with low tree density and pasture with high tree density with respect to mean bird or butterfly species richness or abundance. Overall, live fences emerged as one of the most significant landscape elements contributing to bird and butterfly diversity. This finding is consistent with previous observations that live fences can function both as ecotones and as corridors through agricultural landscapes, attracting a diversity of species that use these habitats for a portion of their life (Lang et al. 2003; Harvey et al. 2005b). These functions may be all the more important in the agricultural zones of the Río Copán watershed, where multi-strata live fences composed of many tree species of various heights are uncommon.

Several factors may explain the observed patterns of butterfly distribution by land use. First, as other investigators have noted, butterflies tend to be especially diverse at the forest-air interface—for example in the forest canopy, forest edges, or light gaps (DeVries 1987; Wood and Gillman 1998). The forest-air interface is a dominant feature in live fences and strips of riparian forest, the two land uses with the highest observed butterfly abundance and species richness. Second, butterfly distribution is strongly influenced by temperature and humidity levels; in the dry season, many species retreat to river bottoms or other moist locations (DeVries 1987). This behavioral tendency could help explain the rich butterfly fauna observed in riparian forests during the mainly dry-season field outings. Third, the relative paucity of species observed in the broadleaf forest plots may be attributable to the young age and high disturbance level that characterizes most of these forests. Neotropical forest butterflies tend to exhibit vertical stratification according to their preference for understory, canopy, or light gap niches (Horner-Devine et al. 2003). However, since most of the forest plots sampled in this study contained neither a well-developed understory nor light gaps associated with tree senescence, it is not surprising that the ground-based transects yielded a relatively low butterfly diversity. Sampling in the canopy of these forests might be expected to reveal many additional species.

To the extent that this study recorded forest-dependent bird species, they were mostly associated with the forest plots, not with managed tree cover in pastures, live fences, or coffee plantations. By contrast, forest-dependent butterfly species were associated with both semi-natural and managed forms of tree cover, including scattered trees and live fences in pasture areas. This finding supports the intuitive conclusion that butterflies respond to smaller-scale habitat features than birds. It also suggests that the complementarity of resources present in the mosaic of agricultural and semi-natural habitats in the study area might provide an acceptable alternative to intact forest for many butterfly species that are ordinarily forest-dependent.

3.5.3 Effects of vegetation, landscape context, and scale

As predicted, higher tree species richness was associated with higher bird abundance and species richness at the plot scale, similar to results from other pasture-dominated landscapes (Cardenas et al. 2003; Harvey et al. 2006b). This finding is consistent with theories of niche differentiation that predict that a greater diversity of habitat resources—characterized by horizontal heterogeneity, vertical heterogeneity, and diverse food resources—will support a more diverse animal assemblage (Schoener 1974). Bird abundance and bird species richness were also associated with tree density; however, contrary to expectations, they were not correlated with total basal area. The implication is that dense assemblages of small trees may have greater value than a few scattered large trees for birds in this landscape.

Butterfly species richness was associated with tree species richness and total tree basal area, consistent with expectations and with the niche differentiation explanation. However, butterfly abundance was unrelated to any of the vegetation metrics evaluated. This observation is consistent with the analyses of butterfly characteristics by land use (Figure 3-3b), which indicates that land uses with high tree cover can have either high (live fences and riparian forest) or low (broadleaf forest) butterfly abundance. As discussed above, butterfly distribution in fine-grained habitat mosaics depends significantly on microclimate and tree configuration, factors that may be poorly captured by aggregate vegetation metrics such as tree density or basal area.

Landscape context had a weaker and more equivocal influence on faunal assemblages than expected, and this effect was limited mainly to the smallest context area evaluated (a 100 m radius, corresponding to a 3.1 ha context zone). This result is consistent with several previous studies of butterflies, hover flies, and other taxa, which found that the scale of influence of habitat features in agricultural landscapes was limited to a few hundred meters at the most (Horner-Devine et al. 2003; Weibull et al. 2003; Kohler et al. 2008). On the other hand, research in Coto Brus, Costa Rica suggested that the influence of a relatively large forest patch on faunal assemblages can extend at least two kilometers for moths (Ricketts et al. 2001) and birds (Luck and Daily 2003). Although my study area contained several sizable forest patches (>100 ha), I found no evidence that these were exerting a 'halo effect' of increased faunal diversity in the surrounding area. On the contrary, the best-fit multivariate models indicated that bird abundance and species richness were negatively correlated with the proportion of tree-covered land uses in the surrounding 100 m radius, while butterfly species richness was negatively correlated with percent forest cover within the same radius.

The reasons for this counterintuitive result are not obvious, but I hypothesize three explanations. First, with regard to birds, the central part of the Río Copán watershed is especially depauperate in forest-dependent species, suggesting that the aggregate quantity and quality of forest habitat in this zone is inadequate to attract many such species that are found in other portions of the watershed. In other words, this agricultural zone may have exceeded a threshold of management intensity beyond which many species cannot persist—a threshold hypothesized by previous modeling studies of habitat fragmentation (Bergman et al. 2004). Since the available pool of species in this zone is both significantly diminished and generally well-adapted to agricultural habitats, most of the species that are present are unlikely to benefit significantly from high levels of nearby tree cover.

Second, with regard to butterflies, the previously discussed results indicate that managed and semi-natural forms of tree cover alike can support a diversity of butterfly species, including forest-dependent species. In addition, many butterfly species have been documented to migrate through pasture-dominated mosaics and across pasture plots, suggesting that a heavily forested landscape context is unnecessary to support such species (Marín et al. 2009). Conversely, given the relatively poor condition of the forest habitats in the central portion of the watershed, this zone may simply lack many of the rarer and more sensitive butterfly species that could benefit from greater tree cover in the surrounding landscape.

Third, landscape-scale species richness in the study area results from the additive contributions of all land uses, each of which contains unique bird and butterfly species. Thus, habitat heterogeneity appears to be a key driver of species richness, and because five of the eight land uses surveyed for birds (and three of the six land uses surveyed for butterflies) are either forested or heavily tree-covered, nearby open areas would tend to increase local habitat heterogeneity. Similar patterns have been observed in suburban landscapes, where complementary habitats can support relatively high overall species richness but few species that depend on large forest patches (e.g., Marzluff 2005). In short, plot-scale habitat variables and agricultural management practices appear to be more important than landscape context in influencing the bird and butterfly communities present at any given location within the study area. However, the mosaic of distinct habitat types results in a high level of beta diversity and thus an overall diverse fauna at the landscape level.

Tasseled-cap wetness was a significant predictor of butterfly abundance in both the univariate and multivariate models, and of butterfly species richness in the multivariate models. This finding may reflect the tendency of many butterfly species to seek moister microclimates during the dry season. Aside from this relationship, however, NDVI and the tasseled-cap metrics contributed little to explaining patterns of bird and butterfly distribution. This result contrasts with a recent study in Coto Brus, Costa Rica, which found consistent relationships between tasseled-cap wetness, tasseled-cap brightness, and bird species richness by group (Ranganathan et al. 2007).

Two factors help explain the weaker predictive power of these remote sensing metrics in Copán. First, the metrics did not strongly track tree cover or forest cover in simplistic ways, a finding that likely reflects a greater biophysical heterogeneity in Copán than in Coto Brus. Overall, relationships between land use and the remote sensing metrics were generally as expected, with forests having relatively high values for NDVI and wetness and low vales for brightness, while pastures, annual crops, and bare soil fell toward the opposite end of the spectrum. However, there were exceptions to these general patterns. In addition, the amount of variation within each land use was high relative to the differences in the mean values among land uses. These complex relationships between the remote sensing metrics and land cover are consistent with earlier theoretical and empirical research on the behavior of the tasseled-cap metrics, which identified intra-annual vegetation cycles, canopy architecture, soil properties, and moisture gradients as important determinants (Crist et al. 1986). In Copán, these properties vary somewhat independently across different land uses and tree cover types, especially during the dry season, when the ASTER image was acquired.

Second, bird and butterfly abundance and species richness were not consistently associated with landscape contexts containing high levels of forest or tree cover. Thus, even to the extent that the remote sensing metrics tracked vegetation density, it does not follow that these metrics would consistently predict bird and butterfly distribution patterns. Overall, my findings with respect to the remote sensing metrics affirm a conclusion of the meta-analysis conducted by Nagendra (2001), who cautioned against over-optimism that remote sensing metrics will be able to predict species distributions consistently, especially in heterogeneous landscapes.

3.5.4 Combinations of factors explaining faunal distribution patterns

Contrary to expectation, this study found categorical land use descriptors generally to be better than continuous habitat descriptors at explaining bird and butterfly patterns in the study landscape. While continuous variables have the theoretical advantage of being able to describe landscape heterogeneity more precisely and less subjectively that categorical ones, in the case of this landscape, land use classifications appear to integrate multiple habitat variables in a manner that is ecologically relevant for birds and butterflies. The significance of land use as a biodiversity correlate is consistent with many prior studies and suggests that land use or agricultural management regime may offer an expedient, cost-effective, and reasonably accurate basis for designing policies and incentives to promote wildlife-friendly agriculture and to monitor the conservation value of agricultural landscapes over time.

Unsurprisingly, the combination of continuous and categorical predictor variables provided the greatest explanatory power, explaining 41-64% of the variability in the different animal assemblage response variables. This finding underscores the significant potential to parameterize species-habitat relationships to aid in the management of human-dominated tropical landscapes. However, it also indicates that there are other important co-variates—many of them likely context-specific attributes that would be difficult to identify, let alone measure—that play an important role in influencing patterns of bird and butterfly diversity. Thus, habitat proxies may offer a suitable coarse filter for biodiversity conservation in agricultural landscapes, upon which conservationists can layer more detailed management and monitoring initiatives focused on particular taxa of concern.

3.5.5 Conservation and management implications

This study corroborates much of the previous research on biodiversity in agricultural mosaics in Central America, which characterizes such landscapes as supporting a wide diversity of plants and animals, but being of limited value for conserving rare or specialized species. While such landscapes have been theorized to contribute to regional conservation by providing functional corridors or compatible matrix habitat for vulnerable species, at this point there is limited empirical evidence to support such roles. A parallel can be seen to the experience in Europe, where the maintenance or re-introduction of semi-natural habitats through agri-environmental payment programs has increased biodiversity to some degree, but mainly among common and widespread species (Kleijn et al. 2006).

On the other hand, the diverse plant and animal assemblages present in the Río Copán watershed provide a range of locally significant ecosystem services—both documented and as yet unknown—and are therefore already contributing important conservation values (Bejarano 2009). Given that the central part of the Río Copán watershed is heavily degraded from the standpoints of native vegetation cover and agricultural impact, the fact that it can support such a diverse flora and fauna offers promise for other degraded pasture-dominated landscapes in Central America. In such landscapes, management practices such as increasing tree diversity, adding multi-strata live fences, and maintaining forest patches on farms can provide both conservation and livelihood benefits. These practices offer win-win opportunities for agricultural landscapes and are worthwhile irrespective of any benefit to rare or endangered species.

LITERATURE CITED

Baillie, J.E.M., C. Hilton-Taylor, and S.N. Stuart. 2004. 2004 IUCN Red List of Threatened Species: a global assessment. IUCN, Gland, Switzerland.

Barlow, J., T.A. Gardner, I. S. Araujo, T.C. Ávila-Pires, A.B. Bonaldo, J.E. Costa, M.C. Esposito, L.V. Ferreira, J. Hawes, M.I.M. Hernandez, M.S. Hoogmoed, R.N. Leite, N.F. Lo-Man-Hung, J.R. Malcolm, M.B. Martins, L.A.M. Mestre, R. Miranda-Santos, A.L. Nunes-Gutjahr, W.L. Overal, L. Parry, S.L. Peters, M.A. Ribeiro-Junior, M.N.F. da Silva, C. da Silva Motta, and C.A. Peres. 2007. Quantifying the biodiversity value of tropical primary, secondary and plantation forests. Proceedings of the National Academy of Sciences of the USA 104: 18555–18560.

Barrance, A.J., L. Flores, E. Padilla, J.E. Gordon, and K. Schreckenberg. 2003. Trees and farming in the dry zone of southern Honduras I: campesino tree husbandry practices. Agroforestry Systems 59: 97-106.

Bejarano, L.F. 2009. Evaluación metodológica del enfoque de ecoagricultura para medir el desempeño de un paisaje con matríz agropecuaria en la subcuenca del Río Copán, Honduras. Master's thesis. Centro Agronómico Tropical de Investigación y Enseñanza, Turriabla, Costa Rica.

Bennett, A.F. 2003. Linkages in the landscape: the role of corridors and connectivity in wildlife conservation. IUCN, Gland, Switzerland and Cambridge, UK.

Bergman, K-O., J. Askling, O. Ekberg, H. Ignell, H. Wahlman, and P. Milberg. 2004. Landscape effects on butterfly assemblages in an agricultural region. Ecography 27: 619-628.

Burnham, K.P., and D.R. Anderson. 2002. Model selection and multimodel inference: a practical information-theoretic approach. 2nd edition. Springer-Verlag, New York.

Cardenas, G., C.A. Harvey, M. Ibrahim, and B. Finegan. 2003. Diversidad y riqueza de aves en diferentes hábitats en un paisaje fragmentado en Cañas, Costa Rica. Agroforestería en las Américas 39-40: 78-85.

Crist, E.P., and R.J. Kauth. 1986. The tasseled cap demystified. Photogrammetric Engineering and Remote Sensing 52: 81-86.

Crist, E.P., R. Laurin, and R.C. Cicone. 1986. Vegetation and soils information contained in transformed Thematic Mapper data. Pages 1465-1470 in ESA proceedings of the 1986 International Geoscience and Remote Sensing Symposium. Ref. ESA SP-254. European Space Agency, Paris.

Dagang, A.B.K., and P.K.R. Nair. 2003. Silvopastoral research and adoption in Central America: recent findings and recommendations for future directions. Agroforestry Systems 59: 149-155.

Daily, G.C., and P.R. Ehrlich. 1995. Preservation of biodiversity in small rainforest patches: rapid evaluations using butterfly trapping. Biodiversity and Conservation 4: 35-55.

Daily, G.C., P.R. Ehrlich, and G.A. Sanchez-Azofeifa. 2001. Countryside biogeography: use of human-dominated habitats by the avifauna of southern Costa Rica. Ecological Applications 11: 1-13.

Daily, G.C., G. Ceballos, J. Pacheco, G. Suzan, and A. Sanchez-Azofeifa. 2003. Countryside biogeography of Neotropical mammals: conservation opportunities in agricultural landscapes of Costa Rica. Conservation Biology 17: 1814-1826.

DeVries, P.J. 1987. The butterflies of Costa Rica and their natural history. Volume I: Papilionidae, Pieridae, Nymphalidae. Princeton University Press, Princeton, New Jersey.

DeVries, P.J. 1997. The butterflies of Costa Rica and their natural history. Volume II: Riodinidae. Princeton University Press, Princeton, New Jersey.

Estrada, A., R. Coates-Estrada, and D. Meritt. 1993. Bat species richness and abundance in tropical rain forest fragments and in agricultural habitats at Los Tuxtlas, Mexico. Ecography 16: 309-318.

Estrada, A., R. Coates-Estrada, and D. Meritt. 1994. Non-flying mammals and landscape changes in the tropical rain-forest region of Los Tuxtlas, Mexico. Ecography 17: 229-241.

Estrada, A., R. Coates-Estrada, and D. A. Meritt. 1997. Anthropogenic landscape changes and avian diversity at Los Tuxtlas, Mexico. Biodiversity and Conservation 6: 19-43.

Estrada, A., R. Coates-Estrada, A. Anzures Dadda, and P. Cammarano. 1998. Dung and carrion beetles in tropical rain forest fragments and agricultural habitats at Los Tuxtlas, Mexico. Journal of Tropical Ecology 14: 577-593.

Estrada, A., P. Cammarano, and R. Coates-Estrada. 2000. Bird species richness in vegetation fences and in strips of residual rain forest vegetation at Los Tuxtlas, Mexico. Biodiversity and Conservation 9: 1399-1416.

Estrada, A., and R. Coates-Estrada. 2001. Bat species richness in live fences and in corridors of residual rain forest vegetation at Los Tuxtlas, Mexico. Ecography 24: 94-102.

Estrada, A., and R. Coates-Estrada. 2002. Dung beetles in continuous forest, forest fragments and in an agricultural mosaic habitat island at Los Tuxtlas, Mexico. Biodiversity and Conservation 11: 1903-1918.

FAO [Food and Agriculture Organization of the United Nations]. 2006. Global forest resources assessment 2005. FAO Forestry Paper 147. FAO, Rome.

FAOSTAT. 2004. Data base of Food and Agricultural Organization of the United Nations. Online: faostat.fao.org.

Ferrier, S. 2002. Mapping spatial pattern in biodiversity for regional conservation planning: where to from here? Systematic Biology 51: 331-363.

Fischer, J., and D.B. Lindenmayer. 2007. Landscape modification and habitat fragmentation: a synthesis. Global Ecology and Biogeography 16: 265-280.

Gallardo, R. Unpublished manuscript. Avian diversity in the Department of Copán, Honduras.

Gardner, T.A., J. Barlow, R. Chazdon, R.M. Ewers, C.A. Harvey, C.A. Peres, and N.S. Sodhi. 2009. Prospects for tropical forest biodiversity in a human-modified world. Ecology Letters 12: 561-582.

Hargis C.D., J.A. Bissonette, and J.L. David. 1998. The behavior of landscape metrics commonly used in the study of habitat fragmentation. Landscape Ecology 13: 167-186.

Hargrove, W.W., and J. Pickering. 1992. Pseudoreplication: a sine qua non for regional ecology. Landscape Ecology 6: 251-258.

Harvey, C.A., and W.A. Haber. 1999. Remnant trees and the conservation of biodiversity in Costa Rican pastures. Agroforestry Systems 44: 37-68.

Harvey, C.A., F. Alpízar, M. Chacón, and R. Madrigal. 2005a. Assessing linkages between agriculture and biodiversity in Central America: historical overview and future perspectives. The Nature Conservancy, San José, Costa Rica.

Harvey, C.A., C. Villanueva, J. Villacís, M. Chacón, D. Muñoz, M. López, M. Ibrahim, R. Gómez, R. Taylor, J. Martinez, A. Navas, J. Saenz, D. Sánchez, A. Medina, S. Vilchez, B. Hernández, A. Perez, F. Ruiz, F. López, I. Lang, and F.L.

Sinclair. 2005b. Contribution of live fences to the ecological integrity of agricultural landscapes. Agriculture, Ecosystems and Environment 111: 200-230.

Harvey C.A., J.G. Gonzales, and E. Somarriba. 2006a. Dung beetle and terrestrial mammal diversity in forest, indigenous agroforestry systems and plantain monocultures in Talamanca, Costa Rica. Biodiversity Conservation 15: 555–585.

Harvey, C.A., A. Medina, D.M. Sanchez, S. Vilchez, B. Hernandez, J.C. Saenz, J.M. Maes, F. Casanoves, and F.L. Sinclair. 2006b. Patterns of animal diversity in different forms of tree cover in agricultural landscapes. Ecological Applications 16: 1986–1999.

Harvey, C.A., O. Komar, R. Chazdon, B.G. Ferguson, B. Finegan, D.M. Griffith, M. Martinez-Ramos, H. Morales, R. Nigh, L. Soto-Pinto, M. van Breugel, and M. Wishnie. 2008a. Integrating agricultural landscapes with biodiversity conservation in the Mesoamerican hotspot. Conservation Biology 22: 8-15.

Harvey, C.A., C. Villanueva, M. Ibrahim, R. Gómez, M. López, S. Kunth, and F. Sinclair. 2008b. Productores, árboles, y producción ganadera en paisajes de América Central: implicaciones para la conservación de la biodiversidad. Pages 197-224 in C.A. Harvey and J. Saénz, editors. 2008. Evaluación y conservación de biodiversidad en paisajes fragmentados de Mesoamérica. Instituto Nacional de Biodiversidad, Heredia, Costa Rica.

Horner-Devine, M.C., G.C. Daily, P.R. Ehrlich, and C.L. Boggs. 2003. Countryside biogeography of tropical butterflies. Conservation Biology 17: 168-177.

Howell, S., and S. Webb. 1995. A guide to the birds of Mexico and Northern Central America. Oxford University Press, New York.

Hughes, J.B., G.C. Daily, and P.R. Ehrlich. 2002. Conservation of tropical forest birds in countryside habitat. Ecology Letters 5: 121-129.

IFAD [International Fund for Agricultural Development]. 2009. Rural poverty in the Americas. Online: www.ruralpovertyportal.org/web/guest/region/home/tags/americas.

Kaimowitz, D. 1996. Livestock and deforestation in Central America in the 1980s and 1990s: a policy perspective. CIFOR, Jakarta.

Kauth, R.J., and G.S. Thomas. 1976. Tasselled cap: a graphic description of the spectral-temporal development of agricultural crops as seen by Landsat. Pages 41-51 in Symposium on Machine Processing of Remotely Sensed Data. National Telecommunications Conference Record, West Lafayette, Indiana.

Kleijn, D., R.A. Baquero, Y. Clough, M. Díaz, J. De Esteban, F. Fernández, D. Gabriel, F. Herzog, A. Holzschuh, R. Jöhl, E. Knop, A. Kruess, E. J. P. Marshall, I. Steffan-Dewenter, T. Tscharntke, J. Verhulst, T. M. West, and J. L. Yela. 2006. Mixed biodiversity benefits of agri-environment schemes in five European countries. Ecology Letters 9: 243-254.

Kohler, F., J. Verhulst, R. van Klink, and D. Kleijn. 2008. At what spatial scale do high-quality habitats enhance the diversity of forbs and pollinators in intensively farmed landscapes? Journal of Applied Ecology 45: 753-762.

Lang, I., L.H.L. Gormley, C.A. Harvey, and F.L. Sinclair. 2003. Composición de la comunidad de aves en cercas vivas de Río Frío, Costa Rica. Agroforestaría en las Américas 39-40: 86-92.

Lawton, J.H., D.E. Bignell, B. Bolton, G.F. Bloemers, P. Eggleton, P.M. Hammond, M. Hodda, R.D. Holt, T.B. Larsenk, N.A. Mawdsley, N.E. Stork, D.S. Srivastava, and A.D. Watt. 1998. Biodiversity inventories, indicator taxa and effects of habitat modification in tropical forest. Nature 391: 72-76.

Li, H., and J. Wu. 2004. Use and misuse of landscape indices. Landscape Ecology 19: 389-399.

Liang, S. 2004. Quantitative remote sensing of land surfaces. John Wiley & Sons, Hoboken, New Jersey.

Lindell, C.A., W.H. Chomentowski, J.R. Zook, and S.A. Kaiser. 2006. Generalizability of Neotropical bird abundance and richness models. Animal Conservation 9: 445-455.

Luck, G.W., and G.C. Daily. 2003. Tropical countryside bird assemblages: richness, composition, and foraging differ by landscape context. Ecological Applications 13: 235-247.

Marín, L., J.L. León-Cortés, and C. Stefanescu. 2009. The effect of an agro-pasture landscape on diversity and migration patterns of frugivorous butterflies in Chiapas, Mexico. Biodiversity and Conservation 18: 919-934.

Marzluff, J.M. 2005. Island biogeograph for an urbanizing world: how extinction and colonization may determine biological diversity in human-dominated landscapes. Urban Ecosystems 8: 157-177.

Mas, A.H., and T.V. Dietsch. 2004. Linking shade coffee certification to biodiversity conservation: butterflies and birds in Chiapas, Mexico. Ecological Applications 14: 642-654.

Mayfield, M.M., and G.C. Daily. 2005. Countryside biogeography of Neotropical herbaceous and shrubby plants. Ecological Applications 15: 423-439.

McGarigal, K., and S.A. Cushman. 2005. The gradient concept of landscape structure. Pages 112-119 in J.A. Wiens and M.R. Moss, editors. Issues and perspectives in landscape ecology. Cambridge University Press, Cambridge, UK.

McGarigal, K., S.A. Cushman, M.C. Neel, and E. Ene. 2002. FRAGSTATS: spatial pattern analysis program for categorical maps. University of Massachusetts, Amherst, Massachusetts. Online: www.umass.edu/landeco/research/fragstats/fragstats.html.

McNeely, J.A., and S.J. Scherr. 2003. Ecoagriculture. Island Press, Washington, D.C.

Medina, A., C.A. Harvey, D. Sánchez Merlo, S. Vílchez, and B. Hernández. 2007. Bat diversity and movement in an agricultural landscape in Matiguás, Nicaragua. Biotropica 39: 120-28.

Nagendra, H. 2001. Using remote sensing to assess biodiversity. International Journal of Remote Sensing 22: 2377-2400.

Oksanen, L. 2001. Logic of experiments in ecology: is pseudoreplication a pseudoissue? Oikos 94: 27-38.

Pagiola, S., P. Agostini, J. Gobbi, C. de Haan, M. Ibrahim, E. Murgueitio, E. Ramírez, M. Rosales, and J.P. Ruíz. 2004. Paying for biodiversity conservation services in agricultural landscapes. World Bank Environment Department Paper No. 96. World Bank, Washington, D.C.

Pérez, A.M., M. Sotelo, F. Ramírez, I. Ramírez, A. López, and I. Siria. 2006. Conservación de la biodiversidad en sistemas silvopastoriles de Matiguás y Río Blanco (Matagalpa, Nicaragua). Ecosistemas 15(3). Online: www.revistaecosistemas.net/articulo.asp?Id=429.

Perfecto, I., A. Mas, T. Dietsch, and J. Vandermeer. 2003. Conservation of biodiversity in coffee agroecosystems: a tri-taxa comparison in southern Mexico. Biodiversity and Conservation 12: 1239-1252.

Perfecto, I., J.H. Vandermeer, G.L. Bautista, G.I. Nuñez, R. Greenberg, P. Bichier, and S. Langridge. 2004. Greater predation in shaded coffee farms: the role of resident Neotropical birds. Ecology 85: 2677-2681.

Pettorelli, N., J.O. Vik, A. Mysterud, J. Gaillard, C.J. Tucker and N.C. Stenseth. 2005. Using the satellite-derived NDVI to assess ecological responses to environmental change. Trends in Ecology and Evolution 20: 503-510.

Ralph, C.J., S. Droege, and J.R. Sauer. 1995. Managing and monitoring birds using point counts: standards and applications. Pages 161-169 in C.J. Ralph, J.R. Sauer, and S. Droege, editors. Monitoring bird populations by point counts. Gen. Tech. Rep. PSW-GTR-149. USDA Forest Service, Pacific Southwest Research Station, Albany, California.

Ranganathan, J., K.M.A. Chan, and G.C. Daily. 2007. Satellite detection of bird communities in tropical countryside. Ecological Applications 17: 1499-1510.

Ricketts, T.H., G.C. Daily, P.R. Ehrlich, and J.P. Fay. 2001. Countryside biogeography of moths in a fragmented landscape: biodiversity in native and agricultural habitats. Conservation Biology 15: 378-388.

Ricketts, T.H., G.C. Daily, P.R. Ehrlich, and C.D. Michener. 2004. Economic value of tropical forest to coffee production. Proceedings of the National Academy of Sciences of the USA 101: 12579-12582.

Ricketts, T.H., J. Regetz, I. Steffan-Dewenter, S.A. Cunningham, C. Kremen, A. Bogdanski, B. Gemmill-Herren, S.S. Greenleaf, A.M. Klein, M.M. Mayfield, L.A. Morandin, A. Ochieng, and B.F. Viana. 2008. Landscape effects on crop pollination services: are there general patterns? Ecology Letters 11: 499-515.

Reitsma, R., J.D. Parrish, and W. McLarney. 2001. The role of cocoa plantations in maintaining forest avian diversity in southeastern Costa Rica. Agroforestry Systems 53: 185-193.

Sánchez, E.P. 2006. Caracterización de sistemas silvopastoriles y su contribución socioeconómica en productores ganaderos de Copán, Honduras. Master's thesis. Centro Agronómico Tropical de Investigación y Enseñanza, Turriabla, Costa Rica.

Sanfiorenzo, A.R. 2008. Contribución de diferentes arreglos silvopastoriles a la conservación de la biodiversidad, mediante la provisión de hábitat y conectividad en el paisaje de la sub-cuenca del Río Copán, Honduras. Master's thesis. Centro Agronómico Tropical de Investigación y Enseñanza, Turrialba, Costa Rica.

Scherr, S.J., and J. McNeely. 2008. Biodiversity conservation and agricultural sustainability: towards a new paradigm of 'ecoagriculture' landscapes. Philosophical Transactions of the Royal Society B 363: 477-494.

Schoener, T.W. 1974. Resource partitioning in ecological communities. Science 185: 27-39.

Schulze, C.H., M. Waltert, P.J.A. Kessler, R. Pitopang, Shahabuddin, D. Veddeler, M. Mühlenberg, S.R. Gradstein, C. Leuschner, I. Steffan-Dewenter, and T. Tscharntke. 2004. Biodiversity indicator groups of tropical land-use systems: comparing plants, birds, and insects. Ecological Applications 14: 1321-1333.

Sekercioglu, C.H., S.R. Loarie, F.O. Brenes, P.R. Ehrlich, and G.C. Daily. 2007. Persistence of forest birds in the Costa Rican agricultural countryside. Conservation Biology 21: 482-494.

Stiles, F.G. 1985. Conservation of forest birds in Costa Rica: problems and perspectives. Pages 141-168 in A.W. Diamond and T.E. Lovejoy, editors. Conservation of tropical forest birds. International Council for Bird Preservation, Cambridge, UK.

Stiles, F. G., and A. F. Skutch. 1989. A guide to the birds of Costa Rica. Cornell University Press, Ithaca, New York.

Suatunce, P., E. Somarriba, C.A. Harvey, and B. Finegan. 2003. Composición florística y estructura de bosques y cacaotales en los territorios indígenas de Talamanca, Costa Rica. Agroforestería en las Américas 10: 31-35.

Tischendorf, L. 2001. Can landscape indices predict ecological processes consistently? Landscape Ecology 16: 235-254.

TNC [The Nature Conservancy]. 2007. Motagua-Polochic system, Guatemala. Parks in Peril End-of-Project Report. TNC, Arlington, Virginia.

Tobar, D., M. Ibrahim, and F. Casasola. 2007. Diversidad de mariposas en un paisaje agropecuario del Pacífico Central de Costa Rica. Agroforestaría en las Américas 45: 58-65.

Vandermeer, J., and I. Perfecto. 2007. The agricultural matrix and a future paradigm for conservation. Conservation Biology 21: 274-277.

Weibull, A-C., O. Ostman, and A. Granqvist. 2003. Species richness in agroecosystems: the effect of landscape, habitat and farm management. Biodiversity and Conservation 12: 1335-1355.

Wood, B., and M.P. Gillman. 1998. The effects of disturbance on forest butterflies using two methods of sampling in Trinidad. Biodiversity and Conservation 7: 597-616.

WRI [World Resources Institute in collaboration with United Nations Development Programme, United Nations Environment Programme, and World Bank]. 2008. World resources 2008: roots of resilience—growing the wealth of the poor. World Resources Institute, Washington, D.C.

Yarbrough, L.D., G. Easson, and J.S. Kuszmaul. 2005. Using at-sensor radiance and reflectance tasseled cap transforms applied to change detection for the ASTER sensor. IEEE Third International Workshop on the Analysis of Multi-temporal Remote Sensing Images, May 16-18, 2005, Biloxi, Mississippi.

Zhang, W., T.H. Ricketts, C. Kremen, K. Carney, and S.M. Swinton. 2007. Ecosystem services and dis-services to agriculture. Ecological Economics 64: 253-260.

CHAPTER 4

BIODIVERSITY CONSERVATION IN PASTURE-DOMINATED LANDSCAPES OF NORTHERN LATIN AMERICA: ARE THERE CONSISTENT PATTERNS? [3]

4.1 ABSTRACT

Numerous studies have documented the potential of Neotropical agricultural mosaics to conserve significant numbers of native plant and animal species. However, the diversity of research methods used across different studies and sites—as well as the dearth of multi-site studies or syntheses—has made it difficult to document non-trivial patterns that can be generalized to larger areas. To address this need, I conducted a coordinated multi-site study of bird and butterfly diversity in four pasture-dominated landscapes: one each in Honduras, Nicaragua, Costa Rica, and Colombia. The study quantifies patterns of bird abundance and species richness in 13 agricultural and semi-natural land uses, and butterfly abundance and species richness in nine such land uses. This study also develops and applies a new tool—the Biodiversity Conservation Value (BCV) metric—to quantify the conservation value of bird assemblages present within these landscapes. Overall, the study documented 404 bird species and 296 butterfly species across the four landscapes. Bird and butterfly species richness and abundance

---

[3] This chapter is adapted from a manuscript of the same title by Jeffrey C. Milder, Diego E. Tobar, Enrique Murgueitio, A. Mijail Pérez, Joel C. Sáenz, Fabrice A.J. DeClerck, and Muhammad Ibrahim. My original contributions to this work included designing the overall study; designing and overseeing butterfly field work in Nicaragua and Colombia, and bird and butterfly field work in Honduras; compiling and normalizing all data sets; conducting all data analysis; and preparing the entire manuscript including all figures and tables. D.E.T. and four field assistants conducted the butterfly field work. My field assistant Cliff Cordy conducted the bird field work in Honduras. E.M., A.M.P., and J.C.S. oversaw the bird field work in Colombia, Nicaragua, and Costa Rica, respectively. My research assistant Andre Sanfiorenzo assisted me in preparing land use maps. D.E.T., F.A.J.D., and M.I. assisted me in acquiring bird datasets for Nicaragua, Costa Rica, and Colombia.

were significantly related to land use across all four landscapes, and, for the most part, were positively correlated with degree of tree cover in the different land uses. However, the faunal assemblage measures were also significantly related to landscape identity, implying that context-specific co-variates also play an important role. Bird assemblages were highly skewed toward non-forest-dependent and low conservation value species, particularly in the agricultural land uses. This pattern was substantially masked by the species richness and abundance measures, but was revealed by BCV, thus indicating the importance of the choice of metric when evaluating the conservation potential of human-dominated landscapes.

4.2 INTRODUCTION

Worldwide, agricultural expansion and intensification is the foremost threat to biodiversity and a leading cause of water pollution, soil erosion, land degradation, and greenhouse gas emissions (Matson et al. 1997; Tilman et al. 2002; Clay 2004; FAO 2006). Recognizing the centrality of agriculture to the future of global land use, conservationists have recently begun to devote increasing effort to reconciling food, fiber, and biofuel production with conservation goals.

The challenge of simultaneously circumscribing the negative environmental impacts of agriculture while meeting the burgeoning global demand for agricultural products has spurred a lively debate about the best way to address these conflicting mandates. Advocates of the “land sparing” paradigm, emphasizing the successes of the Green Revolution, argue that agricultural output should be maximized on productive lands so that the overall physical footprint of farming can be minimized and other lands spared for ecosystem conservation (Balmford et al. 2005; Green et al. 2005). An alternative school of thought advocates “wildlife-friendly farming” whereby farming systems are deliberately managed to retain a diversity of native species in agricultural landscapes, ideally by enhancing synergies and minimizing tradeoffs between food production and wildlife conservation at multiple scales (McNeely and Scherr 2003; Vandermeer and Perfecto 2007).

In northern Latin America, the choice between these alternative land use approaches is being actively negotiated by a range of local, domestic, and international stakeholders. The region has a strong tradition of smallholder agriculture, which remains a key source of food production for subsistence use, local markets, and high-value export crops such as coffee. At the same time, the region has a history of large-scale intensive agriculture, beginning in the early 20th century with foreign-owned banana plantations and continuing today with investment in crops such as oil palm, sun-grown coffee, pineapple, and various horticultural crops for export (Harvey et al. 2005a). Commodity prices, trade agreements, and crop pests and diseases are some of the key macro-scale drivers that continue to shape the region’s farming systems and land use patterns.

In reality, both land sparing and wildlife-friendly farming have a role to play in meeting societal needs at a global level. The relative desirability of each paradigm must be evaluated locally by considering the degree to which agriculture and conservation can be reconciled in specific places—in other words, the shape of the tradeoff curve between agricultural yields and measures of biodiversity and ecosystem function (Fischer et al. 2008). Regions or farming systems that can retain most of their conservation value under moderate-intensity agriculture may be well suited for wildlife-friendly farming, while those that suffer a significant loss of environmental quality even under low-intensity production may be better suited for either intensive farming or conservation reserves. Going forward, society's ability to identify and implement the most effective combination of these approaches will depend upon understanding the nature of the tradeoffs and synergies between conservation and food production in specific locations. Accordingly, parameterizing these relationships for the principal agricultural production systems in each part of the world is a critical research endeavor.

Over the past two decades, considerable research has taken place to understand these relationships for various production systems in northern Latin America, including coffee, cacao, and cattle-based systems (e.g., Estrada et al. 1993, 1994, 1997, 1998, 2000; Daily et al. 2001, 2003; Hughes et al. 2002; Perfecto et al. 2003; Mayfield and Daily 2005; Harvey et al. 2005b, 2006). The conclusions of these studies have been broadly similar in many respects, helping to confirm some intuitive relationships such as the positive effects of tree cover and floristic and structural diversity on native fauna. In other ways, however, the collective results of these studies have been inconsistent or ambiguous, especially with regard to the influence of landscape composition, configuration, and context on native biodiversity.

A major limitation has been the lack of consistent methodologies from study to study, which has made it difficult to compare results and evaluate the broader applicability of place-specific findings. Studies from different locations, or those conducted by different investigators, have each tended to use their own sets of predictor variables (e.g., habitat and landscape attributes) and response variables (e.g., attributes of species assemblages), even when studying similar phenomena. As a result, individual studies have revealed the effect of specific habitat characteristics on specific plant or animal taxa, but usually without evaluating other habitat characteristics that might allow for cross-study comparisons or the testing of alternative hypotheses. Until such broader analyses are conducted, it will be difficult to develop valid yet non-trivial general principles for the effective design and management of agricultural landscapes for biodiversity conservation (e.g., Lindenmayer et al. 2008).

Coordinated multi-site studies and meta-analyses can help reveal broader patterns by synthesizing results across diverse contexts while attempting to control for methodological differences and confounding factors. For example, a recent large-scale study of 25 agricultural landscapes in seven European countries revealed some consistent relationships between land use and species richness for multiple plant and animal taxa (Billeter et al. 2008). On the other hand, research in Costa Rica found that models relating bird assemblages to habitat characteristics in agricultural settings had limited potential to be generalized from one pre-montane landscape to another, indicating that cross-site syntheses may not always yield simple, consistent patterns (Lindell et al. 2006).

An additional methodological challenge addressed in this chapter is the quantification of faunal assemblages for the purpose of studying habitat-wildlife relationships. When such relationships are studied in the context of human-dominated landscapes, the goal is often to understand the value of these landscapes for biodiversity conservation. Assessing conservation value requires making implicit or explicit value judgments about which components of biodiversity matter most (Duelli and Obrist 2003). Most studies of conservation in human-dominated landscapes have used traditional indices of diversity and abundance—such as species richness, total abundance, and the Shannon index—to quantify biodiversity. Since all species contribute equally to such metrics, the implicit assumption is that all species are equally valuable; for example, such metrics do not distinguish between common or non-native species and those that are highly valued because they are rare, threatened, or provide important ecosystem services (Götmark et al. 1986). Other studies of conservation in human-dominated landscapes have partitioned the species richness of animal taxa according to pre-defined habitat associations such as forest-dependent, forest-generalist, or non-forest-dependent (e.g., Stiles 1985). Such analyses usually come to the rather tautological conclusion that forested areas harbor more forest-dependent species and fewer open land species than pastures or cropland, and vice versa. The implication is generally that a greater presence of forest-dependent species indicates greater conservation value in places where forest was the original land cover. However, forest dependence is only one of several attributes that might distinguish animal species as being of high conservation value.

In this chapter, I propose an alternative metric—Biodiversity Conservation Value (BCV)—that quantifies the conservation value of faunal assemblages according to multiple traits of their component species. This approach is based on the premise that the success of landscape-scale conservation efforts depends less on maximizing local species richness than on contributing to regional or global conservation priorities by sustaining biodiversity that is rare, endemic, threatened, or vulnerable. These two approaches (local species richness versus broader conservation contribution) have long offered conflicting mandates for the field of wildlife management, yet the international conservation community has overwhelmingly embraced the latter because of its orientation toward preventing species extinctions and maintaining intact ecosystems. It is therefore appropriate for research on the conservation value of human-dominated landscapes to use species assemblage metrics that reflect this focus.

To this end, the BCV metric quantifies the conservation value of species assemblages for their contribution to the goal of global biodiversity conservation. Systems already exist to rank species according to their conservation status—most notably the IUCN Red List (Baillie et al. 2004)—but most such systems are of limited value in human-dominated landscapes that contain few species of special conservation status (such as Threatened or Endangered). Yet, even among IUCN species of Least Concern, there are significant differences in the relative risk of population decline or future endangerment. Differentiating species that merit an intermediate level of concern from those that are the most demonstrably secure can help guide pro-active conservation initiatives that help keep common species common and prevent less common species from becoming rare or endangered (Mehlman et al. 2004). The BCV metric quantifies conservation value according to multiple species attributes (explained further in the Methods section), similar to the approach developed by Partners in Flight to prioritize North American bird species and evaluate species assemblages for conservation management (Beissinger et al. 2000).

This chapter reports a study that evaluated the consistency of habitat-wildlife relationships in four pasture-dominated landscapes across four countries in northern Latin America. Pasture-dominated land mosaics comprise a major portion of the land base in the tropical forest biomes stretching from southern Mexico through Central America and into northern South America, north of the Andes and the Amazon basin. In Central America, for example, such areas occupy 27% of land, three times the amount devoted to all other agricultural production systems combined (FAOSTAT 2004). Accordingly, the conservation value of pasture-dominated landscapes has tremendous implications for the success of regional conservation objectives and initiatives, such as the Mesoamerican Biological Corridor (Harvey et al. 2008a). In addition, these landscapes provide an excellent laboratory for studying the consistency of habitat-wildlife relationships because they are broadly similar throughout the region, yet exhibit a degree of variation attributable to locally specific biophysical conditions, land use histories, and agricultural management practices.

To study these relationships, I selected birds and butterflies as focal taxa because both groups have been widely used as indicators of habitat condition and because each group responds to different habitat characteristics at different scales. Birds are commonly used in conservation research because they are relatively easy to census and respond to many habitat parameters typically influenced by humans, including changes in vegetation composition and structure, abundance of various food sources, and presence of suitable habitats and routes for dispersal and migration at local to global scales (Temple and Wiens 1989). Whereas many tropical birds respond strongly to tree cover, butterfly distributions are influenced in part by flower availability, which may be high in grassy or shrubby vegetation types, or in the forest understory (Kremen 1992; Ouin et al. 2004; Harvey et al. 2006).

I posed four research questions: 1) How do bird and butterfly assemblages respond to land use patterns in pasture-dominated landscapes of northern Latin America? 2) To what degree can these relationships be generalized throughout the region? 3) To the extent that the relationships vary across the region, which aspects of landscape composition and context appear to drive these differences? 4) Does the BCV metric quantify bird assemblages in a manner distinct from traditional species richness and abundance measures, and is it helpful for understanding the conservation value of bird assemblages?

4.3 METHODS

4.3.1 Study sites and landscape characterization

Field studies and landscape analysis were conducted in four locations: Copán, Honduras; Matiguás, Nicaragua; Esparza, Costa Rica; and Quindío, Colombia (Figure 4-1). All four landscapes are mosaics of pasture, annual and perennial crops, fruit or timber plantations, and forest fragments (see Table 4-1 for a summary of landscape characteristics).

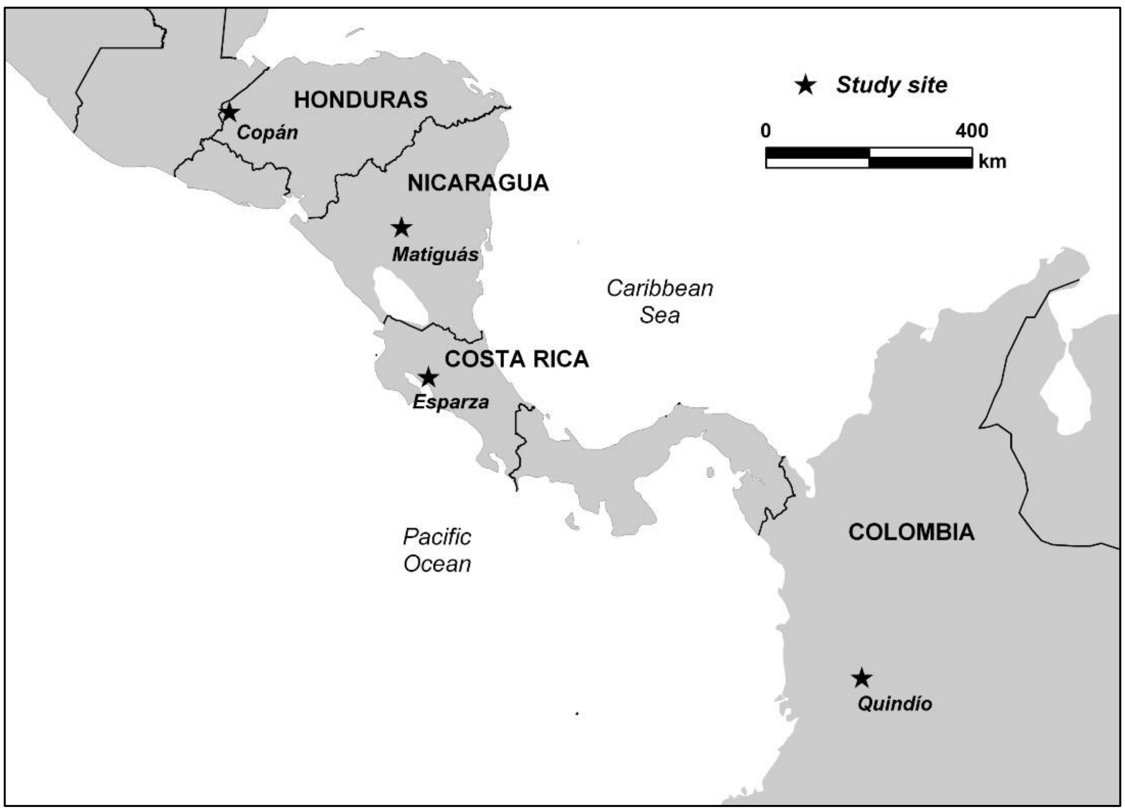


Figure 4-1. Map of northern Latin America showing the study landscapes.

Table 4-1. Characteristics of the study landscapes.

Landscape characteristic	Copán, Honduras	Matiguás, Nicaragua	Esparza, Costa Rica	Quindío, Colombia
Latitude	14°50’ N	12°50’ N	9°59’ N	4°26’ N
Longitude	89°04’ W	85°27’ W	84°38’ W	75°40’ W
Elevation range	600-1200 m	200-400 m	0-870 m	930-1730 m
Annual rainfall	1600 mm	1500 mm	3900 mm	1500 mm
Average temperature	24°C	24°C	27°C	21°C
Landscape size	598 km^2	284 km^2	486 km^2	583 km^2

For each of the four landscapes, land use maps were created based on visual interpretation and selective field verification of high resolution (pixel size ≤ 1 m) Quickbird or IKONOS satellite imagery (see Appendix B). The software program FRAGSTATS (version 3.3; McGarigal et al. 2002) was then used to quantify the composition and structure of each landscape. Composition is reported in terms of the percent of the landscape in each of 13 distinct land use categories. (These 13 categories are based on the 18-category classification system presented in Appendix B; however some of the 18 categories are lumped together to facilitate cross-landscape comparisons.) Structure is reported based on three FRAGSTATS metrics: contrast-weighted edge density, percent core forest area (i.e., percent of the landscape in forest cover excluding an edge zone of the outermost 50 m of each forest patch), and proximity. Proximity is a measure of structural connectivity that is proportional to the size of the patches of the focal land use type and inversely proportional to the square of the inter-patch distance of such patches (McGarigal et al. 2002). I calculated the proximity metric twice: once for forested patches only and once for all tree-covered patches (forest plus successional vegetation, plantations, shaded perennial crops, and pasture with high tree density). Additional detail on the FRAGSTATS analysis method is provided in Appendix B.

4.3.2 Field studies

Within each landscape, bird and butterfly sampling locations were stratified by land use. The study included all land uses that comprised a significant portion of each landscape or represented locally important agricultural management practices. The following six land uses were sampled in all four landscapes: secondary forest, riparian forest, successional vegetation (shrubby or woody vegetation approximately 3-8 years old on fallow crop or pasture land), multi-strata live fence (continuous rows of trees dividing pastures or crop plots), pasture with high tree density (15-30% tree canopy cover), and pasture with low tree density (5-15% canopy cover). Hereafter, I refer to these as the "six common land uses." Additional land uses—including primary forest, pine forest, thin forest (disturbed forest lacking a full tree canopy), shade-grown coffee, fruit or timber plantations, unshaded perennial crops, and pasture without trees—were sampled in certain landscapes where they represented important landscape components. At each location, a central point was identified as the point of orientation for the bird and butterfly sampling.

Birds were sampled by means of a standard point count method in which all individuals heard or seen within 25 m of the central point during a 10-minute sampling period were recorded (Ralph et al. 1995). Point counts were conducted from 0600 to 1000 hours only in good weather conditions without heavy rain or strong winds. Upon arriving at each sampling location, the field technician waited five minutes prior to the start of the counting period to minimize the effect of any initial flushing of birds. Birds observed flying high above the sample plot were excluded from the analysis when they were judged to be unassociated with the habitat from which they were observed. At least eight locations were sampled for each of the six common land uses, except secondary forest in Matiguás, for which only seven locations were sampled. At each sampling location, point counts were repeated at least six times. All point counts were conducted by experienced field ornithologists familiar with the area's birds. Bird sampling in Matiguás, Esparza, and Quindío was conducted from 2003-07 as part of the field monitoring component of a payment for ecosystem services project (Pagiola et al. 2005), and was supervised by senior scientists in the respective countries. I supervised the Copán bird sampling, which was conducted from 2008-09. I also reviewed all bird data prior to analysis.

To sample butterflies, field staff established a 120 m-long transect through the central point at each location. Each transect was surveyed by walking its length at a slow, constant pace for 45 minutes and recording all adult butterflies observed within 2.5 m to either side of the transect. Transects were visited from 0800 to 1600 hours only on days with good weather. To avoid time-of-day biases, both morning and afternoon visits were conducted at each sampling location. Individuals that could not be identified by sight were captured by net for later identification with field guides, keys, and illustrations. The six common land uses were each sampled in five locations in Esparza and in six locations in the other three landscapes. However, due to a lack of access to private land and other logistical constraints, it was not possible to sample secondary forest or successional vegetation in Quindío. In addition, only three successional vegetation plots could be sampled in Matiguás. Ten transect replicates per location were conducted in Esparza and six replicates per location were conducted in the other landscapes. All butterfly sampling was conducted by lepidopterist Diego Tobar and four field assistants. Sampling in Esparza occurred from March 2005 to April 2006. Sampling in the other three landscapes took place between September 2008 and March 2009.

4.3.3 Data analysis

I analyzed the raw field data to provide descriptive statistics of bird and butterfly assemblages at the replicate, plot, land use, and landscape levels of aggregation. For both taxa, I computed species richness and abundance. For birds, I also calculated the BCV metric, introduced above and described more fully in the next sub-section. BCV could not be calculated for butterflies since some of the requisite data on species traits are generally lacking for Neotropical butterflies. To attain normally distributed data and avoid skewed results due to large groups of flocking birds observed at some sample points, I square root transformed all abundance data.

Since the level of sampling effort differed among plots and landscapes, I normalized the faunal assemblage statistics at both these levels of aggregation to enable valid comparisons to be made. At the plot level, I calculated mean values for abundance, species richness, and BCV per sample replicate (i.e., each individual bird point count or butterfly transect sample). At the landscape level, I used the Chao2 estimator to estimate total landscape species richness, including unobserved species (Chao 1987).

For each landscape, I used one-way ANOVA to evaluate differences in bird and butterfly assemblage characteristics by land use. (All of the response variables met the assumptions for ANOVA). Tukey's honestly significantly different (hsd) test was used to evaluate the statistical significance of differences between pairs of land uses. To evaluate the cross-landscape consistency of faunal assemblage characteristics by land use, I used two-way ANOVA with land use and landscape identity as the two factors.

I used the software program EstimateS (version 8; Colwell 2006) to estimate species richness, SAS (version 9.2; SAS Institute 2008) to conduct two-way ANOVAs, and JMP (version 8; SAS Institute 2008) to conduct all other statistical analyses.

4.3.4 Biodiversity conservation value (BCV) metric

As explained above, the BCV metric incorporates multiple factors to quantify the value of each bird species in contributing to the goal of global biodiversity conservation. These factors include: 1) species range size (data from BirdLife International 2009); 2) total global population (data from BirdLife International 2009); 3) sensitivity to disturbance (data from Stotz et al. 1996); 4) degree of forest dependence (data from Stiles 1985 and several bird field guides); and 5) threat status as determined from two other ranking systems (data from Stotz et al. 1996 and Baillie et al. 2004). I first compiled a database of these traits for each bird species observed in one or more of the study landscapes. I then used these data to assign each species a numerical score for each factor based on pre-determined data categories or ranges. For the species range size factor, for instance, endemic species (those with ranges of less than 50,000 km^2) received a score of 4; restricted species (those with ranges of 50,000 to 200,000 km^2) received a score of 2; and all other species received a score of 0. Scoring categories for all five factors are provided in Appendix C.

To determine BCV for each species, I added the individual factor scores for factors 1-4, compared this sum to the score for factor 5, and then selected the higher of the two values. The purpose of this summation rule is to compare two different threat evaluation protocols—each of which may incorporate different data on species conservation value—and select the more sensitive of the two. I then calculated aggregate BCV at the replicate and landscape levels by summing the species BCV for all bird species observed at each of these sampling levels.

4.4 RESULTS

4.4.1 Characterization of bird and butterfly assemblages

The study sampled a total of 38,145 birds of 404 species (Table 4-2, Part A). Landscape-wide bird species richness (both observed and estimated) was highest in Quindío and Matiguás, intermediate in Esparza, and somewhat lower in Copán. Mean bird species richness and mean bird abundance per replicate were substantially higher in Quindío than in the other landscapes, indicating that there are generally more birds present throughout this landscape. Esparza had the lowest mean bird species richness and mean bird abundance per replicate. In general, the four landscapes had fairly similar proportions of forest-dependent, forest-generalist, and non-forest-dependent bird species (Table 4-2, Part A). However, Quindío did have a somewhat higher proportion (12%) and number (23) of forest-dependent species than the other landscapes.

The conservation value of landscape-wide bird assemblages was substantially different among the four landscapes. The total BCV was twice as high in Quindío (171.5) as in Copán (86.5), with intermediate values found in Matiguás and Esparza. Both the mean species BCV and the mean BCV per sample replicate increased monotonically from north to south, with the latter ranging from 1.8 in Copán to 3.4 in Quindío (Table 4-2, Part A). Compared to the entire list of 404 bird species recorded in this study (mean species BCV = 1.01), the species assemblage present in each landscape was disproportionately skewed toward lower conservation value species (mean species

BCV ranged from 0.62 in Copán to 0.90 in Quindío). The five component factors in the BCV metric were uncorrelated with one another ($r < 0.20$), with the exception of factors 3 and 4 (sensitivity to disturbance and degree of forest dependence), which were moderately correlated ($r = 0.55$).

The study sampled a total of 30,334 butterflies of 296 species. (The total number of subspecies is somewhat higher, as many of the butterfly species were present as different subspecies in different landscapes.) Landscape-wide observed butterfly species richness was highest in Esparza (139 species) and slightly lower in Copán (119 species) and Quindío (113 species). Matiguás had by far the fewest observed butterfly species (65). Results from the Chao2 species richness estimator indicate that the total butterfly species richness, including unobserved species, ranged from approximately 88 in Matiguás to 156 in Esparza and 158 in Copán. Comparing only the land uses common to all four landscapes, Esparza had by far the greatest butterfly abundance per sample replicate (40.4). It also had the highest mean species richness per replicate (10.4), but by a smaller margin. Quindío had the lowest mean butterfly abundance (17.6) and species richness (6.5) per replicate. Thus, relative patterns of species richness and abundance across the four landscapes were nearly opposite for butterflies versus birds.

Table 4-2. Characterization of bird and butterfly assemblages in the four study landscapes. Landscape totals and bird species composition include data from all surveyed land uses, while mean values per replicate include data only from the six common land uses.

Faunal assemblage characteristics	Copán, Honduras	Matiguás, Nicaragua	Esparza, Costa Rica	Quindío, Colombia	All four landscapes
A. Bird assemblage characteristics					
Land uses sampled	8	10	10	10	13
Total plots sampled	79	113	121	113	426
Total species (observed)	139	186	153	191	404
Total species (estimated)	161	210	185	222	N/A
Total individuals	3,939	12,225	5,574	16,407	38,145
Total BCV	86.5	136	128.5	171.5	409
Mean species BCV	0.62	0.73	0.84	0.90	1.01
Mean species per replicate	5.1	5.7	4.3	9.4	5.8
Mean individuals per replicate	8.7	12.3	7.5	16.6	10.8
Mean BCV per replicate	1.8	2.2	2.8	3.4	2.5
Composition: % forest-dependent species	8%	4%	5%	12%	10%
Composition: % forest-generalist species	25%	28%	27%	26%	29%
Composition: % non-forest-dependent species	66%	67%	67%	62%	61%
B. Butterfly assemblage characteristics					
Land uses sampled	6	6	9	4	9
Total plots sampled	36	33	51	24	144
Total species (observed)	119	65	139	113	296
Total species (estimated)	158	88	156	140	N/A
Total individuals	5,261	4,552	17,992	2,529	30,334
Mean species per replicate	7.6	8.8	10.4	6.5	8.8
Mean individuals per replicate	24.4	23.0	40.4	17.6	28.9

4.4.2 Landscape characterizations

The four landscapes contained mostly the same component land uses, but proportions and configurations of these land uses varied considerably (Table 4-3). Pasture was a dominant land use at all four sites, ranging from 40% of total land cover in Copán to 72% in Matiguás. Matiguás and Quindío had the lowest proportions of forest (13-16%) and core forest (about 2.5%), and relatively little other wooded land (about 8%). Esparza had a high proportion of forest (44%) but little other wooded land (5%), while Copán had a moderate amount of forest (21%) as well as considerable other wooded land (35%, including land planted in shade-grown coffee). Crops and plantations comprised a minor part of each landscape, with the exception of shade-grown coffee in Copán (23% of land) and sun-grown coffee in Quindío (15% of land).

Esparza was the only landscape with many large, interconnected forest patches, a feature that is reflected in its high values for percent core forest and proximity of forested patches (Table 4-3). However, other wooded land uses contributed to structural connectivity in Copán and Matiguás, as indicated by their moderate to high values for proximity of tree-covered patches. Quindío had the lowest proximity values, indicating that its forested and tree-covered patches were relatively small, fragmented, and isolated. The Copán and Matiguás landscapes contained a relatively high proportion of intermediate-intensity land uses such as forest fallows, shade-grown coffee, and pasture with high tree density. This characteristic resulted in low mean contrast among adjacent patches and correspondingly low values for contrast-weighted edge density (CWED). In Esparza, and to a lesser extent in Quindío, there tended to be more bifurcation between wooded areas and intensively-managed farmland, resulting in higher CWED values.

Table 4-3. Landscape composition and structure of the four study landscapes. Land use percentages add to 100% except where rounding has resulted in small deviations. See the main text for descriptions of the landscape structure metrics.

Landscape characteristic	Copán, Honduras	Matiguás, Nicaragua	Esparza, Costa Rica	Quindío, Colombia
A. Landscape composition (% of the landscape in each land use category)				
Upland forest	14.5	7.7	36.2	3.2
Riparian forest	6.3	5.3	7.7	12.4
Forest subtotal	***20.8***	***13.0***	***43.9***	***15.6***
Thin/disturbed forest	4.1	1.6	0.4	0.4
Successional vegetation	7.8	6.3	2.7	1.9
Timber plantation	0.0	0.2	1.9	3.3
Diversified / shaded perennial	23.1	0.1	0.0	2.0
Other wooded land uses subtotal	***35.0***	***8.5***	***5.1***	***7.5***
Pasture, high tree density	3.3	13.8	1.7	1.8
Pasture, low tree density	31.8	49.0	38.5	33.8
Pasture, no trees	4.7	9.7	2.3	15.9
Pasture subtotal	***39.8***	***72.5***	***42.6***	***51.4***
Unshaded perennial	0.5	0.1	0.1	15.1
Annual crops	0.8	0.4	1.9	1.3
Crop land uses subtotal	***1.3***	***0.5***	***2.0***	***16.4***
Human settlement	2.6	0.5	2.8	1.7
Other / no data	0.5	5.0	3.6	7.3
B. Landscape structure (metrics from the software program FRAGSTATS)				
Contrast-weighted edge density	41.2	43.2	74.6	59.1
% core forest area	10.8	2.4	19.7	2.6
log (proximity of forested patches)	2.98	3.13	5.45	2.91
log (proximity of tree-covered patches)	5.56	4.54	5.47	3.41

4.4.3 Relationships between land use and faunal assemblages

Bird species richness, adjusted abundance, and BCV were all significantly related to land use in all four landscapes ($p<0.0001$ for all response variables in all landscapes). The results for adjusted bird abundance closely mirrored those for bird species richness, so only the species richness results are presented here. In Copán, bird species richness and BCV showed similar patterns: both were relatively high in successional vegetation and shade-grown coffee, slightly lower in secondary and riparian forests, and lowest in pasture and pine forest (Figure 4-2a). In Matiguás, pairwise comparisons revealed no significant differences in bird species richness among land uses, except for pasture without trees, which had a significantly lower value than the other land uses (Tukey hsd test, $\alpha=0.05$). However, the BCV metric revealed striking differences in the conservation value of species assemblages among the land uses. Secondary and riparian forests had the highest mean BCV, followed closely by primary forest and thin forest. Successional vegetation had an intermediate value. BCV in pasture land uses responded to gradients of tree cover: pasture with high tree density had an intermediate BCV, while pasture with low tree density had a lower BCV and pasture without trees was lower still (Figure 4-2b).

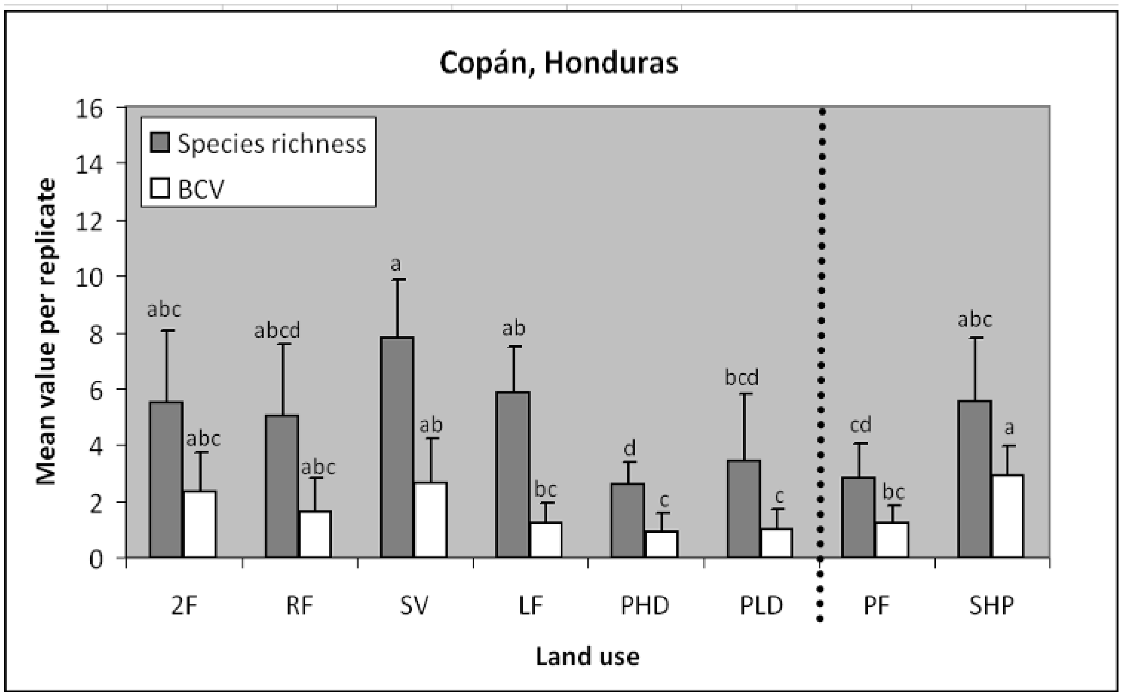


Figure 4-2a. One-way ANOVAs of bird species richness and BCV by land use for the Copán landscape. ANOVA for species richness: $F_{(7,71)} = 7.45$, $r^2 = 0.42$, $p < 0.0001$; for BCV: $F_{(7,71)} = 5.28$, $r^2 = 0.34$, $p < 0.0001$. The six land uses to the left of the dotted line are common to all four landscapes. Error bars indicate one standard deviation. Different letters above each data series denote significant differences between pairs of land uses (Tukey's hsd test; $p < 0.05$). Land use abbreviations are as follows: 2F = secondary forest; RF = riparian forest; SV = successional vegetation; LF = live fence; PHD = pasture with high tree density; PLD = pasture with low tree density; PF = pine forest; SHP = shaded perennial crops.

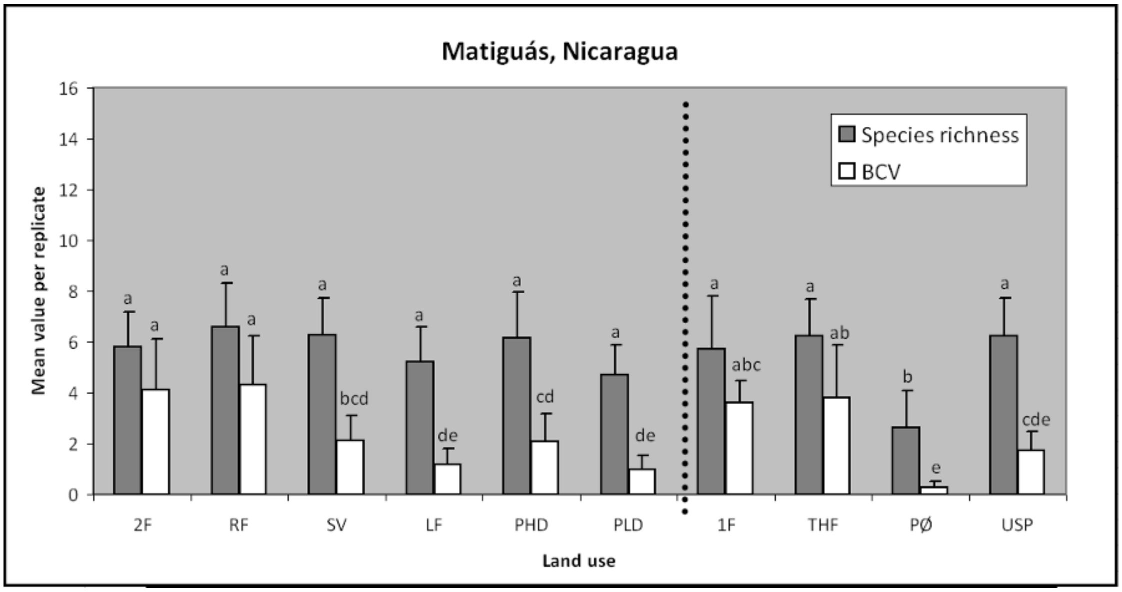


Figure 4-2b. One-way ANOVAs of bird species richness and BCV by land use for the Matiguás landscape. ANOVA for species richness: $F_{(9,103)} = 6.19$, $r^2 = 0.35$, $p < 0.0001$; for BCV: $F_{(9,103)} = 14.61$, $r^2 = 0.56$, $p < 0.0001$. The six land uses to the left of the dotted line are common to all four landscapes. Error bars indicate one standard deviation. Different letters above each data series denote significant differences between pairs of land uses (Tukey's hsd test; $p < 0.05$). Land use abbreviations are as follows: 2F = secondary forest; RF = riparian forest; SV = successional vegetation; LF = live fence; PHD = pasture with high tree density; PLD = pasture with low tree density; 1F = primary forest; THF = thin forest; PØ = pasture without trees; USP = unshaded perennial crops.

In Esparza, mean bird species richness was not dramatically different among the six common land uses. However, the BCV metric differentiated land uses into four distinct clusters: primary forest had the highest mean per-replicate BCV; secondary forest, riparian forest, and successional vegetation had the next highest values; live fences, plantations, and pastures with high and low tree density were lower; and pasture without trees registered a mean per replicate BCV barely above zero (Figure 4-2c). Quindío displayed the greatest degree of distinction among land uses, and these patterns were substantially different for BCV than for bird species richness (Figure 4-2d). Unexpectedly, species richness was highest in agricultural land uses including plantations, pasture with high tree density, and pasture with low tree density. Primary, secondary, and riparian forests all had lower mean species richness per replicate. However, these patterns were almost reversed for BCV, with the forest land uses registering the highest per-replicate scores and pasture land uses having considerably lower values.

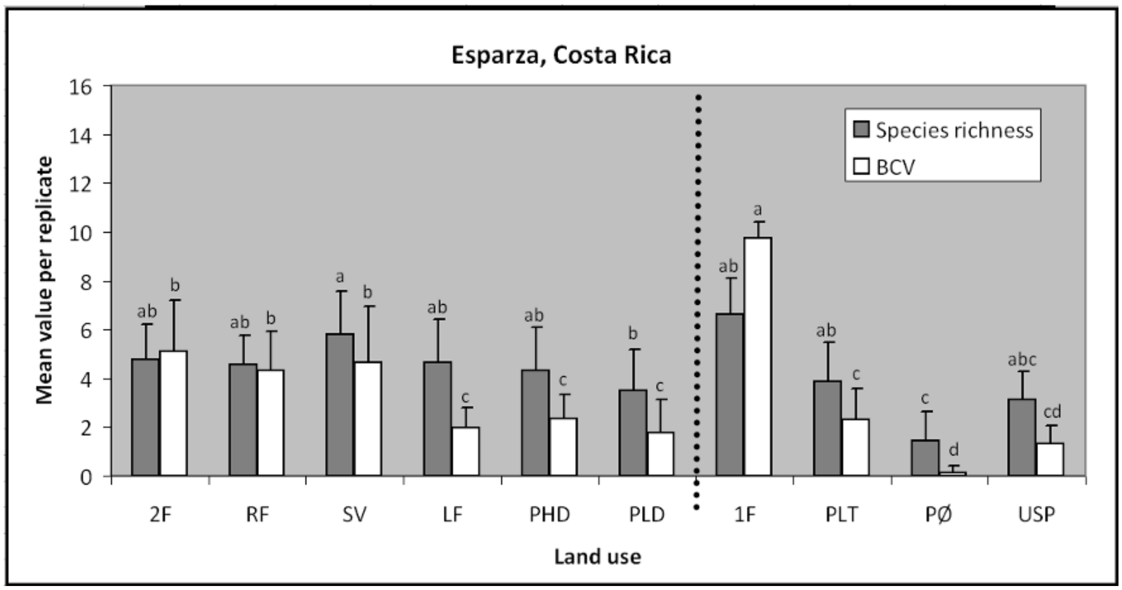


Figure 4-2c. One-way ANOVAs of bird species richness and BCV by land use for the Esparza landscape. ANOVA for species richness: $F_{(9,111)} = 8.09$, $r^2 = 0.40$, $p < 0.0001$; for BCV: $F_{(9,111)} = 23.47$, $r^2 = 0.66$, $p < 0.0001$. The six land uses to the left of the dotted line are common to all four landscapes. Error bars indicate one standard deviation. Different letters above each data series denote significant differences between pairs of land uses (Tukey's hsd test; $p < 0.05$). Land use abbreviations are as follows: 2F = secondary forest; RF = riparian forest; SV = successional vegetation; LF = live fence; PHD = pasture with high tree density; PLD = pasture with low tree density; 1F = primary forest; PLT = plantation; PØ = pasture without trees; USP = unshaded perennial crops.

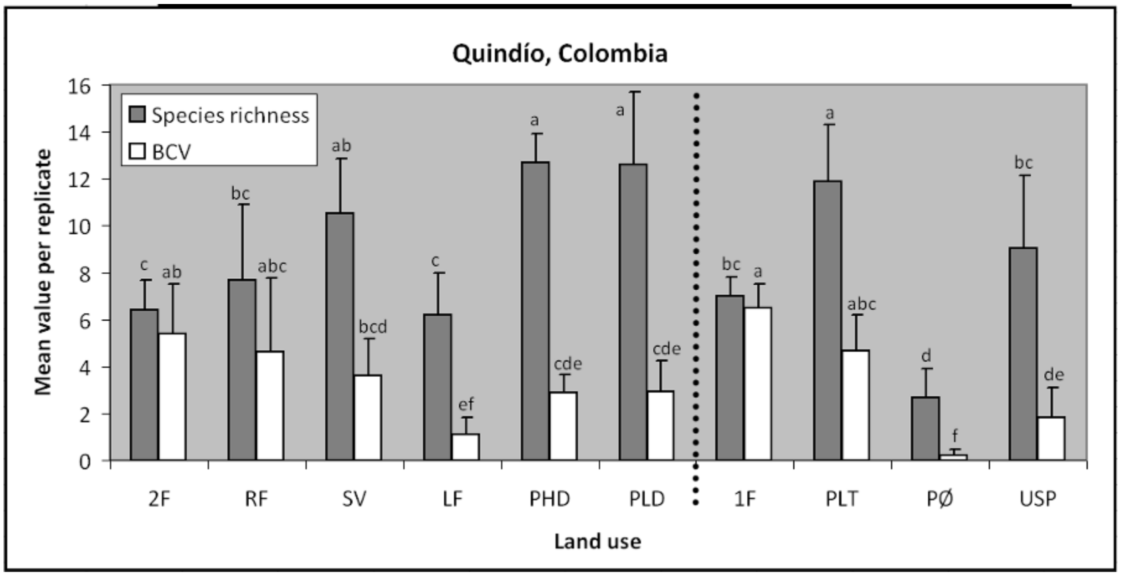
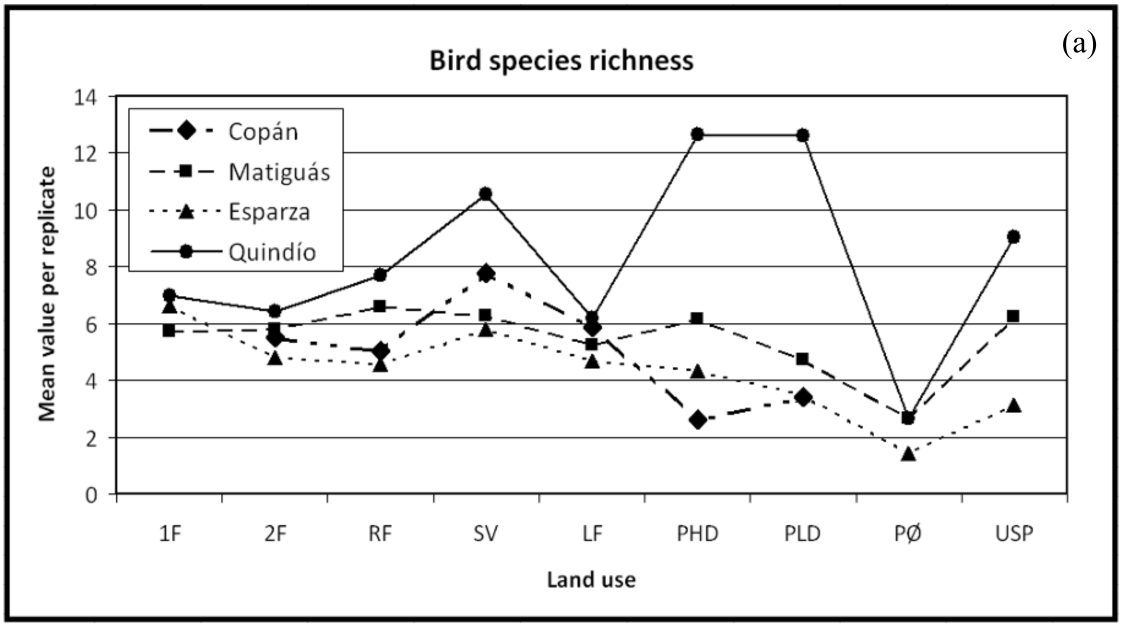
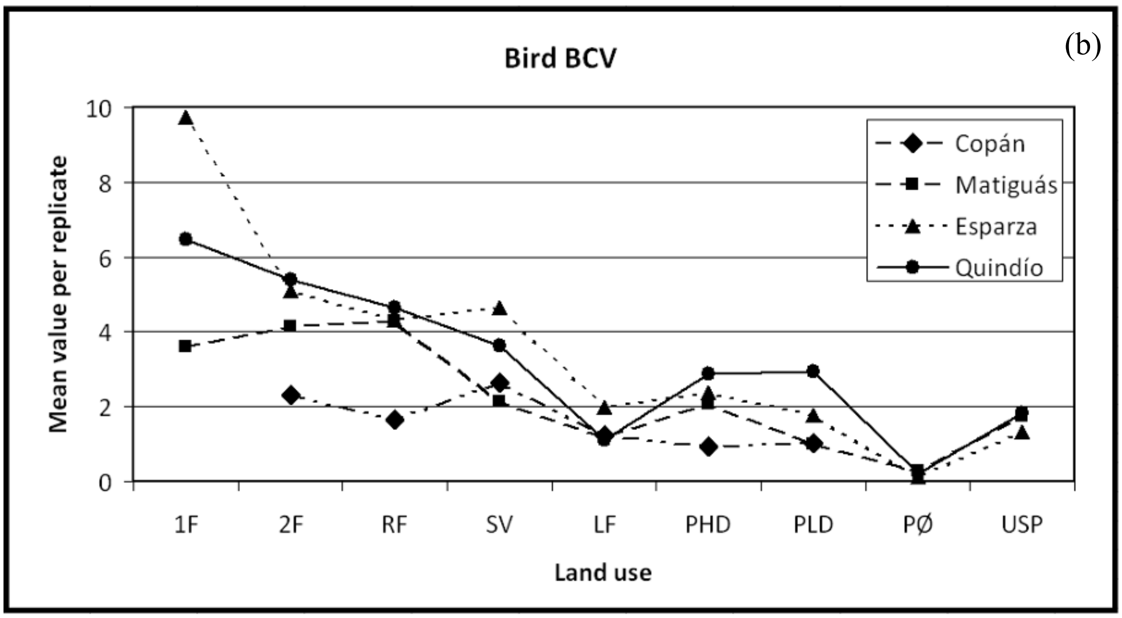


Figure 4-2d. One-way ANOVAs of bird species richness and BCV by land use for the Quindío landscape. ANOVA for species richness: $F_{(9,103)} = 31.88$, $r^2 = 0.74$, $p < 0.0001$; for BCV: $F_{(9,103)} = 24.99$, $r^2 = 0.69$, $p < 0.0001$. The six land uses to the left of the dotted line are common to all four landscapes. Error bars indicate one standard deviation. Different letters above each data series denote significant differences between pairs of land uses (Tukey's hsd test; $p < 0.05$). Land use abbreviations are as follows: 2F = secondary forest; RF = riparian forest; SV = successional vegetation; LF = live fence; PHD = pasture with high tree density; PLD = pasture with low tree density; 1F = primary forest; PLT = plantation; PØ = pasture without trees; USP = unshaded perennial crops.

Across all four landscapes, land use and landscape identity were both significant predictors of bird species richness and BCV (two-way ANOVA: $p<0.0001$ for both response variables and both predictor variables; Figures 4-3a and 4-3b). In other words, bird species richness and BCV varied consistently with land use, but other, landscape-specific, factor were also important in influencing these attributes. For bird species richness, landscape identity was a more important predictive factor than land use ($F = 94.0$ and 23.4, respectively), while for BCV, land use was more important ($F = 32.3$ and 42.4, respectively). These results imply that the BCV metric may reflect habitat conditions more reliably than species richness, which can attain high values due to any combination of forest-dependent, forest-generalist, or open land species.





Figures 4-3a and 4-3b. Two-way ANOVAs of bird species richness (a) and bird BCV (b) by landscape and land use. Results are shown only for land uses that are common to at least three of the landscapes. Bird species richness and BCV are given as means per sample replicate. Land use abbreviations are the same as those in Figures 4-2b through 4-2d. ANOVA for species richness: full model $r^2 = 0.70$; land use $F_{(12,388)} = 23.43$, $p < 0.0001$; landscape $F_{(3,388)} = 94.01$, $p < 0.0001$; interaction $F_{(22,388)} = 10.25$, $p < 0.0001$. ANOVA for BCV: full model $r^2 = 0.64$; land use $F_{(12,388)} = 42.37$, $p < 0.0001$; landscape $F_{(3,388)} = 32.28$, $p < 0.0001$; interaction $F_{(22,388)} = 4.71$, $p < 0.0001$.

Butterfly species richness and adjusted abundance were significantly related to land use in Copán and Esparza, but not in Matiguás or Quindío (Figures 4-4a through 4-4d). In Copán, butterfly species richness and adjusted abundance were highest in the linear land uses (riparian forest and live fences), intermediate in pastures, and lowest in secondary forest (Figure 4-4a). In Esparza, land uses sorted into two clusters with respect to mean butterfly species richness and adjusted abundance. The higher values were observed in secondary forest, riparian forest, and successional vegetation, while lower values were found in the more open pasture and live fence habitats (Figure 4-4c).

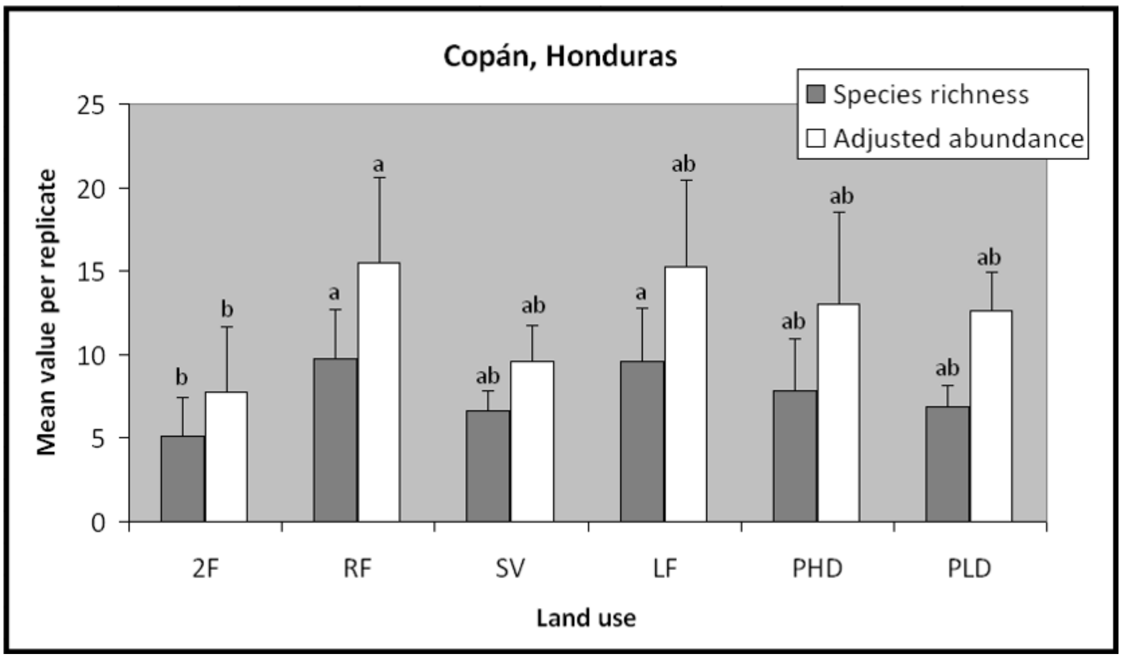


Figure 4-4a. One-way ANOVAs of butterfly species richness and square root adjusted abundance by land use for the Copán landscape. ANOVA for species richness: $F_{(5,30)} = 3.16$, $r^2 = 0.34$, $p = 0.02$; for adjusted abundance: $F_{(5,30)} = 3.11$, $r^2 = 0.34$, $p = 0.02$. Error bars indicate one standard deviation. Different letters above each data series denote significant differences between pairs of land uses (Tukey's hsd test; $p < 0.05$). Land use abbreviations are the same as those in Figure 4-2a.

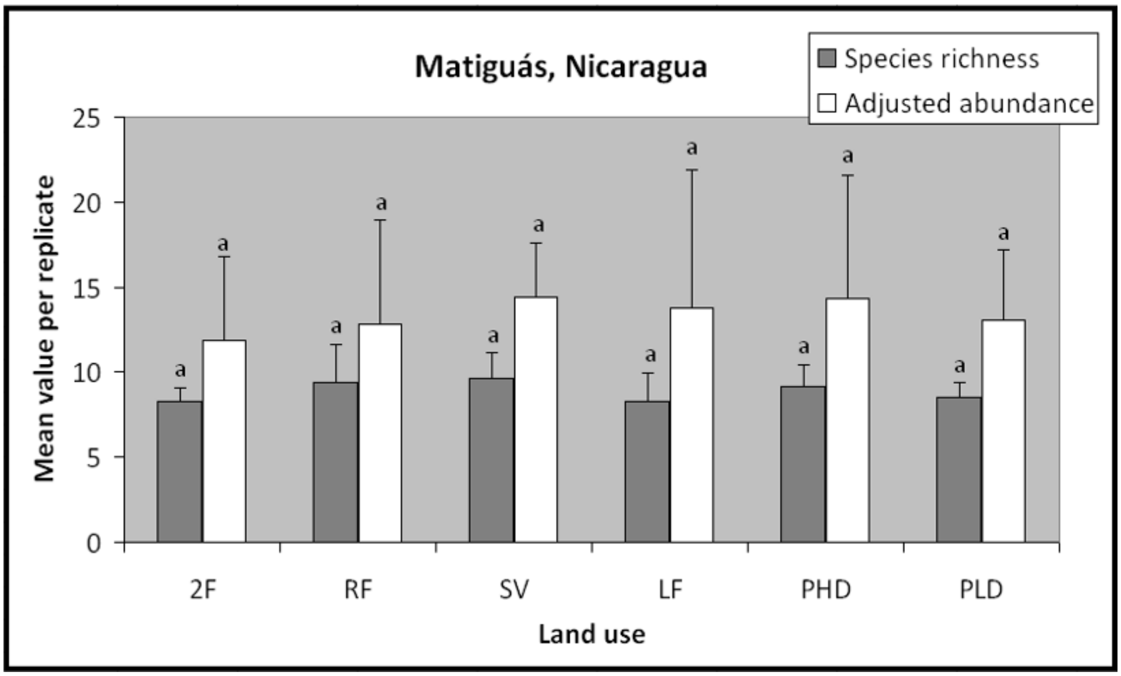


Figure 4-4b. One-way ANOVAs of butterfly species richness and square root adjusted abundance by land use for the Matiguás landscape. ANOVA for species richness: $F_{(5,27)} = 0.81$, $r^2 = 0.13$, $p = 0.55$; for adjusted abundance: $F_{(5,27)} = 0.73$, $r^2 = 0.12$, $p = 0.61$. Error bars indicate one standard deviation. Different letters above each data series denote significant differences between pairs of land uses (Tukey's hsd test; $p < 0.05$). Land use abbreviations are the same as those in Figure 4-2b.

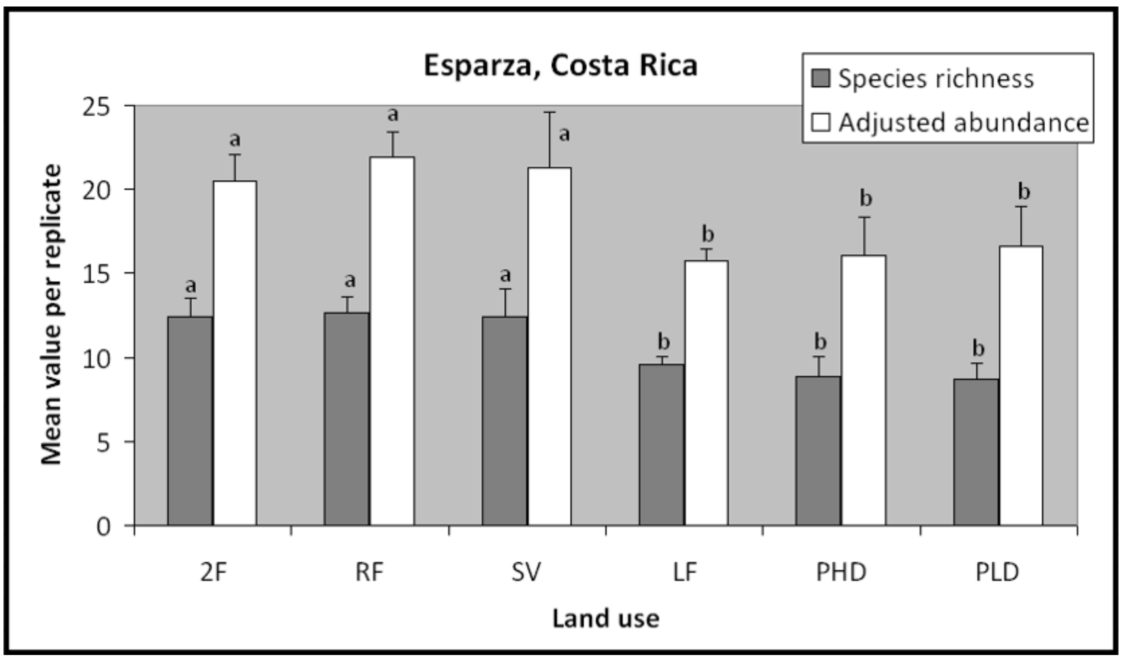


Figure 4-4c. One-way ANOVAs of butterfly species richness and square root adjusted abundance by land use for the Esparza landscape. ANOVA for species richness: $F_{(5,27)} = 16.95$, $r^2 = 0.76$, $p < 0.0001$; for adjusted abundance: $F_{(5,27)} = 9.21$, $r^2 = 0.63$, $p < 0.0001$. Error bars indicate one standard deviation. Different letters above each data series denote significant differences between pairs of land uses (Tukey's hsd test; $p < 0.05$). Land use abbreviations are the same as those in Figure 4-2c.

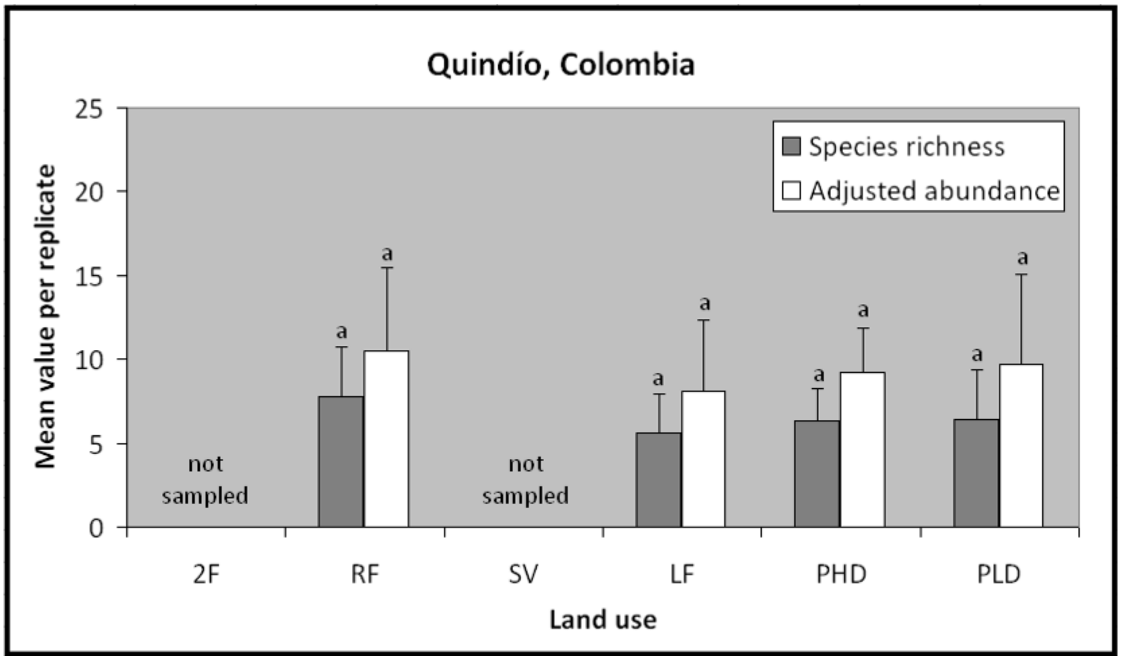
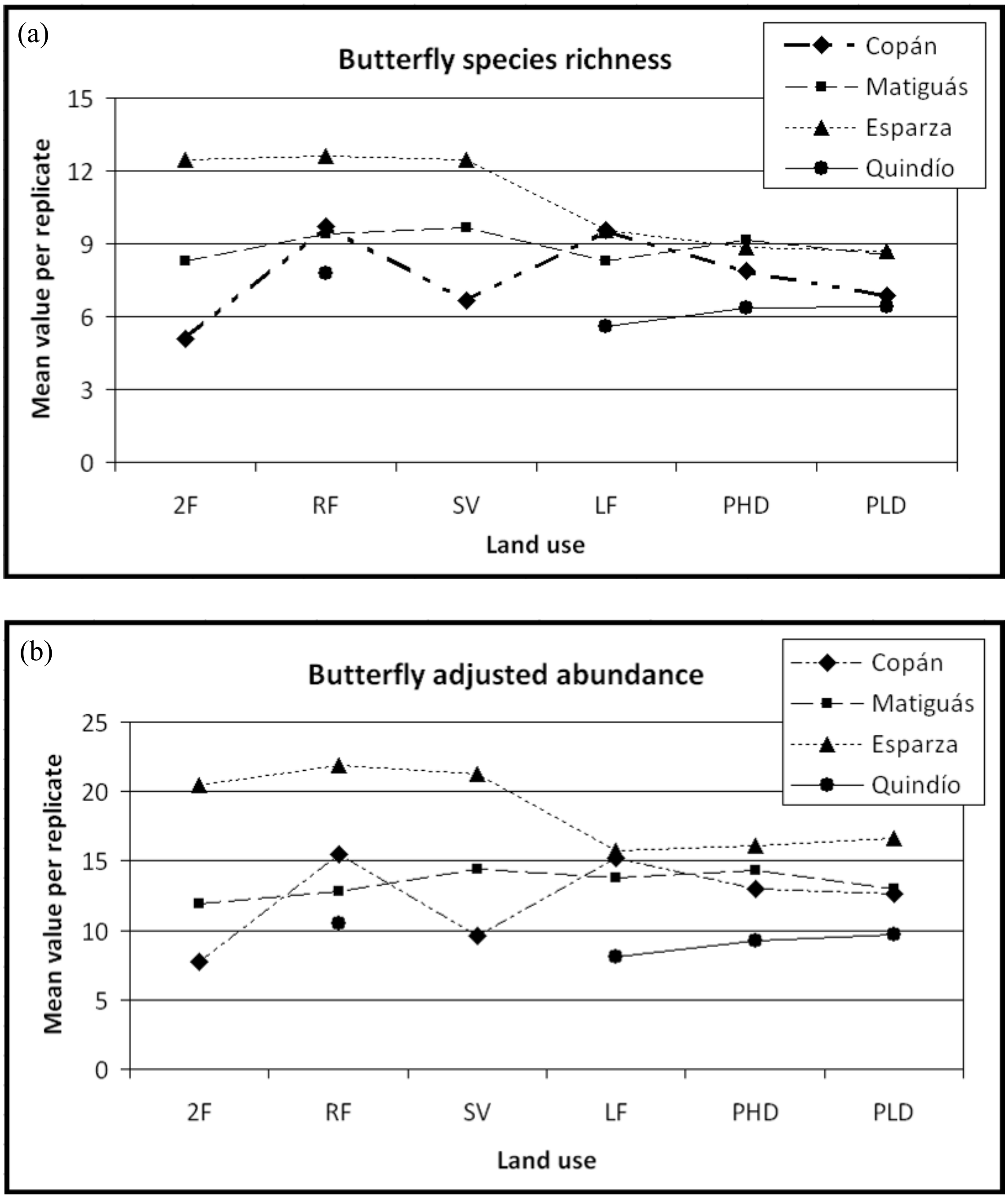


Figure 4-4d. One-way ANOVAs of butterfly species richness and square root adjusted abundance by land use for the Quindío landscape. ANOVA results for species richness: $F_{(3,20)} = 0.74$, $r^2 = 0.10$, $p = 0.54$; for adjusted abundance: $F_{(3,20)} = 0.31$, $r^2 = 0.04$, $p = 0.82$. Error bars indicate one standard deviation. Different letters above each data series denote significant differences between pairs of land uses (Tukey's hsd test; $p < 0.05$). Land use abbreviations are the same as those in Figure 4-2d.

Across all four landscapes, butterfly species richness and adjusted abundance were significantly related both to land use and to landscape identity (two-way ANOVA: $p<0.0001$ for both response variables and both predictor variables; Figures 4-5a and 4-5b). However, landscape identity was a more important predictor than land use for both response variables ($F = 17.7$ and 6.4 for species richness, and $F = 27.1$ and 8.0 for adjusted abundance, respectively). Furthermore, the relative species richness of different land uses was not entirely consistent across all four landscapes. This was particularly true for Copán, which had the lowest values in secondary forest and relatively high values in live fences, in contrast to the other sites.



Figures 4-5a and 4-5b. Two-way ANOVAs of butterfly species richness (a) and square root adjusted abundance (b) by landscape and land use. Land use abbreviations are the same as those in Figures 4-2a through 4-2d. ANOVA for species richness: full model $r^2 = 0.64$; land use $F_{(9,116)} = 6.43$, $p < 0.0001$; landscape $F_{(3,116)} = 17.65$, $p < 0.0001$; interaction $F_{(15,116)} = 4.48$, $p < 0.0001$. ANOVA for adjusted abundance: full model $r^2 = 0.69$; land use $F_{(9,116)} = 7.96$, $p < 0.0001$; landscape $F_{(3,116)} = 27.13$, $p < 0.0001$; interaction $F_{(15,116)} = 4.11$, $p < 0.0001$.

4.5 DISCUSSION

4.5.1 Faunal assemblage characteristics

The bird assemblages observed in all four landscapes were heavily skewed toward non-forest-dependent and lower conservation value species. For example, in Costa Rica overall, 24% of land birds are forest-dependent, 35% are forest-generalists, and 40% are non-forest-dependent (Stiles 1985). By contrast, these categories comprised 5%, 27%, and 67%, respectively, of the bird assemblage observed in Esparza. Nevertheless, some higher conservation value species were present in each landscape. In Quindío, for instance, the sample included four endemic species and eight additional restricted-range species.

The under-representation of less common bird species (including many forest-dependent species) in the samples likely reflects both the actual habitat suitability of these agricultural landscapes and the difficulty in detecting many such species even when they are present. The detection bias explanation is supported by the fact that all landscapes had a relatively high percentage of species that were observed only once or twice. These singletons and doubletons totaled 15-27% of bird species and 19-34% of butterfly species. It is well known that most plant and animal groups tend to have a modest proportion of abundant species and a longer "tail" of uncommon species (Preston 1948). Thus, it is logical to infer that additional uncommon species were present in the study landscapes but not detected.

Nevertheless, the number of additional unobserved species in these landscapes is probably modest. The field methods used and the level of sampling conducted were appropriate for detecting rare species (Nur 1998), and species accumulation curves indicated that the samples detected approximately 83-89% of all bird species and 74-89% of all butterfly species actually present in each landscape (Table 4-2). Furthermore, even if additional high conservation value species were present in the study landscapes, their scarcity implies that these agricultural landscapes may not be providing high-quality breeding habitat. Long-term bird monitoring in Turrialba, Costa Rica, for example, has found that many of the forest-dependent species that enter agricultural land uses do so only as young adult males seeking new breeding territories (F.A.J. DeClerck and A. Martínez, unpublished data). As such, these land uses could constitute either valuable dispersal routes or ecological traps, but not suitable breeding habitat.

4.5.2 Performance of the BCV metric

The results of this study demonstrate the ways in which traditional metrics for quantifying faunal assemblages—particularly species richness and abundance—can mask the paucity of high conservation value species in human-dominated landscapes because they are insensitive to species identity. In contrast, the BCV metric proved to be a helpful tool for differentiating faunal assemblages by composition based on their aggregate conservation value. The five component factors in the BCV metric displayed little autocorrelation and captured many of the key species traits that conservationists tend to value.

In Esparza and Matiguás, BCV was a much more sensitive indicator than species richness of bird responses to habitat conditions. In other words, bird species richness in these landscapes was only moderately different among land uses, and many of these differences lacked statistical significance. The BCV metric essentially amplified the observed patterns in species richness, revealing statistically significant differences not captured by the species richness metric. The results from Quindío were the most noteworthy because species richness and BCV displayed contrasting patterns. Based on the species richness indicator, one would conclude that pastures with trees were the highest conservation value land use, followed by plantations and successional vegetation; primary forest would be among the lower value habitats. On the other hand, BCV closely tracked the level of tree cover, indicating primary and secondary forests to be the highest conservation value habitats. These examples illustrate how the choice of species assemblage metrics can dictate the conclusions reached about the relative conservation value of different habitats. As suggested earlier, species richness can provide a misleading picture of the relative conservation value of human-dominated landscapes.

4.5.3 Relationships between land use and faunal assemblages

Relationships between land use and bird assemblage characteristics were generally consistent across all four landscapes, especially for the BCV metric. As might be expected, BCV was highest in primary, secondary, and riparian forest, and slightly lower in successional vegetation. The samples revealed little distinction between pasture with high tree density and pasture with low tree density, but both of these had substantially higher BCV than pasture without trees. Live fences, which have been previously characterized as having high conservation value in agricultural mosaics (e.g., Lang et al. 2003; Harvey et al. 2005b), had relatively high species richness but low BCV consistent with their open land context.

Relationships between land use and butterfly assemblage characteristics were weaker and less consistent than those for birds. I suggest three explanations for this result. First, there appear to be genuine differences in the butterfly habitat value of different land uses in different landscapes. The most anomalous landscape was Copán, where the linear land uses (riparian forest and live fences) had the greatest mean butterfly abundance and species richness, while secondary forest had the least. (See Section 3.5.2 for further explanation of these patterns.) Second, as suggested in the last sub-section, the species richness and abundance metrics may be insensitive to important differences in species composition among land uses. Some of these differences in composition are revealed by the forest dependence analysis for Copán butterflies presented in Section 3.4.2. Third, it is possible that greater sampling effort would have been needed to reveal clear distinctions among land uses. This explanation is supported by the fact that the two landscapes with the greatest butterfly sampling effort—Esparza and Copán—are also the ones that displayed the clearest distinctions in butterfly assemblage characteristics among land uses.

The consistent patterns of bird responses (and general trends of butterfly responses) to land use and agricultural practices add substantial support to conservation and management recommendations previously advocated on the basis of single-site studies that reached similar conclusions. These recommendations include adding natural vegetation to farm plots, installing live fences or hedgerows between plots, conserving patches of natural forest, and increasing vegetative diversity with elements such as forest fallows, plantations, and agroforestry (e.g., Harvey 2008b). However, while these are all positive steps for conservation, their benefit for sustaining globally significant biodiversity must not be overstated. The fact that four agricultural mosaics characterized by significant forest and tree cover together contained only five observed Red Listed bird species (two Vulnerable and three Near Threatened) suggests that it is probably over-optimistic to expect that Neotropical pasture-dominated landscapes will provide high-quality habitat or functional corridors for many of the species of greatest conservation concern. Such conservation needs are more likely to be met by large protected areas, or perhaps by production systems that more closely resemble native forests, such as extractive reserves or eco-certified forestry.

What these pasture-dominated landscapes do provide is habitat for species of intermediate conservation concern (including many migratory songbirds) and species that contribute important ecosystem services (e.g., seed dispersal, carrion disposal, and charismatic value for ecotourism and local enjoyment). To be most effective, the academic and policy discourses on conservation in human-dominated landscapes must be realistic about the degree to which different conservation objectives can actually be met in such settings. Greater orientation toward ecosystem service values and toward species of intermediate conservation concern—as distinguished by metrics such as BCV—might help conservationists and policy makers better articulate the rationale for conservation in agricultural landscapes while focusing on landscape interventions that hold the greatest prospect for success.

4.5.4 Toward landscapes as the unit of analysis

Understanding the relationships between agricultural management, biodiversity conservation, and ecosystem services requires a landscape perspective, especially in heterogeneous land use mosaics (Tscharntke et al. 2005). Evaluating plot-scale conservation outcomes at many sites across a landscape tells only part of the story; the other part is the ways in which local populations and habitat features interact to create broader patterns of beta and gamma diversity that are tied to properties of the entire landscape mosaic and its context. In this regard, the present study may generate as many questions as answers.

Differences among the four study landscapes hint at some interesting correspondences between landscape factors and assemblages of bird and butterflies. For instance, Quindío has the greatest bird species richness and total BCV although it has among the least forest and tree cover. This bird diversity likely reflects Quindío's location in a global hotspot that contains significantly more bird species than the native habitat types around Copán, Matiguás, or Esparza. However, fine-scale heterogeneity and land cover diversity also appear to be important contributors to landscape-wide species diversity, as documented previously for Copán (see Chapter 3) and as seen in Matiguás, which has relatively little forest but is dominated by small farms containing a high proportion of intermediate-intensity land uses.

While it is tempting to speculate about such explanations, at this point they are mainly hypotheses: given that there are more possible co-variables than landscapes in this study, it is impossible definitively to tease apart the factors resulting in different landscape-wide conservation outcomes. Both the findings and the limitations of the current study point the way toward using landscapes as the unit of analysis in future research on conservation in human-dominated environments (Bennett et al. 2006). Such research would help inform burgeoning efforts to use landscapes as the spatial unit for sustainable land management, eco-certification, and the integration of conservation and livelihood objectives (see Chapter 2). Research that uses landscapes as the sampling unit will require large-scale, coordinated effort and must strike a balance between selecting landscapes that are diverse enough to be widely representative of the systems of interest but not so diverse that the number of co-variates becomes unwieldy. Such research would eventually enable us to draw much broader-reaching conclusions while gaining insight into the combinations of biophysical conditions and land management practices that help sustain biodiversity across larger spatial and temporal scales.

LITERATURE CITED

Baillie, J.E.M., C. Hilton-Taylor, and S.N. Stuart. 2004. 2004 IUCN Red List of Threatened Species: a global assessment. IUCN, Gland, Switzerland.

Balmford, A., R.E. Green, and J.P.W. Scharlemann. 2005. Sparing land for nature: exploring the potential impact of changes in agricultural yield on the area needed for crop production. Global Change Biology 11: 1594-1605.

Beissinger, S.R., J.M. Reed, J.M. Wunderle, S.K. Robinson, and D.M. Finch. 2000. Report of the AOU Conservation Committee on the Partners in Flight species prioritization plan. The Auk 117: 549-61.

Bennett, A.F., J.Q. Radford, and A. Haslem. 2006. Properties of land mosaics: implications for nature conservation in agricultural environments. Biological Conservation 133: 250-264.

Billeter, R., J. Liira, D. Bailey, R. Bugter, P. Arens, I. Augenstein, S. Aviron, J. Baudry, R. Bukacek, F. Burel, M. Cerny, G. De Blust, R. De Cock, T. Diekötter, H. Dietz, J. Dirksen, C. Dormann, W. Durka, M. Frenzel, R. Hamersky, F. Hendrickx, F. Herzog, S. Klotz, B. Koolstra, A. Lausch, D. Le Coeur, J.P. Maelfait, P. Opdam, M. Roubalova, A. Schermann, N. Schermann, T. Schmidt, O. Schweiger, M.J.M. Smulders, M. Speelmans, P. Simova, J. Verboom, W.K.R.E. van Wingerden, M. Zobel, and P.J. Edwards. 2008. Indicators for biodiversity in agricultural landscapes: a pan-European study. Journal of Applied Ecology 45: 141-150.

BirdLife International. 2009. World Bird Database. Online: www.birdlife.org/datazone/index.html.

Chao, A. 1987. Estimating the population size for capture-recapture data with unequal catchability. Biometrics 43: 783-791.

Clay, J. 2004. World agriculture and the environment. Island Press, Washington, D.C.

Colwell, R.K. 2006. EstimateS: statistical estimation of species richness and shared species from samples. Version 8. Online: purl.oclc.org/estimates.

Daily, G.C., P.R. Ehrlich, and G.A. Sanchez-Azofeifa. 2001. Countryside biogeography: use of human-dominated habitats by the avifauna of southern Costa Rica. Ecological Applications 11: 1-13.

Daily, G.C., G. Ceballos, J. Pacheco, G. Suzan, and A. Sanchez-Azofeifa. 2003. Countryside biogeography of neotropical mammals: conservation opportunities in agricultural landscapes of Costa Rica. Conservation Biology 17: 1814-1826.

Duelli, P., and M.K. Obrist. 2003. Biodiversity indicators: the choice of values and measures. Agriculture, Ecosystems, and Environment 98: 87-98.

Estrada, A., R. Coates-Estrada, and D. Meritt. 1993. Bat species richness and abundance in tropical rain-forest fragments and in agricultural habitats at Los Tuxtlas, Mexico. Ecography 16: 309-318.

Estrada, A., R. Coates-Estrada, and D. Meritt. 1994. Non-flying mammals and landscape changes in the tropical rain-forest region of Los Tuxtlas, Mexico. Ecography 17: 229-241.

Estrada, A., R. Coates-Estrada, and D. Meritt. 1997. Anthropogenic landscape changes and avian diversity at Los Tuxtlas, Mexico. Biodiversity and Conservation 6: 19-43.

Estrada, A., R. Coates-Estrada, and D. Meritt. 1998. Dung and carrion beetles in tropical rain forest fragments and agricultural habitats at Los Tuxtlas, Mexico. Journal of Tropical Ecology 14: 577-593.

Estrada, A., P. Cammarano, and R. Coates-Estrada. 2000. Bird species richness in vegetation fences and in strips of residual rain forest vegetation at Los Tuxtlas, Mexico. Biodiversity and Conservation 9: 1399-1416.

FAO [Food and Agriculture Organization of the United Nations]. 2006. Livestock's long shadow. FAO, Rome.

FAOSTAT. 2004. Database of the Food and Agricultural Organization of the United Nations. Online: faostat.fao.org.

Fischer, J., B. Brosi, G.C. Daily, P.R. Ehrlich, R. Goldman, J. Goldstein, D.B. Lindenmayer, A.D. Manning, H.A. Mooney, L. Pejchar, J. Ranganathan, and H. Tallis. 2008. Should agricultural policies encourage land sparing or wildlife-friendly farming? Frontiers in Ecology and Environment 6: 380-385.

Götmark, F., M. Åhlund, and M.O.G. Eriksson. 1986. Are indices reliable for assessing conservation value of natural areas? An avian case study. Biological Conservation 38: 55-73.

Green R.E., S.J. Cornell, J.P.W. Scharlemann, and A. Balmford. 2005. Farming and the fate of wild nature. Science 307: 550-555.

Harvey, C.A., F. Alpízar, M. Chacón, and R. Madrigal. 2005a. Assessing linkages between agriculture and biodiversity in Central America: historical overview and future perspectives. The Nature Conservancy, San José, Costa Rica.

Harvey, C.A., C. Villanueva, J. Villacis, M. Chacon, D. Munoz, M. Lopez, M. Ibrahim, R. Gomez, R. Taylor, J. Martinez, A. Navas, J. Saenz, D. Sanchez, A. Medina, S. Vilchez, B. Hernandez, A. Perez, F. Ruiz, F. Lopez, I. Lang, and F.L. Sinclair. 2005b. Contribution of live fences to the ecological integrity of agricultural landscapes. Agriculture, Ecosystems and Environment 111: 200-230.

Harvey, C.A., A. Medina, D.M. Sanchez, S. Vilchez, B. Hernandez, J.C. Saenz, J.M. Maes, F. Casanoves, and F.L. Sinclair. 2006. Patterns of animal diversity in different forms of tree cover in agricultural landscapes. Ecological Applications 16: 1986-1999.

Harvey, C.A., O. Komar, R. Chazdon, B.G. Ferguson, B. Finegan, D.M. Griffith, M. Martinez-Ramos, H. Morales, R. Nigh, L. Soto-Pinto, M. van Breugel, and M. Wishnie. 2008a. Integrating agricultural landscapes with biodiversity conservation in the Mesoamerican hotspot. Conservation Biology 22: 8-15.

Harvey, C.A. 2008b. Designing agricultural landscapes for biodiversity conservation. Pages 146-165 in S.J. Scherr and J.A. McNeely. Farming with nature. Island Press, Washington, D.C.

Hughes, J.B., G.C. Daily, and P.R. Ehrlich. 2002. Conservation of tropical forest birds in countryside habitat. Ecology Letters 5: 121-129.

Lang, I., L.H.L. Gormley, C.A. Harvey, and F.L. Sinclair. 2003. Composición de la comunidad de aves en cercas vivas de Río Frío, Costa Rica. Agroforestería en las Américas 39-40: 86-92.

Kremen, C. 1992. Assessing the indicator properties of species assemblages for natural areas monitoring. Ecological Applications 2: 203-217.

Lindell, C.A., W.H. Chomentowski, J.R. Zook, and S.A. Kaiser. 2006. Generalizability of neotropical bird abundance and richness models. Animal Conservation 9: 445-455.

Lindenmayer, D., R.J. Hobbs, R. Montague-Drake, J. Alexandra, A. Bennett, M. Burgman, P. Cale, A. Calhoun, V. Cramer, P. Cullen, D. Driscoll, L. Fahrig, J. Fischer, J. Franklin, Y. Haila, M. Hunter, P. Gibbons, S. Lake, G. Luck, C. MacGregor, S. McIntyre, R. Mac Nally, A. Manning, J. Miller, H. Mooney, R. Noss, H. Possingham, D. Saunders, F. Schmiegelow, M. Scott, D. Simberloff, T. Sisk, G. Tabor, B. Walker, J. Wiens, J. Woinarski, and E. Zavaleta. 2008. A checklist for ecological management of landscapes for conservation. Ecology Letters 11: 78-91.

Matson, P.A., W.J. Parton, A.G. Power, and M. Swift. 1997. Agricultural intensification and ecosystem properties. Science 277: 504-509.

Mayfield, M.M., and G.C. Daily. 2005. Countryside biogeography of Neotropical herbaceous and shrubby plants. Ecological Applications 15: 423-439.

McGarigal, K., S.A. Cushman, M.C. Neel, and E. Ene. 2002. FRAGSTATS: spatial pattern analysis program for categorical maps. University of Massachusetts, Amherst, Massachusetts. Online: www.umass.edu/landeco/research/fragstats/fragstats.html.

McNeely, J.A., and S.J. Scherr. 2003. Ecoagriculture. Island Press, Washington, D.C.

Mehlman, D.W., K.V. Rosenberg, J.V. Wells, and B. Robertson. 2004. A comparison of North American avian conservation priority ranking systems. Biological Conservation 120: 383-390.

Nur, N., S.L Jones, and G.R. Geupel. 1999. A statistical guide to data analysis of avian monitoring programs. Biol. Tech. Pub. R6001-1999. U.S. Department of the Interior, Fish and Wildlife Service, Washington, D.C.

Ouin, A., S. Aviron, J. Dover, and F. Burel. 2004. Complementation/supplementation of resources for butterflies in agricultural landscapes. Agriculture, Ecosystems and Environment 103: 473-479.

Pagiola, S., P. Agostini, J. Gobbi, C. de Haan, M. Ibrahim, E. Murgueitio, E. Ramírez, M. Rosales, and J.P. Ruíz. 2005. Paying for biodiversity conservation services: experience in Colombia, Costa Rica, and Nicaragua. Mountain Research and Development 25: 206-211.

Perfecto, I., A. Mas, T. Dietsch, and J. Vandermeer. 2003. Conservation of biodiversity in coffee agroecosystems: a tri-taxa comparison in southern Mexico. Biodiversity and Conservation 12: 1239-1252.

Preston, F.W. 1948. The commonness, and rarity, of species. Ecology 29: 254-283.

Ralph, C.J., S. Droege, and J.R. Sauer. 1995. Managing and monitoring birds using point counts: standards and applications. Pages 161-169 in C.J. Ralph, J.R. Sauer, and S. Droege, editors. Monitoring bird populations by point counts. Gen. Tech. Rep. PSW-GTR-149. USDA Forest Service, Pacific Southwest Research Station, Albany, California.

Stiles, F.G. 1985. Conservation of forest birds in Costa Rica: problems and perspectives. Pages 141-168 in A.W. Diamond and T.E. Lovejoy, editors. Conservation of tropical forest birds. International Council for Bird Preservation, Cambridge, UK.

Stotz, D.F., J.W. Fitzpatrick, T.A. Parker, and D.K. Moskovits. 1996. Neotropical birds: ecology and conservation. University of Chicago Press, Chicago.

Temple, S.A., and J.A. Wiens. 1989. Bird population and environmental changes: can birds be bio-indicators? American Birds 43: 260-270.

Tilman, D., K.G. Cassman, P.A. Matson, R. Naylor, and S. Polasky. 2002. Agricultural sustainability and intensive production practices. Nature 418: 671-677.

Tscharntke T., A.M. Klein, A. Kruess, I. Steffan-Dewenter, and C. Thies. 2005. Landscape perspectives on agricultural intensification and biodiversity-ecosystem service management. Ecology Letters 8: 857-874.

Vandermeer, J., and I. Perfecto. 2007. The agricultural matrix and a future paradigm for conservation. Conservation Biology 21: 274-277.

CHAPTER 5

CONCLUSION

5.1 CONSERVATION IN PASTURE-DOMINATED LANDSCAPES

In the Introduction, I argued that multifunctional landscape management is not a luxury or a dalliance of researchers and innovators; it is a plain necessity if humanity is to meet its goals for poverty reduction, food production, and ecosystem conservation. Yet this premise begs the question of how, exactly, landscapes are to be managed for multiple outcomes. Concepts such as ecoagriculture are often associated with models of wildlife-friendly agriculture in which farm plots, agroforests, and natural vegetation are interspersed in fine-grained mosaic managed by small farmers and rural communities. However, ecoagriculture can also include intensive farming, precisely managed for high yield and low environmental impact, and integrated with protected areas at larger spatial scales. The choice between these paradigms should be negotiated region by region, informed by science and economics, and not dictated by ideology.

To this end, my research has sought to characterize the potential for meeting multiple goals in pasture-dominated regions of northern Latin America by evaluating the conservation values of landscapes that already produce food and support rural communities. I conducted this research in four landscapes: Copán, Honduras; Matiguás, Nicaragua; Esparza, Costa Rica; and Quindío, Colombia, Overall, the four landscapes contained a high diversity of birds and butterflies (404 and 296 species, respectively). Across all the landscapes, bird assemblage characteristics, and, to a lesser extent, butterfly assemblage characteristics, were significantly related to land use. The consistency of these relationships—even in landscapes with disparate biophysical conditions and contexts—supports the application of land use classifications as a reasonably accurate proxy for conservation outcomes in pasture-dominated landscapes. Contrary to my hypothesis, continuous habitat descriptors were not better predictors than categorical land use descriptors for bird and butterfly assemblage characteristics in Copán. However, additional research is needed to evaluate whether the same can be said for the other three landscapes.

Results from this study offer a prime example of a point made previously in the literature but commonly disregarded by subsequent researchers: one's choice of biodiversity metrics strongly influences the conclusions drawn about the conservation value of human-dominated environments (Duelli and Obrist 2003). The study landscapes, for instance, contain a high total diversity of bird and butterfly species, a moderate presence of forest-dependent species and species of intermediate conservation concern, and an extremely low presence of globally threatened or endangered species. The 'conservation value' of these landscapes, then, depends a great deal on which conservation value one has in mind.

Nearly a decade ago, Daily et al. (2001) drew this inference from their study of bird assemblages in an agricultural mosaic in southern Costa Rica: "…countryside habitats may buy time for the conservation of some species; at best, they may even sustain a moderate fraction of the native biota." This general finding has been corroborated by numerous subsequent studies in agricultural mosaics of Central America, including those presented in Chapters 3 and 4. Nevertheless, in recent years an influential ecological narrative emerging from this body of research has been one advocating the benefits and potential of moderate-intensity agricultural systems for conserving native biodiversity and maintaining functional landscape- and regional-scale habitat corridors (e.g., Vandermeer et al. 2008). Certain types of agroforestry systems, such as rustic coffee and cacao, may indeed provide many such benefits (e.g., Moguel and Toledo 1999; Schroth and Harvey 2007). Yet, if Neotropical agricultural mosaics such as the ones included in this study—with diverse forms of tree cover and approximately 15-40% total forest cover—have only modest value for threatened species and global biodiversity conservation overall, then perhaps expectations of the degree to which such landscapes will support this goal are overblown.

Instead, it may be more realistic to focus on the conservation benefits that such landscapes can provide by keeping common species common, by sustaining a moderate abundance of species of intermediate conservation concern, and by fostering species that contribute to ecosystem services and rural livelihoods (Gordon et al. 2003). The first two benefits can be measured and advanced by using tools, such as the BCV metric, that help evaluate the conservation value of species assemblages in an objective and quantitative manner. The conservation initiative Partners in Flight has used a similar approach in North America to improve understandings of bird conservation trends and to target conservation efforts accordingly (Carter et al. 2000). The ecosystem services of wild and managed biodiversity in agricultural mosaics merit much greater study. While some research has been done on taxa such as trees (e.g., Barrance et al. 2003; Restrepo-Sáenz et al. 2004), birds (e.g., Perfecto et al. 2004), and pollinating bees (e.g., Ricketts et al. 2008), overall the ecosystem services and dis-services provided by particular species in agrarian landscapes are little known.

A final conclusion to be drawn from the collective body of research on conservation in Neotropical agricultural mosaics is that the need for additional protected areas is not obsolete. There is ample evidence that moderate-intensity farming systems can provide some conservation benefits, particularly for species of low and intermediate conservation concern. Yet the wildlife-friendliness of many such systems—limited to begin with—is under constant threat from myriad pressures ranging from agricultural intensification to rural-urban migration to climate change. While not without significant challenges of their own, protected areas can provide more stable bastions of conservation, especially for the most vulnerable and threatened species. Thus, protected area strategies must not be neglected, even as conservationists rightly turn a greater share of their attention toward conservation outside protected areas.

5.2 ECOAGRICULTURE AND THE MEANINGS OF 'CONSERVATION'

The preceding conclusions point to the importance of precise and nuanced understandings of the concept of 'conservation value' when evaluating or managing for conservation outcomes in agricultural landscapes. The global conservation community tends to bring one understanding, but it is not always the most important one. Local communities, farmers, government agencies, and NGOs bring other perspectives that are also relevant and that may coalesce around common ground, from which collaborative conservation efforts may be built.

As noted above, much more research is needed to understand the implications of wildlife conservation for agricultural productivity and rural livelihoods. Yet, despite the hopes of many conservationists, there is no assurance that these links will prove to be positive, strong, or consistent. As has been observed in numerous prior projects and studies, there are sometimes synergies between biodiversity conservation and human development goals, but there are also frequently tradeoffs (e.g., Palm et al. 2005).

However, if one takes a broader view of conservation to include the core components of ecosystems such as soils, water, nutrients, primary productivity, and functional diversity of species, then conservation can be much more credibly viewed as a consistent underpinning of rural development and the Millennium Development Goals, as discussed in Chapter 2. Structured multi-stakeholder processes, such as those proposed in the Landscape Measures approach, can help ensure that relevant perspectives on conservation, agriculture, and livelihoods are elicited and discussed in decision-making forums. Doing so is an important step toward adopting management systems that increase synergies and reduce tradeoffs among multiple landscape objectives.

5.3 EPILOGUE

When cities get crowded, people build up. They stack their dwellings and workplaces over stores and subways and sewers. They meld public and private spaces, dissolving the boundaries between commerce, culture, art, and recreation. The result, in the world's great cities, is an intricate use of space: charmingly clever and endlessly evolving, a savory stew formed of laws and customs, innovation and necessity, civic-mindedness and selfishness.

When Earth gets crowded, we also must stack. There is no other way. Human-dominated rural landscapes must provide food, fiber, energy, water, wildlife habitat, carbon storage, beauty, and a viable way of life for their inhabitants. The spatial solutions for doing so are many. As in our cities, we will blunder toward them. As in our cities, the best solutions will be born from science and logic, but also from policies and systems that harness innovation, profit motive, and local custom, and that build capacity to work together. Learning how to stack: this is a great challenge of our era.

LITERATURE CITED

Barrance, A.J., L. Flores, E. Padilla, J.E. Gordon, and K. Schreckenberg. 2003. Trees and farming in the dry zone of southern Honduras I: *campesino* tree husbandry practices. Agroforestry Systems 59: 97-106.

Carter, M.F., W.C. Hunter, D.N. Pashley, and K.V. Rosenberg. 2000. Setting conservation priorities for landbirds in the United States: the Partners in Flight approach. The Auk 117: 541-548.

Daily, G.C., P.R. Ehrlich, and G.A. Sanchez-Azofeifa. 2001. Countryside biogeography: use of human-dominated habitats by the avifauna of southern Costa Rica. Ecological Applications 11: 1-13.

Duelli, P., and M.K. Obrist. 2003. Biodiversity indicators: the choice of values and measures. Agriculture, Ecosystems, and Environment 98: 87-98.

Gordon, J.E., W.D. Hawthorne, G. Sandoval, and A.J. Barrance. 2003. Trees and farming in the dry zone of southern Honduras II: the potential for tree diversity conservation. Agroforestry Systems 59: 107-117.

Moguel, P., and V.M. Toledo. 1999. Biodiversity conservation in traditional coffee systems of Mexico. Conservation Biology 13: 11-21.

Palm, C.A., S.A. Vosti, P.A. Sanchez, and P.J. Ericksen, editors. 2005. Slash-and-burn agriculture. Columbia University Press, New York.

Perfecto, I., J.H. Vandermeer, G.L. Bautista, G.I. Nuñez, R. Greenberg, P. Bichier, and S. Langridge. 2004. Greater predation in shaded coffee farms: the role of resident Neotropical birds. Ecology 85: 2677-2681.

Restrepo-Sáenz, C., M. Ibrahim, C. Harvey, J.M. Harmand, and J. Morales. 2004. Relaciones entre la cobertura arbórea en potreros y la producción bovina en fincas ganaderas en el trópico seco en Cañas, Costa Rica. Agroforestría en las Américas 41-42: 29-36.

Ricketts, T.H., J. Regetz, I. Steffan-Dewenter, S.A. Cunningham, C. Kremen, A. Bogdanski, B. Gemmill-Herren, S.S. Greenleaf, A.M. Klein, M.M. Mayfield, L.A. Morandin, A. Ochieng, and B.F. Viana. 2008. Landscape effects on crop pollination services: are there general patterns? Ecology Letters 11: 499-515.

Schroth, G., and C.A. Harvey. 2007. Biodiversity conservation in cocoa production landscapes: an overview. Biodiversity Conservation 16: 2237-2244.

Vandermeer, J.H., I. Perfecto, S. Philpott, and M.J. Chappell. 2008. Reenfocando la conservación en el paisaje: La importancia de la matriz. Pages 75-104 in C.A. Harvey and J.C. Sáenz, editors. Evaluación y Conservación de la Biodiversidad en Paisajes Fragmentados de Mesoamérica. Instituto Nacional de Biodiversidad, Heredia, Costa Rica.

APPENDIX A

METHOD FOR QUANTIFYING LANDSCAPE COMPOSITION AND STRUCTURE FROM MULTI-SPECTRAL REMOTE SENSING IMAGERY[4]

The method for quantifying landscape characteristics from multi-spectral ASTER imagery consisted of two parts: 1) acquiring and processing the raw satellite imagery into raster-based vegetation metrics, and 2) summarizing these metrics for the land areas surrounding each sample point. These parts are described in sequence below.

ACQUISITION AND PROCESSING OF ASTER SATELLITE IMAGERY

ASTER image processing included eleven steps, divided into the following three phases (Figure A-1):

Steps 1-7: Pre-process ASTER imagery to correct radiometric and geometric errors and to convert raw digital numbers to radiance and reflectance values.

Steps 8-9: Use the reflectance data to calculate vegetation metrics.

Steps 10-11: Create landscape-wide composites of the vegetation metrics that maximize data availability across each landscape.

Step 1: Select and purchase ASTER images

I obtained the ASTER DMO 14 product, which provides orthorectified ASTER scenes as well as a 30 m resolution digital elevation model (Abrams et al. 1999). The DMO

[4] I conducted the analyses described in this chapter in conjunction with my research assistant Andre Sanfiorenzo. Notwithstanding any use of singular pronouns in this chapter, the work presented here represents a joint effort of Andre and me.

14 product provides all 14 ASTER bands, including three visible and near-infrared (VNIR) bands, six short-wave infrared (SWIR) bands, and five thermal infrared (TIR) bands. This product is available free of charge to certain academic users, and at a modest fee to other users, from the U.S. Geological Survey's Earth Resources Observation and Science Center in Sioux Falls, South Dakota.

In cases where there was no single scene that provided substantially cloud-free coverage for a given landscape, I obtained two or more scenes that, together, provided the maximum possible cloud-free coverage for the date range of interest.

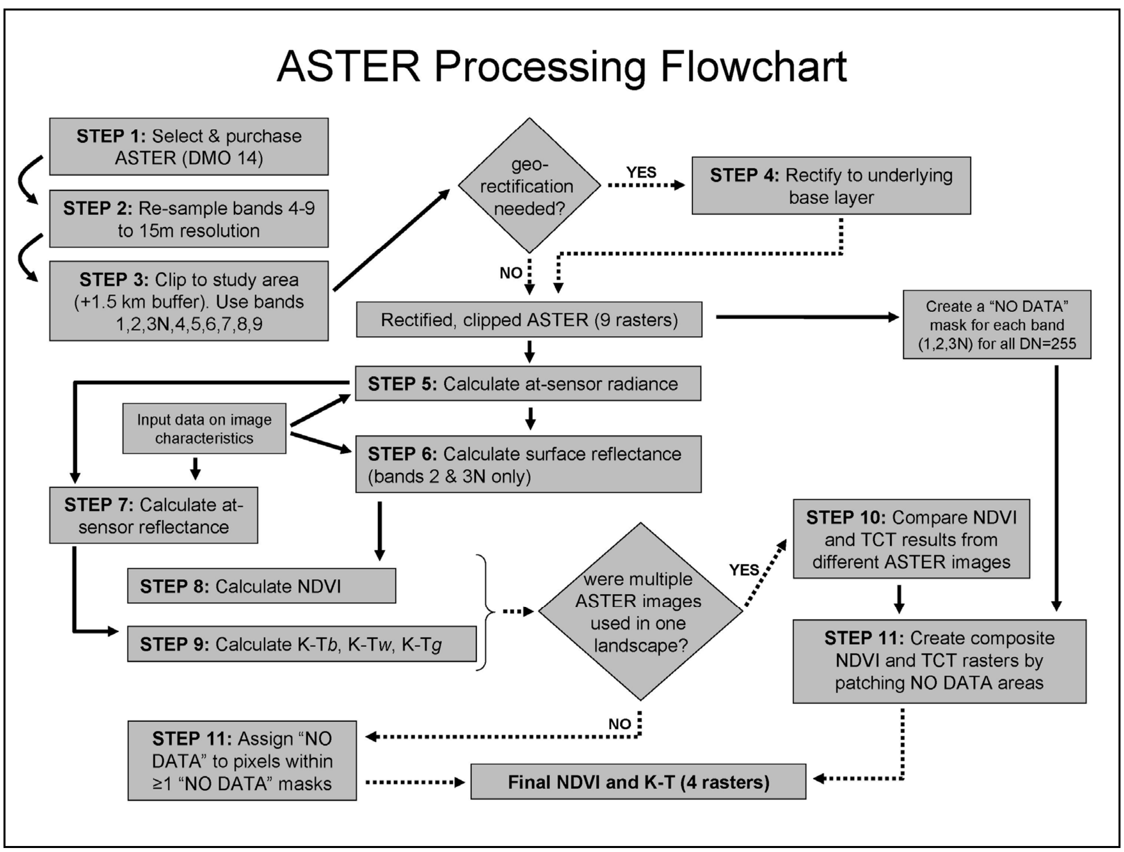


Figure A-1. Flowchart for processing ASTER imagery to generate vegetation metrics. See the text for additional description of each step.

Step 2: Re-sample bands 4-9 to 15 m resolution

The ASTER bands of interest for this project are provided at different resolutions. The VNIR bands 1, 2, and 3N are delivered at 15 m resolution. The SWIR bands 4, 5, 6, 7, 8, and 9 are delivered at 30 m resolution. To attain a consistent resolution for use in subsequent processing, I used ArcGIS 9.2 to resample the SWIR bands to 15 m resolution based on a nearest neighbor algorithm.

Step 3: Clip ASTER images to the study area

To reduce the ASTER file size and processing time, I clipped all the input rasters to the size and shape of the study area, plus a 1.5 km buffer, in each landscape. This was done only for the three VNIR bands and six SWIR bands; the TIR bands were not used in this study.

Step 4: Geo-rectify ASTER image to underlying base layer (if necessary)

Although the ASTER product that I ordered is intended to provide orthorectified imagery, in some parts of some of the landscapes, I found significant discrepancies (several hundred meters) between the ASTER imagery and the underlying GIS base layers, which were based on high-resolution Quickbird or IKONOS imagery. These errors were not due to different projections or coordinate systems. In the instances where such errors were present, I was able to obtain additional ASTER scenes that did not contain the same discrepancies. However, if this had not been possible, it would have been necessary to use control points to geo-rectify the ASTER scenes to the underlying base layer. The intermediate product from Step 4 was geo-rectified, clipped rasters for each of the nine VNIR and SWIR ASTER bands.

Step 5: Calculate at-sensor radiance

Steps 5-7 consisted of image correction to convert the raw ‘digital number’ values recorded by the ASTER sensor into calibrated data that better represent surface features within the study landscapes. These corrections allowed for the use of multiple ASTER images without introducing confounding variables associated with differing properties from the multiple images (e.g., different dates, times of day, and sensor settings).

I used a standard method to convert the ASTER raw digital number to at-sensor radiance (Smith 2007; Yuksel 2008). This calibration corrects for the sensor gain and offset. First, I identified any pixels with a value of 255, which are saturated pixels that exceeded the sensor’s detection limit. Because they are saturated, these pixels do not provide useful data for subsequent analysis. I also identified all cloud-covered areas, which were generally the same as the areas with saturated pixels. Then, for each band, a NO DATA mask was created of all areas with saturated pixels or cloud cover.

Next, for each ASTER band, I created a new raster file of at-sensor radiance by applying the following equation:

$$L_{sat} = (DN-1) * UCC$$

Where:

$\mathbf{L_{sat}}$ = at-sensor spectral radiance.

DN = digital number (the pixel values in the original ASTER files).

UCC = unit conversion coefficient. This is different for each ASTER band, and also depends on the gain setting that was used to acquire the image. To determine which gain setting was used, I consulted the ASTER metadata files (these files accompany the rasters and are denoted by the *.met suffix). Within this file, there is a section called "ASTERGAINS" that lists the gain setting that was used in acquiring each band: NOR = normal; HGH = high; LOW = low gain 1; LO2 = low gain 2. Based on these gain settings, I selected the appropriate UCC for each band from published values, which are reproduced in Table A-1.

Table A-1. ASTER unit conversion coefficients. Source: Abrams et al. 1999.

	Coefficient ((W/m^2*sr*um)/DN)			
Band	**High Gain**	**Normal**	**Low Gain 1**	**Low Gain 2**
1	0.676	1.688	2.25	
2	0.708	1.415	1.89	
3N	0.423	0.862	1.15	
4	0.1087	0.2174	0.2900	0.2900
5	0.0348	0.0696	0.0925	0.4090
6	0.0313	0.0625	0.0830	0.3900
7	0.0299	0.0597	0.0795	0.3320
8	0.0209	0.0417	0.0556	0.2450
9	0.0159	0.0318	0.0424	0.2650

The at-sensor radiance raster used a floating point format to accommodate decimals. The intermediate product from Step 5 was at-sensor radiance rasters for each of the nine VNIR and SWIR bands.

Step 6: Calculate surface reflectance

I calculated surface reflectance to use as the input for the Normalized Difference Vegetation Index (NDVI). Surface reflectance corrects for two sets of factors: 1) variations in solar illumination influenced by properties such as the solar elevation angle and earth-sun distance, and 2) the influence of atmospheric haze and aerosols on the signal detected by the sensor. By correcting for these factors, surface reflectance should characterize the land features themselves. Since surface reflectance was used as an input only for NDVI, I calculated it only for the two ASTER bands that are used to generate NDVI: bands 2 and 3N (red and near infrared, respectively).

To calculate surface reflectance, I applied the following equation from the method of Warner (2008), which is based on earlier methods published by Chavez (1996) and Lu et al. (2002):

$$\rho = (\pi * (L_{sat} - L_{haze}) * d^2) / (Esun_\lambda * (\cos(\theta_s))^2)$$

Where:

ρ = surface reflectance.

L_{sat} = at-sensor radiance (the output raster from Step 5).

d = earth-sun distance, calculated using this equation:

$$d = (1 - 0.01672 * \cos(\mathrm{RADIANS}(0.9856 * (\text{Julian Day} - 4))))$$

$Esun_\lambda$ = a constant that is different for each ASTER band (Table A-2).

θ_s (solar zenith angle) = 90 - Solar Elevation Angle. The Solar Elevation Angle is provided in the “Solar_Elevation_Angle” section of the ASTER metadata file. In the metadata file, this angle is given in degrees. For software programs that calculate cosine based on radians, it is first necessary to convert degrees to radians. Note that in the equation, the denominator contains the term $(\cos(\theta_s))^2$. Some versions of this equation use only the term $\cos(\theta_s)$, but here the second $\cos(\theta_s)$ is used to approximate tau, the atmospheric transmittance. This method is appropriate for humid climates, such as the tropical landscapes in this study (USU 2008; Warner 2008).

Table A-2. $Esun_{\lambda}$ values for each ASTER band. Source: Smith 2007.

ASTER Band	$Esun_{\lambda}$
1	1845.99
2	1555.74
3N	1119.47
4	231.25
5	79.81
6	74.99
7	68.66
8	59.74
9	56.92

$\mathbf{L_{haze}}$ = estimate of upwelling scattered path radiance due to atmospheric haze and aerosols. The subtraction of L_{haze} from L_{sat} is a 'dark object subtraction' approach to determine the portion of the at-sensor radiance that is attributable to ground properties, while subtracting out the portion that is attributable to atmospheric effects. This method has been found to be reasonably accurate, and is the most feasible approach to atmospheric correction when actual atmospheric data from the moment of image acquisition are not available (Chavez 1996; Lu et al. 2002). To determine L_{haze} for each ASTER band, I viewed the histogram of data values for the at-sensor radiance raster and manually selected the value at the toe of the histogram (the point where the histogram began to register a significant number of pixels). This value is generally around the 0.05th to 0.1th percentile of all pixel values, as illustrated in Figure A-2. The lowest value on the histogram generally should not be selected because it may represent outlying noise not representative of a typical dark object on the landscape.

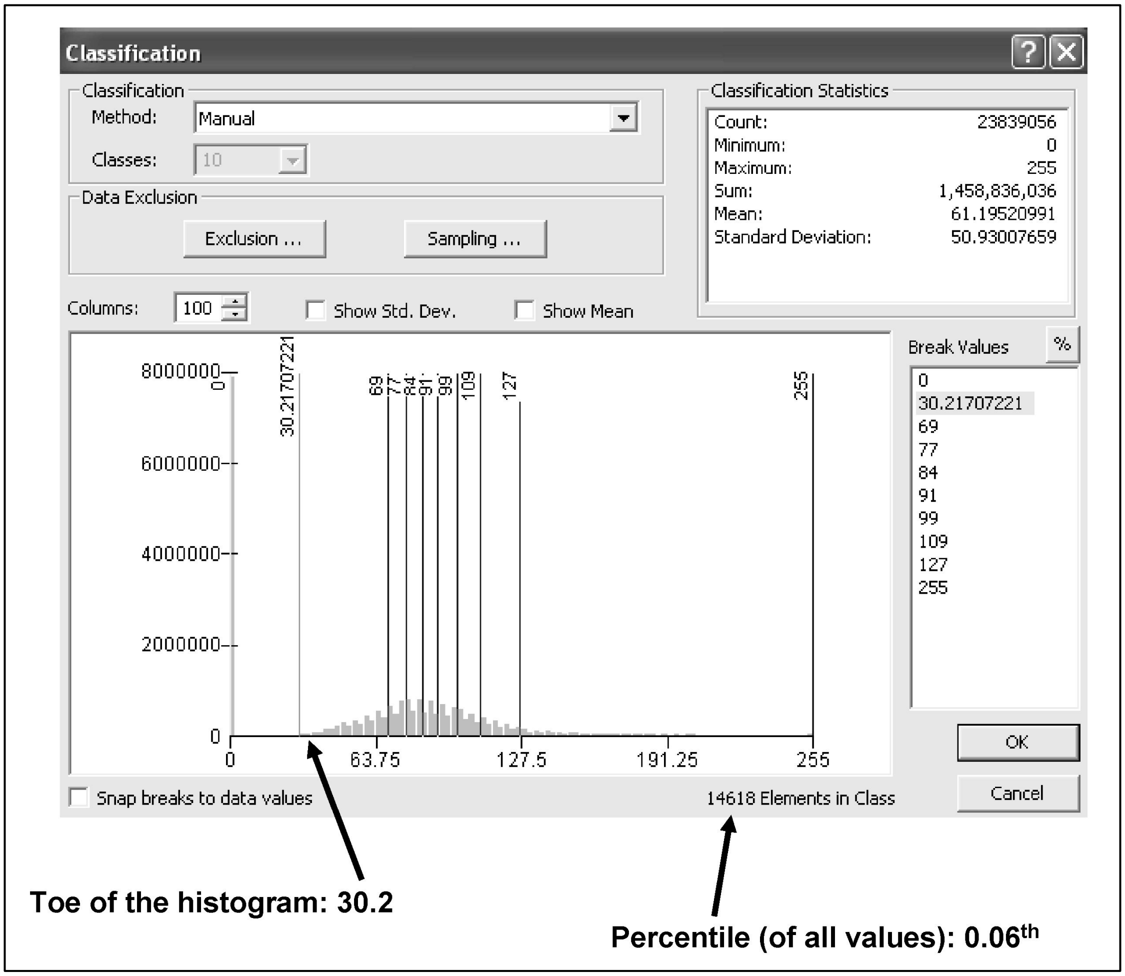


Figure A-2. Screenshot from ArcGIS 9.2 indicating the process for determining the L_{haze} value based on the histogram of data values for the at-sensor radiance raster.

I verified that these dark object pixels indeed represented suitable dark objects by using the software program ENVI to display the pixels that had been selected as dark objects. These pixels were typically shadow areas for band 2 (red) and water for band 3N (near infrared).

The raster of surface reflectance consisted of floating point values. Those few values that were calculated to be <0 were re-assigned to 0. The intermediate product from Step 6 was surface reflectance rasters for ASTER bands 2 and 3N.

Step 7: Calculate at-sensor reflectance

Although surface reflectance provides the best available representation of surface features, it is inappropriate for calculating the second set of vegetation indices, the Kauth-Thomas transformation (K-T). The K-T (which is a generalized version of the more specific "tasseled-cap transformation," a term that is used more commonly but often inaccurately in the literature) is a method for reducing the dimensionality of data from multispectral sensors into a few components that correspond to surface properties of the landscape (Kauth and Thomas 1976; Crist 1985; Crist and Kauth 1986). This transformation can be accomplished by means of a principal component algorithm (e.g., Yarbrough 2006) or a Gram-Schmidt transformation (e.g., Yajuan and Danfeng 2005). Both approaches typically resolve multispectral data into three components—brightness, greenness, and wetness—which have been used as proxies for various vegetation characteristics as well as biodiversity.

Brightness, greenness, and wetness are simply linear combinations of the input bands (in this case, ASTER bands 1 through 9) multiplied by their respective coefficients. Once the K-T coefficients have been derived for a particular sensor, they can be applied in any context. However, this holds true only for K-T coefficients based on at-sensor radiance or at-sensor reflectance. Since surface reflectance depends on the specific atmospheric conditions of each ASTER scene, K-T coefficients based on surface reflectance would need to be derived anew for each scene. The at-sensor reflectance, however, corrects for sensor and planetary effects but not atmospheric effects. Therefore, ASTER K-T coefficients derived with at-sensor reflectance data in one context should apply to such data in any other context. Compared to at-sensor radiance, the at-sensor reflectance data have the benefit of correcting for planetary variables such as the solar elevation angle and earth-sun distance. Trials of different approaches to the K-T transformation confirmed that the at-sensor reflectance algorithm did indeed provide superior results to the at-sensor radiance or surface reflectance approaches. Accordingly, I calculated at-sensor reflectance to use as the input for generating the K-T brightness, greenness, and wetness components.

The equation for at-sensor reflectance is similar to that for surface reflectance, except that it does not include the dark object subtraction (L_{haze}) or the provision for outgoing path radiance (the second $\cos(\theta_s)$) (Smith 2007):

$$R_{TOA} = (\pi * L_{sat} * d^2) / (ESUN_\lambda * \cos(\theta_s))$$

Where:

R_{TOA} = at-sensor reflectance (or top-of-atmosphere reflectance).

L_{sat} = at-sensor radiance (the output raster from Step 5).

d = earth-sun distance, calculated using this equation:

$$d = (1 - 0.01672 * \cos(\mathrm{RADIANS}(0.9856 * (\mathrm{Julian\ Day} - 4))))$$

$Esun_\lambda$ = a constant that is different for each ASTER band (Table A-2).

θ_s (solar zenith angle) = 90 - Solar Elevation Angle. The Solar Elevation Angle is provided in the “Solar_Elevation_Angle” section of the ASTER metadata file.

The rasters of at-sensor reflectance consisted of floating point values. The intermediate product for Step 7 was at-sensor reflectance rasters for each of the nine VNIR and SWIR bands.

Step 8: Calculate NDVI

NDVI is an indicator of the density and photosynthetic activity of living vegetation that has been widely used in ecological studies. It is a function of the relative reflectance in the red and near infrared bands, and is calculated according to the following formula:

$$NDVI = (NIR - RED) / (NIR + RED)$$

Where:

NIR = reflectance in the near infrared band (ASTER band 3N).

RED = reflectance in the red band (ASTER band 2).

I used the raster calculator in ArcGIS 9.2 to calculate NDVI based on the surface reflectance rasters generated in Step 6. The intermediate product for Step 8 was one or more NDVI rasters for each landscape (one raster for each ASTER scene used).

Step 9: Calculate Kauth-Thomas transformation (K-T) components

As discussed above, I used at-sensor reflectance as the input data source to implement the Kauth-Thomas transformation based on previously derived coefficients (Table A-3). Although coefficients are available for nine transformation components (which would transform the original ASTER nine-dimensional space into a new nine-dimensional space), the first three K-T components typically explain 97-99% of the variability in the data (Yarbrough 2006). In addition, these first three K-T components (brightness, greenness, and wetness) are related to biophysical properties of the land and therefore have clear interpretive value. This is not the case with components 4 through 9, so I did not calculate these components.

Table A-3. Kauth-Thomas transformation coefficients. Source: Yarbrough et al. 2005.

K-T component	Band 1	Band 2	Band 3N	Band 4	Band 5	Band 6	Band 7	Band 8	Band 9
Brightness	-0.274	0.676	0.303	-0.256	-0.020	0.415	-0.255	0.073	-0.262
Greenness	-0.006	-0.648	0.564	0.061	-0.055	0.394	-0.193	0.021	-0.249
Wetness	0.166	-0.087	-0.703	0.187	0.040	0.500	-0.287	0.030	-0.318

I used the raster calculator in ArcGIS 9.2 to calculate each K-T component as a linear combination of the nine input bands multiplied by their respective coefficients. For example, the brightness component is calculated as:

> Brightness = (-0.274 * Band 1) + (0.676 * Band 2) + (0.303 * Band 3N) + (-0.256 * Band 4) + (-0.02 * Band 5) + (0.415 * Band 6) + (-0.255 * Band 7) + (0.073 * Band 8) + (-0.262 * Band 9)

The intermediate product for Step 9 was one or more rasters for brightness, greenness, and wetness for each landscape (one K-T brightness raster, one K-T greenness raster, and one K-T wetness raster for each ASTER scene used).

Step 10: Compare NDVI and K-T results from different ASTER images

In landscapes where it was necessary to use multiple ASTER scenes to provide maximum cloud-free coverage of the landscape, I identified areas of overlap between two or more scenes and used these areas to compare the NDVI and K-T results from the different scenes. The purpose of these comparisons was to evaluate the degree to which the results depended on the specific ASTER scenes used in the analysis. Two types of comparisons were conducted:

1) Approximately 100 points were randomly selected within the overlap area. The following pixel values were then recorded for each point: Band 2 surface reflectance, Band 3N surface reflectance, Band 2 at-sensor reflectance, Band 3N at-sensor reflectance, NDVI, K-T brightness, K-T greenness, and K-T wetness. I created scatterplots for each data type to evaluate the correlation among values from the different scenes.
2) For these same data types, I created tables indicating the percentage of pixel values falling within each of 20 equally sized data ranges across the full range of pixel values. I then evaluated the similarity in the data distribution among the different scenes.

Comparisons indicated that the surface reflectance, at-sensor reflectance, NDVI, and K-T values did not depend on which ASTER scene was used.

Step 11: Create final composite NDVI and K-T rasters for each landscape

The final step was to create a composite of data from multiple ASTER scenes and to apply the NO DATA masks generated in Step 5 to create final rasters with the maximum available data coverage.

In landscapes where I used only a single ASTER scene, I overlaid the NO DATA masks on the NDVI and K-T rasters created in Steps 8 and 9, respectively. Pixels that were within a NO DATA mask for one or more input bands were re-assigned to NO DATA.

In landscapes where I used multiple ASTER scenes, I created mosaics of data from adjacent scenes to cover the entire landscape. When possible, I also patched NO DATA areas from each scene (for example, cloud-covered areas) with available data from other scenes. The resulting rasters provided maximum coverage for each landscape. The final product from the eleven-step process was one NDVI raster, one brightness raster, one greenness raster, and one wetness raster for each landscape.

DATA ANALYSIS AND SUMMARIZATION

Next I analyzed the NDVI and K-T rasters for the areas of interest in each landscape. To do so, I first defined the landscape context areas around each sample point by using the buffer tool in ArcGIS 9.2 to create concentric circular buffer polygons corresponding to each of the following radii: 100, 200, 400, 800, 1500, and 3000 m. For each of these context zones, I then used the zonal statistics tool in ArcGIS 9.2 to calculate the mean and standard deviation for NDVI, brightness, greenness, and wetness. I also calculated means and standard deviations for each landscape study area in its entirety. These data were transferred to Microsoft Excel spreadsheets for further analysis.

LITERATURE CITED

Abrams, M., S. Hook, and B. Ramachandran. 1999. ASTER user handbook, version 2. Jet Propulsion Laboratory, Pasadena, California, and EROS Data Center, Sioux Falls, South Dakota.

Chavez, P. 1996. Image-based atmospheric corrections - revisited and improved. Photogrammetric Engineering & Remote Sensing 62: 1025-1036.

Crist, E.P. 1985. A TM tasseled cap equivalent transformation for reflectance factor data. Remote Sensing of Environment 17: 301-306.

Crist, E.P., and R.J. Kauth. 1986. The tasseled cap demystified. Photogrammetric Engineering & Remote Sensing 52: 81-86.

Kauth, R.J., and G.S. Thomas. 1976. Tasselled cap: a graphic description of the spectral-temporal development of agricultural crops as seen by Landsat. Pages 41-51 in Symposium on Machine Processing of Remotely Sensed Data. National Telecommunications Conference Record, West Lafayette, Indiana.

Lu, D., P. Mausel, E. Brondizio, and E. Moran. 2002. Assessment of atmospheric correction methods for Landsat TM data applicable to Amazon basin LBA research. International Journal of Remote Sensing 23: 2651-2671.

Smith, A.M.S. 2007. How to convert ASTER radiance values to reflectance: an online guide. College of Natural Resources, University Idaho. Online: www.cnrhome.uidaho.edu/default.aspx?pid=85984.

[USU] Utah State University Remote Sensing/GIS Laboratory. 2008. Image standardization: at-sensor reflectance and COST correction. USU, Logan, Utah. Online: ftp.nr.usu.edu/imagestd.

Warner, T. 2008. Conversion of Landsat DN to reflectance using the CosT approach. University of West Virginia. Unpublished class notes.

Yajuan, W., and S. Danfeng. 2005. The ASTER tasseled cap interactive transformation using Gramm-Schmidt method. In L. Zhang, J. Zhang, and M. Liao, editors. MIPPR 2005: SAR and Multispectral Image Processing. Proceedings of SPIE 6043: 60430R-1. doi: 10.1117/12.654861.

Yarbrough, L.D., G. Easson, and J.S. Kuszmaul. 2005. Using at-sensor radiance and reflectance tasseled cap transforms applied to change detection for the ASTER sensor. IEEE Third International Workshop on the Analysis of Multi-temporal Remote Sensing Images, May 16-18 2005, Biloxi, Mississippi.

Yarbrough, L.D. 2006. The legacy of the tasseled cap transform: a development of a more robust Kauth-Thomas transform derivation. PhD dissertation. Department of Engineering Science, University of Mississippi, University, Mississippi.

Yuksel, A., A.E. Akay, and R. Gundogan. 2008. Using ASTER imagery in land use/cover classification of eastern Mediterranean landscapes according to CORINE land cover project. Sensors 8: 1237-1251.

APPENDIX B

METHOD FOR QUANTIFYING LANDSCAPE COMPOSITION AND STRUCTURE FROM HIGH-RESOLUTION SATELLITE IMAGERY[5]

This Appendix describes methods for calculating landscape composition and structure measures from high-resolution Quickbird and IKONOS satellite imagery.

ACQUISITION OF SATELLITE IMAGERY

This study uses high-resolution IKONOS and Quickbird satellite imagery (pixel size of 1 m and 0.61 m, respectively) to derive measures of landscape composition and structure. This imagery is sufficiently detailed to discern individual trees, live fences, streams, and other fine-scale landscape features. I obtained imagery from four sources:

1) Quickbird imagery purchased by the Centro Agronómico Tropical de Investigación y Enseñanza (CATIE) for the Matiguás, Esparza, and Quindío landscapes as part of the GEF-Silvopastoral project (Pagiola et al. 2005). These images were acquired in 2003.
2) Supplemental Quickbird imagery that I downloaded from Google Earth for the Matiguás, Esparza, and Quindío landscapes in 2008 (of imagery acquired in 2003). This imagery provided coverage for portions of these landscapes that were within the zone of analysis but beyond the coverage area originally purchased by CATIE. Because of the limitations of Google Earth, these free

[5] I conducted the analyses described in this chapter in conjunction with my research assistant Andre Sanfiorenzo. Notwithstanding any use of singular pronouns in this chapter, the work presented here represents a joint effort of Andre and me.

images needed to be downloaded in 1000 x 1000 pixel scenes, then mosaicked and geo-rectified in ArcGIS 9.2 to align with the existing Quickbird imagery.

3) Supplemental Quickbird imagery that I purchased for the Esparza and Quindío landscapes in 2008 (of imagery acquired between 2003 and 2005), for areas that were not available from CATIE or from Google Earth. This imagery provided supplemental coverage for portions of the landscape that were within the zone of analysis, including coverage for some areas that were cloud-covered in the original imagery. I geo-rectified these new image scenes to align with the existing Quickbird imagery.
4) IKONOS imagery purchased by CATIE for the Copán landscape as part of the World Bank-Bank Netherlands Partnership Program. This imagery was acquired in February 2007.

All imagery was imported into ArcGIS 9.2 and layered to maximize the cloud-free coverage area.

LAND USE/LAND COVER CLASSIFICATION

Within each landscape, the study area was defined as the union of circles of 3000 m radius surrounding each sample point. All landscape analysis was restricted to these context zones. Similar to the ASTER analysis, I calculated each landscape metric for six concentric context zones around each sample point, corresponding to radii of 100, 200, 400, 800, 1500, and 3000 m. These context zones were defined by using the buffer tool in ArcGIS 9.2.

In each landscape, I used heads-up digitizing combined with field verification of selected areas to classify land use and land cover into 18 discrete categories (Table B-1). The minimum mapping unit was 0.25 ha.

Table B-1. Land use/land cover categories. The Biodiversity Score refers to proxy values assigned to each land use based on its estimated value for supporting native biodiversity, on a scale from 0 to 1. The scores shown here follow or extrapolate from those of Murgueitio et al. (2004), which were established for the GEF-Silvopastoral project to allocate ecosystem services payments to farmers based on land use changes undertaken to help conserve biodiversity on their farms.

Land use code	**Land use/ land cover**	**Biodiversity Score**	**Description**
1	Annual crops	0.0	Annual crops
2	Unshaded perennial crops	0.1	Includes sun coffee, banana, and fodder banks
3	Pasture without trees	0.0	Includes degraded, natural, and improved pastures with less than 5% tree canopy cover
4	Pasture with low tree density	0.3	Pasture with 5-15% tree canopy cover
5	Pasture with high tree density	0.5	Pasture with 15-30% tree canopy cover
6	Live fence	0.6	Continuous rows of trees separating pastures or farms
7	Plantation	0.4	Plantations of timber trees, including bamboo; also includes monoculture fruit plantations
8	Diversified perennial crops and agroforestry	0.6	Shade coffee with a tree overstory, typically containing multiple species; also includes diversified fruit plantations
9	Successional vegetation	0.6	Abandoned or fallowed agricultural land containing shrubby or woody vegetation approximately 3-8 years old; known locally as *charral* or *tacotal*
10	Thin broadleaf forest	0.7	Second-growth forest with a sparse canopy or many gaps
11	Thin pine forest	0.7	Pine or oak/pine forest with a sparse canopy or many gaps; present only in Copán
12	Riparian forest	0.9	Forest that abuts and is within 150 m of a river or stream
13	Dense broadleaf forest	1.0	Broadleaf forest with a closed canopy of mature trees; includes second-growth and

Land use code	Land use/ land cover	Biodiversity Score	Description
			primary forest
14	Dense pine forest	1.0	Pine or oak/pine forest with a closed canopy of mature trees; present only in Copán
15	Bare soil	0.0	Bare soil
16	Human settlement	0.0	Towns, villages, clusters of buildings, wastewater treatment plants, major paved roads
17	Water	0.0	Rivers and other water bodies
18	No data	0.3	Clouds, shadows where the land cover is not discernable, and gaps in the imagery; the assigned Biodiversity Score of 0.3 was chosen because it is the modal value for all the landscapes and therefore is likely to result in the least possible error

To evaluate the correspondence of the land use classification to actual on-the-ground conditions, I conducted an accuracy assessment by comparing the results of the heads-up digitizing to plots of known land use that had been classified previously through field visits as part of the GEF-Silvopastoral project. The analysis revealed an overall accuracy of 66%, which is generally considered to be satisfactory for this type of application (Table B-2). Furthermore, some of the inaccuracies are attributable not to errors in interpretation but rather to minor differences in the way in which land use was classified from the satellite images versus on the ground. The 18-category land use classification scheme shown in Table B-1 is an attempt to synthesize and standardize the landscape-specific classification schemes used by field technicians in each landscape. Although all of these schemes are ostensibly based on the same 28-category classification system, in practice there were several deviations and local interpretations of this system at each site. For example, in some cases, the criteria for pasture with low tree density and pasture with high tree density were applied inconsistently in the field whereas this study applied a consistent cutoff of 15% tree canopy cover during the image interpretation. Discrepancies such as this account for some of the errors indicated in the accuracy analysis. When this type of discrepancy is ignored, accuracy rises to approximately 80%.

Table B-2. Accuracy analysis for land use classification for the Quindío, Colombia landscape. Land use codes are as follows: 1 = annual crops; 2 = unshaded perennial crops; 3 = pasture without trees; 4 = pasture with low tree density; 5 = pasture with high tree density; 7 = plantation; 8 = diversified perennial crops and agroforestry; 9 = successional vegetation; 10 = thin broadleaf forest; 12 = riparian forest; 13 = dense broadleaf forest.

	ACTUAL LAND USE (FIELD-VERIFIED)												
	1	2	3	4	5	7	8	9	10	12	13	Total	User's Accuracy
INTERPRETED LAND USE 1	4											4	100%
2		7										7	100%
3		1	10									11	91%
4	3	1		3							1	8	38%
5				5				1				6	0%
7				1		3						4	75%
8						1	6				1	8	75%
9	1						1	5				7	71%
10				1				1			1	3	0%
12								1		10	5	16	63%
13											2	2	100%
Total	8	9	10	10	0	4	7	8	0	10	10	76	**66%**
Producer's accuracy	50%	78%	100%	30%	NA	75%	86%	63%	NA	100%	20%	**66%**	

FRAGSTATS ANALYSIS

Based on the land use classifications, I used the software program FRAGSTATS 3.3 (McGarigal et al. 2002) to analyze a suite of landscape composition and structure variables for each of the six context zones surrounding each sample point. To prepare the land use data for the FRAGSTATS analysis, I generated three input rasters, each of which was used in a separate FRAGSTATS analysis. Following is the sequence of steps to prepare these rasters:

1) I converted the vector land use maps into raster files with a 5 m pixel size by using the vector-to-raster conversion utility in ArcGIS 9.2. This yielded Landscape Raster #1, a raster of pixel values from 1 through 18, with each pixel value signifying the corresponding land use (see Table B-1).
2) I re-classified Landscape Raster #1 to create a binary raster distinguishing forested land uses (land use codes 12 through 14) from non-forested land uses (all other land use codes). This resulted in Landscape Raster #2, which consisted of two different values: 0 for non-forested pixels and 1 for forested pixels.
3) I re-classified Landscape Raster #1 to create a binary raster distinguishing tree-covered land uses (land use codes 5 through 14) from non-tree-covered land uses (all other land use codes). This resulted in Landscape Raster #3, which consisted of two different values: 0 for non-tree-covered pixels and 1 for tree-covered pixels.

Next, I used ArcGIS 9.2 to clip each of these landscape rasters by each of the buffers (100, 200, 400, 800, 1500, and 3000 m) around each of the sample points. I saved these files according to a standard naming convention so that they could be easily input into FRAGSTATS batch files. I also generated an edge contrast matrix, which is required to calculate contrast-weighted edge density. This matrix consisted of contrast values for each pairwise combination of land uses, and was based on the absolute value of the difference in the Biodiversity Scores for each pair of land uses (see Table B-1). For example, pasture with low tree density (Biodiversity Score = 0.3) and riparian forest (Biodiversity Score = 0.9) have a contrast of 0.6. All contrast values were between 0 and 1, inclusive.

Using the previously mentioned input files, I conducted FRAGSTATS analysis for each landscape context zone around each sample point. The FRAGSTATS metrics that I analyzed were selected in advance based on hypotheses about their relevance to birds and butterflies, as discussed in Chapter 3. FRAGSTATS outputs were imported into Microsoft Excel and edited to extract the metrics of interest for further analysis.

LITERATURE CITED

McGarigal, K., S.A. Cushman, M.C. Neel, and E. Ene. 2002. FRAGSTATS: spatial pattern analysis program for categorical maps. University of Massachusetts, Amherst, Massachusetts. Online: www.umass.edu/landeco/research/fragstats/fragstats.html.

Murgueitio, E., M. Ibrahim, E. Ramirez, A. Zapata, C.E. Mejia, and F. Casasola. 2004. Land use on cattle farms: guide for the payment of environmental services. CIPAV, Cali, Colombia.

Pagiola, S., P. Agostini, J. Gobbi, C. de Haan, M. Ibrahim, E. Murgueitio, E. Ramírez, M. Rosales, and J.P. Ruíz. 2005. Paying for biodiversity conservation services: experience in Colombia, Costa Rica, and Nicaragua. Mountain Research and Development 25: 206-211.

APPENDIX C

CRITERIA FOR ASSIGNING SCORES FOR BIRD BCV

CRITERIA FOR DETERMINING BIRD BCV

This Appendix provides the criteria used to assign numerical scores for birds for each factor in the BCV metric (Table C-1).

Table C-1. Criteria for assigning numerical scores for birds for each factor in the BCV metric. The second column indicates the percentage of the 404 bird species observed in this study that fall into each data category or range for each factor.

BCV factor	% of species	Point value
Factor 1: Species range size		
Endemic: range of <50,000 km^2	2%	4 points
Restricted: range of 50,000 to 200,000 km^2	5%	2 points
Not endemic: range >200,000 km^2	93%	0 points
Factor 2: Total global population		
Population size <50,000	3%	3 points
Population size 50,000 - 500,000	14%	1 point
Population size >500,000	83%	0 points
Factor 3: Sensitivity to disturbance (as described by Stotz et al. 1996)		
High sensitivity	5%	2 points
Medium sensitivity	31%	0.5 point
Low sensitivity	64%	0 points
Factor 4: Habitat specificity / forest dependence (data from Stiles 1985 and bird field guides)		
Forest-dependent: Stiles' 1 rating and equivalent	3%	2 points
Forest-specialist: Stiles' 1-2 rating and equivalent	7%	1 point
Forest-generalist: Stiles' 2 rating and equivalent	30%	0.5 point
Non-forest-dependent: Stiles' 2-3 and 3 ratings and equivalent	61%	0 points
Factor 5: Conservation threat or priority status (data from Stotz et al. 1996 and Baillie et al. 2004)		
IUCN Red List Critically Endangered	0%	15 points
IUCN Red List Endangered or Stotz et al. "Urgent" classification	0%	12 points
IUCN Red List Vulnerable	0.5%	9 points
IUCN Red List Near Threatened or Stotz et al. "High" classification	1%	6 points
Stotz et al. "Medium" classification	5%	3 points
Does not meet any of the preceding criteria	94%	0 points

LITERATURE CITED

Baillie, J.E.M., C. Hilton-Taylor, and S.N. Stuart. 2004. 2004 IUCN Red List of Threatened Species: a global assessment. IUCN, Gland, Switzerland.

Stiles, F.G. 1985. Conservation of forest birds in Costa Rica: problems and perspectives. Pages 141-168 in A.W. Diamond and T.E. Lovejoy, editors. Conservation of tropical forest birds. International Council for Bird Preservation, Cambridge, UK.

Stotz, D.F., J.W. Fitzpatrick, T.A. Parker, and D.K. Moskovits. 1996. Neotropical birds: ecology and conservation. University of Chicago Press, Chicago.