FINDING COMMON GROUND: CONSERVATION, DEVELOPMENT AND INDIGENOUS LIVELIHOODS IN THE HUASCARÁN BIOSPHERE RESERVE, PERU

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ABSTRACT

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(Under the direction of Dr. Bruce Winterhalder and Dr. Tom Whitmore)

The Huascarán Biosphere Reserve (HBR) is a tropical Andean protected area managed for "conservation with development," one that is experiencing the common challenge of achieving both. Although there is little question that what was recognized as a global treasure in 1975, is a cultural landscape shaped by a long history of human occupation and management, the sustainability of indigenous agro-pastoral land use is now in question. My research was motivated by a desire to understand the environmental outcomes of indigenous livelihoods in an era in which they are being increasingly transformed by the conservation and development agendas of national and international actors. I argue that failures of people-centered conservation are less to do with failures of local indigenous peoples to sustainably manage resources, and more to do with failures of policy-makers to accept responsibility for their role in shaping this outcome. Through the lens of cultural and political ecology I show that successful biodiversity conservation in the HBR will demand greater attention to the specifics of *common property* management, and to the social, political, economic and environmental contexts in which communal institutions and their constituent decision-makers are embedded. This multi-scaled perspective takes a critical look at tourism and its influence on the herding practices of indigenous agro-pastoral households in the HBR. By drawing on *common property* theory, human behavioral ecology, and ecological fieldwork I show that enclosure in a protected area and the unsustainable growth of adventure tourism have had many unintended consequences. I discuss these consequences throughout the dissertation as they are revealed through analyses of data collected during 2 years of fieldwork, their implications for indigenous livelihoods and biodiversity conservation in the HBR, as well as some potential solutions for avoiding negative outcomes in this unique protected landscape.

DEDICATION

To my family whose love and support made my education, fieldwork and write-up possible. Especially to my husband Barker, for whom I am grateful beyond words.

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As is the case with any effort of this magnitude, or any process as long, I could not have done it alone. Reflecting on the rather substantial number of individuals to whom I owe a debt of thanks, I must simply say that I am extremely fortunate. Some individuals assisted with very specific stages of this process, while some have had a presence throughout. I want to begin by thanking the individuals that have been with me from the beginning. Thanks to my parents, Bob and Lisa Sullivan, for the opportunity to pursue a graduate education and for their moral support over the duration of this project. I especially want to recognize my mother, whose expert editorial and data entry skills were often employed without solicitation or pay. Most importantly, I am grateful for my husband, Barker Fariss, for his unconditional love, tireless support, long hours in the field, and unyielding optimism. Lastly, thanks to my two-year old son Keller, for being my biggest source of inspiration, even if he doesn't yet know it.

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I hope that mine does justice to the lives and challenges of those most intimately connected to this amazing place.

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LIST OF ABBREVIATIONS

ASAM	Asociación de Servicios de Alta Montaña
AATAM	Asociación de Asociación de Trabajadores de Alta Montaña
CEDEP	El Centro de Estudios para el Desarrollo y Participación
CP	Common property
CUP	Comités de Usuarios de Pastos; pasture user-groups
FONCONDES	Fondo Nacional de Compensación y Desarrollo
GIS	Geographic information system
GO	Governmental organization
GPS	Global positioning system
HBE	Human behavioral ecology
HBR	Huascarán Biosphere Reserve
HNP	Huascarán National Park
INEI	Instituto Nacional de Estadística e Informática
INRENA	Instituto Nacional de Recursos Naturales
NGO	Non-governmental organization
PRONAA	Programa Nacional de Asistencia Alimentaría
PRONOMACHS	Programa Nacional de Manejo de Cuencas Hidrográficas y Conservación de Suelos
TMI	The Mountain Institute, Instituto de Montaña; Andean program
TY	Tupac Yupanqui, specifically the sectors of Collón and Pashpa

GLOSSARY

The words in this glossary are italicized throughout the text. Brief definitions are offered here:

Alcalde: mayor of a town or peasant community.

Arriero: mule driver hired to carry gear for visiting trekkers and mountaineers.

Asamblea: community meeting or assembly.

Barrio: area or neighborhood: a collection of houses and fields within a community.

Bofedale: wet pasture created by topographies that allow for water accumulation; a critical dry season reserve that is usually avoided in the wet season.

Campaña chica: the less common dry season planting of the agricultural calendar.

Campaña grande: the most common wet season planting of the agricultural calendar.

Campesino/a: rural man/woman; peasant.

Chacra: agricultural field worked by a single household or collective.

Choza: small round hut of rock or adobe mud brick; dwelling.

Common pool resource: a resource with the characteristics of subtractability (i.e., one person's use subtracts from another's) and excludability (i.e., difficult and costly to exclude others from using). Examples include irrigation networks and rangelands.

Common property: a socially defined relationship to a resource whereby its management is achieved by a clearly defined group of individuals with rights of access and use; contrast with open-access.

Comunero/a: recognized member of the community; an individual entitled with rights to access and use common property in the community.

Comunidades Campesinas: rural indigenous communities.

Cordillera Blanca: a glaciated north–south trending mountain chain in Peru, approximately 21 km wide and 180 km long; translates as the White Range.

Empresa: a business; usually a community-run enterprise.

Gramadale: xeric pasture common to topographies where water tends to run off.

Hacienda: a large land-holding estate granted to Peruvian elites and operated largely by the labor of the rural poor; a system that was dismantled during Peru's land reform of 1969.

Huerto: household garden.

Jalca: high altitude grasslands in the transitional ecotone between wetter páramo grasslands of the northern Andes and drier puna grasslands of the southern Andes.

Jefe: economic head of the household, usually (but not exclusively) the adult male.

Jornal: day laborer.

Laguna: lake.

Lo andino: a way of life characteristic of the Andes.

Minka: reciprocity-based communal work party.

Nevado: mountain.

Open-access: a relationship to a resource whereby rights of access and use are open to all; there are no clear owners and no arrangements for how the resource is used or managed.

Paja: crop stubble utilized as livestock fodder.

Páramo: high altitude grasslands of the northern Andes.

Partible inheritance: system of inheritance whereby property is divided equally among descendents.

Portada: gate to the Ishinca valley used as a mechanism of establishing community-sanctioned openings and closings for high altitude pastures in the national park.

Puna: high altitude grasslands of the southern Andes.

Quebrada: valley.

Siete Cabrillos: Seven goats; the constellation of Pleiades.

Transhumance: seasonal herding patterns dictated strongly by the availability of natural forage; in mountainous landscapes, this movement is often up-valley as the dry season progresses.

Trigo: wheat.

Usufruct: a recognized right to use and secure the benefits of a communally-held resource granted to a comunero.

Vacas silvestres: wild cattle; an unmanaged communal herd often in the national park.

Queñual: local name for the genus *Polylepis*; a tree species of the Rosacea family endemic to the Andes; a conservation priority in the national park.

CHAPTER 1

INTRODUCTION: ILLUSION AND REALITY IN PROTECTED LANDSCAPES OF THE TROPICAL ANDES

As I sat at a small ephemeral lake with the silhouette of the Huascarán Biosphere Reserve's (HBR) highest summit (6768 meters) perfectly mirrored in its still depths, the image before me was as magnificent as any I could conjure in my mind's eye. Such an image evokes certain impressions about a place (see Photo 1.1). The monumental and rugged appearance of the Andean landscape creates the illusion that it is timeless and untouched. But, things are not often what they appear. Although Nevado Huascarán is perfectly reflected in the lake before it, the true nature and complexity of contemporary Andean landscapes is not. Their true nature is first revealed by recognizing that "interaction with nature has…been humankind's most enduring practical concern (Netting 1986). Despite increasing efforts to protect places like these from human influence (Rodriguez et al. 2004), most of this area is neither beyond the influence of local actors, nor by consequence of its protected status, beyond the influence of distant political and economic forces (Cronon 1996, Denevan 1992).

Photo 1.1: Nevado Huascarán reflected in an ephemeral lake above Pashpa (photograph by the author, 2002).

A closer look at the landscape of the HBR reveals a multi-faceted human dimension as complex as its topography (Zimmerer 2000, Zimmerer 2006, Zimmerer and Bassett 2003, Zimmerer, Galt, and Buck 2004). Landscapes of the tropical Andes are cultural ones, shaped not only by a long history of human occupation and use, but recently by the policies of conservation actors with very different practical concerns. While the HBR may hold a variety of meanings for all parties involved (i.e., indigenous residents, conservationists, and tourists), all must agree that it is undeniably shaped by the ongoing dialectic between them and it. The reality below the surface of the illusion in Photo 1.1 is that this protected area is not only a landscape influenced by humans, but one that reflects what these various actors think it is, and what they think it should be. It is the juxtaposition of indigenous livelihoods and conservation interests, and the inability to find common ground between them that have created numerous tensions between people and parks world-wide. In the HBR, a similar tension brews just below the surface—this is its current reality.

PEOPLE AND PARKS

Concerns of environmental degradation in the HBR implicate the region's indigenous communities because they are directly involved in its use and management (INRENA 1990a, INRENA 2002, TMI 1996b, TMI 1997, TMI 2001a).[1] While local resource use is undoubtedly an important factor to consider, simple assumptions that

[1] Much of this newly created conservation territory falls under the category of sustainable use not strict protection. These conservation policies seek to integrate local resource users in strategies of "conservation with development". This has been dubbed the "third wave" of conservation by some researchers (Brandon, Redford, and Sanderson 1998, Zimmerer 2006).

local livelihood strategies are inimical to conservation are short-sighted and inadequate according to most political ecologists (Robbins 2004). Even so, local mismanagement has a long history as the dominant explanation for environmental degradation in the highlands (Eckholm 1976, Ellenberg 1979, reviewed in Forsyth 1998), and protected areas in general (Kramer, van Schaik, and Johnson 1997, Oates 1999). This pervasive attitude manifests itself in a growing trend to exclude humans from protected areas or to diminish their role in managing them (Chapin 2004, Terborgh et al. 2002). In doing so, conservation not only marginalizes but vilifies indigenous peoples and local resource users (Fairhead and Leach 2000, Neumann 1998, Stevens 1997, Wilshusen et al. 2002). My research was motivated by a fundamental concern with viewing environmental change (real or perceived) in the HBR with such a narrow focus, and in doing so, jumping to false conclusions about indigenous peoples, indigenous institutions, and people-centered approaches to conservation. Instead, my research looks beyond proximate causation to the ways in which their political and economic contexts shape decisions regarding the use and management of the reserve and its conservation core. In doing so, I hope to illustrate that the condition of the tropical Andes, for better or worse, is a responsibility which both local and supra-local actors have in common.

SCALING UP IN HUMAN-ENVIRONMENT RESEARCH

My research adopts a cultural and political ecology perspective. By doing so it explores the complexity of interactions between the region's indigenous agro-pastoralists and their recently protected environment. It seeks to understand how this interaction is affected by the creation of the park and its policies of "conservation with development," especially as they relate to the development of tourism. I work from the premise that the various scales of influence shaping the HBR are "nested within one another, with local decisions [e.g., individual, household, community] influenced by regional policies, which are in turn directed by global politics and economics" (Robbins 2004). While I acknowledge that there are a variety of definitions and research agendas emerging in political ecology, my perspective is loosely based on combining "the concerns of ecology with a broadly defined political economy" (Blaikie and Brookfield 1987). A prominent focus in political ecology research is on conservation in the developing world (e.g., Zimmerer and Young 1998). My study follows the same line of inquiry and employs similar methodologies to those grounded in biogeography and ecology, including an attention to place and ecological analyses (e.g., Turner 1998, Zimmerer 1991, Zimmerer 1993, Zimmerer 1996). By conducting this research I hope to provide additional insights into the conservation challenges specifically facing the HBR, as well as to add to examples specific to protected highland landscapes in general (i.e., Brower and Dennis 1998, Byers 2000, Echavarria 1998, Young 1998).

SCALING DOWN IN HUMAN-ENVIRONMENT RESEARCH

Although human-environment researchers realize the need to situate their study of local resource users in a broader context, such a focus does not diminish the necessity to address the dynamics of environmental change at the scale where decisions are acted out upon the landscape (Hewitt 1988, Lauer 1993). Environmental change is the aggregate consequence of the actions of independent decision-makers. The focus of much of the research presented here is on the decision-making processes of the household; the household being an important unit of decision-making in the Andes (Brush and Guillet 1985, Mayer 2002, Netting 1981, Netting 1993, Zimmerer 2004). A cultural-political ecology perspective requires us to examine the contexts in which household decisions are made as well as how they are mediated at scales beyond as well as within the household. This requires exploring the role of conservation NGOs, government agencies, and communal institutions, as well as the role of individuals of different ages, genders, and access to opportunities such as wage-earning and education in influencing household-level decision making and its environmental outcomes.

There is a lack of conclusive theory guiding attempts to explore local decision-making and its environmental outcomes within this multi-scaled framework. This research draws on *common property* (CP) and human behavioral ecology (HBE) as its theoretical framework. This includes the use of game theory as a means of predicting the decisions of multiple users of a commons, and how these decisions are mediated by community rules and sanctions. Persistent attention has been given to *common property* management from cultural and political ecology researchers. CP theory allows us to explore the ways in which communal institutions function to maintain the cooperation of their constituents, as well as the conditions under which they are likely to fail.

Recent decades have seen an increase in conservation initiatives, predominately in tropical countries of the western hemisphere, which have increasingly brought many local actors into real or potential conflict with national and international conservation agendas (Bruner et al. 2001, Zimmerer 2006). Most of these conservation projects are focused on biodiversity, and many seek to achieve its conservation by integrating local resource users into sustainable economies designed to offset their use of key resources, thereby decreasing their potential degradation. At best, these policies may be presumptuous. A number of studies have reported that indigenous peoples and institutions of resource management maintain, and in some cases create biodiversity (Fjeldså, Lambin, and Mertens 1999, Oldfield and Alcorn 1991, Piperno and Pearsall 1998, Robbins et al. 2006).

Human subsistence in the Callejón de Huaylas of the Peruvian Andes extends as far back as 10,000 years bp, as estimated from the oldest cultivated botanical assemblages of the New World found in the region's Guitarrero Cave (Moseley 1992). The imprint of humans on the Andean landscape is prominent (Gade 1992). In the buffer zone of the HBR, this imprint manifests itself in terraces, irrigation canals, and the patchwork of *chacras* (agricultural fields) and livestock corrals of local *comunidades campesinas* (rural indigenous communities); while subtle, but no less significant, imprints are evident at the higher elevations of the HNP. Given this long occupation and the region's status as a biodiversity hotspot (Manne, Brooks, and Pimm 1999, Myers et al. 2000, Rodriguez and Young 2000), it is likely that local communities have played a role in creating the landscape and the diversity conservationist wish to protect; at a minimum, they have not been strongly inimical. If this is so, how can indigenous peoples of the HBR be both defenders and destroyers of the tropical Andes, as it is currently assumed? This question cannot fully be explored without careful consideration of the importance of *common pool resources* in the world's highlands, the institution of *common property* management, and the challenges of maintaining collective action under changing socio-political and economic contexts (Agrawal 2001, Bromley et al. 1992, Lambin et al. 2001, McCay and Acheson 1996, McKean 2000, Ostrom 1992).

COMMON PROPERTY

Many common pool resources play an integral part in rural livelihoods of the highlands (Berkes 1989, Netting 1976, Prakash 1998). The focus of this dissertation is on the *puna* (high elevation grassland), which is managed as *common property* by the reserve's indigenous communities despite its enclosure in a national park (see Photo 1.2).[2]

Photo 1.2: The windswept *puna* of the HBR (photograph by the author, 2002).

[2] An estimate of land cover in the HBR indicates that these high elevation grasslands are the dominant vegetation in the region, accounting for approximately 47% of the reserve's area (INRENA 1990b). Given their importance as a grazing resource, the factors affecting their use and management become critical to conservation concerns for the HBR.

A few key terms and clarifications will be relevant for subsequent chapters. First, a resource is any material good. Those that are diminished in quantity or quality through use (i.e., are subtractable) as well as costly to exclude others from using are *common pool resource*s (McKean 2000). Mountains are particularly rich in these resources, and the study of their use and potential degradation has been a research priority in recent decades (e.g., Casimir and Rao 1998, Orlove 1976, Lynch 2001). *Property* is a social construct whereby people relate to a resource through a set of social arrangements defining the rights and responsibilities governing resource use (Bromley and Cernea 1989, McCay and Acheson 1996). A resource and the social context in which it is embedded collectively define a *property regime*. *Common property* (i.e., property held and managed by a group) is one possible type; while private (i.e., property held and managed by an individual) and state property regimes (i.e., property held and managed by a centralized authority) are others (Stevenson 1991). *Common pool resource*s that are extensive, spatially or temporally heterogeneous, low yielding, unpredictable and offer little potential for intensification, like those of the *puna*, can be optimally managed as *common property* (Berkes et al. 1989, Feeny et al. 1998, McKean 2000, Netting 1976).

Many protected areas were established as a means of transferring access and control of resources from an individual or group to the state. Yet this transfer is not absolute in people-centered conservation approaches. Indigenous communities in the HBR are allowed access to and use of pasture resources within the conservation core of the HNP (del Castillo, Gallo, and Monge 1995). Several community-specific *common property regime*s dictating who has access to these resources, as well as the rules governing their use, were long-established before the park was created. These institutions continue to mediate the use of the reserve's resources the region's indigenous residents despite the ultimate and sometimes ill-defined authority of the state (Pinedo 1999).[3]

Although there are current concerns about the sustainability of community-based management in the HBR and the need for increased state involvement, recent decades have seen the demise of an erroneous assumption that these arrangements are doomed to failure. Garrett Hardin's (1968), "Tragedy of the Commons" was the reigning metaphor regarding the management of property by a group, and still lingers in popular discourse and policy-making today. Yet, it is now widely recognized that *common property* regimes are fundamentally different from *open-access* where such failures are more certain (Ciriacy-Wantrup and Bishop 1975, Feeny et al. 1998). In short, *common property* is akin to private property for a defined group, where accountability and rules of use do exist (McKean 2000). In other words, it is not a situation of *open-access*, where everyone can use a resource and there are no incentives for managing it sustainably. This permits the possibility that *common property* institutions of the Andes may not only avoid tragic outcomes, but also be the most optimal and socially just arrangement that exists for managing the *puna*.

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[3] A common property institution continues to function in the study community despite the creation of the park and the ultimate authority granted to the state. Park authorities recognize comités de usuarios de pastos (pasture user-groups) or CUPs that are granted access to grazing resources in exchcange for keeping and reporting a detailed livestock census, as well as for cooperation in reforestation efforts (Tohan 2000). Currently, management of livestock is largely determined by the operational rules of the community, thus I treat the arrangement as one of common property as opposed to state property (for similar treatment see Pinedo 1999).

This dissertation is based on the premise that indigenous institutions for managing resources in the HBR can be sustainable when supported by appropriate policies. Such policies depend on an enhanced understanding of the importance of *common pool resource*s to the world's highland subsistence communities, the various decision-making arrangements that govern their use, the patterns of use that emerge among individual users of a commons, and the factors that affect them in the present day. Given the prominence of *common property* in the Andes, and the challenges of maintaining sustainable use of a commons in the face of economic diversification and growing heterogeneity among its users, these insights will also be critical for the success of "conservation with development" (Adams et al. 2003, Jodha 1992).

HUMAN BEHAVIORAL ECOLOGY

Many "conservation with development" projects have arguably failed due to a lack of sufficient attention to indigenous livelihoods and institutions of resource management. Conservation in the HBR will require an exploration of the factors influencing local decision-makers so that we may better understand the challenges faced by communal institutions in maintaining their cooperation (Caro 1998). This study draws on human behavioral ecology (HBE) to frame hypotheses of the decision-making processes driving environmental change in the HBR. To date, HBE has contributed in innumerable ways to the conservation literature, and has the potential to shed many insights on the situation unfolding there (Holt 2005, Smith and Wishnie 2000, Tucker 2007, Winterhalder and Lu 1997). In general, HBE operates from an assumption that decisions are driven by economic concerns—by the resource user's consideration of the costs and benefits associated with available options (Winterhalder and Smith 2000).

Given the criticisms of 'rational behavior' by some political ecologists, it is important to note that the approach of HBE does not imply that available options are unrestricted, knowledge of them perfect, or that the decision reached is necessarily the optimal one relative to a particular phenomenon under study. What it does imply is that decision-makers do their best to operate rationally, and where they appear not to, are not 'irrational', but likely operating under the influence of additional constraints that can be revealed through careful study. HBE researchers have included the effects of power inequalities, wealth differentials, coercion, uncertainty and risk in their decision-making models (Winterhalder 1990); in doing so, they have offered us many heuristic devices for generating testable hypotheses of human behavior and resource use (Winterhalder 2002a). Thus, the perspectives of HBE are not necessarily at odds with those of cultural and political ecologists, who are largely concerned with the same research questions focused on understanding the factors affecting indigenous resource management and environmental degradation. Here the aim is to apply both individual and institutional perspectives and scales of human-environment analysis to indigenous decision-makers; who are no less divorced from economic concerns, and are perhaps even more likely constrained geographically, economically, politically, socially and historically.

Innumerable conservation failures justify the need to develop a comprehensive approach such as this; especially when considering the current trend of polarizing indigenous peoples as inherently conservationist or inherently incapable of serving as environmental stewards regardless of the context in which they are embedded (Redford and Stearman 1993). HBE offers an alternative to these stereotypes, as well as a theoretical framework for generating testable hypotheses. An example relevant to this research, and one tested in Chapter 4, applies game theory to the decision-making of transhumant Barabaig pastoralists managing a commons. Ruttan and Borgerhoff-Mulder (1999) present an alternative the "tragedy of the commons" scenario by illustrating that wealthy herders receive a greater share of the benefits from communal resources. Thus, they are more likely to have a vested interest in managing their own herd sustainably. In addition, the authors show that wealthy herders typically have more power and the means to coerce other relatively poor herders to do the same (*ibid* 1999). This finding offers the possibility of an optimistic outcome for conservation in the HBR given the region's growing tourism industry and the associated increase in livestock holdings for these households as described in Chapter 2. However, if conservation is "isomorphic with economic efficiency" for wealthy herders and can be encouraged of poor herders, at some point this cooperation becomes unlikely when wealth inequalities are extreme (*ibid* 1999). Even if rejected, like any model tested against empirical observation, it serves to increase our understanding of the various scales of influence acting on particular resource users and allows us to frame new questions about the factors affecting the decision-making process (Winterhalder 2002b).

RESEARCH OBJECTIVES

The overall objective of this research was to take a multi-disciplinary and multi-scaled approach to study the factors affecting land use, communal resource management, and conservation in the HBR. Although a number of studies have described Andean agro-pastoral land uses and institutions for managing them (e.g., Brush 1977, Guillet 1983, Knapp 1991, Mayer 2002, Orlove 1977), few have focused on their overlap with protected areas and the changes that have ensued as a result of this interaction. Locally relevant exceptions include Byers (2000), Kintz, Young, and Crews-Meyer (2006), Tohan (2000), Young (1993), and Young and León (1993). My research was motivated by a desire to contribute to the literature on Andean livelihoods in an era where many are increasingly transformed by the conservation and development agendas of national and international actors (Bebbington 2000). It was also motivated by claims that indigenous agro-pastoral communities are mismanaging the resources of the HBR, and therefore threatening the biodiversity it seeks to protect.

Recent work in cultural and political ecology has revealed that policies collectively referred to as "conservation with development" can result in significant unintended consequences (see Zimmerer 2006). Although contrary to intention, such policies have often disrupted or even disabled indigenous livelihoods without creating sustainable alternatives. Thus, the research questions explored during my fieldwork were predicated on the possibility that a comprehensive study of over-grazing in the HBR would require looking beyond local resource management to the policies of "conservation with development" that have encouraged the growth of the region's adventure tourism industry (i.e., climbing and trekking). While this frames the dissertation as a critique of current policies in the HBR, a nuanced understanding of how the adventure tourism industry articulates with the land use and resource management decisions of the region's agro-pastoral residents is necessary in order to move beyond a stale debate over people vs. parks, the sustainability of tourism, and the viability of "conservation with development" in the developing world. With these interests in mind, the specific research questions I sought to address during fieldwork were the following:

(1) What is the current state of land use in the HBR? How has enclosure in a national park and the subsequent development of tourism industry affected indigenous resource users and the communal institutions that manage them?

(2) How does tourism involvement influence the management of household herds in the HBR? Do the patterns of use and interaction emerging on common property pastures suggest unsustainable outcomes?

(3) Is there evidence of environmental degradation in the buffer zone or the core of the HBR? Specifically, what is the relative effect of grazing on native plant diversity, and what are the implications of changing land use scenarios for biodiversity conservation in this protected landscape?

These findings should be of interest to ecological anthropologists, geographers, cultural and political ecologists, conservation biologists, *common property* researchers, tourism researchers, and highland tropical ecologists in general; as well as to stakeholders in protected areas, including local (often indigenous) peoples, governments, and international NGOs in the HBR.

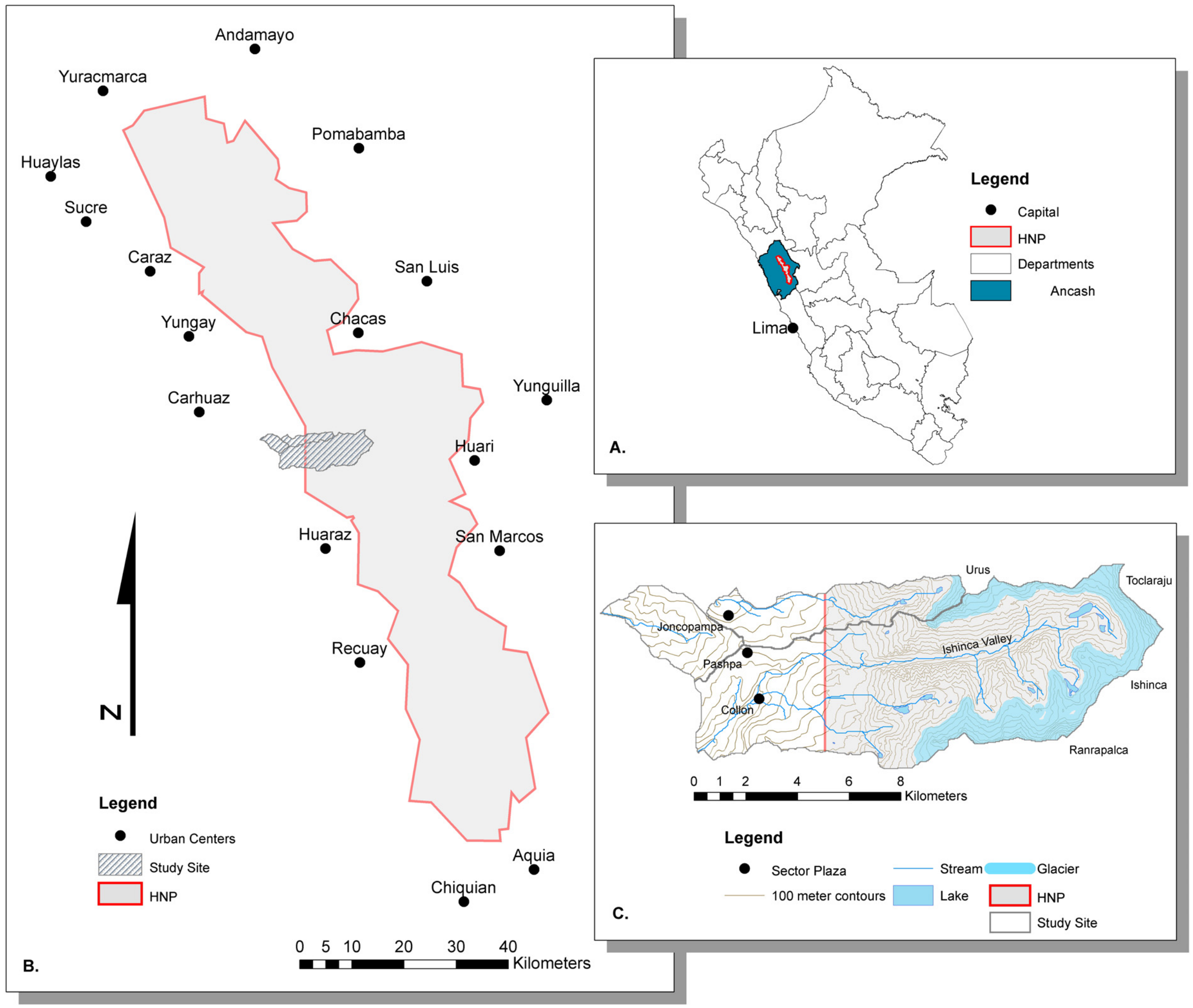
STUDY SITE SELECTION

I returned to the Andes, specifically to the Ishinca valley of the HBR, to answer these questions in 2001. The HNP was established in 1975, and is one of Peru's longer-established national parks. Shortly after recognition as a national park, in 1977, its conservation territory was extended to include several indigenous communities in what is referred to as the Huascarán Biosphere Reserve (HBR). By 1985, the reserve gained international status as a UNESCO world heritage site; and its core (the HNP) is currently categorized as a level II protected area by the IUCN (see Figure 1.1).

The reserve is located in the Peruvian department of Ancash. Its conservation core (the HNP) spans the coordinates 77° 10'–77° 50' W and 8° 30'–10° 00' S. Although tropical in locale, it encompasses 3400 km^2 (340,000 ha) of the highest reaches of the Cordillera Blanca, a north–south oriented mountain chain forming the continental divide between Peru's arid coast and the Amazon basin. The HNP is approximately 180 km in length and only 40 km at its widest in the east–west direction. Despite the prominence of glaciated peaks within its borders, it boasts an amazing diversity of plant and animal life; including 779 plant, 112 bird, and 10 different mammalian species—many rare and endemic to the Andes (Kolff and Kolff 1997, Smith 1988). In addition to these conservation priorities, the buffer zone of the HBR encompasses a number of agro-pastoral Quechua-speaking communities whose enclosure in this conservation territory instantly brought an estimated 226,000 decision-makers into national and international conservation agendas (Byers 2000, INRENA 2002). The juxtaposition of local and supra-local interests that has ensued has not been without conflict. The challenge of reconciling the two is what spurred this research.

I sought to explore this conflict by conducting an in-depth study of land use and resource management by one of the HBR's indigenous communities. This decision was motivated by my anthropological training and my desire to paint an intimate portrait of indigenous life and the realities influencing local resource use. My first trip to park headquarters and the offices of NGOs in Huaraz allowed me to identify several possible communities for study. My criteria for selection included the indigenous community's level of involvement in tourism, and the conservation community's perceptions of environmental degradation on the lands managed by them. Although these criteria allowed for several different possibilities, the Ishinca valley seemed well-suited for the purpose. I consider it representative of the trajectory many other indigenous communities in the region are likely to take given the region's burgeoning tourism industry. The Ishinca valley is often considered degraded by park officials and NGOs; it also reportedly receives a substantial amount of tourism traffic (INRENA 1996, INRENA 2002, Ramirez 2002, Tohan 2000). This prompted me to visit the indigenous community of Tupac Yupanqui, specifically the sectors of Collón and Pashpa. These sectors are responsible for the management of resources in the Ishinca valley. In addition, they are ideally situated to take advantage of a stream of tourism traffic that ensures their integration into the market economy. Pashpa ironically happened to be the same community where I first sat at the ephemeral lake of Cochapampa to observe the reflection of Huascarán. Thus, I sought answers in the very place where I formed my first initial questions about the Andean landscape.

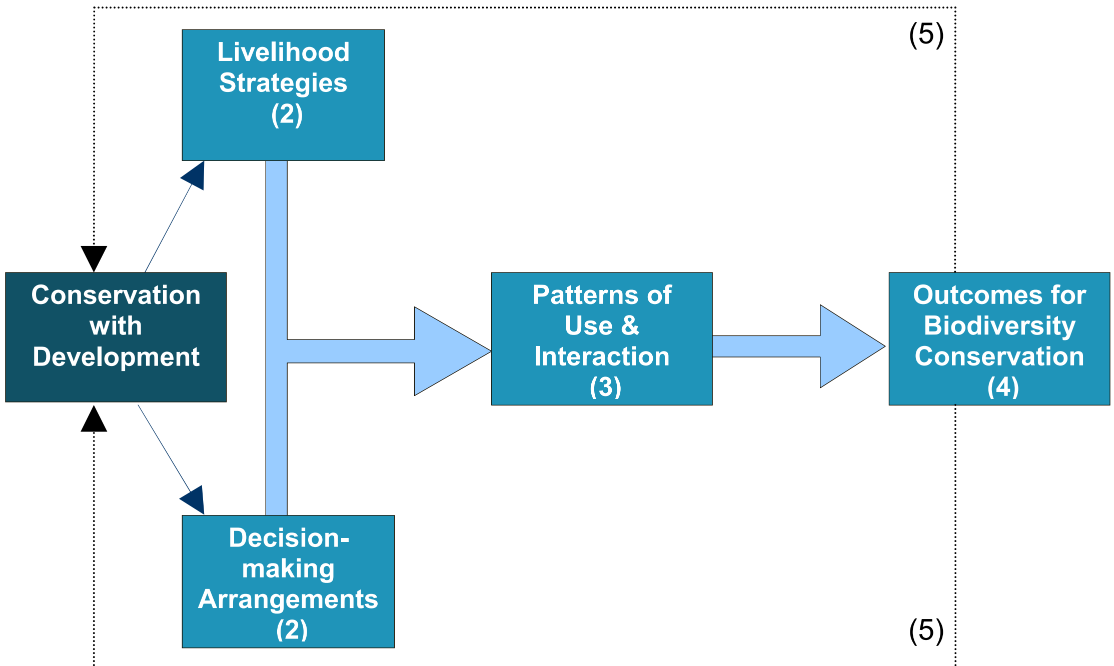
Figure 1.1: Location of the study site (C) in the Huascaran Biosphere Reserve (B) in the Peruvian department of Ancash (A).



ORGANIZATION AND CHAPTER PROGRESSION

This dissertation was written as a series of stand-alone papers progressing from a description of the problem and its context, to exploration of its implications for local resource users, and its consequences for biodiversity conservation. Given the significance of common pool resources to highland subsistence economies, I have organized the chapters loosely after a framework proposed by Oakerson (1992) for evaluating the outcomes of *common property* management (Figure 1.2). This includes describing the physical attributes, technologies and decision-making arrangements defining livelihood strategies in the study site (Chapter 2), exploring the patterns of interaction of individual actors and the factors influencing their decision-making (Chapter 3), quantifying the environmental outcomes of these decisions with regard to native plant diversity (Chapter 4), and making recommendations to help ensure positive outcomes for biodiversity conservation in the HBR (Chapter 5).

Figure 1.2: Framework for evaluating the outcomes of communal resource management modified from one proposed by Oakerson (1992). I modified the original framework to reflect the influence of policies associated with the creation of a protected area, including enclosure and the development of a tourism industry. Chapters related to each element in this framework are indicated in parentheses (#). Dotted lines indicate conclusions and recommendations that may allow for alternative outcomes in the HBR.



Presentation of the research begins with Chapter 2, which provides an important foundation for subsequent chapters. Keeping in the tradition of cultural ecology research, it offers a detailed description of the agricultural and pastoral land uses of variously scaled decision-makers (i.e., individual, household, community, and *empresa*) and the loosely structured and largely implicit *common property regime* through which these uses are mediated. After this background is established, the remainder of the chapter focuses on outlining how a variety of socio-political and economic changes facilitated by the region's enclosure as a national park are creating potentially unsustainable patterns of interaction and failures of cooperation in the management of communal resources. In particular, it explores the effects of (1) interventions by the state, (2) market integration through the development of a tourism industry, and (3) increases in rural population and temporary out-migration. Findings suggest that the establishment of the park boundary, below 4000 meters in the Ishinca valley, has reduced the historical extent of the community's arable land base.[4] Through a combination of land scarcity and increased access to fertilizers and pesticides (granted by wage-earning opportunities and easy access to the Huaraz market), many households are reducing fallow times and intensifying their agricultural production.

Given the coordination of agriculture and pastoralism in the Andes (McCorkle 1987), changes in one have repercussions for the other. With less land in fallow, there has become less land available for grazing in the community. This potentially translates to more dependence on high elevation grasslands in the conservation core of the HNP. In addition to changes in the spatial management of herds, the lucrative tourism industry, which mainly involves transporting the gear of visiting mountaineers with packstock, has created an incentive for households to own more animals. Involvement in tourism, as well as temporary out-migration to work in the growing city of Huaraz, appears to have created the potential to make these investments. The data presented in this chapter indicate that tourism households have larger herds on average. Beyond the measurable inequality in livestock holdings between market and subsistence-oriented households, the nature of the tourism industry is such that the types of animals favored are mostly non-

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[4] The park boundary is generally cited as occurring at 4000 meters, but a series of survey markers scattered throughout the region actually define this boundary, which in some valleys, occurs lower than 4000 meters. This discrepancy can be seen in Figure 2.2.

native. Cattle (requiring little supervision), horse and mule are animals increasingly preferred by all members of the community. With a growing number of non-native livestock and less land in the community to herd them on, sustainable use and cooperation appear more difficult to maintain with the existing institutional framework and the context of uncertainty created by the involvement of actors external to the community. These findings shed important insights on the reasons behind park and community observations that there are more animals in the park than in the past, and raise concerns over the outcomes of such trends.

Chapters 3 and 4 attempt to quantify the patterns described in Chapter 2, their causes, and their outcomes for biodiversity conservation in the HBR by taking a more locally scaled approach. In Chapter 3, a detailed time allocation dataset is used to model household herding labor and the effects of tourism involvement on decision-making by the household. This model tests competing hypotheses from the human behavioral ecology (HBE) literature, in an attempt to explore the effects of market involvement on the household's use of communal resources. Whereas in Chapter 2, I describe the concern of over-grazing in the HBR, in Chapter 3, I illustrate that tourism households have larger herds on average, and that the very industry created to offset the use of the park may have had the unintended consequence of intensifying its use. This chapter is motivated by a desire to understand whether an increase in herd size translates to an increased dependence on communal resources and an increased incentive to manage them sustainably, as suggested by game theory and empirical evidence presented by Ruttan and Borgerhoff-Mulder (1999). Alternatively, does increased market involvement create additional constraints for the household, such as labor conflicts and opportunity costs that result in spending less effort to manage a larger herd? The results of the time allocation study suggest that the latter is an important constraint for many households. Households involved in tourism, with larger herds on average, spent less time actively herding them. This model corroborates the ethnographic observations of absentee herding presented in Chapter 2, a pattern in which some households leave animals unattended on the high elevation grasslands of the HNP.

While these findings would seem to confirm the park's worst fears, the actual outcomes for biodiversity conservation are more complicated, and are mediated by a number of other factors explored in the dissertation—biophysical, social and political. For example, pasture improvement projects by a local NGO in the sector of Collón offer its households a solution to absentee herding in the park. Households of Collón, especially those most involved in tourism, have the option to buy access to improved and enclosed pastures in the community. This access minimizes the conflict between herding and tourism, so that households can manage their herds while maintaining a presence in the community with their packstock if *arriero* work materializes. While this suggests that the actions of local NGOs have offset the community's reliance on the park, Chapters 2 and 3 discuss the need for similar projects in the sector of Pashpa, which makes up the other half of the user-group for resources in the Ishinca valley. The tourism households of Pashpa experience the same opportunity costs, but do not have the same enclosures devoted to herding in their sector. Beyond this, Pashpa is not well-situated (geographically) in the stream of tourism traffic passing through the Ishinca valley. As a result, its households have only moderate involvement in tourism and little means to purchase fodder or access to improved pastures, if they did exist. Thus, it is largely this constituent that is responsible for an increased use of the Ishinca valley. The implications are more than those that are immediately obvious. Sustainable management of a commons requires the continued cooperation of all households from both sectors, and community members of Pashpa feel that they have been disadvantaged. Recommendation for how to improve this situation will be explored throughout the dissertation.

Chapter 4 links the patterns of interaction observed among these diverse households to their outcomes for biodiversity conservation in the HBR. The analysis presented in this chapter is an attempt not only to explore widely held assumptions of over-grazing and environmental degradation, but to tease apart the relative effects of grazing from other abiotic factors structuring vegetation communities. The ecological data presented in this chapter illustrate that plant species frequently cited as indicators of over-grazing, including *Opuntia flocossa*, *Astragalus garbancillo* and *Aciachne pulvinata* (e.g., Poma 2002, Tovar and Oscanoa 2002), occur throughout the lands grazed by the study community. This occurrence has most likely created the numerous and somewhat qualitative perceptions of environmental degradation in the Ishinca valley. Yet, a model of plant species richness, developed from a sample of 12 different pastures throughout the Ishinca valley illustrates that sites with moderate levels of grazing intensity have the highest native species richness. Given the park's focus on biodiversity conservation, there are numerous implications for local land use and resource management. Rather than view the agro-pastoral communities surrounding the HNP as the problem, this analysis argues that they are integral to the creation of the Andean landscape that we wish to protect.

However, an important extension of the analysis in Chapter 4 offers a cautionary tale of assuming that human-environment interactions are static. The patterns of interaction described in Chapters 2 and 3 suggest the potential for unsustainable outcomes by changing the intensity of grazing at certain locales and undermining the prospect for cooperation. Thus, various potential outcomes for biodiversity conservation were modeled in Chapter 4, based on future land use scenarios informed by observation. There was little evidence at the time of fieldwork that the community had responded with new rules to govern these changing patterns of use. Therefore, one prediction was based on the worst-case scenario of doing nothing about a growing pattern of absentee herding in the HNP. The model conditioned with this land use scenario suggests that high altitude pastures in the Ishinca valley may be negatively affected by increased grazing pressure, resulting in the potential loss of a number of plant species at the locales and scales sampled.

Given the lack of baseline ecological data, including detailed climate data, the model developed is admittedly a simplification of reality whose predictions are based on "space for time" substitutions driven by a few key biophysical and anthropogenic variables. However, such an approach offers important initial insights into the role of humans in shaping this landscape and the potential outcomes of policies affecting the decision-making of actors most intimately involved in its use and management. Like any model its purpose is to aid our understanding of human-environment dynamics, while recognizing that the reality is sufficiently more complicated. Chapter 5 concludes the dissertation with a summary of the findings, policy implications and recommendations for people-centered conservation in the HBR.

CHAPTER 2

THE UNINTENDED CONSEQUENCES OF CONSERVATION AND DEVELOPMENT FOR INDIGENOUS COMMUNITIES OF THE HUASCARÁN BIOSPHERE RESERVE, PERU

ABSTRACT

The environment and biodiversity of the Huascarán Biosphere Reserve (HBR) are threatened according to many in the region's conservation community. This chapter explores how social, political, and economic changes in the region have affected agro-pastoral land uses and indigenous institutions for managing them. Here, I examine the extent to which the region's policies of "conservation with development" may ultimately be responsible for these threats. The focus of my analysis is on the conservation of grasslands in the HBR, a resource managed as *common property*. I adopt a cultural and political ecology perspective that relies on *common property* theory to describe the effects of state intervention, tourism involvement, and demographic change on the decision-making of the Ishinca valley user-group, specifically the sectors of Collón and Pashpa in the indigenous community of Tupac Yupanqui (TY). Findings illustrate how conflicts and unintended consequences can emerge when attempting to balance conservation and development, especially where the management of a commons is involved. Data compiled from diverse sources, including census, household and community-level surveys, time allocation observations, ethnography and participant observation suggest that the enclosure of its communal lands with the creation of the

national park, the subsequent integration of many of its households into the market, a young population age structure, and growing number of temporary out-migrants create the following potential problems with respect to biodiversity conservation in the HBR:

(1) reductions in fallow on agricultural lands in the buffer zone;

(2) unanimous desire and widespread ability to increase livestock holdings through wage earnings in tourism;

(3) changes in spatial and temporal management of herds that have increased the presence of non-native livestock in the park;

(4) growing inequalities in livestock ownership and access to pasture resources that may undermine cooperation in the management of grazing commons; and

(5) perceptions of insecurity that may lead households to discount restraint in the use of this protected area

"La vida acá tiene dos lados. La vida de la montaña es duro, pero hermosa. Por eso somos pobres, pero somos ricos también."

Life here has two sides. The life of the mountains is hard, but beautiful. For this we are poor, but we are also rich.

(a comunero of Tupac Yupanqui, Sector Pashpa, 2002)

INTRODUCTION

Despite the beauty of mountain landscapes, one readily associates notions of remoteness, ruggedness and harshness with them. Characteristics such as these suggest that high altitudes provide sparse means to make a living. Yet, life in the mountains can be rewarding as well as difficult, uncertain but consistently so; a contrast realized by all intimately familiar with Andean cultures and Andean landscapes. Contemporary agro-pastoralists of the Huascarán Biosphere Reserve (HBR) recognize the constraints and opportunities that shape their daily lives, and credit their existence to the bounty of the *chacra* (field) and the *puna* (pasture), while many are increasing their participation in tourism. Agro-pastoralism has had a very long and prominent history in this region while tourism has not. A number of concerns have recently been raised over the sustainability of long-standing land use and resource management practices in the HBR—especially in the Ishinca valley. The ensuing study was motivated by the possibility that these concerns are the result of incompatibilities between subsistence and tourism, which brings into question the viability of adventure tourism in the HBR and the fundamental assumptions of the "conservation with development" paradigm that have encouraged it.

This chapter analyzes land use and land tenure in a single indigenous community of the HBR, and how enclosure in a protected area and the subsequent development of a tourism industry, have affected household resource users and the communal institutions for managing them. To meet this objective requires careful consideration of *common property regime*s and the challenges of maintaining sustainable interactions among users of a commons. In Tupac Yupanqui (TY), all land use by the community, *empresa* (community organization/business) or household involves *common property,* and its management is largely governed by a *common property* regime.[5]

Several case studies document the prominence of *common property* regimes in the world's highlands. Examples include the Swiss Alps (Netting 1981), the Nepalese Himalaya (Stevens 1993), the Central Japanese highlands (McKean 1992), the Peruvian Andes (Campbell and Godoy 1992, Trawick 2001), and the highlands of Morocco (Gilles and Hahdi 1992). *Common property* researchers have provided several important insights on these institutions that are relevant to this study. I will comment on two here: 1) *common property* may be the optimal arrangement given the physical characteristics of the resource base, and 2) they can be essential to sound resource management when not aggravated by rapid socio-economic change (Jodha 1987, McKean 1992, McKean 2000, Netting 1981). These insights suggest that the highland resources of the HBR may be best managed communally, yet their sustainable use is ensured only when communal

[5] Given the existence of the national park, a *state property* regime has been layered onto the traditional *common property* regimes of the reserve's indigenous communities. Indigenous peoples have retained their rights to graze and forage in the conservation core of the park, thus its management is still largely determined by these communites. In some places state control is apparent, such as in the community of Catac in the valley of Pastoruri. In others it is less so. Although residents of Colllón and Pashpa make up a pasture user-group that reports their livestock holdings to the park, at the time of fieldwork, the management decisions of household herders were largely governed by the operational rules of the community. Thus, I refer to *common property* management throughout the dissertation.

institutions can keep pace with the inevitably changing circumstances of their constituents.

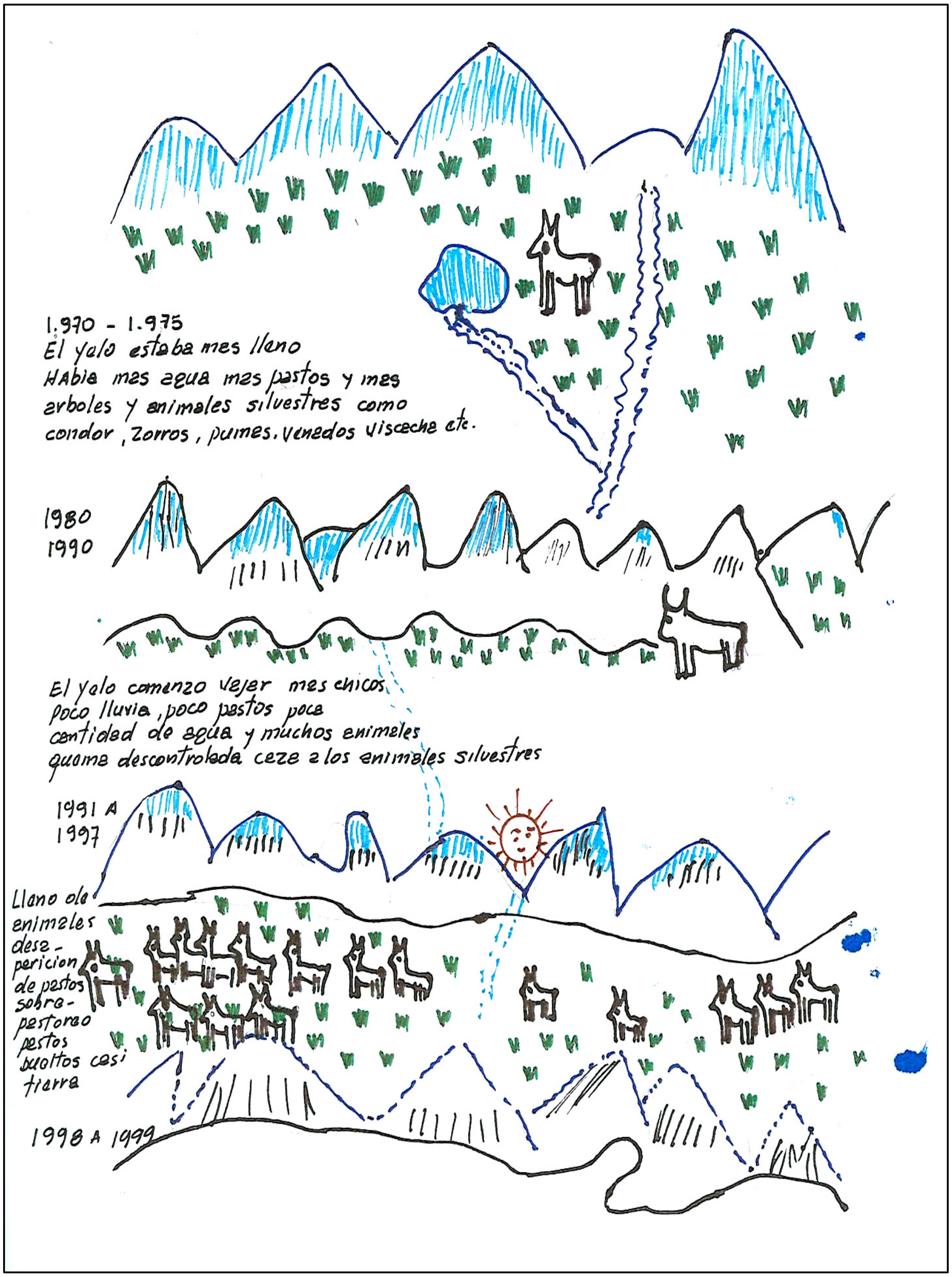
Given the rapidly changing context in which many communities find themselves today, we can not assume that all communities are keeping pace any more than we should assume that they will not. This study works from the assumption that communal institutions for resource management can be sustainable, while seeking to explore the conditions under which this outcome is unlikely.[6] The question then becomes, is conservation occurring now in the Ishinca valley? If not, why and how can we encourage it? This is a perennial question despite the rich literature on land use and resource management that has informed our understanding of the Andean landscape. A pictorial drawn by local informants in Collón suggests that a number of changes experienced in the region have acted synergistically to create environmental degradation in the HBR (see Figure 2.1). This includes climate change and glacial recession, increased numbers of livestock, and declining forage quantity and quality due to excessive numbers of livestock.

It is clear from the summary of the community's perception of environmental change in the Ishinca valley that these changes are not solely caused by local indigenous peoples (see caption to Figure 2.1). For example, glacial recession in the Cordillera Blanca is well-documented (Thompson et al. 2006). While the cause is indisputably the

[6] Researchers of Amazonian communities stress the need to explore the conditions under which conservationist behaviors are more likely to emerge, an emphasis due to the fact that many Amazonian communities have had historically low population densities and abundant resources which have not required the emergence of intentional conservation measures (Holt 2005, Lu 2001). I stress the need to explore conditions under which conservation can be maintained in Andean communities whose land use strategies and institutions have evolved in the context of relatively scarcity.

result of carbon emissions from more industrialized nations, the results are quite localized, affecting changes in plant species dominance and primary productivity which are vital resources on which residents of TY depend. Other changes (i.e., increased herds and changes in herd management that have resulted in a decline in the quantity and quality of forage) may be similarly traced to exogenous actors, specifically to the agendas of national development and global conservation. The aim of this chapter is to explore these possibilities. After describing the study site and methods of data collection, I will discuss: (1) the physical attributes, technologies, and decision-making arrangements associated with farming and herding land uses in the study community, (2) its integration into the market economy through tourism, (3) subsequent changes in household-level land use, and (4) the various ways that social, political and economic changes in the region potentially influence the existing *common property regime*'s ability to manage them sustainably.

Figure 2.1: Natural history of the Ishinca valley drawn by local informants of the sector of Collón. Translation of informant reports are as follows: (1) 1970-1975: glaciers were fuller; water, trees and wild animals were more abundant. (2) 1980-1990: glaciers started to recede; there was less rain and fewer productive pastures in addition to many more animals. (3) 1991-1997: problems of over-grazing and soil erosion start to appear; glaciers continue to recede. This figure was reprinted with permission from The Mountain Institute (TMI 2001a).



STUDY SITE

The study site is situated in the Cordillera Blanca of north – central Peru. This range serves as the continental divide between the west coast and the eastern Amazon basin, and it is the highest tropical mountain chain in the world.[7] The HNP was carved out of this magnificent mountain range in 1975, beginning at an elevation of approximately 4000 meters and extending upward to the region's highest peak, Mt. Huascarán, at 6768 meters.[8] Within its borders are some of the range's tallest summits (60 peaks over 5700 meters), extensive areas of *puna* (high altitude grasslands) and patches of *Polylepis* forests, endemic tree species locally referred to as *Queñual*. This research focuses on the region's high altitude grasslands, which are the dominant landcover of the HNP, as well as a critical resource for the region's indigenous agro-pastoral communities.[9] A population of approximately 226,000 (Byers 2000, INRENA 2002) indigenous peoples reside in a 2310 km^2 buffer zone surrounding the 3400 km^2 national park, in what is collectively referred to as the Huascarán Biosphere Reserve

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7 Thorough reviews of Andean geology are provided elsewhere (Allenby 1987, Brush 1982, Clapperton 1983, Dewey and Lamb 1992, Petford and Atherton 1992).

8 This 4000 meter mark is commonly reported as the park boundary. The actual boundary is determined by a series of survey markers that have been digitized onto a 1:100,000 scale map which creates a legal boundary that sometimes encompasses lower elevations, as commented on by Smith (1988). The significance of this discrepancy will be discussed later in the chapter.

9 The history of community control over this resource and access to it is complex, as discussed in greater detail later in the chapter.

(HBR). A good proportion of this population continues an agro-pastoral lifeway that depends on the *puna* resources in the buffer as well as in the national park.[10]

The HBR is a region of magnificent natural beauty, biodiversity and recreation potential. This is due largely to its geography and topography, as well as the rich cultural traditions and history of the region. The HBR encompasses an ecotonal transition between the bio-geographic provinces of *puna* and *páramo* (Brush 1982). This transition zone is sometimes referred to in the botanical literature as *jalca* (Luteyn et al. 1999) and is dissected by a number of inter-Andean valleys.[11] These valleys span a great altitudinal range, are an important resource for the indigenous agro-pastoral communities living there, and are the only routes to the region's climbing destinations. Agro-pastoralists cultivate lands in the buffer zone, but by government decree, maintain their rights to herd animals within the park. The use and traffic within the inter-Andean valleys of the HNP creates great overlap and potential conflict with its conservation objectives. Thus, a study of the long-standing practice of agro-pastoralism and its uncertain articulation with a relatively new tourism industry is paramount to the success of conservation efforts in this region.

The user-group of Ishinca valley is the focus of this study (see Figure 2.2). Tupac Yupanqui is an indigenous Quechua-speaking community granted legal status by the

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[10] Indigenous land use occurs within the buffer zone as well as within the national park, thus the study site will often be referred to as the HBR, while conservation efforts are more focused on the core of the reserve, which is the former national park (HNP).

[11] The region's residents simply use the term *puna* to designate the higher altitudes beyond the limits of cultivation. This term is used throughout the dissertation as it is consistent with local nomenclature.

Peruvian government in 1938. It is comprised of three sectors that operate somewhat autonomously in matters of land use and resource management. Each sector is a separate village, comprised of a plaza center surrounded by a number of *barrios* (neighborhoods) with varying numbers of households. Residents of Tupac Yupanqui's third sector, Joncopama, which is located farther to the north, utilize the resources of the lesser-impacted and lesser-visited valleys of Urus and Aquilpo. They do not share rights to utilize resources in the Ishinca valley and were therefore not included in this study.

The sectors of Collón and Pashpa are situated at the mouth of the Ishinca valley, and form the Ishinca valley user-group. Like many other valleys in the region, Ishinca is a v-shaped valley cut by a glacier-fed river. The valley bisects the Cordillera Blanca in an east–west direction, is approximately 10 kilometers long and up to ½ km wide at its head, and is surrounded by more than 5 peaks ranging in elevations from 6034 to 6274 meters.[12] It is a popular destination for many of approximately 157,000 annual visitors to the region (INRENA 1996, INRENA 2001, van Es 2002), given the number of accessible summits and the ease of access from Huaraz, which is little more than an hour away by car to the plaza of Collón.

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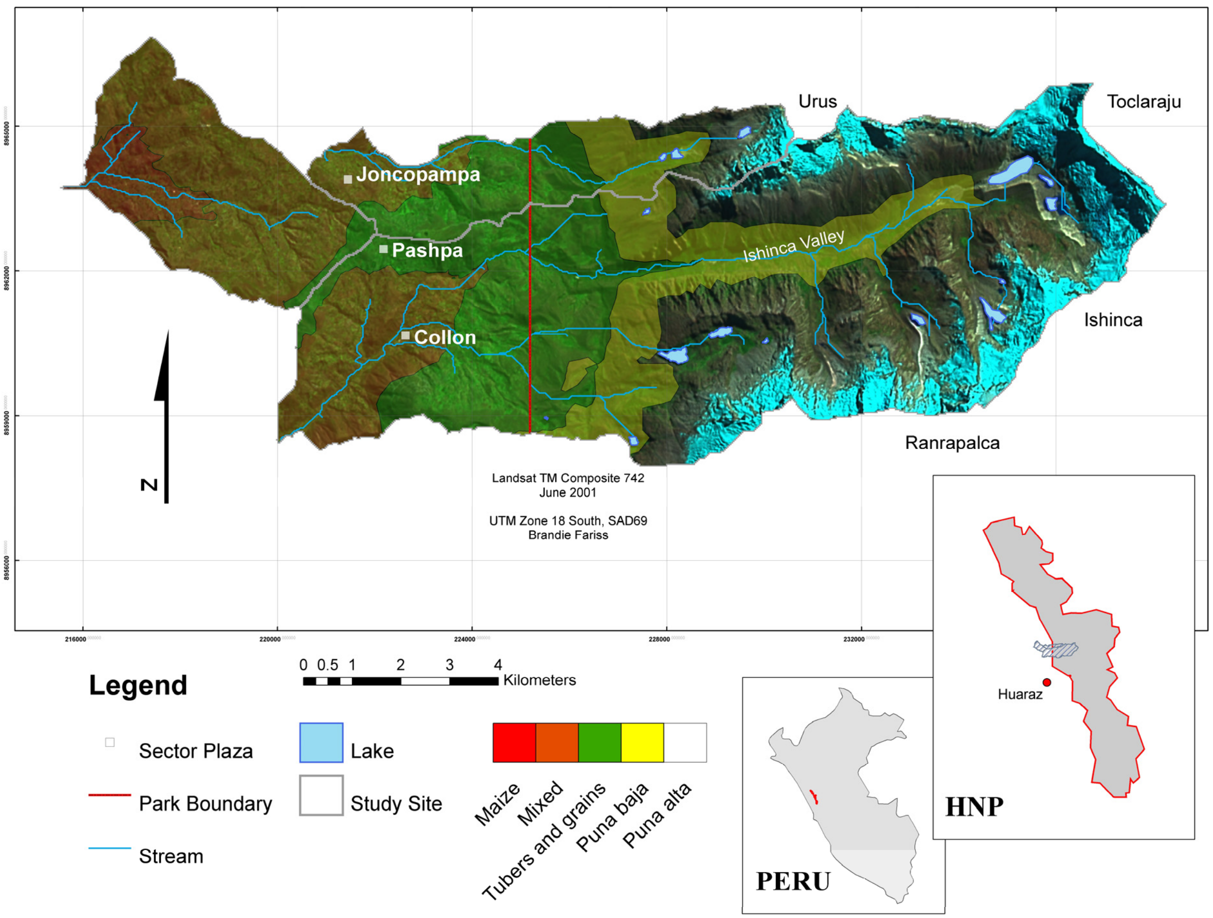
[12] Although several hanging valleys extend from the main valley and culminate in these peaks, the entire complex is often referred to as the Ishinca valley. Residents of Collón and Pashpa identify each hanging valley uniquely; as Chopi-Uran, Myoruri, and Paclaraju (from bottom to top). This entire network of grazing resources will be referred to simply as the Ishinca valley throughout the dissertation, except when discussing the results of vegetation sampling, where unique place names will be utilized (see Chapter 4).

Figure 2.2: Land use zones in the indigenous community of Tupac Yupanqui, emphasizing the extent of land managed by Collón and Pashpa. Land use categories are based on the agro-ecological ranges of central Andean land use utilized by Brush (1976, 1982). These have been overlaid on LandSat TM imagery for a crude approximation of the extent of various land use activities (i.e., doesn't consider actual land cover). Of special note is the tuber zone from 3501–4000 meters, which has been reduced by the creation of the park boundary. Effectively everything east of the park boundary is used only for grazing and limited foraging.



METHODS

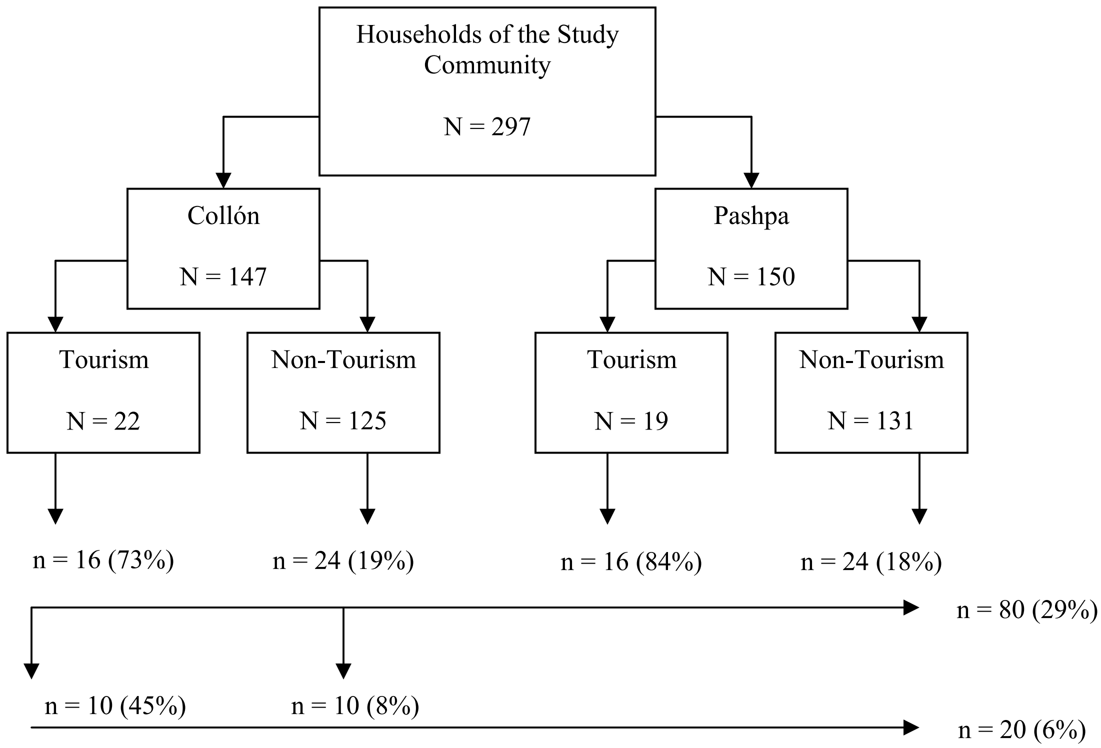
The bulk of the research was carried out during 18 consecutive months of fieldwork from July 2001 through December 2002. A wide range of methods were employed and a number of them generated the results reported in this chapter. Those not discussed extensively in subsequent chapters (i.e., time allocation in Chapter 3 and vegetation sampling in Chapter 4) will be summarized here.

In the first few months of fieldwork, I conducted a community census to describe the user-group for the Ishinca valley. The census included questions about household demographics, livestock holdings, and market involvement. This information was used as the basis for defining a sample of economically diverse households in order to explore the effects of tourism on household-level land use and communal resource management. Figure 2.3 diagrams the sampling strategy. A total of 297 households and 1474 individuals (mean household size = 5 stdev = 2) were registered by the census, from which 80 households and 393 individuals were ultimately drawn. First, all households recorded in the census were stratified by sector, then by their reported involvement in tourism. For each sector, 40 households were randomly drawn from tourism and non-tourism categories in a ratio of 3:2. This was done to include as many tourism households as possible in the 80-household sample, while maintaining a sample that was generally reflective of the overall community structure.[13] This resulted in a sample of

[13] The sample size of 80 was ultimately determined by the logistics of visiting households scattered throughout the community within a single day (as required by the time allocation design). I arrived at the 3:2 sampling ratio in an attempt to capture as many tourism households as possible for research questions

27% of the households using the Ishinca valley, representing 78% of all tourism households and 19% of all non-tourism households in the community.

Figure 2.3: Sample design reflecting the stratified random sample of 80 households (40 from each sector) for time allocation and monthly survey data collection, followed by the 'convenience sample' of 20 households for the comprehensive household survey.



Throughout 2002, time allocation and monthly surveys were conducted with the 80-household sample. Time allocation observations of every individual in the household were made every 6 days from January through December of 2002, using the spot-check method (Borgerhoff-Mulder and Caro 1985). This resulted in a detailed dataset with the

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requiring a comparison by household type (e.g., Chapter 3), while maintaining a more representative sample of the overall population.

ability to describe the general activity profiles of the community, as well as those of specific household types, genders and age groupings. In this chapter, the time allocation data has been summarized by gender to show the annual activity profiles of males and females in farming, herding and tourism. Monthly surveys were also conducted during the same time period with the 80-household sample. These surveys recorded changes in the household herd, monthly herding patterns, household income and purchases.

A number of other forms of data collection took place during fieldwork. A comprehensive household survey was conducted with a smaller subset of tourism and non-tourism households (n = 20) to gain greater detail on agricultural practices and yields, pastoral strategies and herd management, household involvement in tourism, wage-earnings and purchases, and perceptions about environmental change and park relations (see Figure 2.3). Surveys were conducted with the economic head of the household and with the help of local field assistants.[14] These quantitative methods of data collection were combined with ethnography, participant observation, and semi-structured interviews with key community informants such as *alcaldes* (mayors), and *empresas* (community-run businesses). Secondary research was also conducted in regional government and non-government offices. Collectively, these activities provided an opportunity to corroborate information gathered from diverse means and sources, and to gain further insights into the livelihoods of residents in this indigenous community.

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14 Comprehensive household surveys were not completed with all 80 households due to time constraints. The 20 surveyed were largely a 'sample of convenience', reflecting the households that I was able to interview before my departure. These households were exclusively from the sector of Collón, and represented tourism and non-tourism households in equal proportions, 10 from each.

MAINTAINING THE HIGH GROUND

What emerged during my fieldwork in the HBR is a realization that Andean livelihoods are achieved by hard work and ingenuity. This reality is at least partially shaped by the complexity and heterogeneity of the mountain environment. Indigenous communities of the HBR are situated along environmental gradients created by pronounced changes in elevation, thus they must cope with extreme variations in topography, climate and soils (Körner 1999, Troll 1968).[15] These factors affect the primary productivity on which highland peoples depend, and influence adaptations in subsistence and institutions for management that seek to optimize the success of living on high ground. A number of researchers have described the ecology of human-environment interaction in the Andes. These studies emphasize verticality, agro-ecological zonation, and the creation of micro-environmental mosaics of varying land use (Brush 1976, Lauer 1993, Mayer 1985, Murra 1972, Winterhalder and Thomas 1978). Other studies focus on the institutional arrangements governing these diverse land uses, and the importance of multiple scales of governance in highland resource use, including those of the individual, household and community (Brush and Guillet 1985, Flores Ochoa 1977, Guillet 1981, Knapp 1991, Mayer 2002, Young and Lipton 2006). Studies explicitly focused on communal institutions have contributed to this body of knowledge by elaborating on the importance of *common property* management in the highlands (Berkes, Davidson-Hunt, and Davidson-Hunt 1998, Campbell and Godoy 1992, Gilles

[15] A brief discussion of the biophysical characteristics of the study site is provided in Chapter 4, while more extensive reviews are offered elsewhere (de Ferreyra 1979, Fjeldså 1992, Smith 1988).

and Hahdi 1992, Netting 1997). Collectively, these studies have informed our understanding of contemporary highland subsistence strategies and communal institutions for resource management, as well as the biophysical, political, and social and economic contexts that shape them.

Photo 2.1: Adobe bricks drying in front of a *choza* in Collón (photograph by the author, 2002).

Photo 2.2: Woman of Collón spinning yarn while tending livestock (photograph by the author, 2002).

Photo 2.3: Young girls of Collón pausing for a photo upon return from school (photograph by the author, 2002).

Photo 2.4: Women of Collón preparing food for a community event in Collón (photograph by the author, 2002).

Lo Andino: The high altitude economy of Tupac Yupanqui

"Todo que tenemos es agricultura y ganadería.
Es lo que nos sostienen y lo que significa para sernos."

All that we have is agriculture and stockbreeding.
It is what sustains us and what it means to be us."

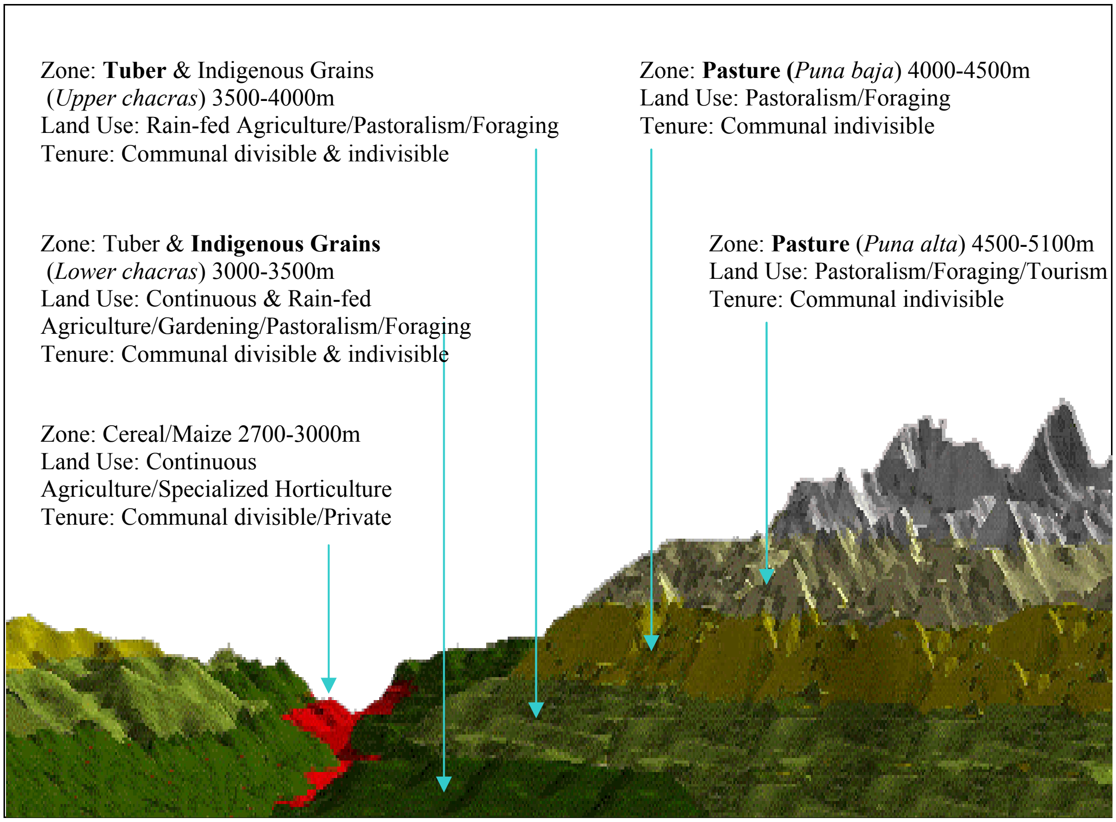
(a comunero of Tupac Yupanqui, sector Pashpa, 2002)

Mountains are often different from other environments, while noticeably similar to one another in the challenges they present for subsistence and the solutions that are manifest. A common solution to living in marginal, heterogeneous, and unpredictable environments is economic diversification (Valdivia, Dunn, and Jette 1996). Residents of Tupac Yupanqui, like many other highland peoples, practice a diverse subsistence strategy of farming and herding. Within these activities they diversify further by cultivating scattered fields (Goland 1993), selecting various crops and varietals (Zimmerer 1996), inter-cropping, and keeping a variety livestock (Kuznar 1991a, Kuznar 1991b, Mace and Houston 1989, Orlove 1980). Agro-pastoralism forms the backbone of the subsistence economy; while gardening, foraging, day labor, and the sale of agricultural surplus have traditionally been important supplemental activities. Moreover, as mountains gain increased attention as conservation targets, new opportunities to diversify the household economy have emerged with tourism. This is especially true of the indigenous communities in the HBR, whose actors have a strong influence in creating the contemporary expression of this Andean landscape. What Gade (1999) refers to as "Lo Andino," or that which is Andean, is increasingly shaped by peoples of the Andes and their widening interaction with people that are not of them, but that are very much interested in preserving what they believe to be their most worthy attributes.

Photos 2.1 through 2.4 and Figure 2.4, accompany the description of farming, herding and tourism in TY presented below. Although some form of land use occurs from approximately 3000 meters to the highest reaches of the Ishinca valley, agriculture extends only to the park boundary, which has effectively reduced the historical extent of cultivation by the community. Herding occurs on fallowed agricultural fields, the *puna baja* and the *puna alta* (high elevation grasslands), which extend into the park.[16] These activities are governed at multiple scales of organization and require coordination across multiple spatial and temporal scales (Browman 1983, Browman 1987, McCorkle 1987), which will be the focus of subsequent description.

[16] Households of the study community do not have extensive holdings (if any) in the "maize" zone (below 3000 meters). Land in this zone is held primarily by the lower communities of Paltay, Lucma and Tarica. Since these communities do not have access to the Ishinca valley, they are not heavily vested in herds, thus they are not considered in this study.

Figure 2.4: Generalized altitudinal profile of land use and land tenure in Tupac Yupanqui created from a cross-section of a 3-dimensional triangulated irregular network (TIN) of the study site. Agro-ecological zones and tenure arrangements are delineated according to the ranges utilized by Brush (1976, 1982) and Guillet (1981). Naming conventions in parentheses reflect the dominant cultivars or designations for each zone used generally by community members of the study site. This tiered model of land use and land tenure is only an aid for organizing the following discussion of socio-environmental patterns of land use and management in the study site, actual patterns are more complex (see Zimmerer 2003).



Farming in Tupac Yupanqui

Nearly all land on which farming, herding and foraging take place in Tupac Yupanqui is communal land. However, collective land use activities and true communal governance (decision-making) apply only to a small fraction of land in the community. Table 2.1 lists the names, sizes and characteristics of communal agricultural fields in Pashpa and Collón. Tenure for these communal *chacras* is indivisible, meaning that all *comuneros* (members) of that particular sector have equal rights of access to them.

Unlike other communal resources in the community, it is the *asamblea* (community assembly) and not the *comunero* that decides on their use. Each communal *chacra* is worked and harvested by those that wish to participate.[17] An agreed upon percentage of the yield is divided among participants, while the surplus is sold to generate capital for community development projects. The communal *chacras* in fallow are grazing resources within the community. During the year of fieldwork an estimated 18 hectares of communal *chacras* were in fallow, 12.5 in Pashpa and 3.5 in Collón. Given their indivisibility, these fallowed fields are an important grazing resource for the community's agro-pastoral households.

[17] In Collón some communal *chacras* are open to all *comuneros* while others are restricted to the residents of certain barrios.

Table 2.1: Communal *chacras* and their characteristics. Grazing is restricted to fallow periods and to *comuneros* of the sector in which the field is located. On occasion grazing is restricted even further by membership in the barrio where the field is located.

NAME	SECTOR	ESTIMATED AREA (HECTARES)	UTILIZATION HISTORY*
Cochacpampa	Collón	1	1
Garbansupampa	Collón	6	4
Panashpampa (a)	Collón	2.25	-10
Panashpampa (b)	Collón	.25	2
Panapampa	Collón	2	3
Rekrishpampa	Collón	2	-5
Ullocpampa	Collón	1.25	0
Subtotal:		**14.7 (47%)**	
Canish Pampa	Pashpa	6.25	-1
Hualcan	Pashpa	6.25	-1
Pamparco	Pashpa	4	0
Subtotal:		**16.5 (53%)**	
Total:		**31.5 (100%)**	

*Positive values reflect the number of years the field had been cultivated prior to 2002. Negative values reflect the number of years the field had been left fallow as a grazing resource prior to 2002.

Household Farmers

While communal farming exists in both sectors, household-level land use is far more prominent. The 297 households of Collón and Pashpa have informal rights (*usufruct*) to agricultural plots ranging from 3000 meters to the park boundary.[18] These rights can be transferred by *partible inheritance*, but sale or informal transfer to a non-*comunero* is forbidden.[19] Despite the lack of true forms of private property, households are largely responsible for the management of land they have been allotted. Guillet (1981) described this as a form of divisible communal tenure, while Campbell and Godoy (1992) describe similar systems in the Andes as *common property regime*s of sectoral fallow. Sectoral fallow is a complex system in which a shifting mosaic of communal grazing land is created on fallowed agricultural fields of the household.[20] In a typically sectoral fallow system, the timing and sequence of planting by a household is decided by the community. The institution for managing agricultural land use in Collón and Pashpa is more akin to a "multiple" and "irregular" form of sectoral fallow that emphasizes loose communal control over household decision-making (Campbell and Godoy 1992). The governing body of the *asamblea* controls the opening and closing of fields in different *barrios*, but its control over individual fields is superseded by household-level decision-making. Within the community-established openings and closings of different areas, it is typically the *jefe* (usually the adult male) of the household that makes decisions regarding agricultural production, while women and children are more likely to assert decision-making authority with regard to the work of herding and foraging.

18 These rights are only revoked when a change in household needs or levels of use for this site indicate redistribution of the land to other *comuneros* that may need it more.

19 Lease or sharecropping of usufruct holdings to other *comuneros* was reported by informants to occasionally happen.

20 Comparative study of sectoral fallow systems argues that such systems emerge under different technological and ecological constraints, but share the commonality of needing to articulate agricultural and pastoral production, both spatially and temporally (Campbell and Godoy 1992, Orlove 1991).

Outside of the area communally farmed, households participating in the comprehensive survey reported *usufruct* rights to an average of 2.7 hectares (min = 0.7, max = 20.0, stdev = 4.3) in 3.8 different agricultural plots (min = 3.0, max = 6.0, stdev = 1.1). Most of these holdings were unenclosed and scattered throughout the range of agricultural production. Throughout the duration of cultivation in a particular sector or barrio, the household's rights to use the *chacras* of this zone are recognized by all members of the community, and the management of this land is solely its responsibility.[21] Such responsibilities include determining what to plant, whether to use fertilizers and pesticides, and to an increasing extent, when and for how long to rest the field. Households in Collón and Pashpa cultivated a combination of the crops listed in Appendix 2.1; including native grains and tubers, as well as various legumes, fodder crops and introduced grains (see Photo 2.5).[22]

[21] The community determines the opening and closing of particular areas in which many households may have holdings. For example, during fieldwork, fields in the barrio of Cachijirca in Pashpa were closed to cultivation. This provides additional grazing resources for the community.

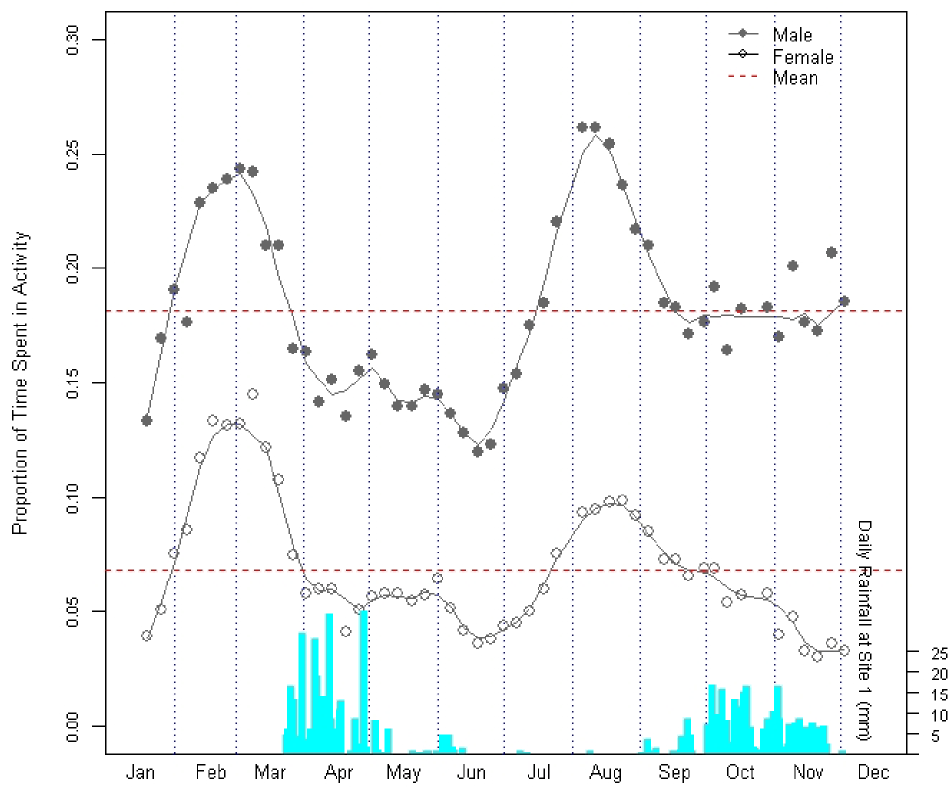
[22] *Huertos* (household gardens) are also cultivated to provide vegetables and various edible and medicinal herbs for household consumption. These gardens are typically planted near the house and do not fall under any form of communal control.

Photo 2.5: *Choza* with a *campaña grande* planting of potatoes in the foreground (photograph by the author, 2002).

There are two annual plantings for these crops, one in the *campaña grande* (wet season) and the other in the *campaña chica* (dry season). Some households cultivate in both, although the majority reported that that they only plant in the *campaña grand,* as many fields are without irrigation. *Campaña grande* planting is a time of high labor demand that typically occurs before the onset of rains in late August or September (see Figure 2.5). Harvest typically occurs in February and March of the following year. The specific timing of planting and harvest is determined by a number of factors including the household's available labor, its environmental knowledge, and more general cues discussed amongst community members, such as the appearance of the "Siete Cabrillos," the constellation of Pleiades (Orlove, Chiang, and Cane 2000). Regardless of the specific timing of these activities, agriculture requires sufficient labor. Household labor is obviously most important, but it is not uncommon for a household to host a *minka* (reciprocity-based communal work party) during peak times in the agricultural calendar.

Results of the time allocation study illustrate the demands of farming on individual labor. Figure 2.5 shows the annual activity profile of males and females for household-specific agricultural tasks. The increased demands of planting and harvesting in the *campaña grande* are evident for both sexes, although the relative proportion of time spent in this activity is consistently higher for males. On average, males spend approximately 18% of their time in household cultivation throughout the year, compared to females who spend only 7%, largely because they must manage a number of other subsistence tasks.

Figure 2.5: Time allocation to farming by gender. Points reflect a moving average of the proportion of time individuals (male or female) were observed doing agricultural work. The moving window for the average was 7 observations, which span every day of the week. The trend lines are based on a lowess regression with a span = 0.1. Multiple climate loggers were positioned throughout the study site, and rainfall data begins in mid-March and extends through the end of November 2002. Daily rainfall values plotted on the right-hand axis were collected by a climate logger installed in the community of Collón at 3300 meters. These data are included to illustrate the increased labor demands of harvesting (February–March) and subsequent campaña grande planting (August) which occurs before the onset of heavy rains.



Herding in Tupac Yupanqui

Livestock have been critical to the region's subsistence and market economies. Further, since the development of the region's tourism industry, livestock have been the means by which *arrieros* carry the gear of visiting mountaineers. Throughout the history of the Andes, domesticates have provided benefits to the household, primarily in the form of meat, wool and hides.[23] Perhaps most importantly, they make valuable contributions to agriculture, including the provision of manure, draft labor and transport of agricultural products to market. Considering the fertilizer contributions of manure alone, agriculture in the marginal soils of the high altitudes is heavily dependent on animal husbandry (Winterhalder, Larsen, and Thomas 1974).

Like agriculture, pastoralism is governed at multiple scales of community, *empresa* and household. Each of these actors relies on communal resources, either those of the fallowed fields in the agricultural zone (between 3000 to approximately 4000 meters in altitude) or those of the *puna* (approximately 4000 meters to snowline). The high altitude grasslands of the *puna* are an indivisible *common pool resource*.[24] The boundaries of this resource are delineated on the Map in Figure 2.1 (see pg. 36), and coincide with the watershed of the Ishinca River and its major tributaries.[25] Lands within the park boundary are utilized jointly by Collón and Pashpa; while lands outside, north or south of the Ishinca River respectively, are utilized by one or the other. The topography of the Ishinca valley effectively excludes the sector of Joncopampa and establishes *comuneros* of Collón and Pashpa as its user-group; with rights, responsibilities, and rules

[23] Milk consumption is not as common, but it does occur. Informants report that they will occasionally fill a *tasa* (cup) for consumption in the field. In addition, other communities in the region have begun small-scale cheese-making operations with the help of local NGOs.

[24] This is a condition that separates the common property regime from open-access or state property, a pre-condition defined by Stevenson (1991) and one that creates the possibility of sustainable use by a group.

[25] Several informants assisted in delineating the boundaries of the common property network utilized by Collón and Pashpa. While the grasslands in this environment are not enclosed, members of the community recognize unique locales. These locales (pastures) were geo-referenced with a GPS unit during hiking trips into the Ishinca valley with local informants. From them a sample was drawn for analyses of grazing impacts on biodiversity, as presented in Chapter 4.

for managing this resource. Because of the spatial extent of herding, the actions of these various actors directly affect the Ishinca valley and the park's goals of biodiversity conservation.

Community and Empresa Herds

Pashpa and Collón each have a number of *vacas silvestres* (community-owned wild cattle) that are in the HNP year-round. According to a semi-structured interview with community officials, there were 179 of these animals, with the majority (76%) owned by Collón. Throughout my time in the community I observed that most were located in the upper reaches of the Ishinca valley and in the hanging valleys of Chopi Uran and Miyu Ruri, while a few from Pashpa were reportedly in the neighboring valley of Urus.[26]

In addition to these animals, a number of community-based organizations owned and managed native camelid livestock that frequently grazed in the HNP as well. Table 2.3 lists the livestock-holding *empresas* of Collón and Pashpa, including ASAM (Asociación de Servicios de Alta Montaña), EcoAgro and the Alpaqueros. Collectively, these *empresas* owned 227 native camelids associated with a variety of development projects. These projects included an ecotourism venture utilizing llamas as pack animals (ASAM), and a commodity production venture involving the rearing of alpaca for wool and meat sales in the Huaraz market (EcoAgro and Alpaqueros). These animals were

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[26] This valley was not included in the study, but is utilized jointly by *comuneros* of Pashpa and Joncopampa. Joncopampa is the third sector of Tupac Yupanqui. It is not a user-group for Ishinca valley, thus it is not included in this study.

maintained on communal pastures of the HBR and members of the group were responsible for their management. Their use exclusively involved use of pastures in the *puna baja* and *puna alta* within the boundaries of the HNP.

Table 2.2: Composition of *empresa* herds in Collón and Pashpa in 2002. The majority of camelids pastured in the Ishinca valley are held by these organizations. Note that far fewer camelids are owned by households. A single household owns the 24 llama shown below, and 15 households owned the 41 alpaca. This situation changed when a decision made by the community during the year of fieldwork resulted in the division of EcoAgro's alpaca herd among its member households. This decision proved to be an unfortunate one that resulted in the death or sale of most of these animals within months of the division.

ORGANIZATION	LLAMA	ALPACA	TOTAL
ASAM	19	0	19
EcoAgro	0	110	110
Alpaqueros	0	98	98
Empresa Subtotal:	*19*	*208*	*227 (78%)*
Household Total:	*24*	*41*	*65 (22%)*
Total:	**43**	**249**	**292 (100%)**

A notable exception to this rule occurred during fieldwork in 2002, when household and *empresa* interests diverged over the management of EcoAgro's alpaca herd. This Collón-based *empresa* was started with the assistance of The Mountain Institute (TMI), and managed its herd for several months of the year within the HNP, often on Winac Pampa (see Photo 2.6).[27] Member households were responsible for herding and corralling the animals on a 2 week rotation that required their presence in the *puna* for an extended period of time. In 2002, EcoAgro members voted to divide the alpaca herd and meld their management into the day-to-day herding activities of the household. The dissolution of communal coordination and specialized management of alpaca in favor of household-level management resulted in the loss of the majority of the original herd over the duration of my fieldwork, as many grew ill or were sold within the year.[28] An inability to maintain *empresa*-wide coordination may be symptomatic of increasing market involvement and economic diversification among its member households. Such situations create labor shortages and hinder the ability to negotiate scheduling conflicts. This was apparent when a member of EcoAgro once commented that it was inconvenient to be responsible for these animals during the peak tourism season. The inability of the *empresa* to sustain coordinated management of the alpaca herd throughout the tourism season suggested to me that the households of the study community may experience similar problems with the management of their own herds as their involvement in the market grew. This is an issue explored in greater detail in Chapter 3.

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27 ASAM and the Alpaqueros maintained their llama and alpaca herds in the *puna*, but with less coordination in their management. In general, it was reported that these animals were only occasionally checked.

28 My fieldwork assistant, who served as a livestock technician and frequent veterinarian for the community, explained that the alpaca were prone to disease and infection, especially when kept at lower altitudes. Because of the common practice of herding diverse types of livestock together, household decisions with regard to where to pasture the herd may not have been optimal for the newly acquired alpaca. Others simply opted to sale their share rather than dealing with the additional herding demands created by these animals.

Photo 2.6: The alpaca herd of EcoAgro, together with a few household animals on Winac Pampa (photograph by the author, 2002).

Household Herds

Given the predominance of grasslands in the park, and the relatively large number of non-native livestock owned and managed by the household, household herders (often women and children) are the ultimate decision-makers affecting conservation in the HBR. Households generally own a variety of livestock including cattle, horses, mules, llamas, alpacas, sheep, goats and pigs.[29] Over 4,014 livestock were recorded in the 2001

[29] Differences in herd size, composition, and management are pronounced among households. Discussion of these differences is provided below, and is an important factor in the analysis presented in Chapter 3.

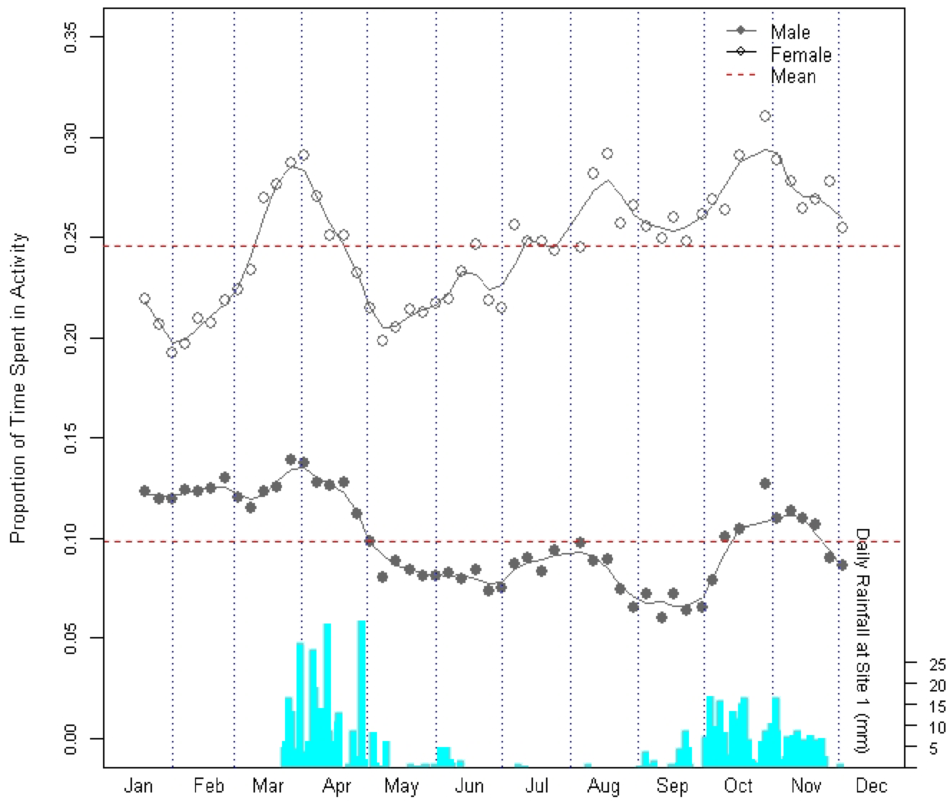
census.[30] Mean herd size for the household was 13 (min = 0, max = 190, stdev = 17), with 75% of the average household herd composed of sheep (75%).[31]

An annual activity profile of household herding labor (shown in Figure 2.6) reveals the importance of this activity. Most households with draft animals utilize them during the labor intensive tasks of field preparation and harvest. The household herd is kept near the community during these times to minimize spatial disjunctions in agriculture and herding (McCorkle 1987). The increased labor demand of herding animals in the community, while keeping them off of cultivated fields is evident in the activity profile of February and March. Herding labor is reduced when crop stubble from the *campaña grande* harvest is grazed in April and May. Once these resources are exhausted, women's' herding labor increases in a trend consistent with dry season *transhumance*; a herding strategy involving the movement of animals to higher pastures as the dry season progresses. On the other hand, men's' herding labor trends downward over the same period. Overall, the labor invested in herding livestock by both men and women is pronounced. Females spend an average of approximately 25% of their time herding throughout the year, significantly more than any other subsistence task, while men spend nearly 10%. Compared to the statistics of time allocation to agriculture, men and women collectively spend more time herding.

[30] The possibility for under-reporting household livestock exists in most pastoralist communities. I was able to cross-check and update these totals during my visits to the household for time allocation observations and monthly surveys.

[31] Pigs were not counted in the livestock census as they are kept at the household or staked near the house and do not rely as much on natural forage or use of the Ishinca valley. Most households had a number of pigs, guinea pigs and chickens which are not represented in this count.

Figure 2.6: Time allocation to herding by gender. Graph characteristics are identical to those of Figure 2.5. Rainfall data illustrate the subtle effects of the dry season on female herding labor, which increases gradually over the months of May through August as many (but not all) herders range farther from the community and higher into the Ishinca valley to seek out suitable forage for their livestock.



Tourism: A fair-weather industry

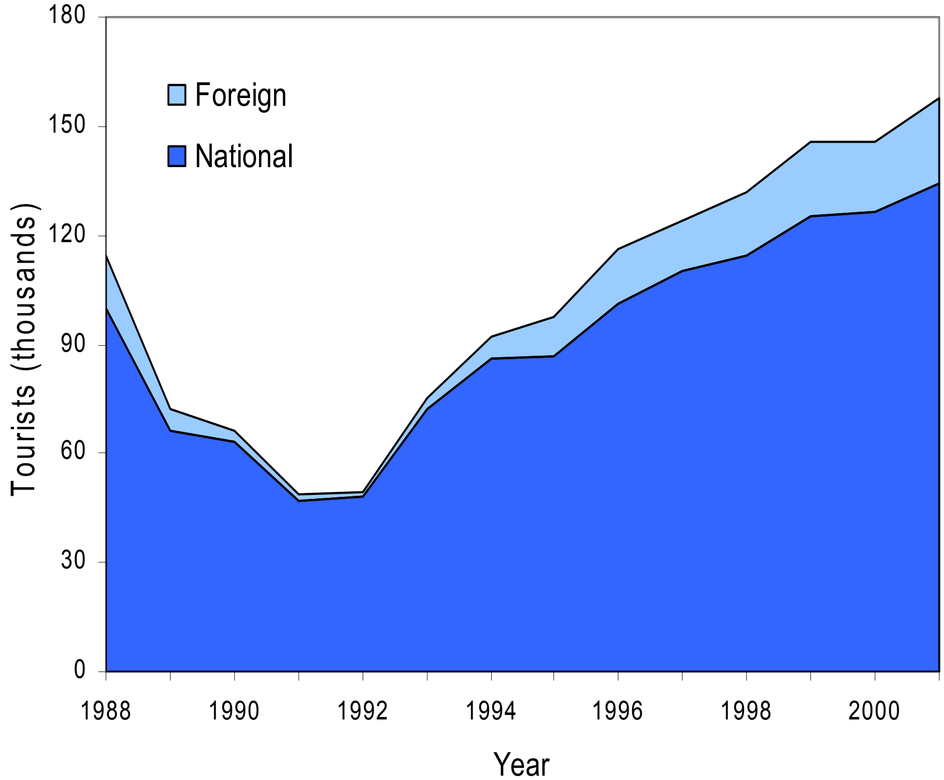
"Yo era el arriero primero para trabajar con turistas.
Acá ellos vienen todavía; algunas veces más y otros menos.
Trabajo cuando llegan, pero necesito mis chacras y mi ganado el mismo.
Ellos me apoyan, siempre están. No es así con las turistas."

"I was the first arriero to work with tourists.
They come here still, sometimes more and others less.
I work when they arrive, but I need my fields and livestock the same.
They support me, they are always here. It is not so with the tourists."

(a comunero of Tupac Yupanqui, reportedly the first arriero, Sector Collón, 2002).

Collectively, mountain tourism makes up approximately one-fifth of the world's tourism revenues (Lynch and Maggio 2000). Travel to the Andes, in particular, has grown substantially in recent years (Mitchell and Eagles 2001). According to statistics published by the region's tourism authority (summarized by van Es 2002), the number of visitors making overnight stays in Huaraz had steadily increased to over 157,000 individuals by 2001. The majority of these overnight stays are Peruvian nationals for which the region serves as a holiday retreat. The economic impact of these tourists is pronounced in Huaraz where money is spent in local cafés, hotels and touring agencies that make day trips into the park's most popular valleys such as Llanganuco and Pastoruri. Yet, a very small proportion of the revenues they generate, if any at all, reach the park's surrounding indigenous communities. It is the lesser but still significant foreign tourists, a group of approximately 23,000 individuals, that are more likely to spend extended periods of time in the region for climbing and trekking, venturing to outlying communities such as TY.

Figure 2.7: Foreign and national tourists making overnight stays in Huaraz from 1988 to 2001. Notice the pronounced decline in tourism during the late 1980s and early 1990s, a time period that corresponds with Shining Path activity in the region. This trend illustrates the fickleness of the industry and its sensitivity to the political climate, discussed in greater detail below.



Many adventure tourists to the HNP utilize the services of *arrieros* from its surrounding indigenous communities.[32] *Arrieros* carry the gear of visiting mountaineers to base camp with the assistance of pack animals, primarily mules and horses. This

[32] The national park registry documents a total of 109,063 entrants to the park in 2000; 95,446 national and 13,617 foreign tourists. Of these, they recorded 4,424 adventure tourists (INRENA 2001). This is likely an underestimate because tourism checkpoints are located at only a few valleys in the entire park, including those of Llanganuco and Carpa (Pastoruri). During fieldwork in 2002, a tourism checkpoint was formally established at Wiliac, a barrio of Collón that most tourists pass through before beginning their trek through the Ishinca Valley. This manned registry will assist in getting better estimates of the traffic through this particular valley, although one can assume that it gets a good proportion of the foreign tourist population due to its popularity as an acclimatization valley.

service brings a lucrative wage to those certified to do the work, with rates of 10 USD/day for the driver, plus an additional 5 USD/day for every animal carrying weight. By comparison to the average daily wage of 2–3 USD/day for the work of a *jornal* or *peon* (day laborer), this income is quite significant. In 2002, there were 46 *arrieros* in the sectors of Collón and Pashpa registered to work with tourists, reflecting the Ishinca valley's popularity for climbing and trekking. Other occupations in the tourism industry for which individuals were actively pursuing certification include cook, porter and guide. These occupations require additional certifications and command higher wages of 25 USD/day for porters and cooks, up to 50–150 USD/day for guides.[33] The majority of households had 1 *arriero*, usually the oldest male and economic head, while the maximum number of *arrieros* within a single household was 4 (a father and his 3 eldest sons). In total, 40 different households (14% of the community) had the ability to earn wages through tourism, allowing for their departure from the other 76% of the primarily subsistence-oriented households in the community.[34]

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[33] Rates were provided by the Casa de Guias of Huaraz (*pers. comm.* 2007). The large range in guiding salary reflects the level of difficulty associated with different climbing pursuits in the region. The base rate of 50 USD/day is for trekking, whereas 120 USD/day is typically charged to guide Alpamayo, a 5947 meter technical climb in the HNP.

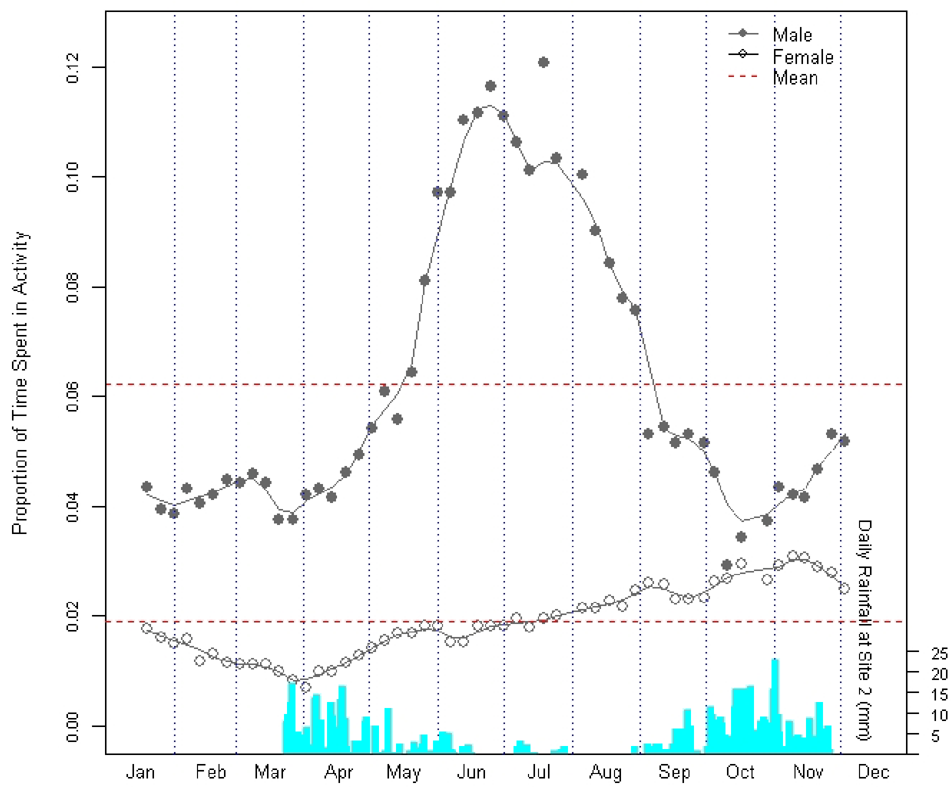
[34] Community members living in the sector of Pasha often commented that that they did not get equal access to tourists entering the Ishinca valley because many usually begin their hike in Williac and bypass them altogether. Many certified to do *arriero* and portering work from Pashpa circumvented this problem by establishing working relationships with trekking agencies in Huaraz. A few individuals rented apartments in Huaraz during the peak climbing season where the odds of contracting with tourists or guiding agencies were greater. During the fieldwork year there was some attempt among the community's *arriero* association to create equal access to tourism opportunities. A rotation schedule was created so that tourists entering the plaza would register at the park's newly created post in Williac, then would contract the next *arriero* in line if their services were required. This plan provides evidence of an attempt by the community to adapt rules for governing access to wage-earning opportunities and creating a sense of fairness. Yet, this plan was only moderately successful, as many times I observed that the *arriero* next in line would not be around in the plaza at the time that tourists arrived, and tourists would go with whomever was available.

While a stark contrast to other wage-earning opportunities, tourism offers little reprieve from the uncertainty inherent in the highland economy. Even the 14% of community's households that are involved in tourism identify themselves, first and foremost, as farmers and herders. As the quote from Collón's eldest *arriero* suggests, community members recognize the sporadic opportunity of tourism and the necessity to make a living from the land. Adventure tourism, while a significant source of income for some households, is insufficient to discourage their dependence on resources within the national park. First, tourism is subject to boom-and-bust cycles reflecting the influence of economic and political processes acting on tourist decision-making. For example, Shining Path activity in the late 1980's and early 1990's discouraged tourists from visiting the region, a trend that can be seen as a dip in registered overnight visitors to the region in Figure 2.7. Although this activity was effectively squelched with the 1992 capture of Abimael Guzman, the political climate of Peru is frequently a deterrent for international travelers whose numbers have recovered slowly since (see Figure 2.7). More recently, protests following on the heels of political events (i.e., a country-wide teachers' strike and subsequent declaration of a "state of emergency" in the department of Ancash) resulted in a noticeable decline in tourism in 2003. According to a local café owner in Huaraz, many believed that the larger political and economic realities affecting the price of oil in 2006 had a bearing on that year's tourism travel as well (*pers. comm.* 2006).

In addition to the larger trends in tourism, the industry also is seasonal at best. Climbing and trekking are exclusive to the dry season. During the dry season months of June through August, *arrieros* from Collón and Pashpa linger in the plaza of Williac (a *barrio* of Collón situated near the entrance into the Ishinca valley) to encounter climbing parties that require their services. Others travel to Huaraz to seek out work by asking tourists walking on the streets, or by contracting directly with one of the many trekking and climbing agencies based there. The success of these efforts is obvious in the average activity profile of wage earning shown in Figure 2.8. This profile illustrates the seasonality of wage earning activities such as tourism, which occupies a significantly greater proportion of males' time during the dry season months when *arriero* work is at its peak.

Although this work is seasonally pronounced, the lesson here is that tourism fluctuates due to a multitude of forces external to the communities that depend on it. Because of the prerequisite of favorable weather for climbing and trekking, it is quite literally a fair-weather industry for residents of TY. Although visitors to the Ishinca valley are many, and the time spent in *arriero* work pronounced, there is only a brief seasonal window over which such opportunities exist. Collectively these conditions result in an industry that does not supplant the traditional agro-pastoral economy, perhaps one that even enforces it; evidence of this at the household-level will be provided below.

Figure 2.8: Time allocation to wage-earning by gender. Graph characteristics are identical to those of Figure 2.5. Rainfall data illustrate the correlation between peak wage-earning activity for men, and prime climbing and trekking weather, which occurs primarily in the dry season months of June through August.



Changes in Household Farming Practices

While households are relatively similar with regard to their crop selection and reliance on rain-fed agriculture, they are more varied in their use of chemical inputs and in the length of time they cultivate a particular plot (Mayer 2002, Zimmerer 2002). Several local informants reported that an increasing number of households are cropping their holdings continuously, a condition for communally coordinated sectoral fallow systems described as a “zero-fallow problem” (Campbell and Godoy 1992). Until recently, it was reported that households would cultivate fields for an average of 3 years before removing them from agricultural production for several more. Figure 2.9

illustrates a typical crop rotation sequence elicited from community informants. The first year of planting is represented in the outer circle, with successive years moving inward. The possibility of fallow is indicated in the innermost circle. If fallow does not occur, the cropping sequence simply starts over. The results of monthly surveys of household purchases support informants' claims of agricultural intensification. Of the 80 households sampled, 51 (64%) made fertilizer and pesticide purchases at least once during the year of observation. Of those 51, 45% made multiple purchases, as many as three times throughout the year (see Photo 2.7).

Photo 2.7: Field assistant, Juan Sanchez Duran, stopping to observe a recently sprayed field of potatoes (photograph by the author, 2002).

The factors commonly cited for reductions in fallow include changes in population density and economic development, especially market penetration (Boserup 1965, Guillet 1987), Collón and Pashpa have experienced both. The establishment of the park boundary in 1975 pinched the arable land base in many of the region's indigenous communities. By doing so the existing population was forced to generate adequate production from a smaller area, much of this has been achieved by reducing fallow as noted above. This is especially true in Tupac Yupanqui where the park boundary dips below 4000 meters and excludes land where agriculture might, on occasion, be practiced (see Figure 2.2). While *comunero*s consider cultivated fields as their property, they view these same fields when fallow as "libre" or free to all members of the community for grazing. 'Zero-fallow' or continuous cropping is a problem in TY because it occurs at the expense of having land available for grazing in the community. Of the 51 households that purchased agricultural inputs to achieve continuous cropping, more than half were tourism households. The loss of fallow land in the community creates spatial disjunctions in farming and herding, as well as the potential for a number of other environmental problems in the HBR.

Returning now to the community's perceptions of declining forage quantity and quality (see Figure 2.1), an argument can be constructed that increased grazing pressure in the park may be linked to agricultural intensification and the loss of grazing resources in the community; and that agricultural intensification, in turn, may be linked to the constraints introduced by the positioning of the park boundary, and the increase in wage earning opportunities in tourism making increased investment in chemical inputs possible. Of course, this may only be part of the picture. The following discussion of changes in household herding will describe how tourism, a policy promoted by national and international conservation organizations, affects changes in herd size, composition and management that contribute to the possibility of over-grazing and environmental degradation in the HBR.

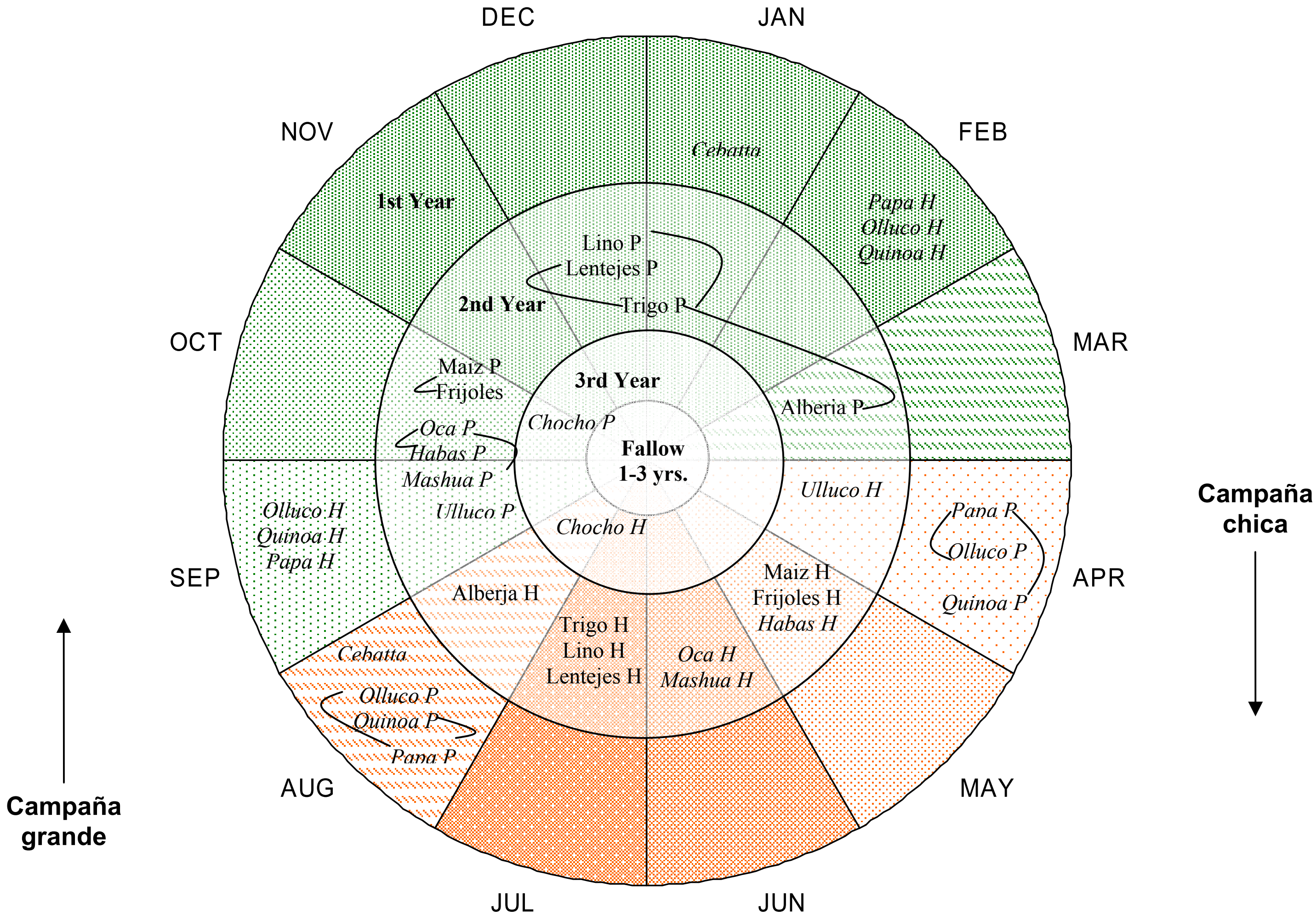


Figure 2.9: Agricultural calendar and crop rotation sequence reported by community informants. Italic face represents crops grown in upper chakras; normal face represents crops grown in lower chakras. Connectors indicate intercropping combinations. Months in green correspond to the campaña grande; orange the campaña chica.

Changes in Household Herding Practices

The park and the community comment that there are simply more animals now than in the past. This perception is echoed by community informants in Figure 2.1 (see pg. 32), as well as in a recent reports by the park and local NGOs (INRENA 2002, TMI 1996a, TMI 1997, TMI 2001a). Case studies in the highlands, in locales as diverse as Bolivia, Peru and the Himachal Pradesh, have documented herd increases or changes in composition due to market involvement (Chakravarty-Kaul 1998, Kuznar 1991b, Orlove 1977, Preston 1998). Tourism in the HBR may have similar effects. Owning pack animals is a necessity for *arriero* work. Once households can perform this work they may reinvest their wages in the agro-pastoral economy; the one that is "always there" according to the community's first and oldest *arriero.*

It was previously shown that households with tourism involvement made more purchases of agricultural inputs (i.e., chemical pesticides and fertilizers) than non-tourism households. Monthly surveys also documented the purchase of 127 livestock of various types by 13 different households (16% of those surveyed). A single tourism household was responsible for the majority of these purchases (75%), which included horses, mules, llamas, alpacas and pigs. Although relatively fewer households purchased livestock than chemical inputs for farming, the comprehensive survey conducted with a smaller subset of households (n = 20) suggests that all would like to make more livestock purchases. These twenty households reported a desire to increase their packstock holdings (horse and mule) by an average of 10 times that of the existing household mean (1, stdev = 1.4). This was followed by sheep at 6 times the existing mean (10, stdev = 15.8), and cattle at 4 times the existing mean (3, stdev = 3.0). The higher preference for large stock is consistent with the increased utility of horse and burro given the nature of tourism work, and the increased utility of cattle that can be left in the *puna* unsupervised or staked in the community and supplemented with fodder. While the preference for sheep may be a reflection of their ease of sale or slaughter when required by the household. In fact, all households consistently reported that they would like to have more of every type of animal with the exception of those that are native to the Andes. Interestingly, the average ideal number of native camelids (llama and alpaca) reported by sampled households did not differ appreciably from the existing mean.

Although these statistics reflect household preferences and not realities, households actually working in tourism seem to have realized these ideals. Tourism households that self-reported *arriero* work in the 2001 census had a significantly greater number of stock equivalents than their non-tourism counterparts. *Arriero* households had an average of 36 stock equivalents compared to 23, a significant difference according to a two-sample t-test with 95% confidence ($p > |t| = 0.05$).[35] Although causality is difficult to establish (e.g., do more animals create tourism work, or does tourism work create more animals), all interviewees reported that there were more animals now than in the past, a perception confirmed by the pictorial in Figure 2.1 (see pg. 32).

[35] Given the variation in herd composition, this comparison is based on stock equivalents reflecting the relative forage requirements of each type of animal. I derived a stock equivalent scale from unpublished documents utilized by TMI (Sotomayor 2000). I set the scale relative to sheep, which require ¼ hectare of good quality pasture to support a single individual per year. Thus the stock equivalent scale reflects the annual requirements of each animal relative to sheep. Sheep = 1.00, Cattle = 4.00, Alpaca =1.48, Llama =2.22, Horse and mule = 3.00. Appendix 2.2 summarizes the stock equivalents for each type of animal in the household herd, which are used throughout the dissertation.

While, an ever-increasing number of cattle was Hardin's metaphor for collective ruin (1968, McCay and Acheson 1996), stocking rate alone is an insufficient determinant of the potential for environmental degradation. Many studies in range management stress the importance of their spatial and temporal management as well (e.g., Adler and Morales 1999, Turner 1998). This issue will here be examined in greater detail, and in Chapters 3 and 4. Different household herding strategies were observed in the community; some were relatively recent according to local informants. Table 2.4 describes three distinct practices. These strategies were not mutually exclusive, as households may combine a number of them to manage different livestock, although most herded the majority of their holdings together, thus one strategy often dominated within the household.[36]

[36] The dominant strategy of the household is influenced by herd size and composition; the availability of fodder; household labor, and the establishment of reciprocal herding networks among neighbors and kin.

Table 2.3: Household herding strategies, their relative labor demands and reliance on the HNP. For labor demand, the number of plus signs (+) indicates the amount of labor required for that particular strategy relative to the others. Three (+++) is the highest. For use of the HNP, wet and dry season are shown separately, with a negative sign (–) indicating that park pastures are not generally used, and a plus sign (+) indicating that they are.

HERDING STRATEGY	PATTERN OF USE	RELATIVE LABOR DEMAND	USE OF HNP
Compressed transhumance	Seasonal movements between community and park pastures	+++	wet - dry +
Community-based herding	Year-round concentration on community pastures	++	wet – dry -
Absenteeism	Year-round concentration of untended animals on park pastures	+	wet + dry +

Transhumance is a common practice reported for middle and high altitude agro-pastoral communities, and is the most prominent and long-standing strategy in The study community (Browman 1984, Guillet 1983, McCorkle 1987, Stevens 1993). After the productivity is wrung from crop residues, many households rely on resources in the *puna* to support their herd.[37] This strategy historically involved movements into the *puna* in the dry season from a second seasonal residence usually higher than the one occupied by the household for the remainder of the year (see Photo 2.8). I refer to the contemporary strategy as 'compressed *transhumance*' because households are no longer permitted to have a residence within the park, although many were historically in the contested space

[37] Households typically save the stems of *trigo* (wheat) for such purposes. This fodder is called *paja*, the amount of which depends on the size of the area cultivated and the yield. Natural forage species are also collected on nearby communal lands to support the household herd.

shown on the map in Figure 2.2.[38] The result is that households may utilize some of the lower altitude pastures of the *puna baja*, but many of the highest ones in the *puna alta* are beyond the reach of a typical herding day, which often begins around 8:00 a.m. from the residence and ends around 4:30 p.m.[39] Pastures in the *puna baja*, many of them *bofedales* (wetlands and critical dry season reserves), would normally be less than a 1-hour walk from the plaza. As the dry season progresses, households practicing this strategy make longer forays, sometimes into the *puna alta*. Pastures in this zone are typically a 1–5 hour walk from the plaza, thus this strategy requires the greatest investment in herding labor relative to the others.

[38] The fact that high altitude *estancias* and corrals are not permitted in the park is a sore subject with many residents, as many commented on the presence of "non-community" structures in the park, such as tourism lodges at the base camp of Ishinca and Pisco valleys that are owned and operated by the Italian Catholic Church.

[39] While this may seem favorable for the park, this does not mean the highest altitude sites were not utilized. Animals were often present but not herders. The implications of absentee herders are explored throughout the dissertation.

Photo 2.8: A *choza* located at the upper limit of cultivation and just inside the formal boundary of the HNP (photograph by the author, 2002).

Strategies that reduce herding labor in the dry season were evident in the study community. Community-based and absentee herding were practiced by many households. Both of these practices require less labor than compressed *transhumance*, and had different implications for the use of the park. During my initial observations in 2001, I noticed that some households herded their animals in the community more than others. In the community-based herding strategy, households would pasture or stake animals on nearby fields and supplement them with stored crop residues, fodder collected from nearby commons, or fodder purchased with wage earnings. In Collón, households could additionally purchase rights to improved pastures that were recently created with the assistance of TMI the year before fieldwork began. This project is an important one that reduces Collón's dependence on the park.[40]

While compressed *transhumance* and community-based herding typically exclude the use of the highest pastures in the Ishinca valley, these areas are not exempt from grazing. As mentioned previously, the *puna* is frequently grazed by communal and *empresa* herds. Moreover, it is increasingly utilized by some households. A third household-level herding strategy is absenteeism. At the time of census, 42% of households in Collón and Pashpa reported frequent use of pastures in the Ishinca valley. Although some households actively herded their animals, unmanaged herds were also observed in the Ishinca valley. This is reportedly a fairly recent occurrence, and one that coincides, among other things, with the region's increase in cattle holdings.[41] Some households left cattle to graze opportunistically, only occasionally checking on them (once a week on average in the dry season, less in the wet season). I observed many untended cattle on my frequent hikes into the Ishinca valley over the 2 years that I lived there. Although these animals could not be discerned (by me) to be from the communal herd or a household herd, I occasionally accompanied household herders on their trips to check animals left in the park. Because it is primarily done only with cattle, households might practice this strategy in combination with compressed *transhumance* or

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40 The risk of degrading communal lands in the agricultural zone, nearby woodlands, and shrublands still exists with this management strategy.

41 This strategy may additionally result from increased spatial disjunctions between dry season residences and high pastures, increased preferences for large stock requiring less supervision and protection from predation (by condor, fox or puma), and decreases in available household labor associated with more off-farm employment.

community-based herding of its remaining livestock. The fact that absentee herding is occurring at all, even if just for a proportion of the household herd, is likely of great concern to park managers.

MANAGING INDIVIDUAL USERS OF A COMMONS

A growing number of non-native livestock coupled with an increased presence of untended animals on sensitive sites in the reserve's core could be a tragedy for biodiversity conservation in the HBR. However, this outcome depends on how household users are mediated by the incentives and sanctions developed within the existing *common property* institution (Feeny et al. 1990, Feeny et al. 1998, Ostrom 1996, Runge 1986). Many characteristics of successful *common property institutions* are apparent in the study community, although they are largely implicit (for similar observations see (Lu 2001). First, rights of access and withdraw (operational rights) are clearly delineated (Schlager and Ostrom 1992). This is achieved in the steep inter-Andean Ishinca valley without difficulty. Collón and Pashpa are the only two sectors of the indigenous community of Tupac Yupanqui (TY) well-positioned to utilize the grazing resources within this valley, while the sector of Joncopampa and neighboring indigenous communities are effectively excluded by the steep terrain and narrow entrance. Beyond exclusion and withdraw rights, there were established regulations affecting how and when *comuneros* from these two sectors could use grazing resources in the Ishinca valley. Chief among these was a communally coordinated opening date for the *portada* (gate) of the Ishinca valley. These collective decisions provide evidence of an attention to resource conservation (1968). Yet, the community herd of *vacas silvestres* and a growing number of household cattle now appear to be violating this regulation without consequence. While this particular regulation may have been relaxed, informants indicate that conflicts among *comuneros* do arise. When they do, I was told, they usually involve livestock trespassing on planted fields, which is resolved by fine or other sanction decided upon by the community. Collectively these operational and collective choice rules are important characteristics of a functioning *common property regime* (McKean 2000, Stevenson 1991). However, their existence alone is insufficient to guarantee its sustainability. During the year of fieldwork I observed no instance of the community enforcing either of these rules, nor hear of any fines levied against defectors, although it reportedly happens on occasion. Lack of cooperation and enforcement is apparently commonplace in empirical studies of indigenous *common property* institutions (Bremner and Lu 2006). Reasons for the failure to sustainably manage diverse users are numerous and complicated, but this outcome is likely fostered by pronounced heterogeneity, mistrust and conflict. The remainder of this chapter will focus on describing the potential effects of social, political and economic changes taking place in the region; including those of park creation, tourism development, and changes in rural and urban population that may undermine collective action and the sustainable management of resources in the HBR.

The Antagonists of Successful Common property Management

An exploration of *common property* institutions and the factors moderating or potentially aggravating them is warranted given the concerns voiced about environmental degradation and biodiversity loss in the HBR. A number of case studies describing *common property* in a variety of environments have shown that such arrangements can encourage conservation, unlike the free-for-all commonly associated with *open-access*

(Berkes 1989, Feeny et al. 1990, Feeny et al. 1998). Other studies have documented the conditions under which these institutions are prone to success or failure (Bromley et al. 1992, Ostrom 1992). If members of a community can maintain cooperation and sustainable use of a resource held in common, which contexts trigger these variable outcomes? In a study of the Swiss alpine community of Törbel, Netting notes that the community had a rule for sustainably managing high altitude pastures which stated that "no citizen could send more cows to the alp than he could feed during the winter" (1976: 139). This particular rule was sufficient for achieving sustainable use of Törbel's grazing commons in 1517 when it was declared, but how would such a rule hold up in the contemporary socio-economic climate of many highlands? If Törbel's households, like many of the world's highland communities, used wage earnings to supplement winter forage, it is feasible that more cattle could be supported over the winter than could be sustainably managed on summer pastures. Here I argue that the contradictions of subsistence and market warrant closer scrutiny as the causation of environmental degradation, not the people or institutions themselves.

These contradictions are especially likely for the "conservation with development" policies of the HBR. The outcomes of such policies depend on a critical evaluation of how the proposed development articulates with existing land use strategies and *common property* regimes. Generally speaking, it is widely recognized that communal institutions can be undermined by rapid social, political, economic and demographic change (e.g., Jodha 1987, Jodha 1995, Ostrom et al. 1999). Most highlands have seen increased intervention by states and other entities concerned with the conservation of unique mountain environments. Coincident with this attention is the growth of tourism in many highland areas. Due largely to these actions, increased accessibility and economic opportunity have led to in-migration to some highlands. A number of these conditions are occurring in the indigenous communities of the HBR, especially in Tupac Yupanqui. The antagonists of *common property* management, namely state appropriation, increased market involvement, technological change, population growth, migration, and the breakdown of traditional values affect a community's ability to sustainably manage common pool resources (Goodland, Ledec, and Webb 1989).

Paradoxically I will argue that these changes often are either directly or indirectly traceable to the policies encouraged by those most concerned with the region's protection. I restrict my discussion here to a few of the dominant forces at work in the study community, namely: (1) those of state intervention in the creation and management of the HBR, (2) encouragement of tourism as a means of poverty alleviation, and as a means to offset dependence on park resources, and (3) demographic changes brought about through population growth and temporary out-migration to take advantage of new wage-earning opportunities in Huaraz. The combined effects of these factors on *common property* management among the Ishinca valley user-group will be explored below with the goal of recognizing that this communal institution must adapt lest the park's concerns of environmental degradation and biodiversity loss be realized. While it may seem at the outset that highlighting the role of the "conservation with development" paradigm in creating these problems would argue against the validity of this approach, I do not. The possibilities of achieving people-centered conservation will be explored throughout the dissertation, as I hope to show that anything less would be a disservice to the region's rich cultural and biological heritage.

No Security from Confiscation: When the state comes marching in

Public interventions and enhanced roles of the state have had a profound effect on communal institutions of resource management. This is especially true of those in the highlands, as many mountain environments are perceived to be in great need of external protections (Ives, Messerli, and Spiess 1997). In 2002, the International Year of the Mountains brought attention to the need to "ensure the wellbeing of mountain and lowland communities by promoting the conservation and sustainable development of mountain regions" (2002). Largely as a result of the attention given to environmental degradation in the world's highlands, many have been integrated into national parks, international world heritage sites, and biosphere reserves that engage state and federal government agencies as well as private and non-profit organizations in local land use and resource management planning. Over twenty organizations in addition to park administration had projects in TY, covering such issues as agro-forestry, reforestation, trout-farming, lodging for tourists, road-building, electrification, vaccinations and health care services, irrigation and potable water, pasture improvement projects, road development, and alpaca rearing for wool and meat in local markets.[42]

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[42] Non-government organizations with active projects in Collón during the year of fieldwork included: CARE, Operation Mato Grosso of the Italian Catholic Church, and The Mountain Institute (TMI). Government agencies included Peru's Ministerio de Agricultura, Ministerio de Trabajo y Promoción del Empleo, Ministerio de Salud, Fondo Nacional de Compensation y Desarollo (FONCODES), Programa Nacional de Manejo de Cuencas Hidrográficas y Conservación de Suelos (PRONOMACHS), and Programa Nacional de Asistencia Alimentaria (PRONAA). The Centro de Estudios para el Desarrollo y Participación (CEDEP) worked with Collón in the past on livestock-related projects, but were not involved

Clear rights of use and ownership for the community are pre-conditions for the sustainable management of a commons (McKean 2000, Stevenson 1991). A perception that use and ownership is secure is undoubtedly undermined by so many stakeholders. Figure 2.10 documents key events that have influenced access and use of communal recourse by the region's indigenous communities. The post-colonial era brought about the dissolution of many indigenous *common property regime*s throughout the Andes when *ejidos* (indigenous communal lands) were assumed into large private land holdings known as *haciendas* (estates). It was not until Peru's Agrarian Reform of 1969, popularly known as "The Day of the Peasant," that these large estates were returned to the descendents of their former owners (Velasco 2005). This move, which made president, Juan Velasco, popular among the region's dispossessed indigenous Quechua communities by reestablishing their access to critical highland resources and facilitating their return to communal control. However, the reform's socialist focus on disallowing ownership (even for a group) was only moderately successful in creating tenure security for Peru's *Comunidades Campesinas* (peasant communities).

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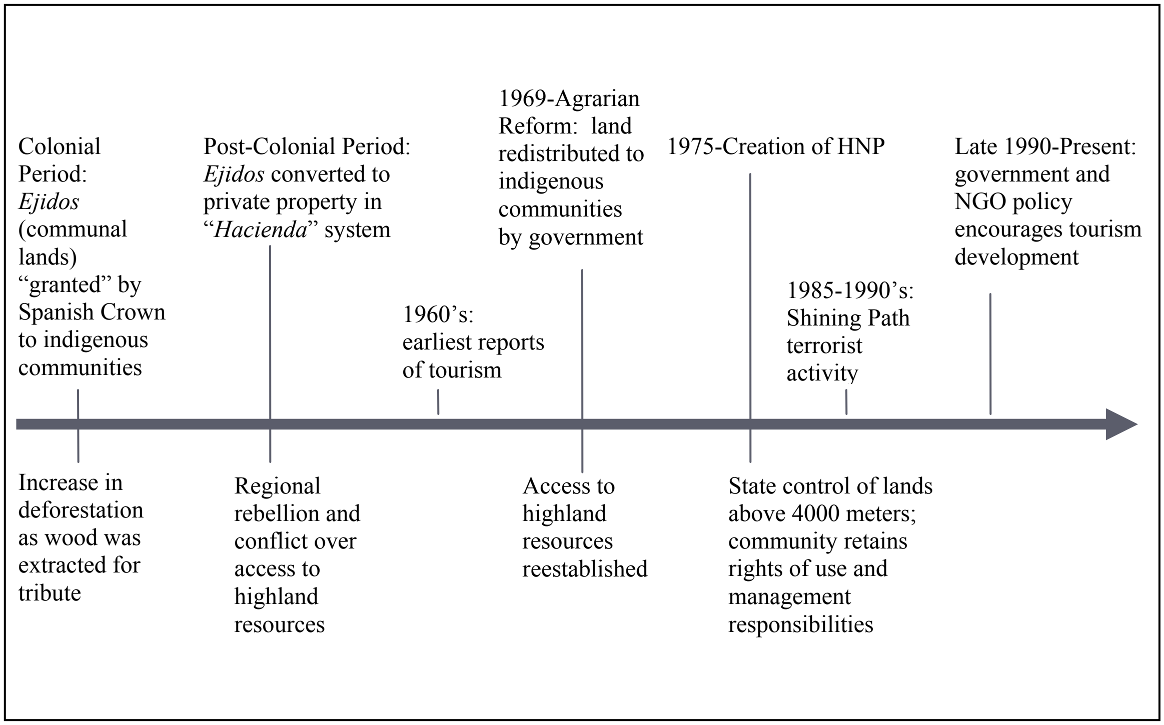
in any at the time of this study. Pashpa had relatively fewer active projects with PRONOMACHS and FONCODES, as well as having begun talks with PRISMA (Proyectos de Informática, Salud, Medicina, Agricultura), a non-government micro-lending institution.

Figure 2.10: Key events in the region. Time periods are provided above the timeline and their effect on highland resource management is described on the bottom.



Ambiguity in ownership and rights of use vex the Ishinca valley user-group today. Although both Collón and Pashpa were actively pursuing formal title to their agricultural lands in 2002, much of their grazing resource occurs within the HNP, and is therefore precluded from title. While the community is permitted to continue grazing and foraging in the national park, its responsibility to sustainably manage these activities is somewhat undermined by state ownership, and the insecurity in access to these resources created by the possibility of state intervention (Banana and Gombya-Ssembajjwe 2000).

Events within the study community will serve to illustrate these points. The first involves the creation of the boundary of the HNP. A boundary delineating the park-community interface was created from several widely-spaced markers that were subsequently digitized to create a park boundary. A number of conflicts between the region's indigenous communities and park administrators have ensued. This is no less so in TY, considering the extent of land below 4000 meters which is now part of the national park (recall Figure 2.2). During fieldwork several households from both Pashpa and Collón plowed *chacras* and planted potatoes in this zone. These fields drew the attention of a number of community members, some of them commenting that households were not supposed to plant there because it was the property of the park. This seemingly went unaddressed for some time, but eventually sparked a dialogue between the community and park administrators. The outcome was unresolved at the time of my departure, but will possibly involve an increased presence of the park in the future, possibly by way of implementing livestock quotas, fees or additional obligations in exchange for use (e.g., use of the Ishinca valley is currently contingent on the household's involvement in reforestation efforts). Whatever the case, it is likely that any resolution will affect the community's sense of responsibility for this territory in the future, and its willingness to manage it sustainably; much of which includes the critical dry season reserves on which an increasing number of community, *empresa* and household herds depend.

Unequal Commoners: The unintended side-effect of well-intentioned development

The cooperation of multiple users with multiple interests is at the heart of successful *common property regime*s. This is best achieved when there is a perception of fairness among these users. Inequalities may undermine fairness by creating a sense of feeling disadvantaged or being denied adequate access to a communal resource (McKean 2000). Inequalities are exacerbated in the study community *via* two primary mechanisms. The first involves tourism, which was shown above to have created an economically diverse user-group with differential capital for exploiting the commons (i.e., various numbers and types of livestock). The second involves community development, namely the pasture improvement projects of Collón, which have unintentionally created differential access to improved pasture resources for only half of the Ishinca valley user-group.

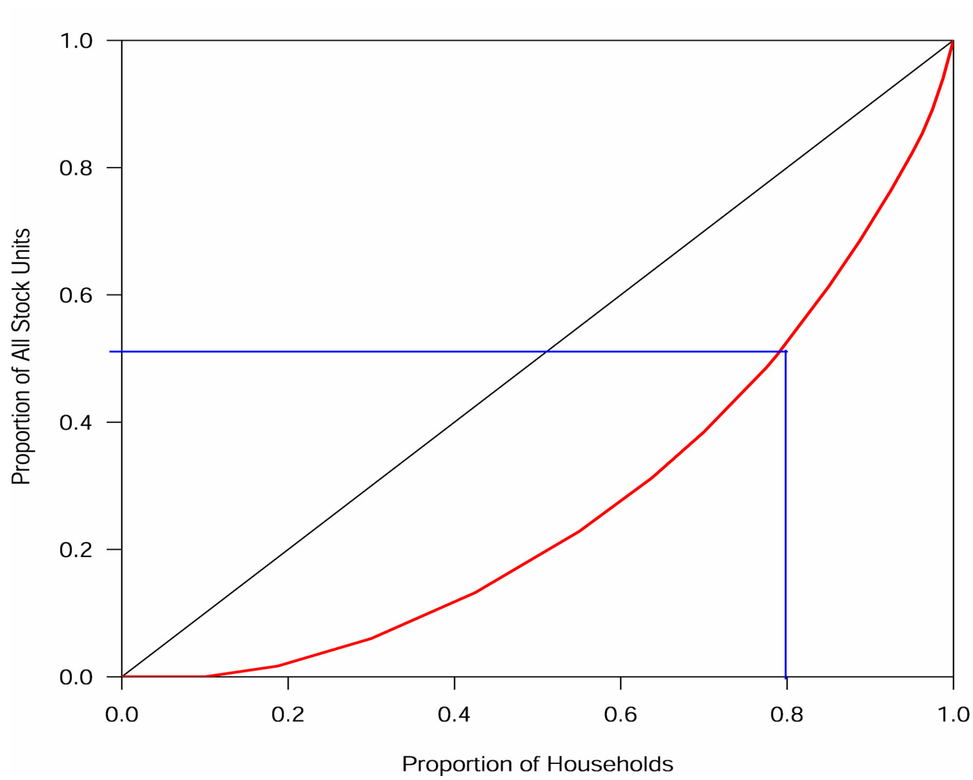
I will begin with the implications of increasing household herds, which not only increases their dependence on the park, but confound the community's ability to sustainably manage them. Differential capital among users of a commons (e.g., variation in numbers of livestock or technologies for resource extraction) is normal, but when extreme may result in a decreased likelihood of cooperation (Singleton 2001). Inequality in livestock holdings in the study community is moderate according to a Gini coefficient of 0.43 (Sen 1973).[43] A Lorenz curve of household stock equivalents graphically illustrates this inequality in Figure 2.11.[44] The 45° line in this figure indicates perfect equality; where 50% of the stock equivalents in the community would be owned by 50% of its households. The curve below this line reveals that livestock holdings are unevenly distributed. A reading of the Lorenz curve at 50% of the community's herd wealth shows that half of all the stock equivalents pastured on the commons are owned by only 20% of

[43] Gini coefficients range in value from 0-1. A value of 0 indicates perfect equality in herd wealth among households of the study community, whereas a value of 1 would indicate that herd wealth was concentrated in a single household. Moderate values such as the one presented indicate that inequality does exist in the community.

[44] Stock equivalents are used to standardize the diverse herd compositions kept by individual households. The justification for conversion of livestock head to stock equivalents is that each has a different demand for and impact on communal resources. All livestock are standardized by the annual forage requirements of sheep (hectares/yr of good quality pasture), as they are the most abundant livestock type kept by households. Appendix 2.2 summarizes stock equivalent values for each type of livestock.

its user-group. These herd-wealthy households are predominately involved in tourism. While it remains to be shown whether or not this disproportionate herd wealth is problematic, the potential exists does exist for cooperation to be undermined by this inequality.

Figure 2.11: Distribution of herd wealth within the study community. The 45° line indicates perfect equality among households, the curve below indicates the cumulative proportion of stock equivalents owned by each successive household. The horizontal line drawn from the y-axis indicates that 50% of all the stock equivalents in the community are owned by only 20% of its households. This illustrates a moderate level of inequality in livestock wealth, as confirmed by the Gini coefficient reported in the text.



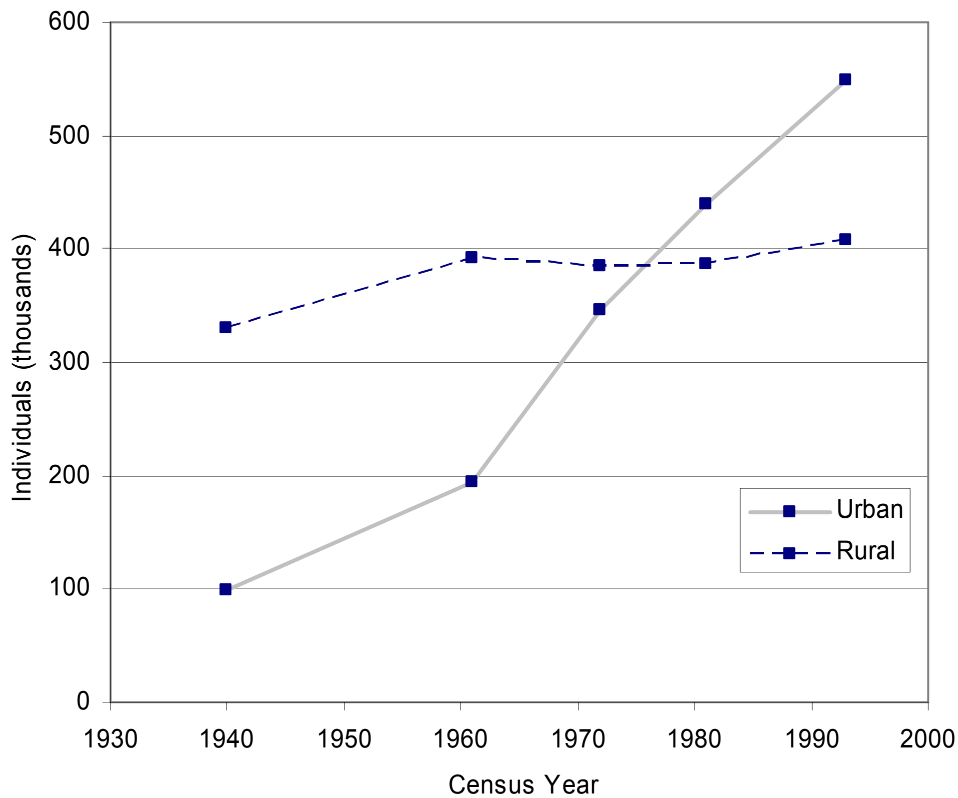
Inequalities in access to resources can result in similar problems for collective action (Young 2001). Community development projects implemented without a clear sense of the structure and function of the existing *common property regime* are problematic in this regard. A case-in-point involves pasture improvement projects in the sector of Collón undertaken by The Mountain Institute (TMI). Although this project was undeniably instrumental in addressing the needs of Collón's *comunero*s, and in creating an alternative to absentee herding for them, Collón is only half of the user-group for the Ishinca valley. The lack of a similar project in Pashpa at the time of fieldwork created envy and resentment among residents of this sector. At one point, a *comunero* of Pashpa approached me to talk to TMI on their behalf, seemingly concerned that TMI felt that Pashpa was not worthy of their attention. Many more suggested that Collón had erroneously represented themselves as the sole users of the Ishinca valley, a misconception residents of Pashpa were eager to dispel. Although unintentional, increasing Collón's access to improved pastures while neglecting the creation of similar opportunities for *comunero*s of Pashpa, have seemingly exacerbated perceptions of unfairness, with the potential to hinder their continued cooperation in the use and management of the park, the extent for which both are responsible. This is an untenable circumstance given the organization's commitment to conservation and sustainable development, and one that will be rectified by their future plans to work with Pashpa, the other half of the user-group.

Shifting Membership in the Commons: Population growth and migration

The growth and migration of populations is a widely explored cause of failures in *common property* management, as such phenomena fundamentally alter the size and characteristics of the user-group, and influence collective action outcomes (Chamberlin 1974, Oliver, Marwell, and Teixeira 1985, Olson 1965, Ostrom 1991). Approximately 10% of the world's human population lived in highland areas in the early 1990's (Grotzbach and Stadel 1997), and at relatively low population densities due to the marginal productivity of higher elevations. Rural to urban out-migration has characterized the net movement of many highland peoples (Escobar and Beall 1982, Turner 1976). Out-migrations were critical for countering natural population increases and for maintaining sustainable levels of use in Törbel (Netting 1981). They have been described as an important strategy for resource conservation in Andean communities as well (Preston 1998). However, in-migrations to the highlands happen as well (Brush 1980). Areas of high biodiversity often coincide with human populations in the Andes (Chepstow-Lusty et al. 1998). Many such areas can expect additional population increases due to in-migration, especially with newly created industries and opportunities born out of the region's designation as a biodiversity hotspot (Myers et al. 2000).

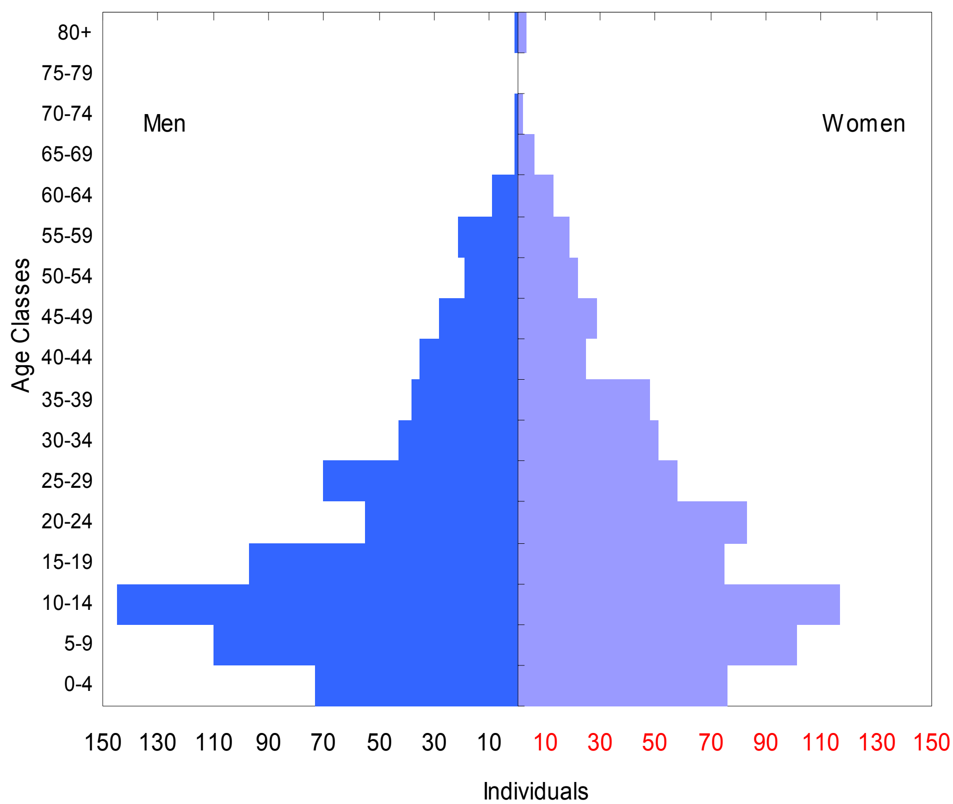
The rugged interior of the north–central Andean highlands was opened by paved highway from the Peruvian capital of Lima, and the HNP was subsequently established in 1975. Tourism and mining have since flourished in the region, and the department of Ancash is witnessing a regional increase in population due, in large part, to these relatively new industries. Expectations of good earnings are a powerful draw on surrounding indigenous populations and even on the populations of urban centers such as Lima (for discussion of push and pull migration factors in the Andes see Brush 1980, Escobar and Beall 1982). Residents of Huaraz, the departmental capital and hub for excursions into the HNP, frequently comment on the region's rapid population growth and economic development in recent years. At present, approximately 337,408 inhabitants live in and around the reserve (INRENA 2002). Figure 2.12 shows that much of the department of Ancash's population growth stems from increases in its urban population over the last 60 years. This growth has been fueled partially by natural population increase, and partially by immigration from surrounding indigenous communities (rural-urban migration) or urban centers (urban-urban migration). A more recent demographic survey suggests that 28.1% of the department's population now live in cities of ≥ 20,000 such as Huaraz (INEI 2001).

Figure 2.12: Trends in urban and rural population in the department of Ancash (INEI 1994). The average annual growth rate for the urban population is 3.3%, as calculated from the data. This growth rate is substantially higher than the estimated national average of 1.3% (IDB 2006).



Despite growth in the region's urban population, it appears that there has been little change in the rural population as a whole. The rural population of the study community mirrors the regional trend. All households surveyed about community demographics (n = 20), perceived very little change in population in recent decades. Although informants reported that the population had changed very little in recent years, most simultaneously expressed concerns about their children's futures, and hopes that they would be able to stay on in the community and make a living. Such preoccupations are warranted given the large number of individuals currently within the pre-reproductive ages of 0–19 (see Figure 2.13), and the restricted land base on which to practice agriculture, much of which has occurred within informants' lifetimes.[45]

Figure 2.13: Age structure of the study community, created from the census conducted during fieldwork in 2001.



[45] This potential for population increase suggested by the proportion of the population in this age range may be countered by a declining total fertility rate suggested by the relatively small proportion of individuals born since 1997 (the 0-4 age set). However, under-reporting for this age class can not be ruled out, as indicated by age-accurracy index and Whipple's index calculations (Whipple 1907).

While population growth can exacerbate failures of *common property* management, pressure on communal resources in the Ishinca valley may occur with or without an increase in the population of the user-group. Note the paucity of men aged 20–24 and women aged 15–19 in the population pyramid of Figure 2.13. At the time the census was conducted, individuals living and working at least half-time in Huaraz were not considered as permanent residents of the community, explaining the shortage of individuals in these age classes. Over the course of fieldwork I learned that these individuals did not lose their status as *comunero*s, or their rights of access to agricultural fields and grazing resources. In nearly all cases, they were simply members of a larger family extending their reach into the urban labor-market. This loss of household labor was offset by the remittances provided from working in Huaraz. These remittances were often spent as the head of the household saw fit. The work done by these individuals included permanent part-time employment in local bakeries, restaurants and homes; or temporary employment as *jornals* (day laborers) for large landowners, mining companies and tourism operators. The wages earned, as with those earned directly from the tourists visiting the Ishinca valley, were invested in a variety of household goods. Those often reported included agricultural fertilizers and pesticides, livestock, and vaccinations or remedies to keep existing herds healthy. Population growth and increased temporary out-migrations thus combine to offer the possibility of over-exploiting and degrading communal resources. This would not result from the size and movement of the indigenous population itself, but from the inability of existing communal institutions to manage a growing number of users whose time away from the community in the bustling city of Huaraz makes them not only increasingly diverse economically, but socially as well (Adger et al. 2002).

CONCLUSION

The persistence of many Andean landscapes and agro-pastoral livelihoods suggests that indigenous institutions for resource management have been sustainable historically (e.g., Orlove 1976). Many characteristics of successful *common property* management are apparent in the study community, although they are largely implicit. For example, evidence of an attention to resource conservation exists in the communally coordinated opening dates for the Ishinca valley *portada* (gate), and informant reports of occasional fines levied against defecting households. In addition to these rules, the physical boundaries of the Ishinca valley easily exclude other communities from this resource and clearly define Collón and Pashpa as its user-group. All are important characteristics of a functioning *common property regime* (McKean 2000, Stevenson 1991), yet alone they are insufficient to guarantee its sustainability.

Property regimes are a social construct involving arrangements that govern individual uses of a communal resource. In the Andes, as in many of the world's highlands, these communal institutions evolve in a specific context from a necessity to efficiently and sustainably managing relatively scarce highland resources. This context is not static; it requires that communities modify decision-making arrangements and rules of use accordingly. The concerns of environmental degradation voiced by park and community alike, beg the question as to whether the existing *common property* institution has adapted to the current social, economic and political context in which its constituent households operate. If not, why and how can we encourage it?

This chapter has outlined possible unintended consequences associated with the region's policies of "conservation with development,' including some that foreshadow unsustainable outcomes. These include changes at the level of the household decision-maker and changes at the level of the communal institutions for managing them. Some are supported by data, while others are hypothesized from *common property* theory and a number of empirical studies of *common property* management. Changes in household-level land use confirmed by my data include reductions in fallow, increases in non-native livestock, and the emergence of new spatial and temporal herding strategies. This analysis argues that environmental change in the HBR is the result of complex feedbacks between park management, NGO agendas, and indigenous livelihoods that collectively define the Andean landscape. The ability of the community to sustainably manage these changes is similarly affected. This includes the creation of ambiguous rights of use and control over park resources, inequality exacerbated by differential involvement in tourism, inequality in access to community development assistance, and demographic change affecting the size and characteristics of the Ishinca valley user-group. Collectively these conditions may act to undermine communal institutions, and the possibility of protecting the integrity and character of Andean cultures and landscapes.

Yet specific outcomes must be evaluated empirically, as environmental degradation in particular, is often too simply assumed. This will involve specifying the mechanisms by which tourism affects household-level decision-making, and exploring the actual effects these decisions are having on the environment. Too little attention to the specifics of communal resource management in “conservation with development” has likely contributed to failures of both. The remainder of the dissertation will be devoted to correcting this oversight. The following chapter will specifically test how changing herd wealth and opportunity cost influence household decisions regarding the management of their herds. In doing so it seeks to identify factors entering in the decision-making process that have resulted in the changing herding practices observed in the community, especially those involving the increased use of high altitude pastures by untended livestock. Chapter 4 will explore the implications of such decisions for the conservation of biodiversity in the HBR, an analysis that relies on vegetation sampling, while recommendations for encouraging sustainable use will be explored in Chapter 5.

CHAPTER 3

EXPLORING THE EFFECTS OF TOURISM ON THE MANAGEMENT OF HOUSEHOLD HERDS IN THE HUASCARÁN BIOSPHERE RESERVE, PERU

ABSTRACT

Common property (CP) researchers wish to understand how cooperation can be sustained in the management of a commons, and the factors that contribute to failures of *common property* management and environmental degradation. Rapid socio-economic change is often cited as a primary reason for problems of collective action to emerge, a condition precipitated by the development of adventure tourism in the Huascarán Biosphere Reserve (HBR). Ironically, the rationale for encouraging this industry is supported by the “conservation with development” paradigm, of which biosphere reserves like the HBR are part. Indications of over-grazing in the HBR, and their potential root in regional policies of economic development, motivate this study of the effects of tourism on household herding practices. In marginal and fragile environments such as those of the Andes, active herd management allows agro-pastoralists to coordinate production schedules, to maximize their returns by herding animals on appropriate sites, and to minimize environmental degradation by dispersing animals or excluding them from seasonally sensitive areas (e.g., wet season pastures or active agricultural fields in the community). Herding labor is therefore critical for the sustainable use of the HBR. However, competing hypotheses of opportunity cost and

herd wealth suggest that tourism may have very different effects on household herding practices, with different outcomes for the conservation of the reserve. Opportunity cost predicts that households with more lucrative uses of their time are likely to spend less time herding, defecting in their responsibility to manage herds sustainably. Game theory predicts that herd wealthy households, often those most involved in tourism, have a vested interest in the pasture resources on which they so heavily depend, and therefore may actually invest more effort in herd management. I evaluate these contrary outcomes using a multilevel model of household herding labor determined via time allocation observations of a sample of economically diverse households comprising the Ishinca valley user-group. I find that:

(1) involvement in tourism strongly influences household herding practices;

(2) moderate involvement in tourism exerts a significant negative effect on household herding labor; and

(3) households with the highest levels of tourism involvement show improved herding practices, providing a hopeful outlook for conservation in the HBR should policies be restructured to create equal opportunities for participation in the tourism industry.

Photo 3.1: Young herder of Musho, north of the Ishinca valley (photograph by the author, 2002).

Photo 3.2: Young *arriero* of Collón (photograph by the author, 2002).

Photo 3.3: The high altitude pasture of Miyu Pampa (base camp), with the summit of Toclaraju in the background (photograph by the author, 2002).

Photo 3.4: An *arriero* transporting gear to base camp amidst climber's tents and *vacas silvestres* (photograph by the author, 2002).

"Creo que tenemos muchas turistas—
Pero hay más arrieros."

"I believe that we have many tourists—
But there are more arrieros."

(an anonymous wife of an arriero in Tupac Yupanqui, sector Pashpa, 2002)

INTRODUCTION

The objective of the Huascarán Biosphere Reserve (HBR) is to conserve biodiversity and to alleviate poverty—conservation with development. To this end, policies of tourism development have been promoted in the tropical Andes of north–central Peru, primarily since the late 1970's. A growing number of mountaineers and trekkers now flock to the region annually, but the conservation benefits it was assumed would follow, have not been as forthcoming. Recent assessments of threats to conservation in the HBR have raised concerns of over-grazing and deforestation (INRENA 1990b, INRENA 2002, INRENA 2003, TMI 1996a, TMI 1997, TMI 2001a). Many of these concerns are currently speculative in nature (for exceptions see Byers, 2000 #1086, Tohan 2000). Nonethelss, they lead to an implicit assumption that indigenous residents have become inimical to the conservation of this Andean landscape.

This situation, which is common to many biosphere reserves, fuels a lively and contentious debate occurring today in conservation literature about the compatibility of people and parks (Chapin 2004, Terborgh and van Schaik 2002, van Schaik and Rijksen 2002). Before weighing in on the merits and the weaknesses of people-centered conservation and the biosphere reserve as a model of protection, careful study should be made of the dependence of indigenous peoples on protected areas, and the particulars of *common property* management that make its sustainability challenging. Such particulars include how communal institutions and their constituents articulate with introduced market activities, how they respond to changes in wealth and labor, and how they negotiate growing inequalities among commons users.

In this chapter, I test the well-known paradigm of the "tragedy of the commons," which implies that commons, such as the grasslands of the HBR, are destined for degradation due to the self-interests of individual actors (Hardin 1968, Ostrom 1998). A failure to sustainably manage the high altitude *puna* (a common pool resource) is of great concern given the fact that it is the single most abundant land cover in the reserve (Byers 2000, INRENA 1990b), containing an abundance of endemic and endangered plant species (Kolff and Kolff 1997, Smith 1988). Fortunately, a number of case studies and examples from the *common property* literature provide an alternative to the tragic outcome (McCay and Acheson 1996, McKean 1992). *Common property* can be an effective arrangement for marginally productive and spatially heterogeneous resources, such as those of mountainous environments (Netting 1976). The high altitude *puna* circumscribed by the HBR has been managed communally by indigenous agro-pastoralists for centuries and remains in their management today. Although a long history of use and management suggests that indigenous peoples have achieved a substantial run of sustainability, preliminary evidence from Chapter 2, the speculations of reserve administrators and NGOs, and the perceptions of environmental degradation by indigenous communities themselves, provide mounting support for a legitimate concern that traditional institutions of resource management may now be failing.

Failures of *common property* management are fundamentally due to problems of collective action and a breakdown in cooperation among users of a commons (Olson 1965, Runge 1981, Runge 1984). While this frames the problem as an internal one, collective action problems are exacerbated by rapid exogenous change (Jodha 1996, Ostrom 1998). Perhaps then, it is ironic that "conservation with development" works from the fundamental assumption that conservation is best achieved in conjunction with economic development.[46] The key to such approaches is that economic development must be sustainable, a condition too often presumed of tourism (Lindberg, Enriquez, and Sproule 1996, McLaren 1998). Various definitions of sustainability exist with regard to tourism (Hunter 2002), as do a number of tourism practices with varying emphasis on sustainability. This paper was motivated by what I perceived as a potential problem with *adventure* tourism in the HBR, and by the prospect that the very industry intended to offset the use of the reserve's core might have the unintended consequence of facilitating its degradation.[47]

A detailed portrait of land use and resource management in Collón and Pashpa (the Ishinca valley user-group) was presented in Chapter 2. The objective of this chapter is to move beyond description and speculation to a formal analysis of tourism's effect on the management of herds in and around the Ishinca Valley, a valley which many believe

[46] For policy origins one can refer to the World Conservation Strategy of 1980, The Bruntland Report of 1987, and the Global Biodiversity Strategy of 1992 and 2002.

[47] Administrators of the reserve recognize that adventure tourism may not be a sustainable form of tourism in the HBR. They consider it, as well as misuse by local indigenous communities, as potential threats to the reserve although they discuss these laregely as independent issues. My research focuses on the interconnectedness of the two.

is degraded. I accomplish this by testing competing hypotheses about how household-level changes associated with tourism involvement (e.g., increasing herd wealth and increasing opportunity cost) affect their herding practices. Data for this effort consist of a community census and a full year of time allocation observations. Time allocation data were obtained from a sample of 80 households yielding a total of 3,270 household observations. A total of 393 individuals made up these households, yielding 16,070 individual observations. Time allocation data have the potential to offer tremendous insight into indigenous livelihoods, yet analyses of time allocation data has largely ignored its hierarchical structure.[48] A secondary objective of this chapter is to illustrate the utility of a multilevel (random effects) modeling framework as a means of accounting for the nested design of the time allocation dataset.

STUDY SITE

The Huascarán Biosphere Reserve (HBR) was created in 1977 and includes a previously designated protected area known as the Huascarán National Park (HNP). It is located in the north central Andes, in the Department of Ancash, approximately 8 hours by bus from the Peruvian capital of Lima. Collectively, the HBR encompasses an area of 5710 km^2 , including 3400 km^2 of core conservation area in the former HNP, and an additional 2310 km^2 buffer containing many of the region's indigenous communities. An

[48] In my dataset, individuals of randomly sampled households are observed over time. As a result, not all sets of observations are equivalent. The presence of observational heterogeneity in a dataset can manifest itself as a correlation among the observations made on the same hierarchical unit (i.e., individual or household). Such correlations violate a basic assumption of ordinary regression analysis. In order to draw valid conclusions from nested time allocation observations, I use a multilevel modeling framework, as discussed in the following text.

estimated population of 337,408 individuals lives in this region (INRENA 2002); approximately 226,000 are Quechua agro-pastoralists with continued grazing and foraging rights in the HNP (see Photo 3.1 on pg. 98). This area is rich in natural resources, cultural heritage and archeological significance, as well as one of magnificent monumental scenery (see Photo 3.3 on pg. 99). It is also a famed climbing destination due to a favorable dry season climate and access to over 60 peaks of the Cordillera Blanca with elevations greater than 5500 meters.

Tupac Yupanqui (TY) is one of many indigenous communities in the reserve. Collón and Pashpa, two sectors of this community, were chosen for dissertation fieldwork primarily because of their responsibility for the use and management of the Ishinca valley, a popular acclimatization valley within easy commuting distance of the department capital of Huaraz.[49] Ishinca is heavily trafficked by a good proportion of the foreign tourist population (approximately 23,000) that visit the region annually, thus the households of Pashpa and Collón are some of the most market-integrated in the region. A total of 1474 individuals and 297 households (mean household size = 5, stdev = 2.0) from these sectors form the exclusive user-group of grazing commons in and around Ishinca valley, as well as the labor pool for visiting mountaineers seeking *arrieros*, porters, guides and cooks to assist them in making the summits of Urus, Toclaraju, Ishinca, Palcaraju, Ranrapalca and Ocshapalca (see Photo 3.4 on pg. 99).

[49] Other sectors of TY, such as Joncopampa, are not part of the user-group for the Ishinca Valley, thus they are not included in the study.

In this study population, as in most of the Andes, use of the commons is organized primarily at the level of the household (Brush and Guillet 1985, Mayer 2002). [50] In general, the household keeps a variety of livestock including cattle, horse, mule, sheep, llama, alpaca and goat. Of the 297 households recorded during the census in 2001, mean herd size was 13, with a range of 0 to 190 livestock (stdev = 17.3). Given the variability in herd composition among households, I report household herd data in stock equivalents.[51] This calculation is based on the annual forage requirements of sheep, the most abundant livestock type in the community. Thus, sheep are set to 1 stock equivalent, and all other livestock units are determined by their average forage requirement relative to sheep. Using this conversion, mean household herd size is 25 stock equivalents, with a range of 0 to 246 (stdev = 25.2). Figures 3.1 and 3.2 summarize the distributions of household size and herd wealth among the study population. This is the population from which the 80-household time allocation sample was drawn, and the one for which inference from the analysis is directed.

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[50] Commons of the HBR are also used by community herds of each sector. Collón has approximately 179 cattle, and a limited number of *empresa* herds totaling 19 llamas (held jointly with Pashpa) and 208 alpacas, formed by various GO and NGO development projects in the region. Pashpa has approximately 42 cattle, and currently very few *empresa* herds though they would like to increase their involvement with local GO and NGO organizations in the future. The focus of this study is on household herds and household-level management decisions affecting the commons.

[51] This stock equivalent scale was derived from forage requirements found in unpublished documents utilized by TMI (Sotomayor 2000). I used information from this document to derive my stock equivalent scale. I set the scale relative to sheep, which require ¼ hectare of good quality pasture to support a single individual per year. Thus the stock equivalent scale reflects the annual requirements of each animal relative to sheep. Sheep = 1.00, Cattle = 4.00, Alpaca = 1.48, Llama = 2.22, Horse and Mule = 3.00. Appendix 2.2 summarizes the stock equivalents for each type of animal in the household herd.

Figure 3.1: Distribution of household size in the study community. The number of individuals in the households is normally distributed between the minimum household size (1 individual) and the maximum (10 individuals).

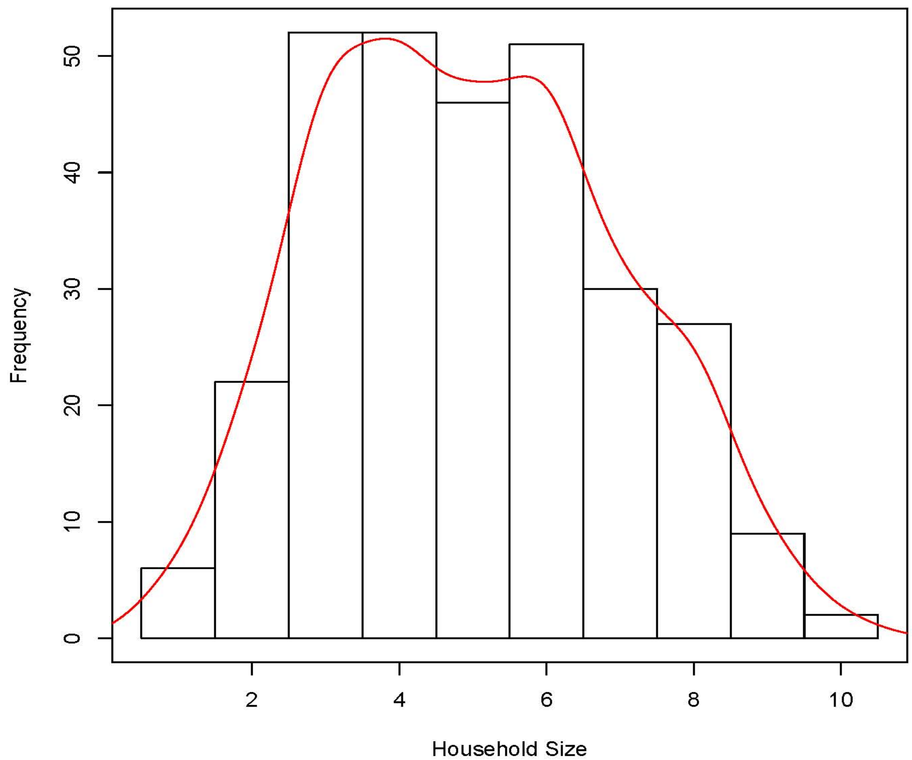
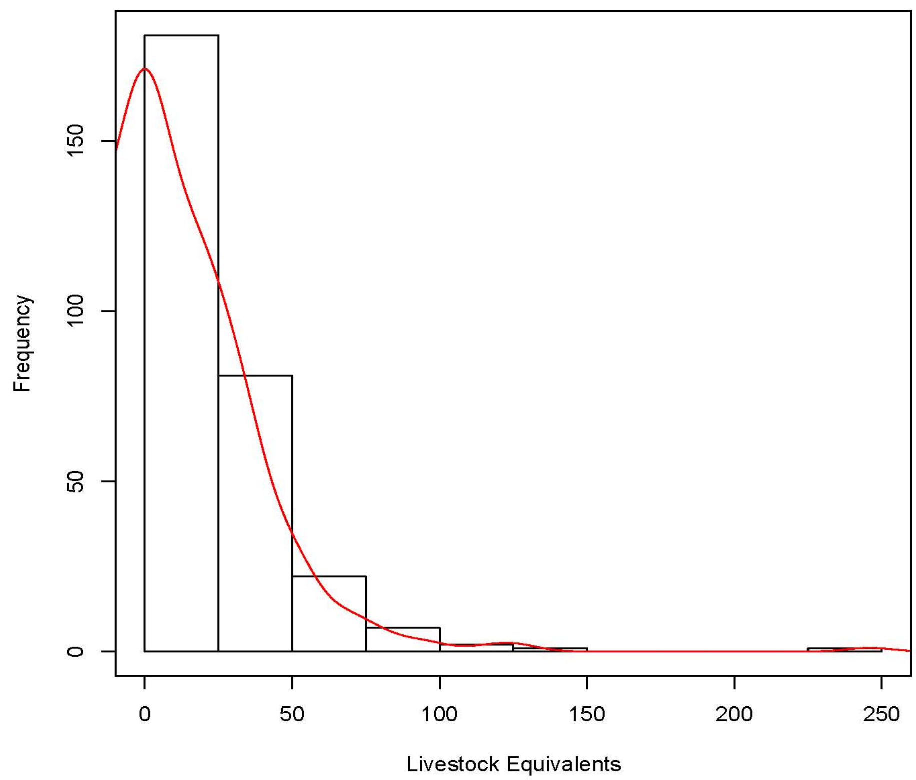


Figure 3.2: Distribution of herd size (in stock equivalents) in the study community. The number of stock equivalents owned by the household is heavily skewed between the range of 0 and 246. The household outlier was not included in the sample drawn for time allocation study.



Patterns of land use among these diverse households are largely dictated by diversity and verticality. Land tenure in Tupac Yupanqui involves divisible[52] communal ownership of agricultural lands in mid-altitudes and indivisible[53] communal rights to pasturing lands above 4000 meters (Guillet 1981). The grazing commons of Collón and

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52 Divisible communal lands are communally held lands that are divided among members of the community for exclusive use by a particular household during cultivation. When fallow, this land typically reverts to indivisible common property.

53 Indivisible communal lands are communally held lands that can be used by all recognized members of the community, sector or barrio.

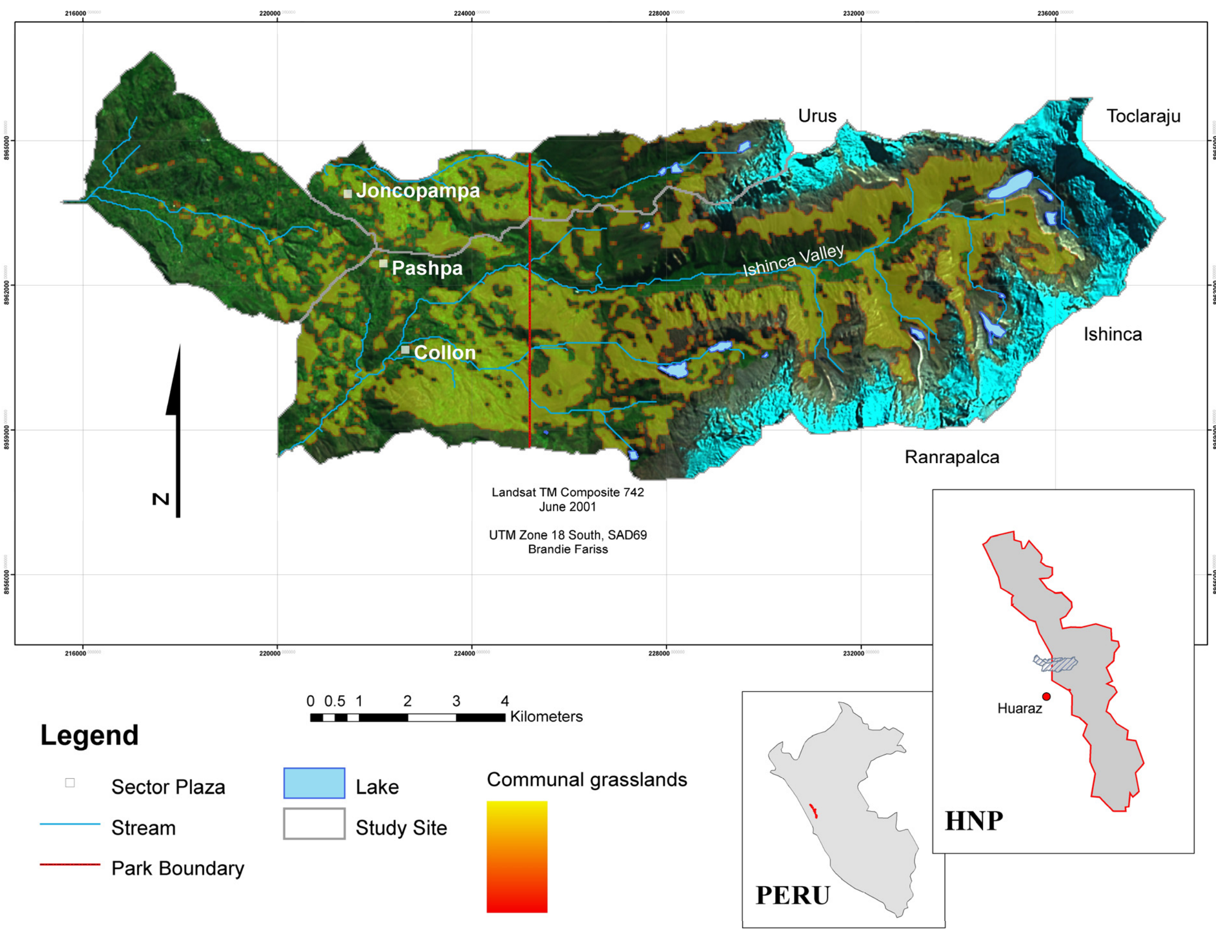
Pashpa are comprised of an estimated 60 hectares in the agricultural buffer zone, and additional 90 hectares in the national park, for a total of approximately 151 hectares.[54] Outside of the national park boundary, in the agricultural buffer zone, rights to land north and south of the Ishinca River are assigned to one of the two sectors. Pashpa controls land north of Ishinca River, and Collón controls land south of it. Both sectors collectively use and manage land within the Ishinca Valley. In the wet season, households pasture livestock primarily on fallowed fields communally controlled by each sector. Pastures above 4000 meters, referred to locally as the *puna*, are used primarily in the dry season by both sectors. This important ecological resource is circumscribed by the political boundary of the HNP, thus the reserve's conservation core becomes a critical dry season resource in addition to an outlet for relieving pressure on pastures near the community that are heavily utilized in the wet season (see Figure 3.3).

Although agriculture and pastoralism continue to be the basis of the household economy, I estimate that 14% of the households in Collón and Pashpa are involved in tourism. Tourism opportunities vary, but the predominant tourism-related occupation is *arriero* work, which involves transporting gear to base camp using pack animals such as horses, mules or (rarely) llamas. As discussed in Chapter 2, households involved in tourism are herd wealthy relative to those that are not. Households reporting arriero work in 2000 owned an average of 36 stock equivalents compared to the average for non-*arriero* households of 23, a significant difference according to a two-sampled t-test ($p >$

[54] I estimate these values from a landcover classification of Landsat TM imagery from 2001 (Lipton 2007). The total area I consider a grazing resource includes grassland, mesic grassland, and fallowed agricultural fields. The latter varies from year to year and there are additional landcover types that may be utilized by some livestock (e.g., scrub and forest). Thus, this is only an approximation.

$|t| = 0.05$). A continued reliance on the pastoral economy, coupled with the economic diversification of the Ishinca valley user-group provide an excellent opportunity to explore the ways in which the concomitant effects of market integration factor into household decisions regarding the use and management of this protected landscape.

Figure 3.3: Extent of common property managed by households of Collón and Pashpa. Communal grasslands represent landcover types primarily utilized for grazing (i.e., grassland, mesic grassland, and fallowed fields). This data was obtained from a landcover classification of 2001 LandSat TM imagery (Lipton 2007). I further restrict it here to slopes of less than 45°.



METHODS

This paper utilizes a year of TA observations on a sample of *common property* users in an attempt to quantify the proportion of time they spend herding and the factors affecting their efforts to sustainably manage common pool resources of the HBR. Details of the census and time allocation dataset will be provided below, as the model variables described in subsequent sections were derived from this data.

Census

A census of Collón and Pashpa was conducted at the beginning of fieldwork in 2001. The goal of the census was to describe the user-group of the Ishinca Valley and to stratify the population in order to select a sample of households that captured variation in tourism involvement and herd wealth. A stratified random sample of 27% of the population, comprised of 80 households and 393 individuals was drawn for the time allocation study.[55] An equal number of households (40 from each) was selected due to potential differences in herding related to the availability of fallowed fields in the agricultural zones of each sector, and their differential involvement in various livestock-related GO and NGO projects. Households of each sector were further stratified by their self-reported involvement in tourism over the previous year. Households from these strata were then sampled in a ratio of 3:2. For every three non-tourism households, two tourism households were drawn. I arrived at this sampling ratio in an attempt to capture

[55] Justification for the sample size is provided in Chapter 2 as well as a graphic of the sample design (see Figure 2.3).

as many tourism households as possible, while still maintaining a sample that would be generally representative of overall community structure. This sampling strategy resulted in the household sample summarized in Table 3.1.

Table 3.1: Characteristics of the time allocation sample. Note that there is very little difference in the mean household size between tourism strata. Differences in mean herd wealth between these strata are more pronounced. Discussion in the text and in Chapter 2 indicates that these differences are significant, and their implications are explored herein.

REPORTED TOURISM INVOLVEMENT IN 2000		**MEAN HOUSEHOLD SIZE (SE)**	**MEAN STOCK EQUIVALENT WEALTH (SE)**
Collón (n = 40)	Non-tourism (n = 24)	5.5 (0.3)	30.3 (4.1)
	Tourism (n = 16)	5.7 (0.5)	33.6 (5.6)
Pashpa (n = 40)	Non-tourism (n = 24)	4.8 (0.4)	28.0 (5.9)
	Tourism (n = 16)	5.2 (0.5)	42.6 (8.5)
Combined (n = 80)	Non-tourism (n = 48)	5.2 (0.2)	29.2 (3.6)
	Tourism (n = 32)	5.5 (0.4)	38.1 (5.1)

Time Allocation

Time Allocation (TA) is a quantitative technique for exploring the activity patterns of individuals, households or other entities of interest (e.g., females, children, subsistence vs. market-oriented households) with significant potential to address a variety of research questions. Time allocation observations were made for all sample households using the "spot check" method (Borgerhoff-Mulder and Caro 1985). Observations of every individual in the household were attempted once every six days [56] over a consecutive period from January through December 2002. The maximum number of observations made on each individual was 44 for households of Pashpa and 52 for Collón. These observations were spread across days of the week, months, and seasons; factors with an expected bearing on household production and market-related activities. Observations occurred between the hours of 6:00 a.m. to 6:00 p.m.[57]

Observations were made at the household using a form pre-printed with the individual's sex, age, and relation to the household head to aid in identification (see Appendix 3.1). The location and activity of each individual was recorded using modified versions of the activity codes proposed by Johnson and Sackett (1998) and those employed by Winterhalder et.al (2007) in their time allocation study of Andean agro-pastoralist of Cuyo Cuyo. The whereabouts and activities of individuals who were not directly observed were inquired about and their activities recorded as well subsequently verified by direct observation when possible.

[56] Initial attempts to sample each household at a completely random time proved too difficult. Due to the terrain and the organization of households in the community it was necessary to randomize observations using an alternative method. Observation dates were systematic. The choice of a six day cycle allowed for collection of observations on different weekdays. Similar approaches have been employed in time allocation studies with agro-pastoralists in southern Peru (Winterhalder et al. 2007). Randomization was introduced in the timing and order of households visited. First, I grouped households into barrios as reported in the census. Barrios were then assigned to one of two groups reflecting their relative locations in the valley. On every observation day, groups were assigned to either a.m. or p.m. observation schedules. After the time of day was determined, barrios within the group were assigned a random order. Finally, households within each barrio were visited in the order in which they were encountered. Randomization in time of day, barrio order, and encounter sequence resulted in a range of observation times for each household and individual in the sample.

[57] Estimates based on daylight hours undoubtedly underreport social and domestic activities at home. The focus of this analysis is on herding activites that occur primarily between the hours of 8:00 a.m. and 4:00 p.m.; thus the bias seems to be of little consequence for the analysis presented.

Four levels of observation were made for each individual (see Appendix 3.2). The first level of observation was the individual's location. Individual locations were recorded as: at the house, in the community, outside of the community, or undetermined either because all members were absent from the household at the time of the visit or because the whereabouts of an individual could not be confirmed by another household member. Activities were then assigned to the individual in increasing level of specificity. Level-2 activities include: agriculture, care-giving, domestic, individual, livestock, market, social, wage-earning and educational activities. Level 3 activities specified a type of activity within the general category (e.g., agriculture: harvesting). For level 3 activity categories of *a priori* interest to the research, such as those related to herd management, a tertiary code was also assigned to distinguish between herding practices with different implications for sustainable use of the commons (i.e., active herding vs. checking on unsupervised herds). For example, an individual may have been (1) away from the community, (2) performing a livestock-related task, (3) managing the herd, and (4) actively pasturing animals. This particular code represents the response of interest for the analysis. Comments relevant to the code, such as the locations or intended destinations of the herders were included where possible (see Appendix 3.2 for an example).

Analytical Approach

The majority of time allocation analyses report simple frequencies or tests of significance based on an erroneous treatment of repeated observations as independent samples. However, a few contemporary anthropological studies have pioneered attempts to model time allocation data with intriguing results. Good examples include Tucker's

(Tucker and Young 2005) study of foraging efficiency among the Mikea, and Godoy's (2002) study of spousal leisure sharing among the Tawahka. The analysis presented in this chapter utilizes a multilevel modeling framework similar to Godoy's (*ibid* 2002), whereby I attempt to model herding practices while accounting for the repeated sampling design inherent in time allocation observations.

The approach I take is novel in that it seeks to test competing hypotheses rather than simple differences from a null hypothesis. Beyond this, I avoid *ad hoc* theorizing by testing *a priori* hypotheses about the effects of tourism on household management decisions. Such an approach is more in keeping with the classical scientific method (Chamberlin 1995). By minimizing the temptation to engage in data dredging, this approach reduces the possibility of obtaining specious statistically significant results (Anderson, Burnham, and Thompson 2000).

MODEL DESCRIPTION

A multilevel model (also called a random effects model, see Kleinbaum 2002) can be used to quantify the observational heterogeneity arising from unknown or unmeasured variables that are related to group membership, such as household or community affiliation, as well as to provide accurate standard error estimates adjusted for repeated observations made on the same unit (Mauny et al. 2004).[58] Parameter estimates for this analysis were obtained using maximum likelihood estimation. A dichotomous response

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[58] A comparison of models with various assumptions about the correlation of time allocation observations is presented in Appendix 3.3. Statistical tests confirm that the multilevel modeling framework produces superior results compared to models in which the inherent structure of the data is ignored.

of household herding labor was derived from time allocation observations, required the use of a logit model. Model runs were made using the Correlated Data Library in S-PLUS 7.0.6 (Insightful 2005a, Insightful 2005b). Random and fixed coefficients were simultaneously estimated using the Laplace maximum likelihood estimator. For readers interested in the statistical details of the multilevel logistic model, the specification of random effects, and the estimation approach, I present them in the following section. Others may wish to skip directly to the discussion of the response, control, and explanatory variables of interest (see Table 3.2 on pg. 123).

Multilevel Logistic Regression

An attempt to model household herding labor from the time allocation data requires the use of a logistic model. Let $p_{ij} = P(y_{ij} = 1)$ be the probability that household *i* was observed herding at time *j*. Thus $y_{ij} \sim \text{Bernoulli}(p_{ij})$ with probability density function:

$$f(y_{ij}\,; p_{ij}) = p_{ij}^{\;y_{ij}} (1 - p_{ij})^{1-y_{ij}} \; .$$

The Bernoulli distribution is a special case of the binomial distribution in which the total number of trials is 1. So, we can also write:

$$y_{ij} \sim \text{Binomial}(n_{ij} = 1,\; p_{ij}).$$

Probabilities are proportions that are bounded between 0 and 1. Thus, they require special treatment in statistical modeling. A standard solution is to not model the probability p_{ij} directly but instead to model some function of p_{ij}. The logit function,

$$\text{logit}(p_{ij}) = \log\left(\frac{p_{ij}}{1-p_{ij}}\right),$$

is a popular choice because it can be interpreted as log odds. The logit function maps the interval [0, 1] onto the interval $(-\infty, \infty)$ in a one-to-one fashion. As a result, the boundedness constraint of a probability is avoided and we can proceed to model the logit as a linear function of a set of predictors.

Formally then I assume $y_{ij} \sim \text{Bernoulli}(p_{ij})$ and construct the model:

$$\text{logit}(p_{ij}) = \log\left(\frac{p_{ij}}{1-p_{ij}}\right) = \beta_0 + \sum_{k=1}^{p} \beta_k x_{kij} \;.$$

[eqn. 3.1]

Here x_{kij} is the value of predictor x_k for household *i* at time *j* and β_k is the corresponding parameter. This equation is easily inverted to yield the probability of herding:

$$p_{ij} = \frac{\exp\left(\beta_0 + \sum_{k=1}^{p} \beta_k x_{kij}\right)}{1+\exp\left(\beta_0 + \sum_{k=1}^{p} \beta_k x_{kij}\right)},$$

or more succinctly using vector notation:

$$p_{ij} = \frac{\exp(\mathbf{x}'_{ij}\boldsymbol{\beta})}{1+\exp(\mathbf{x}'_{ij}\boldsymbol{\beta})}.$$

[eqn. 3.2]

Here $\mathbf{x}_{ij}$ is a vector of predictors (with 1 being the predictor associated with the intercept β_0) and $\boldsymbol{\beta}$ a vector of parameters. The right-hand side of equation [3.2] is called a logistic function and when plotted yields an S-shaped curve that is bounded between 0 and 1. In a more sophisticated model of time allocation data we can allow each parameter to have different values for different households (so that β_k should be written as β_{ki}). In the model presented herein, x_{kij} can be either time-varying (i.e., changing with both *i* and *j*) or time-invariant (i.e., changing only with *i*).

Introducing Random Effects

The model in equation [3.1] does not account for the fact that multiple observations exist for the same household. A simple and often adequate way to accomplish this is to include a subject (household)-specific term in the model as follows:

$$\text{logit}(p_{ij}) = \log\left(\frac{p_{ij}}{1-p_{ij}}\right) = \beta_0 + \sum_{k=1}^{p} \beta_k x_{kij} + u_{0i} \ .$$

[eqn. 3.3]

The u_{0i} term represents the random effects of the household, which are independent and assumed to be normally distributed. In other words, $u_{0i} \sim N(0, \sigma^2)$. Because observations of individuals within the same household share the same value of u_{0i}, this induces a correlation structure in the observations which permits for more sophisticated treatment of time allocation data, and a more nuanced statistical model.

Conditional on the value of the random effects, we assume that the individual observations, y_{ij}, are independent. This assumption permits the construction of a likelihood function for our model given the following: let $k(\mathbf{y},\mathbf{u};\,\boldsymbol{\beta},\sigma^2)$ denote the joint probability density of the data vector, **y**, and the vector of unobserved random effects, **u**. Because I have a random sample of households, this joint density factors into a product of individual densities, one for each household, where for some density function *h:*

$$k(\mathbf{y},\mathbf{u};\,\boldsymbol{\beta},\sigma^2)=\prod_{i=1}^{n} h(\mathbf{y}_i,u_{0i};\,\boldsymbol{\beta},\sigma^2).$$

[eqn. 3.4]

Using the definition of conditional probability, I condition on the value of the random effects, and thus express each individual household density as the product of a conditional density and a marginal density,

$$h(\mathbf{y}_i,u_{0i};\,\boldsymbol{\beta},\sigma^2)=f(\mathbf{y}_i\,|\,u_{0i};\boldsymbol{\beta})g(u_{0i};\sigma^2).$$

Where *f* is a Bernoulli probability density (mass) function and *g* is a normally distributed probability density function,

$$g(u_{0i};\sigma^2)=\frac{1}{\sqrt{2\pi\sigma^2}}\exp\left[-\frac{u_{0i}^2}{2\sigma^2}\right].$$

Now by assumption, once we condition on the value of the random effects, the repeated observations on individuals coming from the same household are independent. Thus the joint probability density function for the multiple observations made on the same household, $f(\mathbf{y}_i\,|\,u_{0i};\boldsymbol{\beta})$, factors into a product of individual densities,

$$h\left(\mathbf{y}_i, u_{0i}; \boldsymbol{\beta}, \sigma^2\right) = \prod_{j=1}^{n_i} f\left(y_{ij} \mid u_{0i}; \boldsymbol{\beta}\right) g\left(u_{0i}; \sigma^2\right).$$

[eqn. 3.5]

Combining equations [3.4] and [3.5] yields the following expression for the joint probability density function:

$$k\left(\mathbf{y}, \mathbf{u}; \boldsymbol{\beta}, \sigma^2\right) = \prod_{i=1}^{n} \prod_{j=1}^{n_i} f\left(y_{ij} \mid u_{0i}; \boldsymbol{\beta}\right) g\left(u_{0i}; \sigma^2\right).$$

[eqn. 3.6]

Because the random effects are unobserved, the expression in equation [3.6] needs to be rewritten. I use the law of total probability in order to eliminate the random effects from the calculation:

$$P(A) = \sum_{B} P(A|B) P(B).$$

In the continuous realm the sum in the law of total probability becomes an integral as shown below:

$$l\left(\boldsymbol{\beta}, \sigma^2; \mathbf{y}\right) = \int k\left(\mathbf{y}, \mathbf{u}; \boldsymbol{\beta}, \sigma^2\right) d\mathbf{u} = \prod_{i=1}^{n} \int \prod_{j=1}^{n_i} f\left(y_{ij} \mid u_{0i}; \boldsymbol{\beta}\right) g\left(u_{0i}; \sigma^2\right) du_{0i}.$$

[eqn. 3.7]

This last expression, when viewed as a function of the parameters $\boldsymbol{\beta}$ and σ^2, is called the marginal or integrated likelihood function and is denoted $l\left(\boldsymbol{\beta}, \sigma^2; \mathbf{y}\right)$. All inference for multilevel models is based on this quantity.

Finally I replace the generic notation $f\left(y_{ij} \middle| u_{0i};\boldsymbol{\beta}\right)$ with the formula for the Bernoulli density function to yield the following expression for the marginal likelihood,

$$l\left(\boldsymbol{\beta},\sigma^2;\mathbf{y}\right)=\prod_{i=1}^{n}\int\left(\prod_{j=1}^{n_i}\left[\frac{\exp\left(\mathbf{x}'_{ij}\boldsymbol{\beta}+u_{0i}\right)}{1+\exp\left(\mathbf{x}'_{ij}\boldsymbol{\beta}+u_{0i}\right)}\right]^{y_{ij}}\left[\frac{1}{1+\exp\left(\mathbf{x}'_{ij}\boldsymbol{\beta}+u_{0i}\right)}\right]^{1-y_{ij}}\right)g\left(u_{0i};\sigma^2\right)du_{0i}\,.$$

[eqn. 3.8]

Maximizing this expression with respect to the unknown parameters yields maximum likelihood estimates for the parameters $\boldsymbol{\beta}$ and σ^2.

Estimation Approach

The presence of the multiple integrals in equation [3.8] makes finding an exact solution to the optimization problem impossible, although a number of numerical approximations are available. These approximations proceed in one of two ways, either by approximating the integrand or by approximating the integral (Murphy and Dunne 2005). Laplace's method approximates the integrand with a function that is based on the density function for a normal distribution. This yields an expression whose integral can be found analytically. Alternatively, adaptive Gaussian quadrature evaluates the integrand in equation [3.8] at a number of different values of u_{0i}, then uses these values to approximate the integral. Although adaptive quadrature is the more accurate of the two methods (assuming a large number of quadrature points are being used to approximate the integral), I found little substantive difference in the solutions obtained with the two methods. Because it tended to yield fewer convergence problems, the model results I present are the maximum likelihood estimates of the parameter vector $\boldsymbol{\beta}$ and the variance component σ^2 that were obtained using the Laplace approximation. This method was implemented in the Correlated Data Library of S-PLUS 7.0.6 (Insightful 2005a).

Table 3.2: Definition and summary statistics for variables in the multilevel model of household herding labor, as explained in subsequent text.

RESPONSE VARIABLE:								
NAME	DEFINITION	TYPE	OBS.	MEAN	STDEV	MIN	MAX	LEVEL
herding labor	At least one member of household herding animals during observation	Dichotomous	3270	0.46	0.50	0	1	Time
CONTROL VARIABLES:								
NAME	DEFINITION	TYPE	OBS.	MEAN	STDEV	MIN	MAX	LEVEL
head age	Age of the economic head of the household	Continuous, quadratic	80	37.59	12.20	21	80	Household
farming labor	Proportion of time spent in agriculture	Proportion	80	0.30	0.12	0.05	0.65	Household
dry season	May-August dry season dummy variable	Dichotomous	3270	0.40	0.49	0	1	Time
*farming labor * dry season*	Interaction term	Proportion	3270	0.31	0.11	0.05	0.65	Household *Time
EXPLANATORY VARIABLES:								
NAME	DEFINITION	TYPE	OBS.	MEAN	STDEV	MIN	MAX	LEVEL
tourism involvement	Proportion of time spent in tourism	Categorical: Low Moderate High	 65 7 8	 X X X	 X X X	 0.00 0.03 0.06	 0.03 0.06 0.12	Household
*tourism involvement * dry season*	Interaction term	Categorical: Low Moderate High	 2953 343 334	 X X X	 X X X	 0.00 0.03 0.06	 0.03 0.06 0.12	Household *Time
RANDOM EFFECTS VARIABLE:								
NAME	DEFINITION	TYPE	OBS.	MEAN	STDEV	MIN	MAX	LEVEL
household	Unique identifier for the household	ID	80	X	X	X	X	Household

Household Herding Labor

Data collected via time allocation provide a measure of the household's herding effort. The choice of herding labor as the model response is based on the premise that active herd management is necessary to maintain "landesque capital" in the commons and to prevent long-term declines in resource productivity and environmental degradation (Blaikie and Brookfield 1987, Turner 1999). The labor-environment connection is especially strong in the Andes, where constant care must be taken to maintain a marginally productive and relatively fragile resource base (Zimmerer 1993). Mobility, achieved by actively herding animals, allows the household to maximize returns by moving to the best quality sites, while minimizing the concentration of animals on those seasonally prone to degradation. This logic assumes that animals left to their own devices would not produce similarly sustainable patterns of use. Several studies suggest that this is true by offering a number of mechanisms by which cows, in particular, can negatively affect high altitude grasslands when left to graze for extended periods of time. These include prolonged grazing pressure which can increase turf exfoliation, soil erosion and compaction (Molinillo and Monasterio 1997, Perez 1993, Perez 1998), as well as encourage the formation of needle ice which prohibits seedling establishment (Perez 1987).

Given the potential effects of excessive livestock concentrations in the high altitude environment, herding labor is critical for the sustainable use of the HBR. Thus, I assume that herding observations are a reflection of the household's overall commitment to managing grazing resources in the HBR sustainably. A measure of herding effort was created from the household's time allocation observations in which at least one member of the household over 6 years of age was observed herding.[59] The ***herding labor*** variable dichotomizes households as "herding" or "not herding" for a particular day where:

$$Y_{ij} = \begin{cases} 1, & \text{if an individual of household } i \text{ is herding at time } j \\ 0, & \text{otherwise} \end{cases}.$$

I assume that $Y_{ij} \sim \text{Binomial}(1,\ p_{ij})$, thus I develop a logistic model of household herding practices, where the probability of herding for a household = p_{ij}. Out of a total of 3270 household observations, 1495 (46%) had at least one individual over the age of 6 herding at the time of observation. Non-herding activity was recorded for the remaining 54% of the observations. This fairly substantial percentage raises concerns for the sustainable use of the HBR given the predominance of the agro-pastoral economy, and the large number of animals that continue to utilize this protected landscape. These non-herding observations prompted me to seek explanations, as they may serve as indicators that problems of collective action may be emerging. The following section presents the explanatory variables that I considered.

Control Variables

Herding practices are undoubtedly influenced by a number of factors other than those of interest to this study. Some variables have direct effects on the household's labor

[59] Herding was one of many possible activities recorded during time allocation. For this analysis, I define herding specifically as active management of animals by at least one member of the household over 6 years of age (individuals younger than this were typically accompanying others). Observations of non-herding do not exclusively imply that households are managing livestock poorly. Most households have crop residues which provide limited foddering resources in the early part of the dry season. Beyond this, reciprocal herding arrangements with neighbors or extended family do occur (Brush and Guillet 1985, Guillet 1980). Even so, given the dearth of such observations in the time allocation dataset, and the observations of community-based and absentee herding discussed in Chapter 2, it is likely that these alternatives do not entirely compensate for all instances of non-herding by the household.

allocation to herding, and others simply confound the effects of variables of interest. A number of theoretically plausible variables were considered for inclusion in the model as controls. Appendix 3.4 lists all the variables I considered and their predicted effects. I considered various demographic and economic characteristics of the household (e.g., household size, dependency ratio); herd demographics (e.g., total, proportion large stock, proportion small stock); climate (e.g. daily rainfall which may have affected a household's decision to herd, season); time (e.g. day of the week, time of day); and community affiliation (e.g. sector, barrio, member of livestock empresa). Of these variables, those that yielded significant Wald-tests or that modified the coefficients or standard errors of the explanatory variable of interest (i.e., tourism) in appreciable ways were included as controls in the following candidate models. Variables meeting these criteria included: the age of the head of the household, the proportion of time a household spends farming, and season. Each of these variables is described below.

Age of the Economic Head of the Household

Empirical evidence, primarily from industrialized nations, shows that individuals are likely to work hardest in midlife (Ghez and Becker 1975). Godoy tested the effects of life-cycle on leisure activity among the Tawakha Amerindians of Honduras, and found that it was a strong predictor of leisure time in rural societies as well (2002). Agro-pastoralists of Collón and Pashpa appear to have similar age-specific work and leisure patterns. The mean age of the economic head of the household is 38 years (stdev = 12.2), ranging from 21 to 80 years of age. As predicted by life-cycle theory, households with economic heads near the mean of 38 years appear to allocate more time and labor to herding. This is presumably because they often have obligations to support a greater number of dependents, and perhaps more available labor than younger or older households at the extremes of 21 years (a very young, newly married couple without children) and 80 (an elderly couple without children living at home). While an interesting effect in and of itself, I include ***head age*** only as a control for household herding labor. Since life-cycle theory predicts that its relationship to herding is quadratic (resembling an inverted-u), mean-centered and mean-centered and squared versions of this variable were included in the model.[60]

Farming

Many studies highlight the complimentarity of agriculture and pastoralism in the Andes (e.g., Browman 1987, McCorkle 1987, Orlove 1980). Combining each allows the household to utilize different ecological niches and to minimize risk by diversifying household production (Valdivia, Dunn, and Jette 1996). Thus, the importance of livestock to agriculture in the Andes cannot be overstated. Animal labor is important for the agricultural tasks of preparing, planting, and harvesting, as well as for transport of agricultural products to market. In addition to labor, animal dung is an important input to maintain the fertility necessary to sustain cultivation on the marginal soils of the Andes (Winterhalder, Larsen, and Thomas 1974). Livestock also offer a form of savings on the hoof, and means to capitalize on the marginal productivity of the highlands; not only spatially, by utilizing extensive *puna* resources above the limits of cultivation, but also

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[60] Mean centering reduces the correlation between the linear and quadratic terms and does alter model coefficients.These terms must be interpreted together in the output of the logistic regression.

temporally, by continuing to graze in the dry season when fields (the majority without irrigation) are not often planted.

Even though most households primarily plant only once a year, the importance of agriculture to households of Collón and Pashpa is apparent in the proportion of agriculture-related observations in the TA data. The proportion of time each household spent farming ranged from 0.05–0.65, with a mean of 0.30 (stdev = 0.12). Given the relative dependence of one on the other, it is reasonable to imagine that the proportion of time spent farming may be positively correlated to the household's herding effort. Although causality is difficult to establish, I assume that households heavily involved in agriculture will have a greater need for livestock and their services (e.g., manuring, plowing, transporting), and a greater need to herd them. Thus I include household ***farming labor*** as a control, with the expectation that it will positively affect the response. This variable is derived from the time allocation dataset by dividing the sum of the occurrence of agricultural observations for the household by its total number of observations.

Season

The seasonality of many household activities is marked in the tropical Andean environment. Dry season observations account for 41% of the observations in the time allocation dataset. As shown in the time allocation graphs of Chapter 2, most household activities (e.g., subsistence, market) are strongly seasonal. Thus I include the effects of season in the model. The categorical dummy variable ***dry season*** defines dry season months as May–August. This variable is included in the model, while its predicted effects on other model variables are discussed in greater detail below.

Seasonal Trends in Farming

The agricultural calendar bears an obvious relation to the season. The wet season, when precipitation and frequent frosts are not limiting factors for cultivation, is known as the *campaña grande*. Agricultural labor peaks at the beginning and end of the *campaña grande* during planting and harvest. The dry season months of May through August separate the harvest of one year's *campaña grande* in April, followed by the planting of the next in September. The dry season is referred to in the diminutive as the *campaña chica*. Very few households plant crops in the *campaña chica* due to its relatively low productivity and the increased occurrence of nighttime frosts. Thus, there is an observable lull in agricultural labor over the dry season months (refer to Figure 2.5). In the dry season when the labor demands of agriculture are at their lowest, it is feasible to assume that there is less conflict in the production calendar and more household laborers available for the task of herding. Thus, I predict that the dry season will act as a positive effect modifier for farming. In other words, the greater the proportion of time spent farming by the household, the more it should herd. Beyond this, I predict that such households are more likely to herd in the dry season when there are fewer conflicts in the production schedule. This logic justifies the inclusion of ***dry season*** as a control variable in the model, as well as the inclusion of an interaction term between it and ***farming labor***.

Seasonal Trends in Herding

Herding is also seasonal. The productivity of natural vegetation and the availability of livestock forage declines in the dry season at lower elevations. With regard to its effect on the model response, one could expect that the dry season requires a greater labor investment as animals are moved to higher pastures in search of suitable forage.[61] This interpretation would be consistent with the dominant, traditional strategy of *transhumance* practiced in the community. Given this herding practice, I predict that dry season by itself will have a positive effect on the response. However, as shown in Chapter 2, other herding strategies are emerging in the community (see Table 2.3 for a summary). Community-based and absentee herding strategies require less household labor in the dry season. I speculate that these newer strategies may be linked to the household's involvement in tourism, which is highly seasonal as well. Given the emergence of these new herding practices, and the concurrence of the peak tourism season with the dry season in the HBR, the effects of season on herding become less clear, but are no doubt, important to consider. I attempt to look into this issue further by defining the household's involvement in tourism as my explanatory variable of interest.

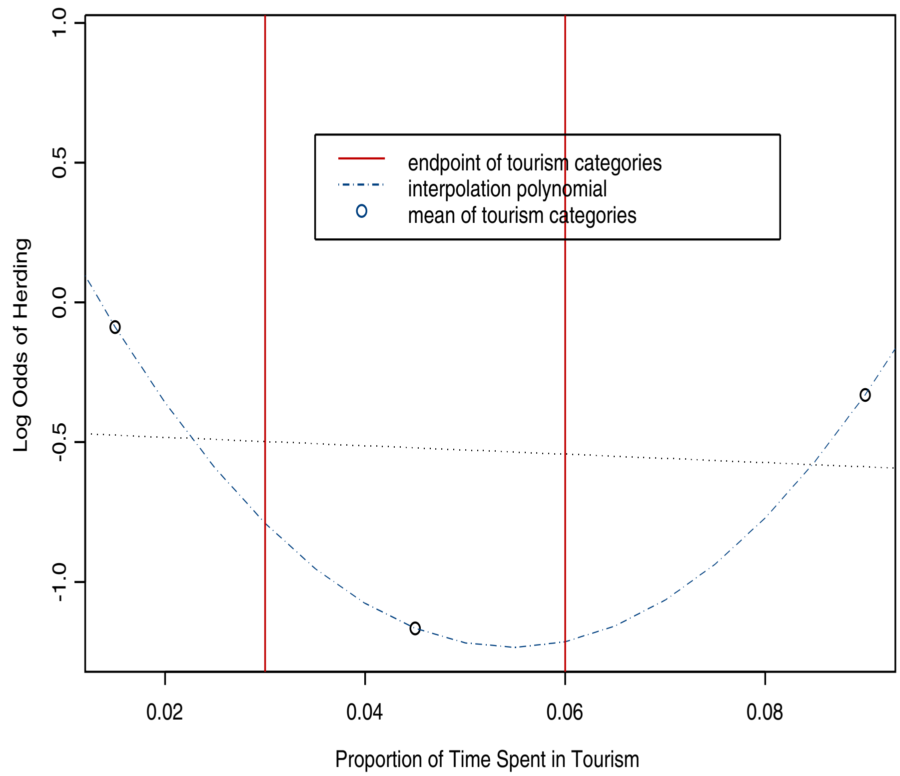
The Explanatory Variable of Interest: Household involvement in tourism

An *a priori* interest in tourism guided the selection process for an additional explanatory variable. Given the uncertainty of tourism's effect on the model response, and the lack of previous empirical observations or studies to inform a prediction, I plotted the model response against the proportion of time households spent in tourism-related work throughout the year of fieldwork. The interpolation polynomial in Figure 3.4 illustrates a weak trend. Categorizing this data into low, moderate and high ranges of tourism involvement captures a strong non-linear trend. Thus the variable ***tourism involvement*** represents different levels of engagement in the tourism industry. This

[61] Alternatively, one could argue that labor in the wet season might be just as high. During certain wet-season months animals must be carefully monitored to prevent them from wandering onto planted fields and destroying crops. Overall, it is clear that herding is a year-round effort that requires a constant commitment from the household.

variable divides households into low (0–3%), moderate (3–6%), and high (6–12%) levels of tourism involvement based on the proportion of the household's time allocation observations in which tourism-related work by at least one of its members was recorded. These observations were predominately of *arriero* work, while few were of work as a cook or porter on various climbing expeditions. Eighty-one percent of the households in the time allocation sample have none to low levels of involvement in tourism; while the remaining 19% are divided equally between moderate and high levels of tourism involvement. This variable was treated as a factor in the model, with contrasts in herding labor for households of moderate and high levels of tourism involvement set relative to those with the lowest levels of involvement. Competing hypotheses about the effects of this increased involvement on household herding practices are discussed below.

Figure 3.4: Plot of the functional form of household herding labor relative to continuous and categorical versions of the tourism predictor only.



Opportunity Cost

Common property literature abounds with the study of conditions under which communal management is prone to failure. Such studies often include explorations of the effects of market integration on commons users (e.g., Agrawal and Yadama 1997, Becker and Leon 2000, Jodha 1996, Lambin et al. 2001, McCay and Jentoft 1998, Ostrom 1992). The rapid growth of the tourism industry in the HBR could potentially have the negative effect of drawing labor away from important management tasks or creating pronounced wealth inequalities that introduce divergent interests and discord among parties responsible for managing grazing commons in and around the Ishinca valley. The concept of opportunity cost provides a mechanism by which these failures of management may occur. Opportunity costs are defined as "the value of some alternative forgone because resources are invested in [one activity] rather than the alternative" (Winterhalder 1983).

In a predominately subsistence-based economy there are fewer competing uses of time than in mixed economies. Before policies that sought to encourage the growth of the tourism industry in the HBR, residents of Collón and Pashpa had little else to compete with the production tasks of farming, herding, and foraging. Now lucrative wage earning opportunities play a stronger role in household decision-making. Once alternative uses of time exist, especially ones as profitable as tourism, opportunity costs rise. As opportunity costs rise, there is an increased utility in having time free to pursue wage-earning activities. This feasibly affects household decisions regarding how much time they are willing to spend herding and who they are willing to offer up for it.

Chapter 2 describes the overall trends of tourism involvement in the study community. Here, it suffices to say the dry season months of May through August coincide with the tourism season, thus a household's opportunity costs are high in the months when climbing and trekking work is more likely. Since the activities of individuals are mutually exclusive, a glance back at Figure 2.8 can serve to illustrate that a substantial number of men, and to a lesser extent women, forgo subsistence tasks in favor of wage-earning activities in the dry season. This is no great surprise considering the wage of 15 USD/day for *arriero* work compared to an average income from *jornal/peon* work of 2–3 USD/day. Due to the direct relationship between tourism involvement and opportunity cost, I use this variable as a surrogate for this fairly new constraint on household decision-making.

The opportunity cost hypothesis predicts that herding will be negatively effected by the proportion of time households spend in tourism because of the additional constraints to herding realized by these households Any household with significant wage-earning potential is likely to have higher opportunity costs than a non-tourism household. Beyond this general assumption, those households with more than one individual working as an *arriero* (e.g., father and son) or many pack animals that can work simultaneously are expected to have higher opportunity costs than those with just a single individual or animal to perform the work. Although herding is primarily the task of individuals other than *arrieros* (i.e., women and children), when *arriero* work is a possibility for the men of the family, livestock that can perform this work must remain nearby. This situation results in an additional burden for herders, given the fact that many households lack sufficient labor to divide their herd and manage them separately (Azhar-Hewitt 1999, Talle 1988). Such a burden may often result in staking and foddering the household herd in the community, or in letting some of these animals go untended. Thus, I predict that households with the highest levels of tourism involvement are likely to herd less, with unsustainable outcomes. I hypothesize that this effect will be more prominent over peak tourism months when the potential for earning wages and the conflict with herding animals to the *puna* are at their height.

Beyond this prediction, the opportunity costs associated with education may influence household herding practices as well. Children are more likely to go to school when the household has sufficient income to send them. I view the option to attend school as an opportunity cost disproportionately affecting child herders of wealthier households. Regardless of the specific opportunities considered by the household (e.g., work as *arriero* vs. herd, go to school vs. herd), the logic of opportunity cost predicts less active herding by wealthy households than that predicted by the herd wealth hypothesis discussed below. An exploration of the opportunity cost hypothesis requires including the ***tourism involvement*** variable in the model, as well as an interaction term between it and the ***dry season*** variable.

Herd Wealth

The "conservation with development" paradigm offers as more optimistic view of the effects of tourism, the assumption being that economic development and poverty alleviation offer a means to achieving conservation. Many conservation efforts in the developing world reflect this assumption, including those of the HBR. As shown in Chapter 2, the wages earned from tourism and the incentives it creates to own packstock (i.e., horse and mule) have resulted in an overall increase in herd size within the study community. This is an unintended outcome for policy makers promoting tourism as a way to offset reliance on resources in the national park. Yet game-theoretic models from human ecology suggest the possibility that increasing herd wealth may actually promote the conservation of these resources by increasing the household's dependence on them and creating a vested interest in their continued productivity.

A largely theoretical, but empirically supported game theoretic model by Ruttan and Borgerhoff-Mulder (1999) shows that herd wealth creates a larger dependence on grazing commons, therefore a greater return on the labor relatively wealthy households will invest in managing their herds so as not to degrade them. Since conservation is assumed to be costly (Alvard 1993, Smith and Wishnie 2000), this logic creates an economic rational for wealthy herders to invest in the labor intensive practice of herding animals sustainably (e.g., moving animals off of critical dry season reserves like *bofedales* in the wet season, or allowing pastures in the community to rest that are otherwise heavily utilized in the wet season). The predictions of their game theoretic model have support from ethnographic observations of herding behavior among Barabaig herders in east Africa (Borgerhoff-Mulder 1991) Maasina Fulβe herders in the Sahel (Turner 1999) and highland herders in the Andes (Preston 1998). Similar investments in herding labor could be predicted for herd wealthy households of Collón and Pashpa despite their growing involvement in tourism and their increasing opportunity costs. Although the inverse of this logic suggests that relatively herd-poor households are more likely to defect (in this case herd less), the authors find that by including mechanisms of enforcement and coercion, their cooperation can be expected if inequalities in herd wealth are not too pronounced.

Herd wealth is correlated to the household's involvement in tourism. Figure 3.5 illustrates the strength of the correlation between tourism involvement and household stock equivalents. An initial test of the homogeneity of variance in herd wealth across different levels of tourism involvement indicates that variances between tourism categories are not significantly different from one another (Levene's test = 0 .0562, Bartlett's test = 0.0741). An anova test of the mean herd wealth across tourism categories shows that there is a significant difference in herd wealth between these categories ($p > |t| = 0.0031$), while decomposition of the tourism sum of squares into linear and quadratic trends illustrates that this trend is significantly positive with increased tourism involvement (p > |t| = 0.0260). Results were similar for a test of the geometric means (log transformed to correct for unequal variances) and medians for tourism categories. Collectively, these statistics support my use of the household's involvement in tourism as a surrogate for its opportunity cost as well as its herd wealth.

The herd wealth hypothesis suggests that tourism involvement will have a reciprocal effect on herding from that predicted by opportunity cost. If herd wealth exerts a significant effect on household decision-making, then herding should be positively affected by the household's involvement in tourism, as opposed to the negative effect predicted of opportunity cost. Herd wealthy households should continue herding in the face of increasing transaction costs because they have more animals dependent on the commons, and more to gain from investing in their careful management. Furthermore, I predict that herding labor will be more pronounced for herd wealthy households relative to those that are herd poor in the dry season when the impacts of unsustainable practices like staking animals in the community or leaving them untended are greater. The interaction of ***tourism involvement*** with ***dry season***, in this case, should capture increases in the cost of herding in the dry season that differentiate household decisions regarding herd management.[62]

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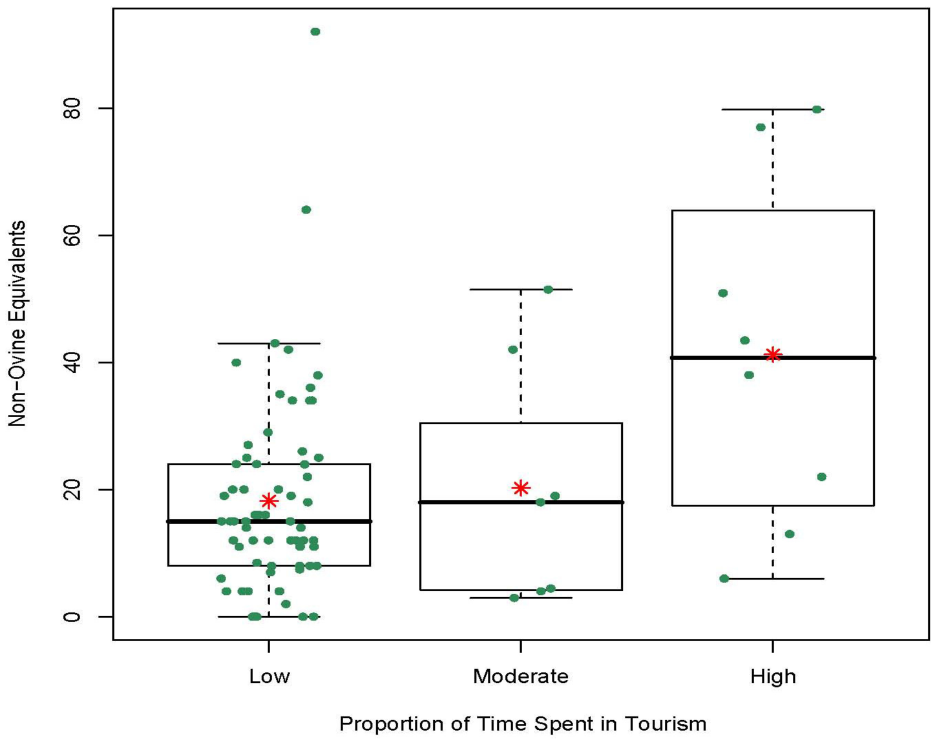
[62] The interaction term between ***tourism involvement and dry season*** is important to include given the hypotheses tested. Both hypotheses suggest that in the dry season, the herding response (positive in the case of herd wealth and negative in the case of opportunity cost) will be magnified.

Figure 3.5: Distributions of herd wealth relative to various levels of involvement in tourism. Sheep were excluded in this calculation because of household outliers in the low tourism category (mean = 12, se = 1.8; three households had 120, 60, and 46 sheep respectively). In addition, sheep are ubiquitous in the household herd, yet they are the only type of animal for which no hypothesized relationship to tourism exists. All other livestock either have utility to serve as packstock (horse, mule and llama), require less labor due to needing less protection from predation (cattle), or produce goods such as wool and meat that can be marketed to tourists in Huaraz (alpaca). I expect these types to be especially responsive to regional growth in tourism. Asterisks (*) denote means. Bars (–) denote medians. The left and right edges of the box represent the first and third quartiles (the middle 50% of the values).



The discussion above illustrates that different aspects of tourism involvement imply very different outcomes for the sustainability of *common property* management in the HBR. Since tourism involvement is correlated with opportunity cost and herd wealth, the strength and direction of the model coefficients of the tourism variable will be used to test these competing hypotheses, as well as confirm the characteristics of households most responsible for influencing their herding practices.

FITTING MODELS OF HOUSEHOLD HERDING LABOR

A formal test of the competing hypotheses of opportunity cost and herd wealth depends on having inter-household variability in herding labor. The modeling effort began with this determination by fitting two simple multilevel models that included the random effects of household membership and season, without any substantive predictors (Singer and Willett 2003). This step is critical in establishing: (1) whether or not there is substantial variation in household herding practices worth modeling, and if so, (2) whether this variation resides in the intercept for each household (i.e., between households), or in the slope of repeated observations on the household over time (i.e., within households).

Model UME is an important baseline and is frequently referred to as an unconditional means model or an intercept-only model which lacks predictors at every level (see Table 3.3a for a taxonomy of all candidate models; Table 3.3b summarizes the models in composite form).[63] Model VCM is a variance components model in which only level-1 (time-variant) predictors are added. ***Dry season*** is the only level-1 predictor in the time allocation dataset (refer to Table 3.2 for a summary). The CONTROL model builds on VCM by adding additional level-2 control variables including ***head age, head age***^{2}, ***farming labor***, and an interaction between ***dry season and farming labor***. TOURISMa and TOURISMb include all the variables of CONTROL plus the level-2

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[63] A simple model whose hallmark is the absence of predictors at all levels. In an ordinary linear mixed model the unconditional means model serves to describe and partition the outcome variation. In a logistic regression mixed model, its primary purpose is to serve as a null model. It is a model that accounts for the data structure, but at the same time is agnostic about the relationship of the response to putative predictors. Thus, the purpose of the unconditional means model is to serve as a baseline model in model comparisons.

predictor of interest, ***tourism involvement***, and its interaction with ***dry season***. The only difference between these models is their assumption of random effects. TOURISMa includes the random effects of household and season, as do all previous models. TOURISMb includes only the random effects of the household.

Table 3.3a: Taxonomy of multilevel models fitted to the herding labor data.

MODEL	LEVELS	RANDOM EFFECTS
UME	Level 1: $\log\left(\frac{p_{ij}}{1-p_{ij}}\right) = \beta_{0i}$ Level 2: $\beta_{0i} = \beta_0 + u_{0i}$	$u_{0i} \sim N\left(0,\ \sigma_0^2\right)$
VCM	Level 1: $\log\left(\frac{p_{ij}}{1-p_{ij}}\right) = \beta_{0i} + \beta_{1i}\ \text{dry season}_{ij}$ Level 2: $\beta_{0i} = \beta_0 + u_{0i}$ $\beta_{1i} = \beta_1 + u_{1i}$	$\begin{bmatrix} u_{0i} \\ u_{1i} \end{bmatrix} \sim N\left(\begin{bmatrix} 0 \\ 0 \end{bmatrix}, \begin{bmatrix} \sigma_0^2 & \sigma_{01} \\ \sigma_{01} & \sigma_1^2 \end{bmatrix}\right)$
CONTROL	Level 1: $\log\left(\frac{p_{ij}}{1-p_{ij}}\right) = \beta_{0i} + \beta_{1i}\ \text{dry season}_{ij}$ Level 2: $\beta_{0i} = \beta_0 + \beta_2 \text{centered head age}_i + \beta_3 \text{centered head age}_i^2 + \beta_4 \text{farming labor}_i + u_{0i}$ $\beta_{1i} = \beta_1 + \beta_7 \text{farming labor}_i + u_{1i}$	$\begin{bmatrix} u_{0i} \\ u_{1i} \end{bmatrix} \sim N\left(\begin{bmatrix} 0 \\ 0 \end{bmatrix}, \begin{bmatrix} \sigma_0^2 & \sigma_{01} \\ \sigma_{01} & \sigma_1^2 \end{bmatrix}\right)$

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MODEL	LEVELS	RANDOM EFFECTS
TOURISMa	Level 1: $\log\left(\frac{p_{ij}}{1-p_{ij}}\right) = \beta_{0i} + \beta_{1i}\,\text{dry season}_{ij}$ Level 2: $\beta_{0i} = \beta_0 + \beta_2\text{centered head age}_i + \beta_3\text{centered head age}_i^2 + \beta_4\text{farming labor}_i$ $\ldots\ldots\ldots\ldots + \beta_5\text{tourism labor1}_i + \beta_6\text{tourism labor2}_i + u_{0i}$ $\beta_{1i} = \beta_1 + \beta_7\text{farming labor}_i + \beta_8\text{tourism labor1}_i + \beta_9\text{tourism labor2}_i + u_{1i}$	$\begin{bmatrix} u_{0i} \\ u_{1i} \end{bmatrix} \sim N\left(\begin{bmatrix} 0 \\ 0 \end{bmatrix}, \begin{bmatrix} \sigma_0^2 & \sigma_{01} \\ \sigma_{01} & \sigma_1^2 \end{bmatrix}\right)$
TOURISM b	Level 1: $\log\left(\frac{p_{ij}}{1-p_{ij}}\right) = \beta_{0i} + \beta_{1i}\,\text{dry season}_{ij}$ Level 2: $\beta_{0i} = \beta_0 + \beta_2\text{centered head age}_i + \beta_3\text{centered head age}_i^2 + \beta_4\text{farming labor}_i$ $\ldots\ldots\ldots\ldots + \beta_5\text{tourism labor1}_i + \beta_6\text{tourism labor2}_i + u_{0i}$ $\beta_{1i} = \beta_1 + \beta_7\text{farming labor}_i + \beta_8\text{tourism labor1}_i + \beta_9\text{tourism labor2}_i$	$u_{0i} \sim N(0, \sigma_0^2)$

Note: Where β_0 = intercept, β_1 = slope, u_0 = random effects on intercept, u_1 = random effects on slope, and ρ_{ij} = household probability of herding. As indicated by the subscript i on the parameters β_0 and β_1, households have their own intercept and slope.

Table 3.3b: Composite versions of the models of herding labor shown in Table 3.3a.

MODEL	COMPOSITE
UME	$\log\left(\frac{p_{ij}}{1-p_{ij}}\right) = \beta_0 + u_{0i}$
VCM	$\log\left(\frac{p_{ij}}{1-p_{ij}}\right) = \beta_0 + \beta_1 \text{ dry season}_{ij} + u_{0i} + u_{1i}\text{dry season}_{ij}$
CONTROL	$\log\left(\frac{p_{ij}}{1-p_{ij}}\right) = \beta_{0i} + \beta_1 \text{ dry season}_{ij} + \beta_2\text{centered head age}_i + \beta_3\text{centered head age}_i^2$ $+ \beta_4\text{farming labor}_i + \beta_7\text{dry season}_{ij} \times \text{farming labor}_i + u_{0i} + u_{1i}\text{dry season}_{ij}$

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MODEL	COMPOSITE
TOURISMa	$\log\left(\frac{p_{ij}}{1-p_{ij}}\right) = \beta_{0i} + \beta_1 \text{ dry season}_{ij} + \beta_2 \text{centered head age}_i + \beta_3 \text{centered head age}_i^2 + \beta_4 \text{farming labor}_i + \beta_5 \text{tourism labor mod}_i + \beta_6 \text{tourism labor high}_i + \beta_7 \text{dry season}_{ij} \times \text{farming labor}_i + \beta_8 \text{dry season}_{ij} \times \text{tourism labor mod}_i + \beta_9 \text{dry season}_{ij} \times \text{tourism labor high}_i + u_{0i} + u_{1i} \text{dry season}_{ij}$
TOURISMb	$\log\left(\frac{p_{ij}}{1-p_{ij}}\right) = \beta_{0i} + \beta_1 \text{ dry season}_{ij} + \beta_2 \text{centered head age}_i + \beta_3 \text{centered head age}_i^2 + \beta_4 \text{farming labor}_i + \beta_5 \text{tourism labor mod}_i + \beta_6 \text{tourism labor high}_i + \beta_7 \text{dry season}_{ij} \times \text{farming labor}_i + \beta_8 \text{dry season}_{ij} \times \text{tourism labor mod}_i + \beta_9 \text{dry season}_{ij} \times \text{tourism labor high}_i + u_{0i}$

Collectively, these candidate models provide valuable information about household herding practices. Variance components of the VCM model are non-zero (σ_0^2 = 1.0 and σ_1^2 = 0.3), suggesting variation in herding practices between households, as well as variation across seasons. These statistics provide justification for modeling household herding labor. Additional statistics summarized in Table 3.4 suggests that successive models including a number of additional characteristics of the household (CONTROL, TOURISMa and TOURISMb), explain a significant amount of this variation. For example, a likelihood ratio test confirms that CONTROL is a significant improvement over VCM ($p > |z| = 0.0001$).[64] Continuing with this measure of model comparison, TOURISMa and TOURISMb are stronger models than the CONTROL ($p > |z| = 0.0007$ and $p > |z| = 0.0060$ respectively). In sum, the household's involvement in tourism appears to be an important predictor of its herding practices, even when controlling for the effects of other household characteristics.

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[64] The likelihood ratio statistic, $-2\log\Lambda$, can be used to test for the significance of the predictors in the full model against the reduced model. Where L denotes the likelihood function:

$$-2\log\Lambda = -2\log\left[\frac{L(\text{reduced})}{L(\text{full})}\right] = -2\left[\log L(\text{reduced}) - \log L(\text{full})\right].$$

$-2\log\Lambda$ has an asymptotic chi-square distribution with degrees of freedom given by, df (full model – df (reduced model).

Table 3.4: Evaluation of the candidate models of household herding labor. Model evaluation was done with information theoretic measures. Likelihood ratio tests are shown for comparison. The likelihood ratio statistic, $-2\log\Lambda$, was calculated between nested models and is given on the line of the fuller model. AIC can be used as an alternative way of evaluating candidate models. The lowest AIC indicates the strongest model. Akaike weights reported in the last column are the probability of being the model selected with repeated sampling.

MODEL	LOG LIKELIHOOD	$-2\log\Lambda$	P-VAL	# OF ESTIMATED PARAMETERS	AIC	Δ_i	AKAIKE WEIGHT
UME	-2026.18	X	X	2.00	4056.37	70.31	0.00
VCM	-2019.41	vs. UME: 2.82	0.1689*	5.00	4048.81	62.75	0.00
CONTROL	-1986.67	vs. VCM: 65.48	<0.0001	9.00	3991.33	5.27	0.04
TOURISMa	-1980.57	vs. CONTROL: 13.47	0.0007	13.00	3987.14	1.08	0.35
TOURISMb	-1982.03	vs. CONTROL: 9.27	0.0060**	11.00	3986.06	0.00	0.60

*This p-value suggests that the variance components model (VCM) is not a significant improvement over the intercept only model (UME). *Dry season* is the only level-1 predictor and I retain it in subsequent models despite this statistic; because of its importance to the hypotheses tested, and because it appreciably modifies the other variables of the model (e.g., *farming labor*).

**The likelihood ratio test between TOURISMb and CONTROL requires a different assumption about the distribution of the likelihood ratio statistic from that cited in the footnote on the previous page. For a likelihood ratio test between nested models with a different number of random effects, a mixture distribution of χ_0^2 and χ_1^2 is required (Verbeke and Molenberghs 2000), such that the p-value of the test is given by:

$$p = \frac{1}{2}P\left(\chi_0^2 > \Lambda\right) + \frac{1}{2}P\left(\chi_1^2 > \Lambda\right).$$

At this point, I introduce the use of information theory as a way of alternatively ranking the candidate models. Akaike's Information Criterion (AIC) is useful for model comparison because it is likelihood-based, but unlike likelihood ratio tests, can be used to compare models that are not nested (Burnham and Anderson 2002).[65] Proponents of one method of model evaluation would argue against the use of the other, thus they are usually not presented together. I do so here given the relative novelty of information-theoretic analysis within my discipline, permitting the reader to follow model evaluation using the criteria they feel most familiar with. The AIC statistic is defined as:

$$\text{AIC} = -2\log L\left(\hat{\theta}\right) + 2K$$

[eqn. 3.9]

In the equation above, the log likelihood function for a given model is evaluated at the maximum likelihood estimate of the parameter set θ. K is the number of parameters estimated in the model. These quantities are multiplied by 2, an arbitrary value chosen by the equation's author, to yield a value that can be used to compare models against one another. In general, models with smaller AIC values are better models of the response. Akaike weights translate as probabilities of being the model that would be selected with repeated sampling. I use this measure as my criteria for model

[65] The advantages of AIC are easy to imagine when considering the possibility that competing hypotheses may be best represented by different predictors, which creates the need to evaluate models that are not nested. In this case, my competing hypotheses of opportunity cost and herd wealth were best represented by the same tourism variable. Although I could rely on a test of the likelihood ratio statistic alone, I retain the information theoretic measures as a way of illustrating their ease of use and their versatility for model comparison.

selection, which shows that the TOURISMb model would be selected 60% of the time.[66] Note that the second place model (Akaike weight = 0.35), also includes the effects of the household's involvement in tourism. Thus, the model I select for discussion below can effectively be written as follows:

$$\log\left(\frac{p_{ij}}{1-p_{ij}}\right) = \beta_{0i} + \beta_1 \text{ dry season}_{ij} + \beta_2 \text{centered head age}_i + \beta_3 \text{centered head age}_i^2$$
$$+ \beta_4 \text{farming labor}_i + \beta_5 \text{tourism labor mod}_i + \beta_6 \text{tourism labor high}_i$$
$$+ \beta_7 \text{dry season}_{ij} \times \text{farming labor}_i + \beta_8 \text{dry season}_{ij} \times \text{tourism labor mod}_i$$
$$+ \beta_9 \text{dry season}_{ij} \times \text{tourism labor high}_i + u_{0i}$$

[eqn. 3.10a]

Where, I assume only random effects for the household:

$$u_{0i} \sim N\left(0,\, \sigma_0^2\right)$$

[eqn. 3.10b]

RESULTS

The discussion of results focuses on interpretation of the model specified by equations [3.10a] and [3.10b]. Figure 3.6 displays the coefficients of a Bayesian version of the model graphically, while a table of coefficients is provided in Appendix 3.5.[67] The direction and relative strength of the effect of each predictor can be compared in Figure

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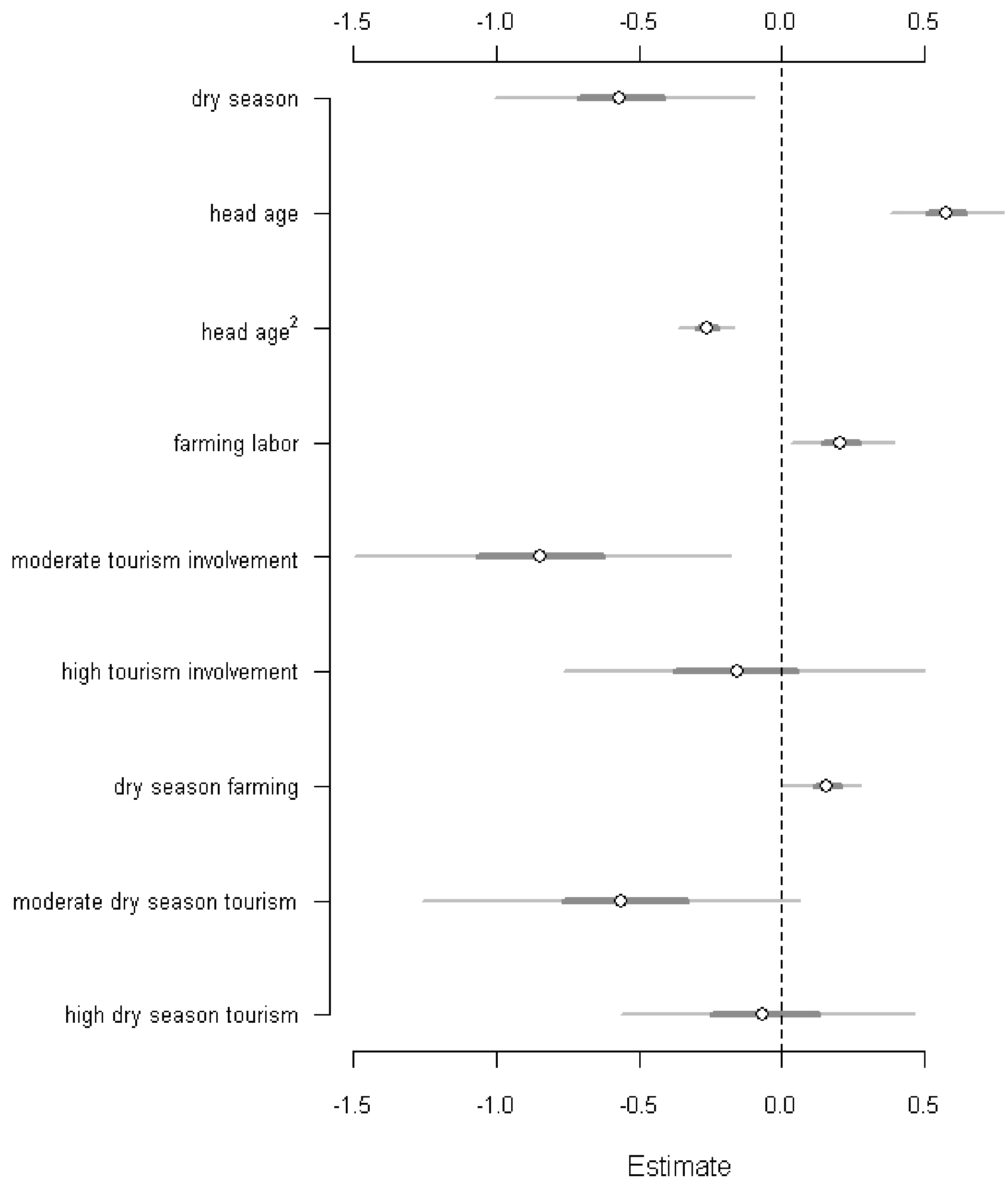
[66] Statistical tests confirmed the need to include the random effects of household and season before modeling began (see Appendix 3.3). However, a final comparison of AICs for TOURISMa and TOURISMb suggested that only the random effects of the household were needed once additional level-2 predictors had been added. Thus I selected the model with fewer estimated parameters for discussion.

[67] An exploration of the assumption of normality is presented in Appendix 3.6.

3.6. For example, the proportion of the time households spent in agriculture had a positive effect on herding (i.e., the estimated coefficient of ***farming labor*** is less than zero). The effect of ***head age*** is parabolic. Specifically, the quadratic term ***head age***^{2} is negative, indicating that the parabola opens downward. These effects were expected, as discussed in the section describing each of the control variables. More importantly, this figure illustrates that ***tourism involvement*** has a negative effect on herding. The sign of this effect lends support to the opportunity cost hypothesis, while reciprocally rejecting the hypothesis of herd wealth, a point I return to in the discussion.[68] The estimates obtained for the interaction of ***tourism involvement*** with ***dry season*** suggests that the months of peak tourism activity (May–August) result in a stronger negative effect on herding. Although the effect is more pronounced at moderate levels of tourism involvement, the possibility that tourism may undermine sustainable herding practices for such households in the wet season, dry season or both, is concerning.

[68] Further support for this interpretation comes from a series of attempts to define herd wealth independent of tourism involvement. Regardless of the herd wealth regressor, none produced significant coefficients indicating a positive effect on herding in the dry season. In fact, the only models with statistically significant coefficients associated with herd wealth produced trends suggesting that herd wealthy households herd less (see Appendix 3.7).

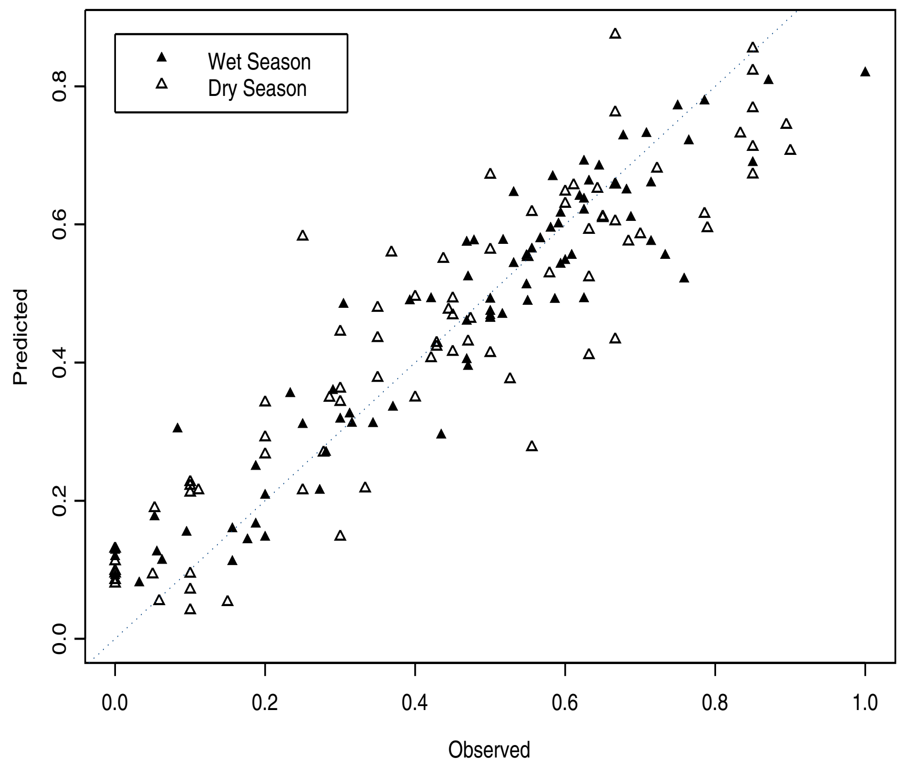
Figure 3.6: Estimates, 50% and 95% credibility intervals for the regression coefficients of predictors in the multilevel model of household herding labor. Parameter estimates were obtained within a Bayesian framework using non-informative priors and are the estimated means of the posterior distributions of the parameters. The interval endpoints are the estimated 0.025, 0.25, 0.75, and 0.975 quantiles of the corresponding posterior distributions. These intervals can be treated as probability statements about the true values of the parameters given the data.



This dissertation was motivated by exploring many unintended effects of the tourism industry, some of which were suggested by the discussion in Chapter 2. There was little doubt at the beginning of the modeling effort that this industry would exert a measurable effect on the household economy. In this chapter, I attempted to quantify the effects of tourism on the herding behaviors of households utilizing grazing commons in the HBR. The model discussed in this chapter considers the household's level of involvement in tourism, along with a small suite of controls including the age of the economic head of the household, its commitment to farming, and season. This model explains 65% of the variance in herding labor between households.[69] The tourism predictor alone accounts for 40% of this variance. Figure 3.7 plots the model's predictions of household herding behaviors in the wet and dry season against observations from the time allocation data, suggesting fairly strong agreement.

[69] The pseudo R-Square statistic can be used to summarize the proportion of variance in the response explained by the inclusion of additional predictors. For multilevel models it must be calculated separately for both random intercepts and random slopes if they exist. The TOURISMb model has only random intercepts, therefore only a level-2 variance was calculated. This value can be interpreted as the percent of variance between households that is explained by the fuller model.

Figure 3.7: Observed vs. predicted probabilities of herding by the household in the wet and dry season.

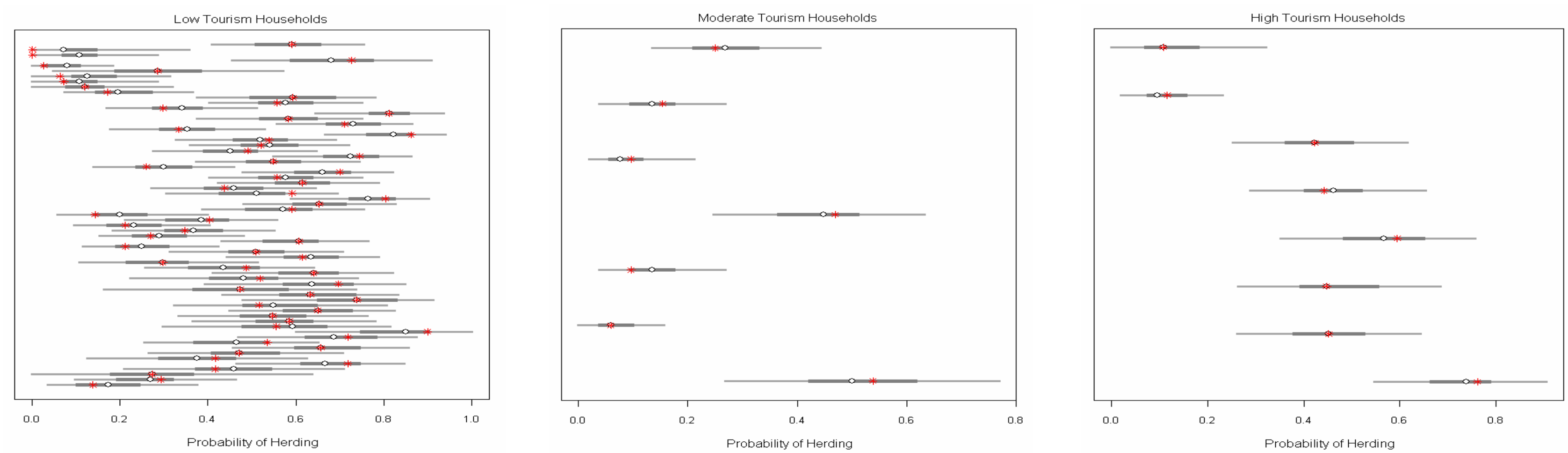


Using Predictive Simulation to Evaluate the Household Herding Model

Predictive simulations at the household level were also run to more formally explore model fit. There are not set rules for predictive simulation, with the exception of choosing a comparison that is relevant to the inferences one attempts to make. Given the hypothesized effects of tourism on household herding practices, I ran simulations that utilized the posterior distributions generated by the Bayesian version of the TOURISMb model summarized in Appendix 3.5b (and plotted in Figure 3.6) to simulate the probability of herding throughout the year for every household. Model simulations can then be compared to actual observations of herding to assess whether the observed datum appear to be a typical member of the simulation set. If so, this can be taken as evidence in support of the model.

Figure 3.8 plots the probabilities of herding for every household, grouped by level of tourism involvement. Taken collectively, herding observations simulated by the model are in agreement with actual time allocation observations for 95% of the sample households. In other words, simulated probabilities of herding throughout the year agree with actual probabilities of herding for 76 of the 80 households. Note that the 4 households whose actual herding behaviors are not predicted well by the model (i.e. fall outside of the 95% credibility interval of the simulated value) are all households with low levels of tourism involvement. In every case, the actual probabilities of herding by these households at least fall within the 50% credibility interval of the simulated value. This predictive simulation, though just one of many possible analyses, provides preliminary evidence that the model does a decent job of capturing household-level herding responses. An evaluation of the model outliers does show a slight bias with regard to tourism involvement; all are of the same tourism class. Fortunately, no simulation outliers exist among the tourism categories I highlight in this study. Accurate model predictions for household of moderate and high tourism involvement help validate the claim I make below that the herding practices of households with moderate tourism involvement, in particular, are significantly different from the others.

Figure 3.8: Household probabilities of herding grouped by level of tourism involvement. Open circles represent estimated probabilities of herding, red asterisks (*) represent actual probabilities calculated from time allocation observations. Smears represent 50% and 95% credibility intervals for the estimated probabilities. These intervals can be treated as probability statements about their true value given the data.



The Odds of Herding Relative to Tourism Involvement

Although most often reported, the log odds plotted in Figure 3.6 and summarized in the typical output of logistic regression (see Appendix 3.5) bear little resemblance to reality. Log odds can be transformed into odds, a practice common in disciplines such as epidemiology, but not in ecological studies. Odds provide a meaningful interpretation of the effects tourism on household herding practices. Table 3.5a shows the odds of herding for households, as affected by their tourism involvement, a measure referred to generally as an odds ratio.[70] This table presents different measures for the fixed and random effects of the full model. For fixed effects, I calculate a subject-specific odds ratio, which can be interpreted as a change in the odds of herding occurring within a single household, and what I am calling a marginal odds ratio, a population-averaged version which can be interpreted as a difference in the odds of herding between households. For the random effects of households, I calculate a median odds ratio, where a value greater than 1 indicates statistically significant differences in the odds of herding between households (see Table 3.5b).

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70 In a multilevel modeling framework, odds ratios are complicated by the random effects of each level. The results presented in Tables 3.5a and 3.5b are based on recommendations for dealing with these complexities during model interpretation (Anderson and Burnham 2002).

Table 3.5a: Odds ratios for variables of interest in the best multilevel model of household herding labor.

ODDS RATIO	PARAMETER	SUBJECT-SPECIFIC*	MARGINAL**
Wet Season			
Moderate vs. low tourism	$\exp(\hat{\beta}_1)$	0.42	0.45
High vs. low tourism	$\exp(\hat{\beta}_2)$	0.85	0.86
Dry Season			
Moderate vs. low tourism	$\exp(\hat{\beta}_1 + \hat{\beta}_4)$	0.24	0.27
High vs. low tourism	$\exp(\hat{\beta}_2 + \hat{\beta}_5)$	0.82	0.83

*Subject-specific odds can only be interpreted between households if random effects are identical. Since this is unlikely, the only interpretation of the subject-specific odds ratio in this case, is the difference in the odds of herding for a household of that may change as its level of tourism involvement changes. Note that these values are similar to the marginal odds between households which I focus on in the text.

**Marginal odds were calculated in order to compare the odds of herding between households. This is a population-averaged version of the standard odds ratio, adjusted by the average random effect of households. The marginal odds ratio can be interpreted as a difference in the odds of the response between households. Confidence intervals cannot be calculated for the marginal odds ratio without the use of a bootstrap technique, not explored here as it is currently experimental.

Table 3.5b: Median odds ratio for the best multilevel model of household herding labor.

PARAMETER	MEDIAN*
σ^2	6.06

*Median Odds Ratios >1 = significant variation in the odds of herding between households.

In Table 3.5a, all marginal odds ratios are less than one. This indicates that households with moderate and high levels of tourism involvement are less likely to herd than those with the lowest levels of involvement. Figure 3.6 shows that moderate levels of tourism involvement produced the strongest negative effects on herding. In general, households with little to no involvement in tourism are approximately 2 times more likely to herd than those that are moderately involved.

Differences in herding exist across seasons as well. Dry season odds ratios show that households with low levels of tourism involvement are 4 times more likely to herd from May–August than those with moderate tourism involvement (1/0.24). Odds ratios are not restricted in the values they can take; thus these statistics may seem trivial. An example will highlight that they are not. There are approximately 123 days of herding in the dry season months of May through August. This is a time when heavily utilized pastures in the community are prone to degradation if not rested. This requires actively moving animals off of community pastures and into the higher pastures of the *puna*. Being 4 times more likely to herd, this odds ratio implies that households with little to no involvement in tourism may manage their herds all 123 days of the dry season, whereas a household moderately involved in this industry may actively herd their animals only 31 days of it (123/4). On the remaining 92 days in which this household does not herd its animals, alternatives include making reciprocal herding arrangements, staking animals in the community, or practicing a strategy of absentee herding which involves moving livestock into the park and leaving them unsupervised for an extended period of time. Given the dearth of reciprocity-based herding observations in my time allocation dataset, and the potential for the breakdown of such arrangements with increasing heterogeneity, a good proportion of these non-herding observations are likely to result in a concentration of livestock on sensitive sites. This result posits a mechanism by which emerging absentee and community-based herding practices discussed in Chapter 2 can be explained, as well as means of identifying the household characteristics most strongly associated with them. Collectively these results suggest that the changes brought about by involvement in tourism have had a disproportionately negative effect on household herding practices, with potentially negative outcomes for the conservation of the reserve. This would certainly be an unintended consequence of encouraging tourism in the HNP.

DISCUSSION

A suite of easily measured variables that offer reasonably accurate predictions of household herding behaviors is useful in itself. However, the primary objective of this analysis was to understand the more ambiguous direction of the effect of tourism and its implications for the management of common pool resources in the HBR. Tourism involvement creates a number of changes for the indigenous agro-pastoral household, thus a number of challenges for the community institutions that coordinate them. With regard to the household's ability to negotiate the continued demands of subsistence, some of the changes wrought by tourism may be beneficial while others may be harmful. This study focused on two aspects of tourism involvement that have very different implications for conservation in the HBR. Opportunity costs, as affected by tourism involvement, were seen as a mechanism that would undermine sustainable use of the commons by negatively affecting the herding practices of households most involved in tourism. On the other hand, game theory predicts that their relative herd wealth may have a positive effect. Unfortunately, the model supports the opportunity cost hypothesis. Households involved in tourism spent less time herding, thus are more likely to rely (at least occasionally) on unsustainable herding alternatives like staking animals in the community or leaving them untended in the national park. This finding suggests that the game theoretic model proposed by Ruttan and Borgerhoff-Mulder (1999) should be modified to include the additional costs that confront wealthier households, many of which are relatively wealthy by community standards because of their involvement activities that reduce household labor. Including the opportunity costs of herding for households engaged in off-farm work, but still predominately dependent on their own labor, would likely diminish the region over which conservation is predicted of them.[71]

A secondary claim advanced by the game theoretic model is that cooperation can be coerced of the relatively herd poor if inequality is not too pronounced (*ibid* 1999). In this case, it seems we need a mechanism for coercion of the herd-wealthy by the herd-poor, a theoretically untenable situation. Predominately subsistence-based households have less power, and are less likely to have the means to offer the incentives necessary to maintain the cooperation of a small group of tourism households. It is feasible that households with the highest levels of tourism involvement, also the most wealthy, may be more successful in coercing these households into cooperation. Yet, the few households to which this applies have effectively reduced their dependence on natural forage despite their greater herd wealth. The result is that they are less likely to invest in the effort required to encourage others to cooperate in the management of a resource upon which they no longer as heavily depend. This is a dilemma that will require careful attention and support from all parties involved in achieving sustainable tourism and sustainable land use in the HBR.

[71] In Appendix 3.8, I explore additional reasons that my empirical observations don't support this theoretically intriguing possibility.

These findings do not support tourism in its current form as a development option conducive to conservation in the HBR. The significant negative effects of moderate tourism involvement on household herding practices are alarming considering the current perceived crisis of over-grazing in the HBR. In Chapter 2, I show that tourism involvement has created an incentive to own more animals, as well as the income to actually make such purchases (also see Figure 3.5). This suggests that tourism has increased the household's investment in livestock, but how does this investment translate to the management of *common property* in the HBR? At moderate levels of tourism involvement, households have more animals, but still rely predominately on natural forage and household labor. The analysis presented shows that such households are devoting less labor to the management of a growing herd.

This pattern of non-herding by households with moderate involvement in tourism warrants discussion for a number of theoretical and practical reasons. First, the u-shaped response of herding with increasing involvement in tourism mirrors the predictions of the Environmental Kuznets Curve (Panayotou 1992). A study of the effects of market integration on the extraction of non-timber forest products in the Amazon showed that use of these common pool resources, thus the potential to abuse them, is heightened at moderate levels of market involvement (Godoy 2001, Godoy, Brokaw, and Wilkie 1995). The effects of moderate levels of market integration seem to apply equally as well to the herding behaviors of Andean agro-pastoralists. The practical implication for managers of the HBR is that preliminary forays into the tourism industry may actually intensify the use of grasslands in this protected landscape, and the possibility of their abuse.

Even so, sweeping generalizations must be avoided. While increasing opportunity costs appear to offer an explanation for my observations of unsustainable herding practices such as absentee herding, these costs do not seem to similarly affect households with the highest levels of involvement tourism. These households show an improved herding response (compare the effects of each in Figure 3.6). Although a larger herd suggests that their investment in the pastoral economy is high, the impacts of this herd are somewhat offset by the income generated from the household's wage earning activities. These households may achieve sustainable herd management by: (1) hiring non-household members to herd for them, (2) purchasing fodder, or (3) buying access to improved, fenced pastures in the community that minimize spatial disjunctions between herding and opportunities to encounter tourists seeking the services of an *arriero*.[72] The last option is responsible for many of the herding observations I recorded for high-tourism households, unfortunately it was only an option in Collón. Households of this sector can manage a relatively large herd without disproportionately burdening the pastures of the community or those of the highest reaches of the Ishinca valley because of recently implemented pasture improvement projects. Existing projects required households to pay for access to improved and fenced pastures, a situation more likely with higher wage-earning potential. Providing similar opportunities for households in Pashpa, and making these pastures accessible to households of either sector with less

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[72] This is an option only for households of Collón, as several pasture improvement projects were newly developed in that sector with the assistance of The Mountain Institute (TMI 2001b). Implications of supporting only a proportion of members in a common property user-group are explored in Chapter 2.

disposable income, is an important recommendation for encouraging sustainable herd management in the HBR.

CONCLUSION

Given the importance of the *puna* to the region's subsistence economy, its geographic inclusion in a protected area, and the ecological and economic rational for maintaining it as a commons, understanding *common property* management and the factors affecting it become of critical import to the success of conservation efforts in the reserve. A number of households in the Ishinca valley *user-group* have increased the size of their herds while simultaneously herding them less. I have shown that this is largely the result of the household's level of involvement in tourism. Moderate involvement in tourism is significantly correlated with decreased odds of herding in the wet and dry seasons. This may partially explain the emergence of unsustainable herding practices such as absentee herding, which involves the year-round use of the conservation core (the HNP). Beyond the immediate impact of such practices, growing wealth inequalities and divergent interests could further exacerbate problems of collective action and undermine cooperation in the management of resources in and around the Ishinca valley.

While this situation suggests that a potential tragedy is brewing in the HBR, my analysis has also shown that households with the highest levels of tourism involvement are maintaining their herding effort (at least as much as those with little involvement in the industry). This finding is encouraging considering the projected growth of tourism in the region, and the possibility that it could proceed more sustainably in the future. Unfortunately, tourism has not been an industry that all community members have equally benefited from to date. This is especially obvious in the number of comments made by community members that there are currently too many *arrieros* and not enough tourism opportunities to go around (e.g., see quote at the beginning of the chapter). It is hard to say whether or not the majority of households in Collón and Pashpa will ever surpass more than occasional involvement in tourism. Considering the negative effect this has on household herding practices, the conservation community faces a substantial challenge in achieving sustainable tourism development and sustainable herd management in the HBR. Much effort has recently been invested into working toward this outcome (e.g., INRENA 2002, Ramirez 2002, TMI 2001a). It is hoped that the results of this study will contribute to this effort by drawing out the connections between conservation and development, and by exploring the specific ways in which the development unfolding in the HBR affects household decision-makers and their use of this protected landscape. Now that I have quantified the effects of tourism on the herding practices of the region's households, the remaining question I explore in Chapter 4 is what the impacts of grazing have been, and what they might be given the changing herding practices suggested by this analysis.

CHAPTER 4

THE IMPACTS OF GRAZING ON NATIVE PLANT DIVERSITY IN THE HUASCARÁN BIOSPHERE RESERVE, PERU

ABSTRACT

The Huascarán Biosphere Reserve (HBR) is a tropical Andean protected area managed for "conservation with development," one that is experiencing the common challenge of achieving both. The objective of this study is to test a prevailing assumption that the reserve is over-grazed by quantifying native plant diversity relative to long-established patterns of grazing by resident agro-pastoralists in and around the Ishinca valley. I fit multilevel models from vegetation samples, community surveys, informant reports, and topographic analyses conducted using a Geographic Information System (GIS). The integration of statistical modeling and GIS creates a powerful tool for quantifying the role of humans in shaping this landscape, and for projecting the outcomes of changing land use and resource management scenarios for biodiversity conservation in the HBR. My findings suggest that moderate grazing intensities in the Ishinca valley have helped maintain native species richness. Yet the changing land use scenarios associated with tourism involvement may have negative and irreversible effects. Of particular note is absentee herding, a strategy that has led to increased livestock presence within the reserve's conservation core, the Huascarán National Park (HNP). The predictions of the multilevel model developed herein suggest that emerging herding practices such as these may result in a reduction of 8–33% of the native plant diversity extant on pastures in the

Ishinca valley, if its user-group does not respond by adapting rules of use and management to changing household circumstances. In making recommendations to avoid negative outcomes for biodiversity conservation in the HBR, I seek to:

(1) support indigenous rights to access and use resources in the reserve;

(2) stress the importance of moderate levels of grazing for maintaining native species richness;

(3) identify management options that may help redirect potentially problematic land use scenarios; and

(4) encourage park administrators and NGOs to develop policies sensitive to the challenges of maintaining cooperation in the management of *common property*.

INTRODUCTION

Highland ecosystems are amazingly diverse and immensely important, given that they make up a relatively small percentage of total land area globally (Körner, Nakhutsrishvili, and Spehn 2006). High biodiversity coupled with monumental scenery and a growing recognition that the health of mountain ecosystems influences the health of all other downstream and downslope ecosystems, have led to the establishment of many protected areas charged with the conservation of mountain biodiversity and the preservation of important mountain environments and ecosystem services. The Huascarán Biosphere Reserve (HBR) is one such protected area attempting to achieve "conservation with development." It is also one of many experiencing the challenge of achieving both. A 5,102 km^2 stretch encompassing the highest peaks of the Cordillera Blanca of North-Central Peru, the reserve, which includes the conservation core of the Huascarán National Park (HNP) and a surrounding buffer zone, is rich in biodiversity and embedded in a 10,000-year history of indigenous land use (Moseley 1992). Although there is little question that what was recognized as a global treasure worthy of protection in 1975, is a cultural landscape shaped by a long history of human activity, the sustainability of subsistence land use is now in question.

Agriculture and pastoralism are the foundation of many highland subsistence economies (Mayer 2002, Netting 1981, Stevens 1993, Winterhalder and Thomas 1978). High altitude pastures and forests in the HBR are a vital resource to the agro-pastoralists living in its borders, and their character and health are largely determined by indigenous land use practices. By government decree, indigenous residents are allowed to graze in the HNP. Access to this resource is a necessity for almost every household living in the reserve. Yet, this arrangement is increasingly tenuous. In recent history, many of the world's highlands have undergone dramatic socio-economic and environmental changes, sometimes making sustainable land use difficult to maintain. Globalization, market integration, population growth, government interventions, and climate change potentially affect mountain biodiversity, the integrity of mountain ecosystems, and the livelihoods of all those that depend on them.

These challenges are quite real in the HBR. Reserve administrators note a decade-long increase in non-native livestock and a growing failure of indigenous residents to manage them sustainably (INRENA 2002). Interviews with community members and pictorials drawn by informants of the indigenous community of Tupac Yupanqui (TY) suggest that their concerns are not unwarranted. Most households in the community believe that there are substantially more livestock now than in the past and that the current condition of pastures in the community is poorer. A study conducted by The Mountain Institute (TMI), an NGO active in the region, echoes these sentiments (TMI 2001a). Key informants in TY feel that many aspects of the environment have changed for the worse, while acknowledging that poor land use practices may at least be partially responsible (refer to Figure 2.1). Over the years of 1970–2000, community members noted an increase in livestock, a decrease in quality forage, and the retreat of permanent glaciers in the Ishinca Valley. Photographs taken in this valley during fieldwork from 2001–2003 would also seem to indicate that environmental degradation may be occurring (see Photo 4.1). These suggestions are discouraging considering the reserve's objectives of biodiversity preservation and the community's continued dependence on it for their economic livelihoods.

Photo 4.1: Visible terracettes and bare ground on Miyu Pampa (photograph by the author, 2002).

Photo 4.2: An attempt to sample vegetation with untended cattle above the Ishinca valley base camp (photograph by the author, 2002).

Intense grazing pressure can result in environmental degradation, including loss of productivity, usefulness and important ecological functions (Adler and Morales 1999, Molinillo and Monasterio 1997). Quantitative evidence presented elsewhere in the dissertation suggests that the threat of over-grazing in the reserve is real, yet a formal study of the impacts of grazing on plant biodiversity in the HBR is lacking. The findings of Chapter 2 lend support to reserve administrators' concerns that non-native herds are growing in surrounding indigenous communities. I have shown that they are likely to continue to grow with increasing involvement in tourism and trekking. The model presented in Chapter 3 suggests that some market-integrated households have increased herd size but are also spending less time herding them. Such findings have important implications for the protection of the HBR if policies continue to inadvertently encourage these land use practices while undermining the incentive for communal institutions to mitigate them.

The objective of this paper is to test the prevailing assumption of environmental degradation in the HBR by exploring the effects of historic grazing practices on native plant biodiversity, a conservation priority of the reserve. Secondarily, my objective is to explore how conditions may change in response to newly emerging land use practices. Linear and random effects models were fit with variables derived from community surveys, informant reports, vegetation samples, and topographic analyses conducted with a Geographic Information System (GIS). The integration of multilevel modeling and GIS allows us to explore the varying spatial and temporal scales at which grazing operates and to visualize predicted changes in biodiversity, identifying areas at risk of being negatively and irreversibly affected by changing herding practices.

STUDY SITE

Huascarán National Park (HNP) was added to Peru's protected area network in 1975 and was later designated a UNESCO Biosphere Reserve in 1977 (see Figure 4.1). In total, the Huascarán Biosphere Reserve (HBR) has an area of 5710 km^2; 3400 km^2 making up the conservation core and an additional 2310 km^2 buffer encompassing a number of indigenous communities (INRENA 2003). The HBR is unique in that it circumscribes the highest peaks of the Cordillera Blanca, a tropical Andean Mountain chain predominately oriented in a north–south direction and forming the continental divide of northern Peru. The reserve's tropical climate, position at the ecotone between wet *páramo* and dry *puna* biomes (Luteyn et al. 1999, Smith 1988), and the range of elevations it encompasses make it one of the world's great treasures of biodiversity. Early visits to the region by the great Andean biogeographers Troll (1968) and Holdridge (1967) were some of the first to document the region's diversity. However, it is largely in the past decade that the tropical Andes have gained substantial recognition as a biodiversity hotspot, and in turn have become the focus of increasing conservation efforts (Rodriguez and Young 2000).

As a conservation entity, the reserve seeks to protect the region's environment and a large number of endemic or endangered plant and animal species while permitting indigenous communities to continue land use within its borders. Photos 4.3 through 4.6 on the following pages highlight some of the more well-known conservation targets in the reserve. However, plant inventories have documented many more; over 779 species and 339 genera, many of them endemic, and many of economic importance to resident agro-pastoralists (Hammond et al. 1998, Palomino 2000, REPAAN 1993, RERUMEN

1992, Turin 2001). Given the reserve's focus on biodiversity preservation and the indigenous population's dependence on pastoralism, the continued health and productivity of economically important species such as *Calamagrostis, Festuca* and *Stipa* species should be of equal conservation value, yet have been relatively less studied.

Like many "conservation with development" projects, conservation success in the HBR has been limited, and local land use is often seen to be to blame for environmental degradation. The sectors of Collón and Pashpa in the indigenous community of Tupac Yupanqui (TY) were chosen for this study primarily due to their association with the Ishinca valley, a valley often assumed by reserve authorities and local NGOs as one degraded due to the mismanagement of livestock (INRENA 1990a, Ramirez 2002, TMI 2001a, TMI 2001b). The Ishinca valley is a typical inter-Andean valley accessed from the western side of the park that terminates in the glaciated peaks of Urus, Toclaraju, Ishinca, Paclaraju, Ranrapalca and Ocshapalca. Variability in elevation, topography, exposure, slope and substrate contribute to high biodiversity in this valley as in many others of the reserve. Through the 10 kilometers of its length, elevation ranges from approximately 3600 to the summit of Palcaraju at 6274 meters (Sharman 1995). A number of vegetation types are encountered throughout the valley and to snowline (approximately 5100 meters). Most notable is a 4–6 km long stand of *Polylepis* forest which dominates the lower third of the valley (see Photo 4.3). However, *bofedales* and *gramadales* (wet and dry grasslands) predominate elsewhere (see Photo 4.6). These grasslands are collectively referred to as the *puna*, and are the focus of this study, as are the fallowed fields within and around the community. Figure 4.1 shows the extent of the study site and the locations where vegetation sampling took place.

Photo 4.3: Large stand of *Polylepis* near the entrance of the Ishinca valley (photograph by the author, 2002).

Photo 4.4: Toclaraju, a popular climbing pursuit in the Ishinca valley (photograph by the author, 2002).

Figure 4.1: Sites where vegetation sampling took place.

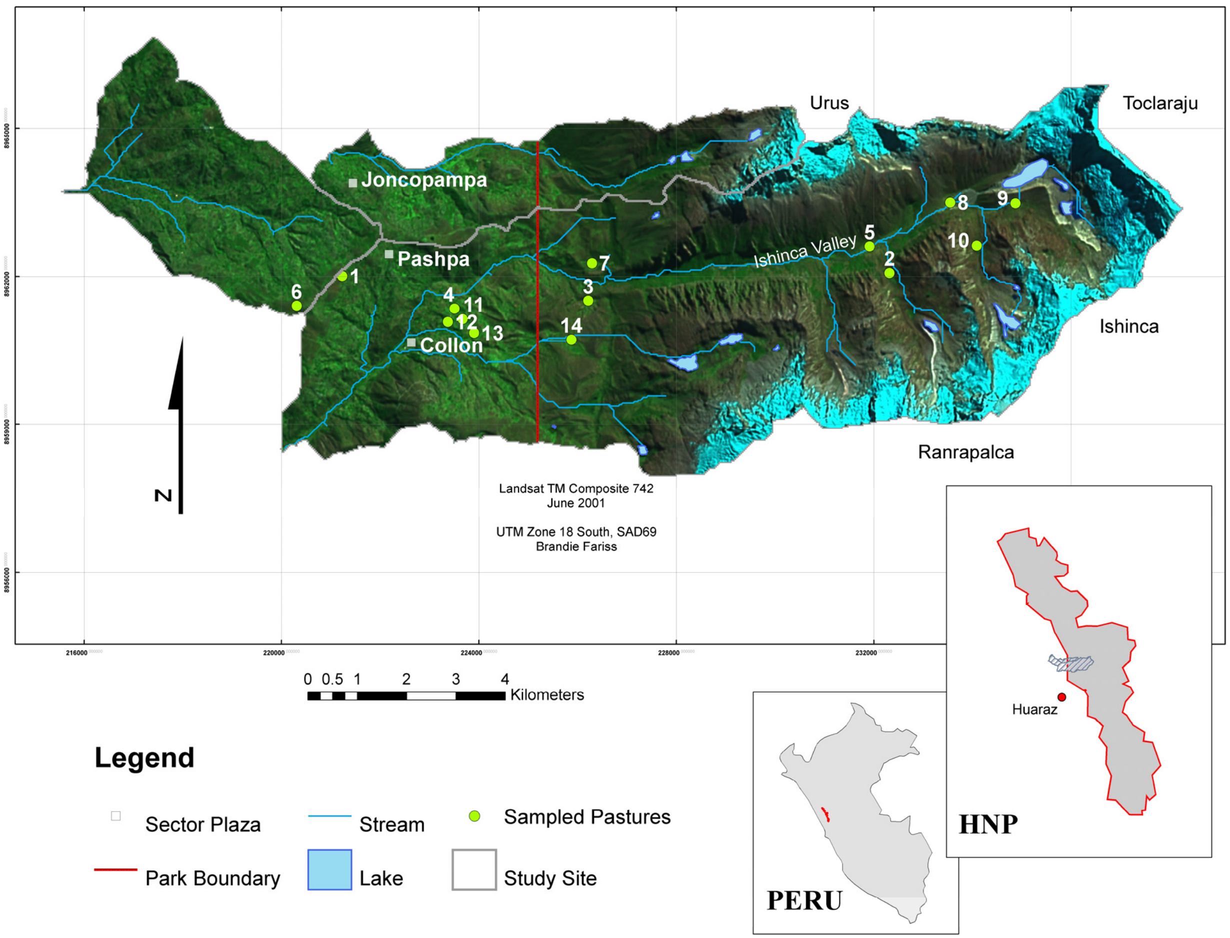


Photo 4.5: *Puya ramondii*, an endemic species found in the valley of Pastoruri to the south of the study area (photograph by the author, 2002).

Photo 4.6: Llama on Winac Pampa in the *puna baja* (photograph by the author, 2002).

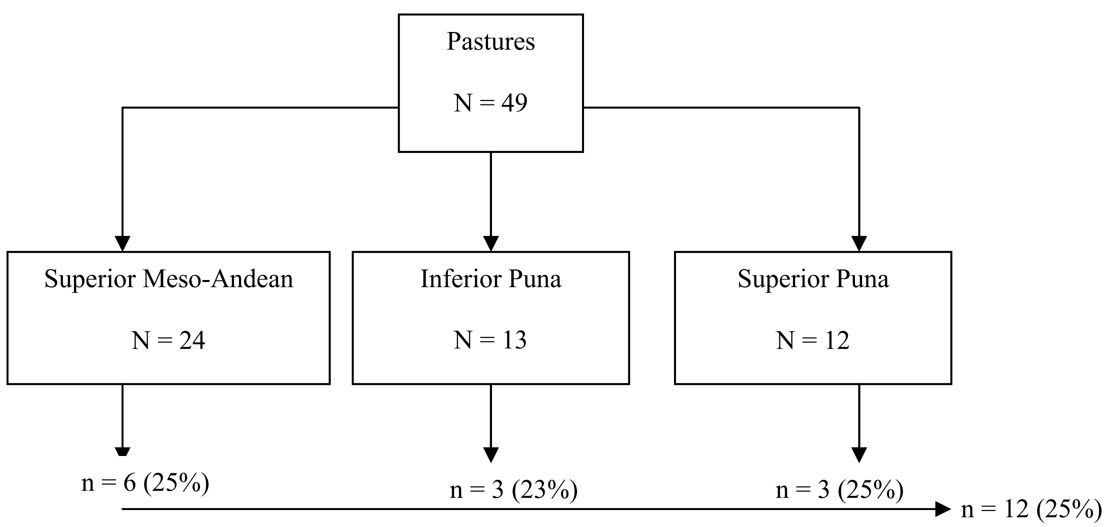
METHODS

Data for the analysis were derived from vegetation sampling and digital terrain analysis in GIS, as well as from household and community level surveys of herding. Each will be described below.

Vegetation Sampling

Sites of vegetation sampling were selected with two primary objectives. The first was to sample a statistically representative proportion of pastures throughout the *common property* network managed by Collón and Pashpa, and the second was to obtain a stratified random sample that spanned a range of grazing intensities and environmental conditions likely to affect plant communities. The pasture sampling scheme is shown in Figure 4.2.

Figure 4.2: Pasture sampling strategy.



Forty-nine pastures were identified after forays into the valley with local informants from Collón and Pashpa at the beginning of fieldwork in 2002. In total, 12 of these were randomly selected for vegetation sampling, representing approximately 25% of all pastures.[73] These 12 sites were drawn from the bioclimatic zones described by Tovar and Oscanoa (2002), which serve as a proxy for temperature and precipitation gradients assumed to be the dominant drivers of vegetation in mountainous terrain. These strata included the: (1) superior meso-Andean zone between 3400m and 3850m, (2) inferior *Puna* between 3850 and 4500, and (3) superior *Puna* from 4500m to 4900m. Due to the verticality of land use in the study site, I assumed that these bioclimatic strata may also serve as means of capturing various grazing intensities. Pastures were randomly drawn from each of these strata in the proportions relative to their representation among the 49 total sites (2:1:1). Thus, 6 pastures were drawn from the superior meso-Andean zone surrounding the community, 3 from the inferior puna, and 3 from the superior puna (n = 12). Table 4.1 lists each sampled pasture and its general characteristics.

[73] The number of pastures sampled reflects the maximum number of sites for which I could collect vegetation data over three consecutive months, so as to minimize the effects of changing seasonal conditions. Additional pastures that were part of a pasture improvement project conducted by The Mountain Institute (TMI) were also sampled (TMI 2001b). These pastures are not included in the analysis, although their general characteristics are summarized in Appendix 4.1 for the purposes of providing information to this organization whose assistance with this fieldwork was extremely valuable and much appreciated.

Table 4.1: Key characteristics of sampled pastures (sorted alphabetically by pasture name).

PASTURE* (# ON MAP)	LOCATION	BIOCLIMATIC ZONE	ESTIMATED AREA (HECTARES)**
Canish Pampa (1)	Pashpa	1	11
Chacuashachina Ruri (2)	Collón	3	6
Chopi Uran (3)	Ishinca	1	5
Cochan Pampa (4)	Collón	1	4
Condor Mashanan (5)	Ishinca	2	6
Hualcan (6)	Pashpa	1	9
Lachoc (7)	Pashpa	1	7
Miyu Pampa (8)	Ishinca	2	6
Pacclish (9)	Ishinca	3	5
Pacha Pampa (10)	Ishinca	3	4
Quisuar Pampa (12)	Collón	1	2
Winac Pampa (14)	Collón	2	20

*Pastures are not fenced thus their boundaries are not discrete, but local herders do recognize unique locales. Quechua place names for these pastures often translate as descriptors of the site or its suitability for grazing. For example, Pocran Pampa (not sampled) describes a treeless flat area (*pampa*) and *pocran* describes a type of animal infirmity. Most herders will utilize this site seasonally, but believe that animals are likely to grow ill and die if left for extended periods of time. In the list above Condor Mashanan describes a site where condors commonly circle overhead, "basking in the sun". Miyu Pampa describes a pasture with dirty water. These place names allow for a cultural memory which is likely to affect general herding practices and levels of use of different pastures throughout the study site.

**Area estimates were derived from pasture boundaries digitized from a false color composite (432) of LandSat TM imagery taken on July 2001. They are only approximations

Sampled pastures span three elevation classes (by sample design), vary in size from 2 to 20 hectares, and encompass two humidity regimes which distinguish each as a *bofedale* or *gramadale*. *Bofedales* are mesic pastures. Thus they are critical dry season reserves that are used extensively by camelids and packstock. Because of the increased incidence of disease and livestock illness in the wet season, *bofedales* are often avoided by herders at these times (Alzerreca et al. 2006). The primary pastures recognized as *bofedales* by local herders include Chuacuachachina Ruri (2), Lachoc (7), and Winac Pampa (14), 25% of the sample. These sites range from humid areas that are not permanently inundated to sites with year-round saturated soils. Matt-forming species such as *Distichia muscoides* and *Plantago rigida* are common, as are forage species such as *Festuca dolichophylla*. All are located in the middle to lower bioclimatic zones of the study site and are used primarily in the months of May through September by horse and mule in Pashpa, and in Collón, by large *empresa* herds of alpaca and llama. *Gramadales* make up the rest of the sample. These pastures are typically more xeric sites of subtle or steep slopes with convex topography where water tends to run-off. Soils at these sites, as in general, tend to be poorly developed loams or silty loams (Brush 1982, Smith 1988). These sites are often dominated by *Scirpus rigidus, Aciachne pulvinata*, and bunch grasses like *Stipa ichu*.

On each of the 12 pastures I used multi-scaled Modified Whittaker plots to sample vascular plant diversity and abundance (Stohlgren, Falkner, and Schell 1995).[74] Random coordinates within the pasture were used to determine an origin for the placement of the largest 1000m^2 plots (50m x 20m). Plots were fixed at the randomly chosen origin, and oriented so that the long axis traversed any apparent environmental

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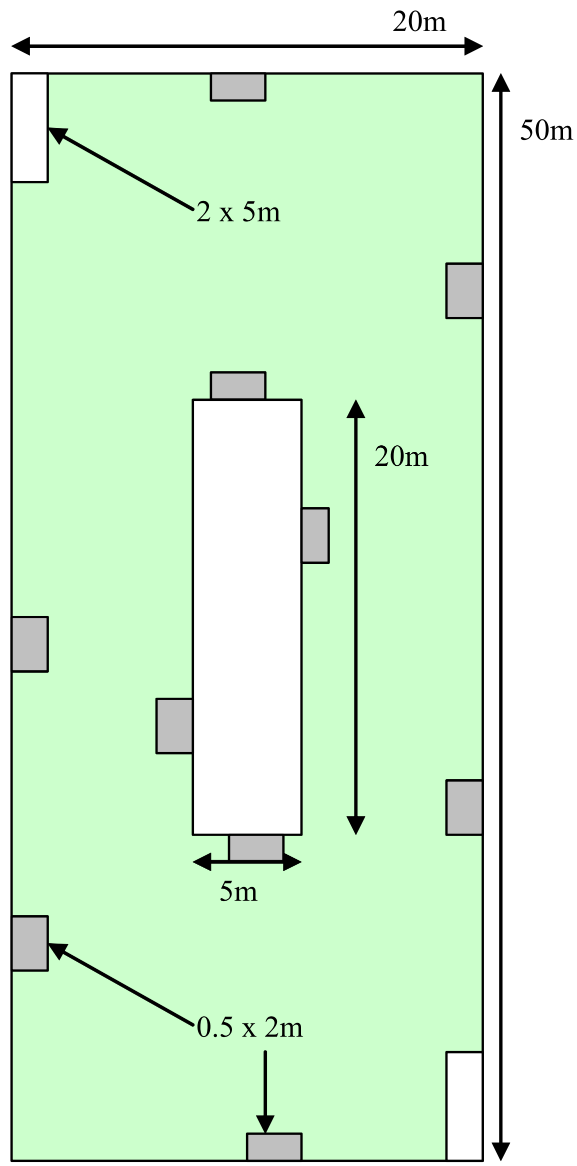
[74] Parker transects were also done to supplement the more time-consuming data collection for the Modified Whittaker plots. Only the plot data are considered for this analysis.

gradient. Nested within the largest plot were a single $100m^2$ plot (5m x 20m), two $10m^2$ plots (2m x 5m), and ten $1m^2$ plots (0.5m x 2m). A diagram of the plot layout is shown in Figure 4.3.

Figure 4.3: Diagram of the Modified-Whittaker plot. Plot-level analysis is based on the smallest $1m^2$ plots of the diagram. Pasture-level analysis is based on the largest $1000m^2$ plot of each pasture.



Vegetation sampling took place from April through July,[75] and during that time the foliar cover of each species and the cover of features such as bare ground, rock, moss, leaf necromass, dung and water were all estimated to the nearest percent in each of the $1m^2$ plots. Species presence was recorded by a scan of the largest $1000m^2$ plots. Voucher specimens of plant species that could not be identified by either me or my field assistant were collected and identified with the assistance of faculty from the herbarium of the Universidad Nacional de San Marcos in Lima. If collection was not possible, as was the case within the park, specimens encountered were keyed to genus and remain in the dataset as unknown individual species. Collectively these unknown species comprise less than 2% all the species encountered during data collection. Additional data were recorded for the larger plots and areas outside of the sampling scheme, including biomass and soil samples, as well as observations of current stocking rates.

Analytical Approach

Modified Whittaker plots have a nested structure, making a multilevel model similar to that employed in Chapter 3 appropriate for modeling the data collected by the sampling design. The benefits of multilevel modeling over a standard generalized linear model include the ability to conceptualize predictors at multiple scales as well as to improve model estimates and standard errors for the response of interest by accounting for the pseudo-replication issue inherent in nested sampling schemes (Laird and Ware

[75] Logistics and limited manpower excluded the possibility of completing all vegetation sampling within a narrower window of time. I completed lower pastures earlier in this range, and higher ones later.

1982).[76] Utilizing the terminology of multilevel modeling, each $1m^2$ plot generates a level-1 response of native species richness influenced by the random effects of the pasture from which it was sampled (a level-2 unit), as well as a number of additional explanatory variables operating at either the plot or pasture-level. The LME (Linear Mixed Effects) package in R was used to fit multilevel models of native species richness at the plot-level (Bates and Sarkar 2007, R Development Core Team 2007). WinBugs 1.4.2 was also used to fit Bayesian versions of the same models (Spiegelhalter et al. 2007). The pasture-level model of native species richness, which is based on the occurrence of species in the largest of the Modified-Whittaker plots ($1000m^2$), was fit in R using ordinary linear regression. Table 4.2 summarizes the variables considered in the following models, while the statistical details and descriptions of each are provided below.

[76] Appendix 4.2 compares a standard linear model against a multilevel model of native species richness for the plot-level response. The reader will notice that the likelihood ratio test statistic is significant, suggesting that there is a meaningful difference between the two modeling approaches. In addition, Akaike's Information Criterion (AIC) favors the multi-level model despite the additional parameters estimated.

Table 4.2: Definition and summary statistics for variables in the multilevel model of native species richness. The response was also modeled at the pasture-level using an ordinary linear regression. Variables included in the pasture-level model are indicated by an (*).

RESPONSE VARIABLE:								
NAME	**DEFINITION**	**TYPE**	**OBS**	**AVG**	**STDEV**	**MIN**	**MAX**	**LEVEL**
*native species richness**	Number of native species encountered in sample unit	Continuous Plot Pasture	120 12	9.6 35.2	3.7 5.9	0 27	20 45	Plot Pasture
EXPLANATORY VARIABLES:								
NAME	**DEFINITION**	**TYPE**	**OBS**	**AVG**	**STDEV**	**MIN**	**MAX**	**LEVEL**
cover	The proportion of vegetative cover in each plot	Proportion	120	0.7	0.2	0	1	Plot
dominant cover	The proportion of plot dominated by a single species	Proportion	120	0.3	0.2	0	0.9	Plot
wetness	Topographic wetness index	Continuous	12	3.3	1.9	0	6.3	Pasture
aspect	Sine transformed aspect	Continuous	12	0.1	0.7	-1	1	Pasture
EXPLANATORY VARIABLES OF INTEREST:								
NAME	**DEFINITION**	**TYPE**	**OBS**	**AVG**	**STDEV**	**MIN**	**MAX**	**LEVEL**
plot-level grazing intensity	Recent intensity of grazing within the plot determined by the mean palatability of species within it	Continuous	120	-1.4	0.5	-2.3	0	Plot
*pasture-level grazing intensity**	Accumulated intensity of grazing for a particular pasture determined from GIS analysis of herder preferences	Categorical Low Moderate High	4 4 4	X X X	X X X	X X X	X X X	Pasture
LEVEL-2 UNIT:								
NAME	**DEFINITION**	**TYPE**	**OBS**	**AVG**	**STDEV**	**MIN**	**MAX**	**LEVEL**
pasture	Unique identifier for the pasture	ID	12	X	X	X	X	Pasture

MODEL DESCRIPTION

Native Species Richness

Grazing remains the only prevalent land use in the conservation core of the HNP, and it undoubtedly interacts with a number of biophysical conditions to affect the vegetation of the reserve. The question motivating this analysis is how and to what extent? Over-grazing is an untested assumption made by reserve administrators, local NGOs, and even the communities themselves. As a frequent and often times inappropriately used descriptor, it implies that grazing has had a measurable negative effect on the environment. The measures of environmental change typically explored in studies of grazing impact include biomass, species composition, abundance and diversity, as well as soil compaction and erosion. Given the reserve's concerns for biodiversity protection, I focus on the effects of grazing on native species richness.[77] Given park and community perceptions of over-grazing, it may be hypothesized that native diversity has been negatively affected by the grazing practices of local indigenous communities.

My objective was to quantify the effects of grazing on native species richness at both plot and pasture-level scales. A measure of native species richness for each scale was created from the $1m^2$ and $1000m^2$ plots by summing the number of unique species encountered in them. This resulted in two different measures of diversity. Plot-level

[77] The economic value of individual species is of greater importance than overall diversity when considering the perspective of resident agro-pastoralists who rely on pastures to support their herds. In addition to an interest in preserving biodiversity and the economic value of that diversity, land degradation is of concern to park and community alike, as it jeopardizes the health of the ecosystem and its productivity, a threat to the conservation objectives of the park as well as the livelihoods of local indigenous people. The effects of grazing on measures other than native species richness, including various measures of diversity and soil erosion are currently being analyzed. Appendix 4.3 summarizes some of them relative to pasture-level grazing intensity.

richness is simply the sum of unique native species encountered in each $1m^2$ plot. Pasture-level richness contains a minimum of all of the unique native species encountered in these plots, as well as a number of species that were encountered during a scan of the remaining area within the $1000m^2$ plot. Therefore ***native species richness*** is a discrete random variable bounded by 0 at two different scales. Native species richness at the plot-level averaged 10 species (min = 0, max = 20, stdev = 3.7) and at the pasture-level averaged 35 species (min = 27 max = 45, stdev = 5.9). Appendix 4.4 provides a comprehensive list of species encountered throughout the study, which includes 213 species of 119 genera and 44 different families. The families most commonly represented by these species include Poaceae (27%), Asteraceae (23%), Cyperaceae (8%), Fabaceae (8%), Juncacea (8%), and Gentianaceae (6%).

Figure 4.4 illustrates the overall distribution of native species richness at the plot-level. I consulted standard recommendations for choosing an appropriate probability model for this response (Snijders and Bosker 1999). Given the distribution and the range of counts represented in the histogram, I assume that the response is normally distributed.[78] I explored this assumption further by comparing normal and Poisson[79] models of native species richness at the plot-level using the "lme" and "glme" libraries in S-PLUS 7.0.6 (Insightful 2005b).[80] A comparison of these models suggests that the

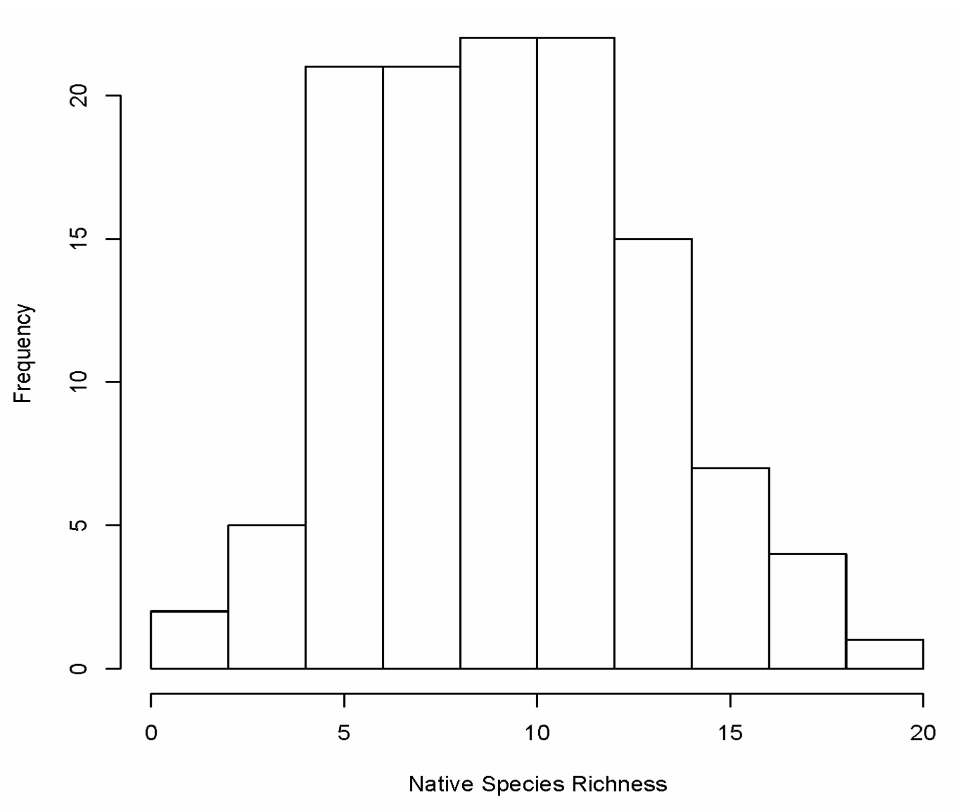
[78] The histogram had to be generated on the data in aggregate because there are not enough plots per pasture to generate anything but a spotty distribution at this scale. While this is a bit dishonest considering the nested design of the data collection, model comparison using AIC provides additional support for this assumption.

[79] A discrete probability distribution that is good for small counts.

[80] Model BOTH, which is specified below, was used for this comparison.

normal assumption is better, according to AIC (571.13 vs. 602.39).[81] Given these preliminary findings, I treat native species richness as a continuous normally distributed variable for the modeling effort. Although transformations of moderate counts such as these are commonplace (e.g., Freeman-Tukey, square root or log transformation), the additional complications they introduce for interpretation and inference caused me to avoid doing so (for a critique of data transformations see McArdle and Anderson 2004).

Figure 4.4: Distribution of plot-level native species richness observations.



[81] For a discussion of AIC and its interpretation please refer to pg. 147. Smaller AIC values indicate better models of the response.

Grazing as a Scaled Disturbance

Grazing, like diversity, can be measured at multiple spatial scales. The grazing regime of the Andes is typically characterized by seasonal *transhumance*. This pattern of resource use provides a model of landscape-scale movements determined by the herder, and a means of deriving historical accumulated grazing intensities for each pasture. However, once animals are driven to selected pastures, localized grazing intensities are dictated more by herding practices (e.g., herd composition and herd management, including whether or not to disperse or stake animals within the pasture), as well as livestock responses to the relative quality and abundance of forage species in different patches. The concentration of livestock on a certain place within a pasture creates heterogeneous grazing intensities within the pasture that are not captured by landscape-scale *transhumance*. Given the desire to model ***native species richness*** at both the plot and pasture-level, it seemed logical to consider grazing as a disturbance that operates at each of these scales. The specific variables defined to represent each scale of grazing are described in greater detail below.

Pasture-Level Grazing Intensity

The variable ***pasture grazing intensity*** attempts to capture long-term, accumulated grazing intensities experienced on each pasture by considering the traditional patterns of *transhumance* practiced by most indigenous residents and the frequently cited preferences of herders in determining which pastures they use. Preferences for different pastures were defined through interviews with local herders (n =

80). The two most frequently cited factors influencing where to herd included the pasture's distance from the community and its proximity to water.[82] Other authors have used similar proxies to derive measures of historical grazing intensity in the absence of information on historic stockings (e.g., Andrew 1988, Beever, Huso, and Pyke 2006, Turner 1998). Here I rank the 12 pasture sample into low, moderate and high accessibility to grazing based on herder preferences and cost-weighted distance algorithms in ArcGIS and Spatial Analyst 9.1. I describe how this assignment was made in the text and figure that follow.

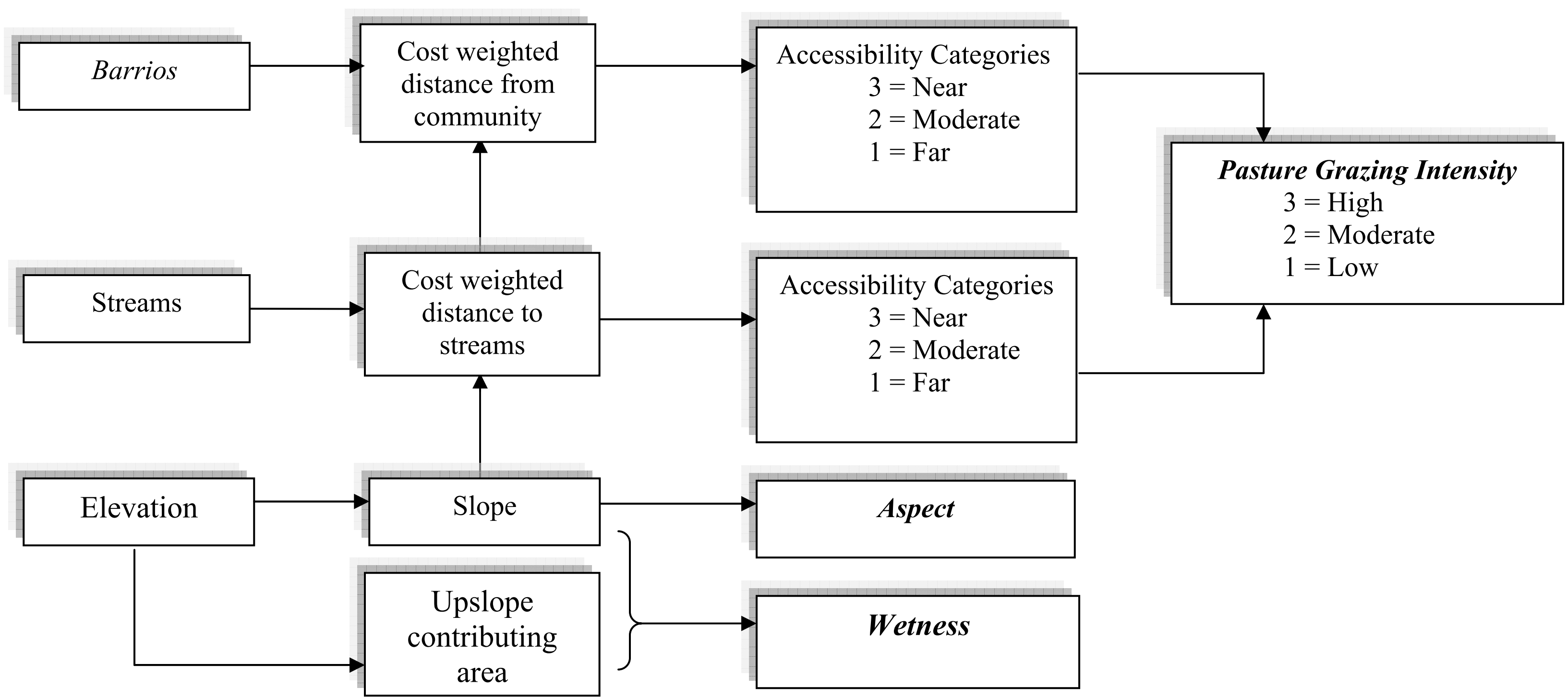
Figure 4.5 documents the derivation of ***pasture-level grazing intensity*** in GIS. First, distances to *barrios* and watering sources were weighted by slope for an estimate of the travel-costs associated with using each pasture. It is assumed that pastures closest to the community and closet to permanent water sources are more intensely utilized than those that are not. The use of pastures equally distant from community and water is determined by the terrain. Since travel costs increase with slope, pastures situated in flatter terrain are assumed to be more intensely utilized than those of steeper terrain.

The ***pasture-level grazing intensity*** variable was created by dividing the distance surfaces obtained above into three classes of equal interval. Thus the range of distances from barrios was divided into low, moderate and high; then given an ordinal ranking of 3, 2 and 1 respectively. The same process was applied to the range of distances from water, resulting in an ordinal ranking of 3 for near water, 2 for moderately near, and 1 for far

[82] Forage quality was another factor cited as important in herder decision-making. Given the variable and subjective nature of how this is assessed by resident herders I exclude it when establishing the overall grazing intensities experienced at the pasture over longer time horizons.

from water. The ordinal rankings for these two characteristics were then summed for every pasture. This resulted in a score of 6 for a pasture that was near the community and near water (i.e., a pasture assumed to have a high accumulated grazing pressure), and a score of 2 for a pasture that was far from both. Lastly, the range of grazing scores was divided into three classes of equal count so that the original 12 pasture sample was categorized into pastures of low, moderate and high grazing intensity. Given the predictions of the Intermediate Disturbance Hypothesis (Connell 1978, Huston 1979), I had reason to consider the possibility of a peaked diversity-disturbance relationship. Thus, I treated the variable as a factor in the model, with contrasts for the model coefficients of moderate and high pasture grazing intensity set relative to pastures of the lowest grazing intensity.

Figure 4.5: GIS implementation flow chart emphasizing the calculation of pasture-level predictors. Variables for the statistical models were extracted from the GIS by overlay of geo-referenced Modified-Whitaker plots and the use of the "extract values by points" command in ArcGIS 9.1. The accumulated grazing intensity experienced on each pasture was estimated by deriving cost-distance surfaces from the community and from streams using slope as the cost surface. Each surface was reduced to a classification of far (1), intermediate (2) and near (3) by dividing the range of cost-distance values into classes of equal size. I then performed a raster addition so that the ordinal variables of cost-distance to community and cost-distance to water were summed to derive a proxy for accumulated grazing pressure. For example, pastures that had low travel cost from the community and to water were calculated as follows: 3+3 = 6. The resulting variable, pasture-level grazing intensity, groups the range of values obtained from this process into 3 equally sized classes of low (1), moderate (2) and high (3), which are treated as factors in the modeling effort. I chose to group this variable rather than treat it as continuous due to its non-linear nature and the fact that distance to water and distance to community ranges were not equivalent. Plot-level predictors were derived from field measurements and are not shown here.



Plot-Level Grazing Intensity

An evaluation of species richness at the plot-level requires us to consider other factors operating at the same scale. Once herders determine that they are going to utilize a particular pasture, livestock are likely to select patches within it that have a higher abundance of quality forage (for discussion of optimal foraging theory see MacArthur and Pianka 1966, Pyke, Pulliam, and Charnov 1977). Furthermore, the herder may occasionally stake animals on a pasture. These actions create plot-level grazing intensities that vary within pastures, influencing vegetation responses at that particular locale, which can be used as a proxy for the plot's recent grazing history.[83]

The correlation between livestock concentration and the increased occurrence of toxic species or low palatability species has been shown elsewhere (Adler et al. 2004, Adler, Raff, and Laurenroth 2001, Hernandez and Monasterio 2006). I derive the level-1 variable ***plot-level grazing intensity*** to proxy for recent histories of grazing within pastures by taking an average of the relative palatability scores for species encountered in each plot. Information on species palatability was compiled from diverse sources, including Mollinillo and Monasterio's study of the Venezualen *páramo* (1997), a key of grasses in the HBR (Tovar and Oscanoa 2002), unpublished databases created and managed by The Mountain Institute (e.g., Sotomayor 2000, TMI 2001b), and the reports

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83 Other authors have utilized dung as proxy for recent and more localized grazing histories (e.g., Beever, Huso, and Pyke 2006). However Turner (1998) illustrates that changes in species composition operate on spatial and temporal scales longer than that captured by relatively short-lived dung deposits. A secondary issue in the Andean landscape is that dung decomposes slower on higher altitude pastures. Thus I argue that it is a less reliable and consistent indicator of grazing history for studies that span a wide range of elevations.

of local informants. Toxic and invasive species, with the exception of *Pennisetum clandestinium* (a palatable invasive), have a palatability score of 0, while those of increasing palatability were assigned a maximum score of 3.[84] These palatability scores were used to calculate ***plot-level grazing intensity (PLGI),*** where X_{ij} is the abundance of species in plot *i* with a palatability score of *j* using the following formula:

$$PLGI_i = \frac{(0)X_{i0} + (-1)X_{i1} + (-2)X_{i2} + (-3)X_{i3}}{X_{i0} + X_{i1} + X_{i2} + X_{i3}}$$

[eqn. 4.1]

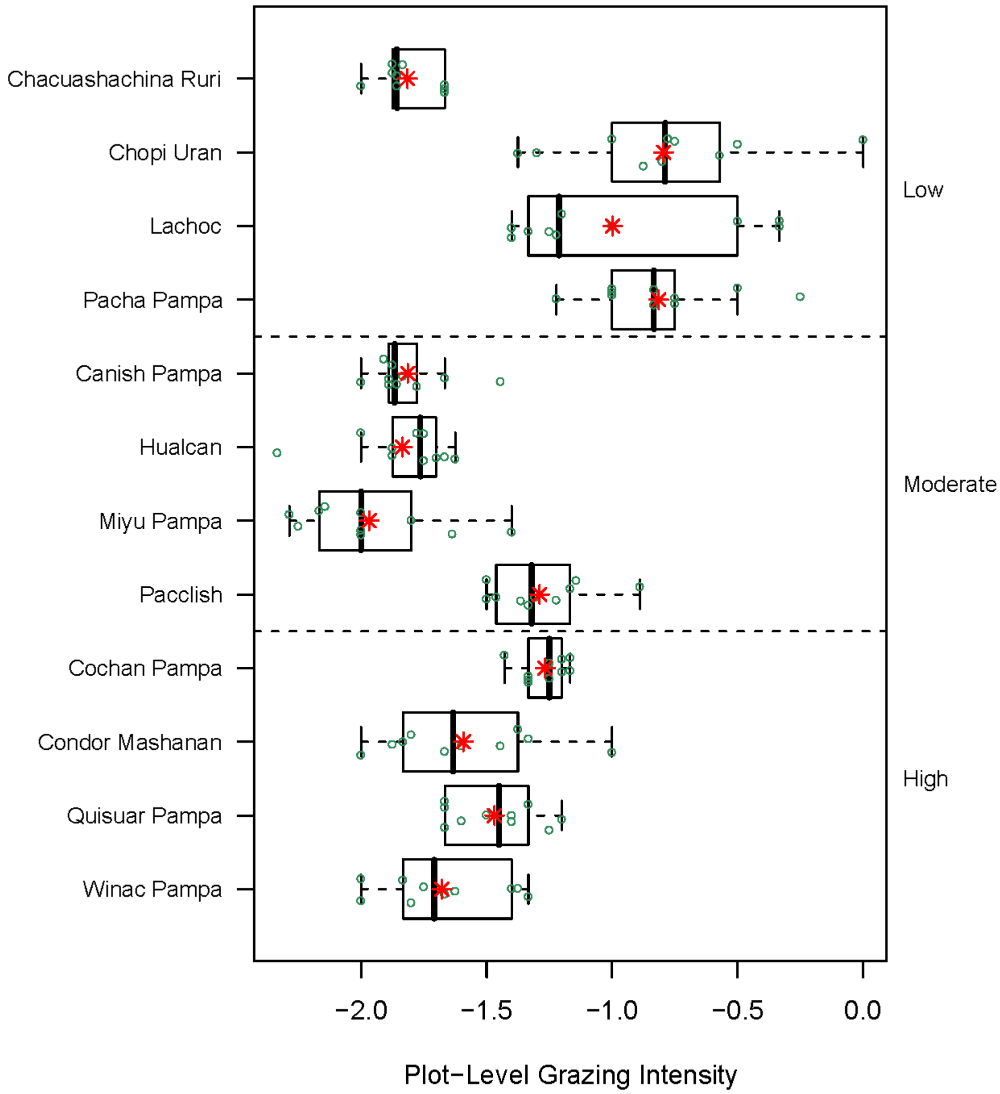
Values for this continuous variable ranged from a minimum of -2.3 representing a palatable plot with relatively little past grazing, to a maximum of 0 representing a plot with an intense grazing history. Mean plot-level grazing intensity was - 1.4 (stdev = 0.5). This was similar across pastures of varying levels of use, as shown in Figure 4.6. Although there is an ongoing debate over the scale at which the IDH applies (Collins and Glenn 1997), I considered the possibility that a peaked diversity-disturbance relationship might exist at the plot-level as well. To determine the best form for this predictor, I fit separate multilevel models with linear and quadratic versions of plot-level grazing. An analysis of the linear model residuals shows a subtle quadratic pattern (see Appendix 4.5). Furthermore, a comparison of model AICs suggests that the quadratic version is

84 *Festuca dolichophila* and *Trifolium repens*, two highly palatable species, were planted in two of the pastures in my sample during a pasture improvement project in 2000. Only *Festuca dolicophilla* was encountered during sampling, I exclude it from the calculation of plot-level grazing intensity due to its potential to bias the score as a proxy for historic levels of use.

stronger. Given these findings I include ***plot-level grazing intensity*** as a quadratic predictor in the model.[85]

[85] Quadratic predictors require two terms. The first term is mean-centered; the second is mean-centered and squared. Mean-centering reduces the correlation between the linear and quadratic terms, and does not alter model fit. Both variables must be considered to interpret their effect on native species richness.

Figure 4.6: Distribution of plot-level grazing intensity scores for sampled pastures. Pastures are grouped by their pasture-level grazing intensity, as shown on the right axis. The left and right edges of the box represent the first and third quartiles (the middle 50% of the values) while the vertical bars represent medians and red asterisks represent means. Open circles indicate individual plot-level estimates obtained within each pasture.



Additional Control Variables

Additional environmental characteristics were included as controls. A suite of variables were considered after a literature review of predictors common to studies of vegetation in mountainous terrain (Urban et al. 2000, Whittaker 1960) and models of grassland species richness in general (for numerous examples see Spehn, Liberman, and Korner 2006; also see Adler and Morales 1999, Cingolani et al. 2003). Preliminary analyses ruled out the use of some measures that were collected, as they showed no significant variation across sample units. Others were ruled out later in the modeling process because they were found to have an insignificant effect on the response and their inclusion in the model did not appreciably alter the coefficients or standard errors of the grazing variables of interest. Control variables considered included plot-level biomass and soil characteristics;[86] pasture-level variables included curvature, slope, relative slope position, yearly relative potential radiation, and elevation. Appendix 4.6 discusses all these variables in greater detail.

Plot-Level Control Variables

Plot-level controls that were used in the multilevel model include the proportion of the plot in vegetative cover (***cover***) and the proportion of the plot dominated by a single species (***dominant cover***). The average proportion cover within a plot was 0.7 (min = 0, max = 1, stdev = 0.2), while the average proportion of a plot dominated by a single species was 0.3 (min = 0, max = 0.9, stdev = 0.2). Species exhibiting frequent dominance of plots varied from site to site, but those most abundant overall included species such *Scirpus rigidus*, *Muhlenbergia ligularis* and *Werneria nubigena* (see Appendix 4.7 for a complete listing). Correlation tests and trellis plots revealed that the raw versions of these two variables were highly correlated with the intercept. Thus,

[86] Soil characteristics could not be included because I was not able to do soil analyses for all sampled sites.

mean-centered versions were employed to improve model stability, which has no effect on the estimates or standard errors of the model.

Pasture-Level Control Variables

Pasture-level controls retained for the multilevel model include ***aspect*** and ***wetness***. These factors affect plant community and structure, and the relative stresses imposed on them, especially during the dry season months. For example, precipitation is variable with respect to the orientation of the slope to up-valley air currents; while micro-scale topographic features influence where this precipitation is likely to accumulate as well as a number of other edaphic conditions (e.g., soil texture, nutrient availability, organic matter, needle ice formation).

Recall that Figure 4.5 summarizes how I created each of the pasture-level variables in ArcGIS 9.1. All were derived from the standard algorithms of Spatial Analyst utilizing a 30 meter digital elevation model built from CAD drawings of 1:25,000 scale topographic maps.[87] Maps were obtained in previous field seasons from the Instituto Nacional Geografica (ING), and CAD drawings were obtained from Barrick, an international mining company active in the region. Once calculated, this and all other GIS-derived variables were extracted from the GIS using the geo-referenced locations of the Modified-Whittaker plots. This process generated a tabular dataset linking all explanatory variables to their respective sample plots. This was then imported as a text file to statistical software packages for modeling.

[87] Given the 30m resolution of the DEM, slope, aspect and wetness varied only subtly within pastures and are considered level-2 variables in the multilevel model.

Aspect

The variable ***aspect*** was derived from the elevation model using the surface analysis option of Spatial Analyst. Aspect is simply defined as the direction of steepest downhill descent from each cell to its immediate neighbors, thus it is reported as a circular statistic ranging from 0° to 360° (both due north). A value of -1 is reserved for pixels in the digital elevation model (DEM) that are surrounded by other pixels of the same elevation (i.e., flat areas where slope = 0). Degrees are a circular measure, and as a result, aspects of 1° and 359° are very similar topographically-speaking but drastically different on a linear scale. These values had to be transformed in order to be meaningful in the model. I chose a sine transformation to maximize differences between eastern and western aspects. This was done primarily because of the east–west orientation of this steep v-shaped valley and the predominant valley atmospheric circulation patterns that carry air and moisture into and up through the valley daily (Smith 1988). This simple transformation involved converting the degrees of aspect to radians and multiplying by the sine function to transform aspect values of 0 through 360 to a range of -1 to 1.

Figure 4.7 shows that aspects are similar within pastures but diverse between them.[88] Given the 30m resolution of this variable, I take an average at the pasture-level (indicated by the asterisk in the figure) and include it as a level-2 variable in the model. Values ranged from -1 to 1 (mean = 0.1, stdev = 0.7). Figure 4.8 plots the sine-

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[88] Elevation was a logical choice to consider as a control. Sampled pastures ranged in elevation from 3435 to 4659 meters (mean = 3984, stdev = 466). As stated in the text, elevation did not have a statistically significant effect on native species richness, nor did it alter the coefficients of the grazing variables in any appreciable way. Because additional parameters that offer little explanatory power are penalized using AIC, I dropped it from the models fitted below.

transformed values for each degree of aspect, which can be used as an aid in understanding these values.

Figure 4.7: Average sine-transformed aspects for sampled pastures. Note that pastures within each category of use have a diverse range of aspects.

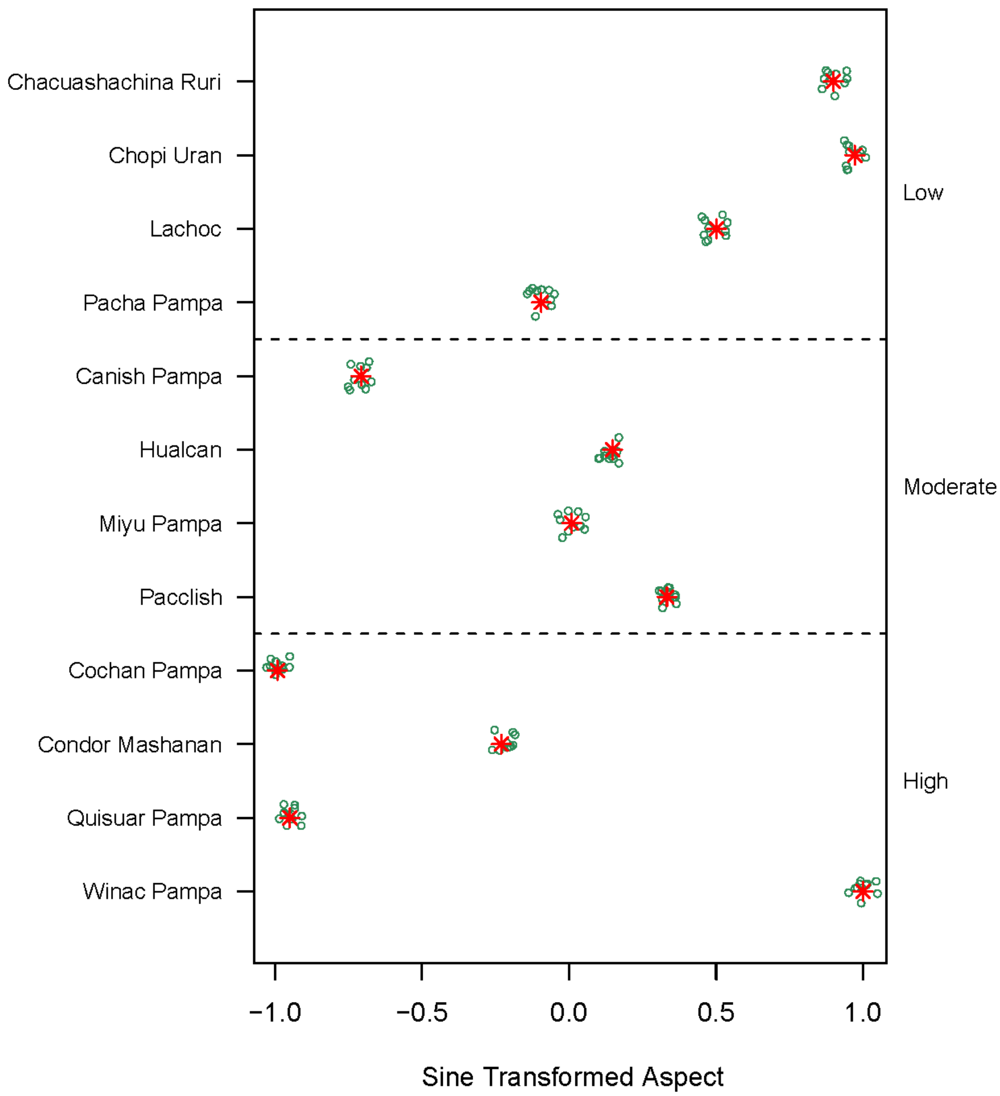
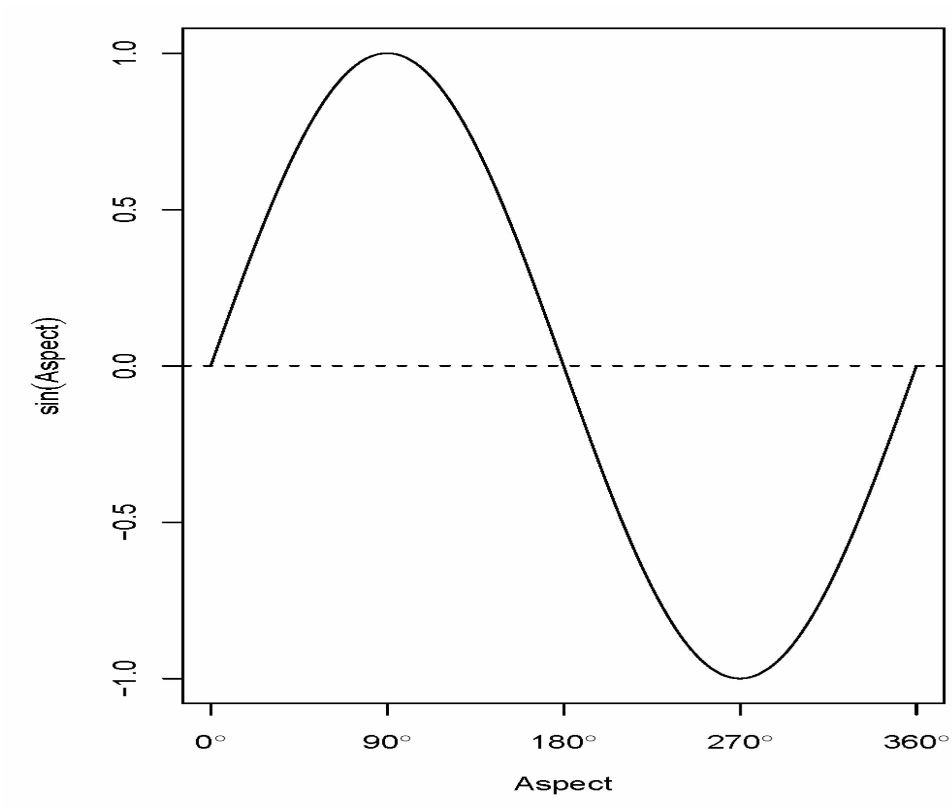


Figure 4.8: Interpretation of sine-transformed aspect values. Note the maximum and minimum values associated with 90° (due east) and 270° (due west). Aspects trending toward the north or south approach zero.



Wetness

The variable ***wetness*** is specifically the topographic wetness index (TWI), which was calculated to serve as a proxy for soil moisture (Beven and Kirkby 1979, Wolock and McCabe 1995). This index can be easily derived from an elevation model in GIS. Where A = upslope contributing area and β = the degrees of slope converted to radians,[89] the conventional form of this variable can be defined as:

[89] The input values to the Tan function in GIS are interpreted as radians. If the desired input is in degrees, the values must be divided by the built-in constant, DEG, to convert the degree values into radians. The value of DEG is 180/π, or ~ 57.296.

$$TWI = \ln\left(\frac{A}{\tan(\beta)}\right)$$

[eqn. 4.2]

Figure 4.9 is a 3-dimensional plot that can be used as an aid in understanding how this variable is influenced by surrounding topography.

Figure 4.9: Interpretation of topographic wetness index values. The wireframe plot illustrates the contribution of slope and catchment area on site specific wetness. Note that drier sites (indicated by red in the far bottom-right corner) are predicted where slopes are steep and there is no upslope contributing area, as is likely to occur on a ridge. Reciprocally, flatter areas with a large upslope contributing area draining into them are wetter (indicated by dark blue in the upper-left corner).

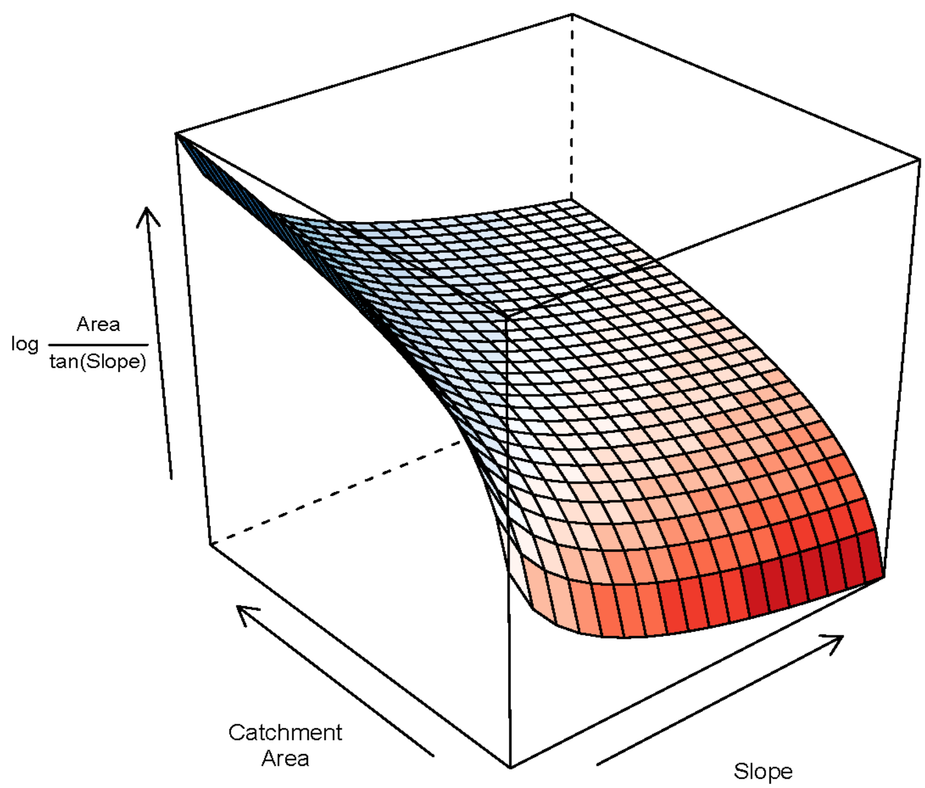
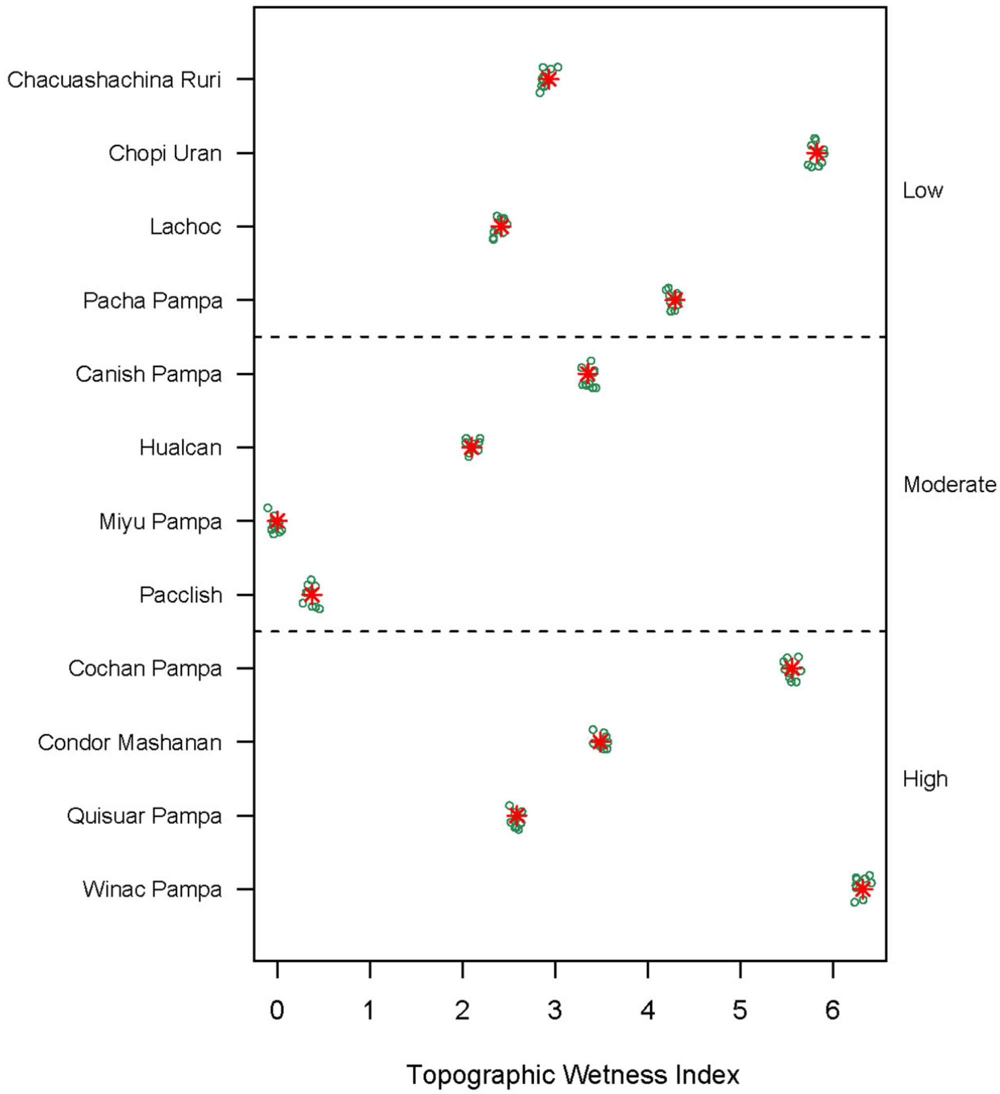


Figure 4.10 shows that wetness values are diverse between pastures. To avoid the problem of having undefined wetness values because of division-by-zero (i.e., flat slopes), I first take the pasture's average slope and contributing area values to calculate wetness. Thus, similar to aspect, I include it as a level-2 variable in the model. Wetness values range from 0.0 to 6.3 (mean = 3.3, stdev = 1.9). Higher values are associated with wetter sites. Note that the driest pastures are of moderate use, including Miyu Pampa, Pacclish and Hualcan.

Figure 4.10: Average wetness for sampled pastures grouped by pasture-level grazing intensity. Note that pastures within each category of use have a diverse range of wetness values, but similar overall means.



PRELIMINARY OBSERVATIONS OF PASTURE CONDITION

Of the 213 species I encountered during vegetation sampling, 123 occurred in the largest of the Modified Whittaker plots sampled on each pasture (n=12) and 116 were encountered in the smaller $1m^2$ (n = 120). Table 4.3 summarizes the occurrence of species considered over-grazing indicators, as well as average plot and pasture-level estimates I obtained for native species diversity.

Table 4.3 shows that invasive species and indicators of over-grazing occurred on the majority of the pastures I sampled. This may partially explain park and community perceptions of over-grazing. Species reported as indicators of over-grazing include *Pennisetum clandestinium*, *Aciachne pulvinata*, *Opuntia flocossa* and *Astragalus garbancillo* (Poma 2002, Tovar and Oscanoa 2002). Both *Pennisetum clandestinium* and *Aciachne pulvinata* are invasive. Although the former is a good forage species, *Aciachne pulvinata*, known as “hinca poto”, “stinging grass” to local Quechua herders, is of poor forage quality. Other indicators of environmental degradation are native species that have growth habits discouraging herbivory, including toxicity, spines and prostrate growth. Species of note include *Opuntia flocossa* and *Astragalus garbancillo*, members of the families Cactaceae and Fabaceae, respectively.

Pennisetum clandestinium and *Aciachne pulvinata* were observed individually or together on 75% of the pastures sampled. *Pennisetum clandestinum* is a ubiquitous species in the region, and although it is a palatable one, its presence constitutes a threat to the native flora of the HBR. This invasive alone occurred on 42% of the pastures sampled. On the pastures where it occurred, *Pennisetum clandestinum* was observed in 32 plots, accounting for a total area of 8.5m^2. Tuspin Pampa (a pasture not included in the random sample, see Appendix 4.1) had the greatest proportion of this total area, which may explain its selection for a pasture improvement began a year before by The Mountain Institute (TMI). *Aciachne pulvinata* or “stinging grass” occurred on half of the pastures sampled, yet this species was only encountered in the smaller plots of four pastures. Chauchashachina Ruri, in particular, had an occurrence of this species on 50% of its plots for a total of 0.3m^2. Interestingly, *Aciachne pulvinata* and *Pennisetum clandestinium* tended to occur at different sites, with an overlap of occurrence only on Canish Pampa and Lachoc, both utilized by the sector of Pashpa.

Native indicators of over-grazing were somewhat less prevalent on sampled pastures. *Astragalus garbancillo* was encountered only in the largest plot of Canish Pampa. On the other hand, *Opuntia flocossa* was present on half of the pastures sampled, and was abundant on Pacclish in particular. Pacclish is a high-altitude pasture moderately used by household and community herds, but during fieldwork I observed a large number of communal cattle at this site (see Photo 4.2).[90] Perhaps due to the continual grazing pressure of these animals, *Opuntia flocossa* was encountered in 60% of the smaller 1m^2 plots sampled there, accounting for a total area of 0.8m^2.

These four indicator species, half of them invasive, have implications for resident agro-pastoralists, as many of them are of little or no economic value. Yet, it is still unclear what they indicate for biodiversity conservation. Perhaps unexpectedly, the highest native species richness occurred on Canish Pampa and Hualcan. These pastures have a presence of indicator species, as well as a moderate level of use by resident herders. Canish Pampa was the only pasture of those sampled that had all four species of concern, yet also maintained the highest native species richness. On the other hand, the highest plot-level species richness occurred on Miyu Pampa, also a pasture of moderate

[90] Approximately 179 cattle held by the sectors of Collón and Pashpa graze often at these altitudes. On occasion some household cattle may also utilize this site.

grazing intensity within the conservation core, but one that has a fair amount of climbing and trekking traffic in the dry season as well.

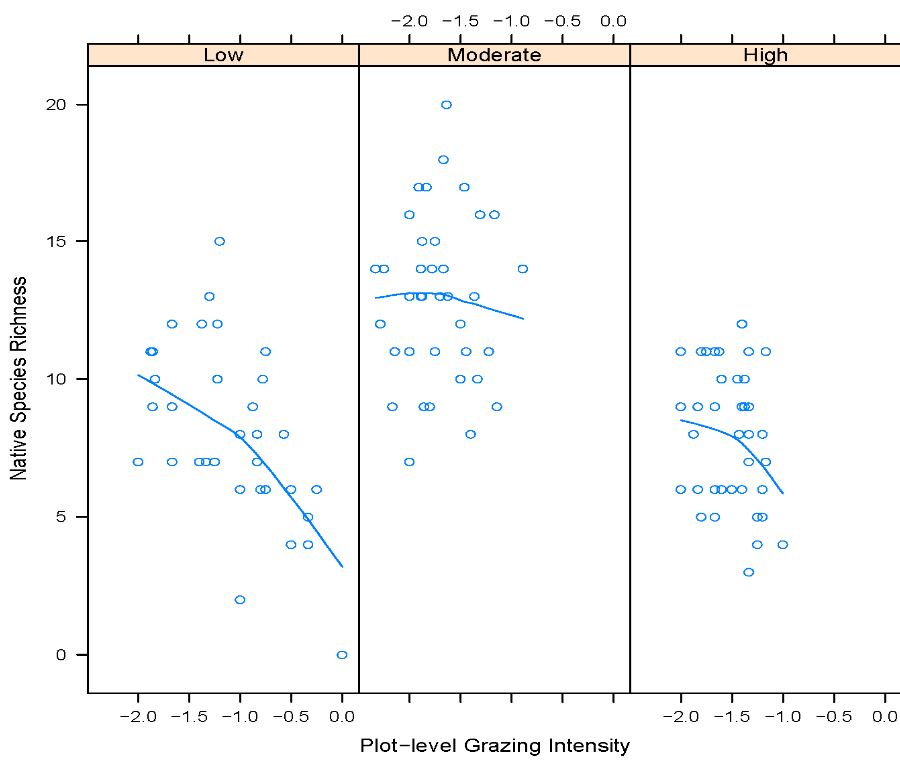
Generally (and qualitatively) speaking it seems that the presence of these indicators is not strongly correlated with native diversity. The lowest native species richness was found on pastures very near the community in the sector of Collón, specifically Quiswar Pampa, Cochan Pampa and Winac Pampa; all are communal pastures with high intensities of use. Yet, relative to other pastures, I encountered fewer over-grazing indicators or invasives in their plots. Quiswar Pampa and Cochan Pampa were part of a pasture improvement project headed by a local NGO and were seeded with *Festuca* and *Trifolium* species the year before my study was conducted. Although these species were encountered in sampled pastures, their low abundance within the $1m^2$ plots of these pastures suggest that pasture improvement efforts have not yet had a marked effect on the composition or overall diversity of species in them.

Table 4.3: Summary of pasture characteristics, with a focus on native diversity and the presence of invasive species and indicators of over-grazing. Pastures are ordered by pasture-level grazing intensity and native richness. Sector codes are as follows: C = Collón, I = Ishinca, P = Pashpa. Species codes are as follows: PC = *Pennisetum clandestinium*, AP = *Aciachne pulvinata*, AG = *Astragalus garbancillo*, and OF = *Opuntia flocossa*. An (+) in the species column indicates that it was recorded on the pasture, while (++) indicate that it was present in the 1m^2 plots where proportion cover was also recorded.

PASTURE NAME (# ON MAP)	**SECTOR**	**PLOT-LEVEL NATIVE SPECIES RICHNESS MEAN (STDEV)**	**PASTURE-LEVEL NATIVE SPECIES RICHNESS**	**PASTURE-LEVEL GRAZING INTENSITY**	**INVASIVE SPECIES**		**INDICATOR SPECIES**	
					AP	PC	AG	OF
Chacuashachina Ruri (2)	C	9.6 (1.7)	32	1	+			
Pacha Pampa (10)	I	7.3 (1.3)	33	1	+			+
Chopi Uran (3)	I	7.7 (4.2)	35	1	++			
Lachoc (7)	P	7.4 (3.5)	35	1	++	+		+
Pacclish (9)	I	12.8 (2.9)	36	2	++			++
Miyu Pampa (8)	I	11.4 (3.7)	41	2				+
Canish Pampa (1)	P	14.4 (2.8)	45	2	++	++	+	+
Hualcan (6)	P	13.6 (1.3)	45	2		++		
Quisuar Pampa (12)	C	6.3 (2.3)	27	3		++		
Cochan Pampa (4)	C	7.4 (1.8)	28	3		++		
Winac Pampa (14)	C	10.1 (1.7)	30	3				
Condor Mashanan (5)	I	7.3 (2.3)	36	3				+
% of pastures with occurrence of an over-grazing indicator:					***50%***	***42%***	***8%***	***50%***

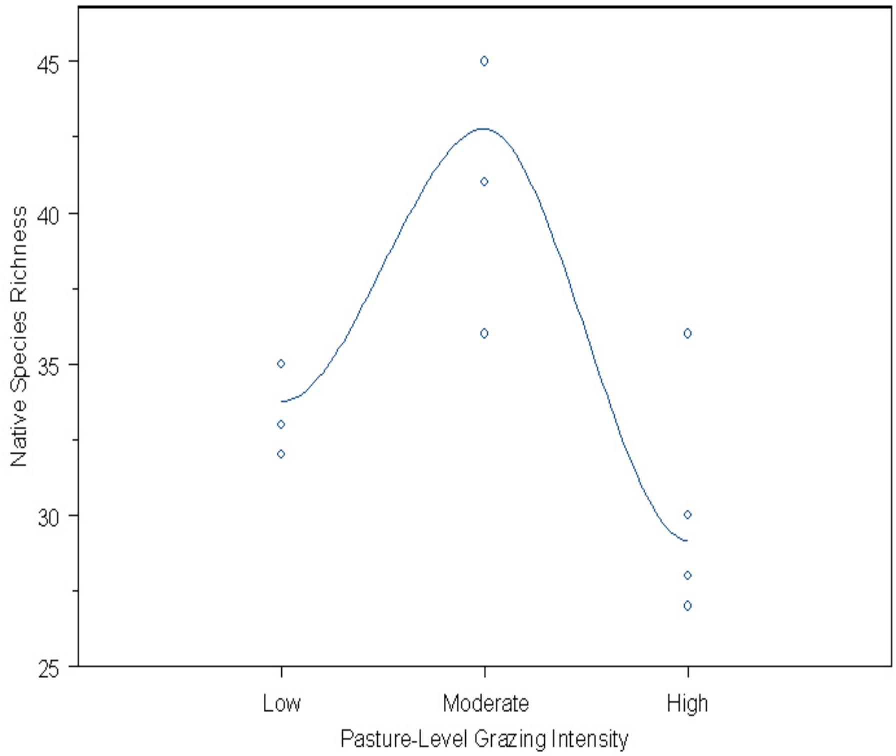
The objective of this analysis was to quantify the effects of grazing on native species richness rather than to rely exclusively on qualitative assumptions or simple indicators such as the ones above. Beyond this objective, approaching the issue with sensitivity to the various scales over which both grazing and diversity can be measured will result in greater insights into the factors influencing environmental change in the HBR. Figure 4.11 summarizes the response of plot-level ***native species richness*** to various scales of grazing intensity. The effect of ***pasture-level grazing intensity*** (defined by herder decisions) can be seen by looking at the mean for the distribution of points in each panel. Doing so reveals that moderate levels of use for a particular pasture (the middle panel) result in the highest plot-level native diversity. This is presumed to be due to the fact that grazing in general creates opportunities for less competitive species to establish themselves, which supports the greatest number of species (Grime 1973).

Figure 4.11: Response of plot level native specis richness to multiple scales of grazing. The structure of the data is ignored in this graph, as plots are nested within pastures. Each panel separates plots by pasture-level grazing intensity only. Within each panel plot-level native species richness is plotted in response to plot-level grazing intensity. Looking at native species richness across panels, one can see that moderately grazed pastures tend to have plots with higher numbers of native species. Within panels, plot-level grazing intensities appear to have a subtly negative or quadratic effect, as shown by the loess curve fit to plots of pastures with the same level of grazing intensity. The strength and significance of these grazing effects relative to other controlling variables is discussed in greater detail below.



A similar effect is seen when looking at native species richness within the 1000m^2 plots of each sampled pasture (n = 12). Figure 4.12 summarizes the effects of pasture-level grazing intensities on native species richness at this larger scale. The same bell-shaped trend can be seen, with moderate use of the pasture encouraging the highest overall native species richness on the pasture.

Figure 4.12: Response of pasture-level native species richness relative to pasture-level grazing intensity.



Returning to Figure 4.11, native species richness appears to show more of a negative trend with regard to plot-level grazing intensity. This is illustrated by the loess curve fit to the plot estimates within each panel. In other words, on plots where toxic and low palatability species have established themselves, suggesting a history of high grazing intensity at that locale, there appears to be a subtly quadratic or negative effect on native species richness. The significance of this trend will be born out in the modeling effort below, but the effect appears strongest on pastures infrequently visited by herders, the majority of which are in the highest reaches of the HNP (see the far-left panel of Figure 4.11). Such sites are notoriously fragile and localized concentrations of animals on them may have negative effects on diversity with the least chance of recovery. I interpret this trend as a reason for recommending that herders encourage the dispersal of their animals on pastures regardless of how intensely a particular pasture is used, especially when those pastures are in low productivity, sensitive sites within the conservation core of the HNP. This is something that can only be done with active herd management, a situation made more difficult with the increased labor demands of non-farm employment. Thus, the practice of leaving animals untended in the higher altitude pastures of the HNP (discussed in Chapters 2 and 3) becomes even more problematic in light of viewing grazing at these two scales. Moreover, this finding implies that the relative palatability and functional characteristics of species on a pasture may be better indicators of over-grazing than the presence of *Astragalus garbancillo*, *Aciachne pulvinata*, *Opuntia flocossa* and *Pennisetum clandestinum* alone.

FITTING MODELS OF NATIVE SPECIES RICHNESS

The patterns of Figure 4.11 suggest that both scales of grazing affect native plant diversity. To quantify this effect I fit and evaluated a set of candidate models that include various combinations of predictors. Some models were formulated with the grazing variables of interest and some were formulated without them. In addition, some were formulated with various combinations of the grazing predictors (for a similar approach see Beever, Huso, and Pyke 2006).

First, let y_{ij} be the ***native species richness*** of plot *j* in pasture *i*. I assume that $y_{ij} \sim N(\mu_{ij,} \sigma^2)$. Modeling began by fitting unconditional means and random intercept models which exclude the grazing variables of interest. The model UME is the unconditional means model (see Table 4.4a, Table 4.4b shows each model in composite form).[91] RIM is the random intercept model, a model in which only level-1 (plot-level) controls are added. These include the proportion of the plot in vegetative cover (***cover***) and the proportion of the plot dominated by a single species (***dominant cover***). The CONTROL model builds on RIM by adding the level-2 (pasture-level) control variables of ***aspect*** and ***wetness***. Grazing variables first appear in the PLOT model. This model includes all the variables of CONTROL, plus ***plot-level grazing intensity***, a measure defined for every plot from the average palatability of species present in it. The PASTURE model includes all the variables of CONTROL, plus ***pasture-level grazing intensity***, a measure defined for every pasture from herder reports of the factors most important in dictating which pastures they are likely to use. BOTH is the most comprehensive model, which includes both ***plot and pasture-level grazing intensity*** in addition to all controls.

91 A simple model whose hallmark is the absence of predictors at all levels. In an ordinary linear mixed model the unconditional means model serves to describe outcome variation. It also serves as a baseline model for model comparisons.

Table 4.4a: Taxonomy of multilevel models fitted to native species richness data, where random effects are the same for each model, namely $u_{0i} \sim N\left(0,\, \sigma_0^2\right)$.

MODEL	LEVEL1	LEVEL2
UME	$\mu_{ij} = \beta_{0i}$	$\beta_{0i} = \beta_0 + u_{0i}$
RIM	$\mu_{ij} = \beta_{0i} + \beta_1 cover_{ij} + \beta_2 domcover_{ij}$	$\beta_{0i} = \beta_0 + u_{0i}$
CONTROL	$\mu_{ij} = \beta_{0i} + \beta_1 cover_{ij} + \beta_2 domcover_{ij}$	$\beta_{0i} = \beta_0 + \beta_5 aspect_i + \beta_6 wetness_i + u_{0i}$
PLOT	$\mu_{ij} = \beta_{0i} + \beta_1 cover_{ij} + \beta_2 domcover_{ij} +$ $\beta_3 plotgraze_{ij} + \beta_4 plotgraze_{ij}^2$	$\beta_{0i} = \beta_0 + \beta_5 aspect_i + \beta_6 wetness_i + u_{0i}$
PASTURE	$\mu_{ij} = \beta_{0i} + \beta_1 cover_{ij} + \beta_2 domcover_{ij}$	$\beta_{0i} = \beta_0 + \beta_5 aspect_i + \beta_6 wetness_i +$ $\beta_7 pasturegraze2 + \beta_8 pasturegraze3 + u_{0i}$
BOTH	$\mu_{ij} = \beta_{0i} + \beta_1 cover_{ij} + \beta_2 domcover_{ij} +$ $\beta_3 plotgraze_{ij} + \beta_4 plotgraze_{ij}^2$	$\beta_{0i} = \beta_0 + \beta_5 aspect_i + \beta_6 wetness_i +$ $\beta_7 pasturegraze2 + \beta_8 pasturegraze3 + u_{0i}$

Note: The variables *pasturegraze2* and *pasturegraze3* refer to moderate and high pasture-level grazing intensities respectively.

Table 4.4b: Composite versions of the models of native species richness shown in Table 4.4a.

MODEL	FORMULATION
UME	$\mu_{ij} = \beta_0 + u_{0i}$
RIM	$\mu_{ij} = \beta_0 + \beta_1 cover_{ij} + \beta_2 domcover_{ij} + u_{0i}$
CONTROL	$\mu_{ij} = \beta_0 + \beta_1 cover_{ij} + \beta_2 domcover_{ij} + \beta_7 aspect + \beta_8 wetness + u_{0i}$
PLOT	$\mu_{ij} = \beta_0 + \beta_1 cover_{ij} + \beta_2 domcover_{ij} + \beta_3 plotgraze_{ij} + \beta_4 plotgraze_{ij}^2 + \beta_7 aspect + \beta_8 wetness + u_{0i}$
PASTURE	$\mu_{ij} = \beta_0 + \beta_1 cover_{ij} + \beta_2 domcover_{ij} + \beta_5 pasturegraze2_{ij} + \beta_6 pasturegraze3 + \beta_7 aspect + \beta_8 wetness + u_{0i}$
BOTH	$\mu_{ij} = \beta_0 + \beta_1 cover_{ij} + \beta_2 domcover_{ij} + \beta_3 plotgraze_{ij} + \beta_4 plotgraze_{ij}^2 +$ $\beta_5 pasturegraze2_{ij} + \beta_6 pasturegraze3 + \beta_7 aspect + \beta_8 wetness + u_{0i}$

Note: The variables *pasturegraze2* and *pasturegraze3* refer to moderate and high pasture-level grazing intensities respectively.

Non-zero variance components for the RIM model justify the modeling effort. Variance in native species richness between pastures (σ_0^2) was 5.1, and within pastures (σ_1^2) was 5.0 after controlling for the effects of ***cover*** and ***dominant cover***. As suggested by these values, variance in plot-level native species richness is nearly equally partitioned between as well as within pastures. Beyond the justification this provides for the modeling effort, further statistics summarized in Table 4.5 suggest that the inclusion of grazing as a multi-scaled disturbance explains a significant amount of this variation

Table 4.5: Evaluation of the candidate models of native species richness. Model evaluation was done with information theoretic measures. Likelihood ratio tests are shown only for comparison. The likelihood ratio statistic, $-2\log\Lambda$ is calculated between nested models and is presented on the line of the fuller model. Akaike weights reported in the last column suggest that the model that includes both scales of grazing has a 100% probability of being the best model selected under repeat sampling.

MODEL	LOG LIKELIHOOD	$-2\log\Lambda$	P-VAL	# OF ESTIMATED PARAMETERS	AIC_c	Δ_i	AKAIKE WEIGHT
UME	-301.04	X	X	3.00	608.29	65.74	0.00
RIM	-281.79	vs. UME: 38.50	<0.0001	5.00	574.11	31.56	0.00
CONTROL	-279.35	vs. RIM: 3.05	0.2170*	8.00	575.83	33.28	0.00
PLOT	-273.92	vs. CONTROL: 12.67	0.0018	10.00	567.87	25.32	0.00
PASTURE	-267.67	vs. CONTROL: 25.19	<0.0001	10.00	555.35	12.80	0.00
BOTH	-258.82	vs. PLOT: 30.22 vs. PASTURE: 17.70	<0.0001 0.0001	12.00	542.55	0.00	1.00

*This value suggests that the addition of pasture-level control variables is not a significant improvement over the random intercept model that includes only plot-level controls. Yet these variables do reduce overall variance in addition to subtly influencing the effect of the grazing variable. Fitting a model where aspect and wetness are dropped from the BOTH model results in an AIC_c roughly 8 points higher (550.85 vs. 542.55) than the BOTH model above. Thus I retain these pasture-level variables as controls.

I use Akaike's Information Criterion (AIC) to evaluate the candidate models shown in Table 4.5. As mentioned in Chapter 3, this statistic can be used to compare models containing different predictors, whereas likelihood ratio tests can only be performed on models that are nested. Burnham and Anderson suggest that when the ratio of the number of observations to predictors drops below 40 (in this case 120:6 = 20), that AIC_c should be used (Burnham and Anderson 2002).[92] This statistic is defined as:

$$AIC_c = -2\log L(\theta) + \left(\frac{2K(K+1)}{n-K-1}\right)$$

[eqn. 4.3]

In general, models with smaller values are better models of the response. The model BOTH, which includes plot and pasture-level grazing intensity variables has an Akaike weight of 1.00 (see Table 4.5). This statistic indicates that the model that treats grazing as a scaled disturbance would be ranked best among other options 100% of the time. These results suggest that grazing is an important factor affecting biodiversity in the HBR. Beyond this observation, the likelihood ratio test calculated between models with one scale of grazing (PASTURE or PLOT) and the CONTROL model suggest that pasture-level grazing is perhaps more important (e.g., the likelihood ratio statistic is roughly double that of the PLOT model). These statistics argue that herder decisions with respect to which pasture to use are especially influential in shaping native plant diversity in the HBR. The model coefficients discussed below will show that this influence is not entirely negative, as some have assumed.

[92] In the AIC equation, the log likelihood function for a given model is evaluated at the maximum likelihood estimate of the parameter set θ. K is the number of parameters estimated in the model.

RESULTS

To summarize the results of the model selection process above, y_{ij} is the ***native species richness*** of plot *j* in pasture *i*, where $y_{ij} \sim N(\mu_{ij,}\sigma^2)$. The best model defined above can effectively be written as follows:

$$\mu_{ij} = \beta_0 + \beta_1 \operatorname{cov} er_{ij} + \beta_2 dom \operatorname{cov} er_{ij} + \beta_3 plotgraze_{ij} + \beta_4 plotgraze_{ij}^2 + \beta_5 pasturegraze2_{ij} + \beta_6 pasturegraze3 + \beta_7 aspect + \beta_8 wetness + u_{0i}$$

[eqn. 4.4a]

Where, for the random effects I assume that:

$$u_{0i} \sim N(0,\, \sigma_0^2)$$

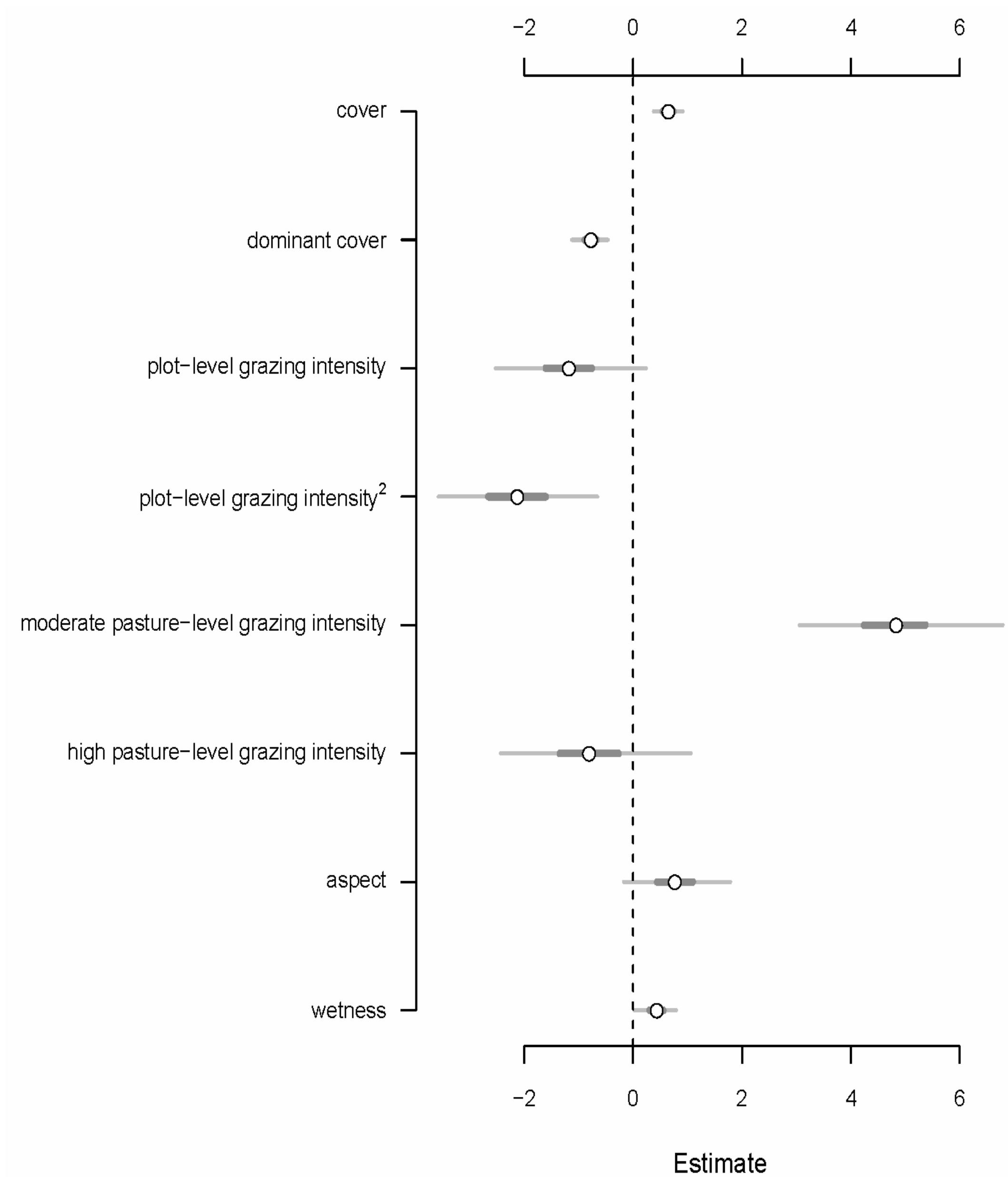
[eqn. 4.4b]

The following results are restricted to this model. Estimates obtained by both frequentist and Bayesian versions of the model are summarized in Appendix 4.8.[93] I display the Bayesian estimates graphically in Figure 4.13 in order to allow for a quick assessment of the relative effects of different variables and their magnitudes. Recall that virtually every pasture sampled had at least one of four indicators or invasive species occurring on it; still those moderately grazed had the highest plot-level native species richness (see Table 4.3). Figure 4.13 confirms this by showing the contrast in native

[93] I ran a Bayesian version of this model in WinBugs as a way of cross-checking estimates obtained from the "LME" library in R (a frequentist approach). I did so because I discovered that R generated erroneous variance estimates for the pasture-level, as it was likely converging to a local solution rather than a global one. These two methods produced very similar estimates, but WinBugs allowed me to generate accurate variance components for the calculation of Pseudo-R squares, as well as posterior probabilities of the estimates which can be used for predictive simulation. I do so below as a means of exploring the model's appropriateness as a tool for predicting changes in native species richness with changes in grazing intensity.

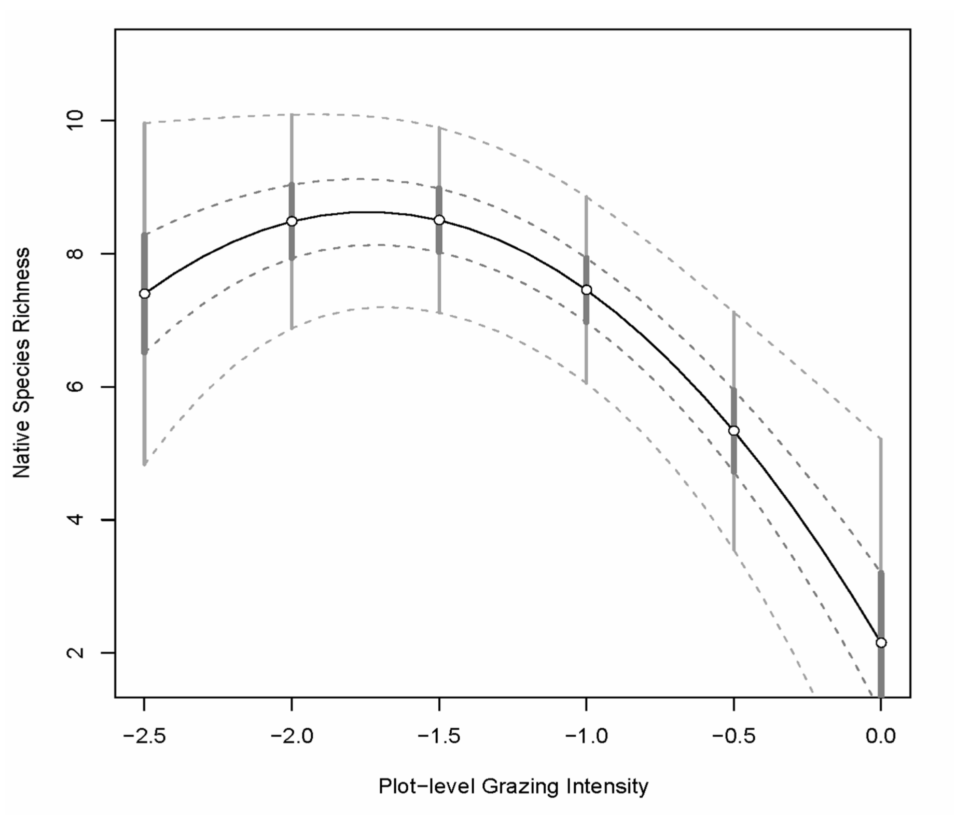
species richness between pastures of moderate and high grazing intensity relative to low. Notice that the effect of ***moderate pasture-level grazing intensity*** is positive and relatively strong compared to other variables. Plots within pastures of moderate use had approximately 5 more native species on average than plots of pastures less utilized by resident herders (stdev = 1.0). Additionally, pastures with the highest grazing intensities show no appreciable differences in native species richness from those with the lowest.

Figure 4.13: Estimates, 50% and 95% credibility intervals for the regression coefficients of predictors in the multilevel model of native species richness. The parameter estimates were obtained within a Bayesian framework using non-informative priors and are the estimated means of the posterior distributions of the parameters. The interval endpoints are the estimated 0.025, 0.25, 0.75, and 0.975 quantiles of the corresponding posterior distributions. The intervals can be treated as probability statements about the true values of the parameters given the data. The proportion variables of *cover* and *dominant cover* are scaled so that every 10% change results in the change in native species richness shown on the estimate scale. For example, with every 10% increase in *cover* plots gain one additional native species on average. *Plot-level grazing intensity* and its quadratic term (*plot-level grazing intensity*^{2}) must be treated together for interpretation (see Figure 4.14).



Plot-level grazing intensity is a continuous variable with a quadratic relationship to the response. An interpretation of its effect on native species richness is complicated by the fact that the two terms in the model must be interpreted together. The quadratic term ***plot-level grazing intensity***^{2} is negative, therefore the parabola opens downward. This produces the same bell-shaped trend seen with pasture-level grazing intensity. However, because its relationship to the response is quadratic, the effect of changing plot-level grazing by a fixed amount will vary depending on its current value. Figure 4.14 must be used to interpret this effect.

Figure 4.14: Effect of plot-level grazing intensity on native species richness. Fences represent 50% and 95% confidence intervals. All other predictors of the model are set to 0, thus the scale on the y-axis is only relative.

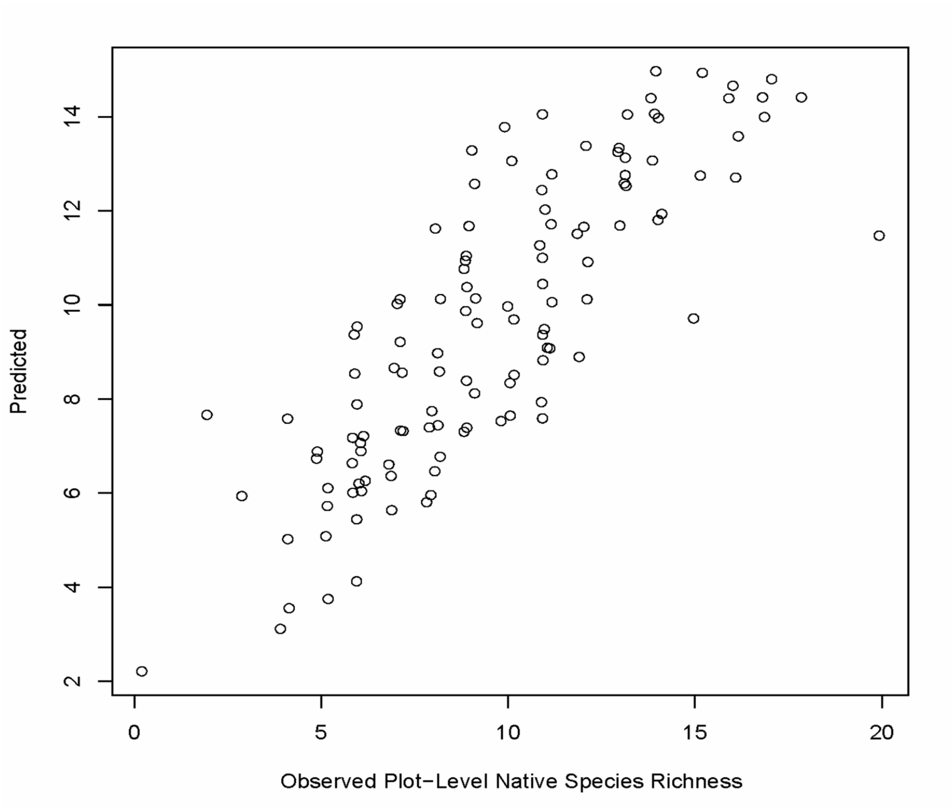


Note that plot-level grazing exerts a subtle effect on the response until it gets sufficiently high (see the parabola in Figure 4.14). Compare estimates of native species richness at a plot-level grazing intensity of 0 (toxic plots assumed to have a history of intense use) and -1.0 (less intense use) for example. The average difference between these two plots is approximately 5 species. Since the 95% confidence intervals (the widest bands) surrounding these estimates do not overlap, this difference is statistically significant. Thus, as plot-level grazing intensity increases beyond a threshold of moderate intensity, there is the potential to lose a number of native species.

The effects of the remaining biophysical controls can be estimated from Figure 4.13. ***Cover is*** positively correlated with native species richness, ***dominant cover*** is negatively correlated, and the pasture-level variables of ***aspect*** and ***wetness*** show subtle positive effects. Recalling Figures 4.8 and 4.9 to aid in the interpretation of these pasture-level predictors, pastures with aspects tending toward the east, which many authors suggest are warmer (Smith 1988, Smith 1977), had more native species than those with west-tending aspects. While wetter pastures (*bofedales*) had more native species than drier ones (*gramadales).* Again these variables were simply included as controls, but the magnitude of their effect can be compared to the grazing variables of interest in Figure 4.13. Doing so clearly illustrates that the relative effects of both scales of grazing are stronger than any of the other variables considered.

A pseudo R-square calculation reveals that the plot-level grazing variable alone accounts for 8% of the variance in native species richness within pastures, while the pasture-level grazing variable explains 94% of the variability in native species richness between pastures.[94] In total, this model has a correlation R-square of 68%, a statistic that quantifies the strength of the correlation between observed and predicted values shown in Figure 4.15.

Figure 4.15: Observed vs. predicted estimates of native species richness.



[94] Consideration of additional level-1 variables would undoubtedly improve the amount of variance in plot-level native species richness within a pasture. Soils play an important role in determining localized vegetation response. At this scale it is likely that exchangeable cations, soil pH, organic matter and suitable conditions for needle-ice formation (e.g., medium textured soils, moisture, and slope) may be partially responsible for the residual variance in species richness. Future work will seek to include such measures as well as the effects of climate and other abiotic drivers common to stochastic environments.

Pasture-level Native Species Richness

The majority of the modeling effort focused on native species richness at the plot-level, yet Figure 4.11 suggests that pasture-level diversity is similarly responsive to grazing. At the scale of the largest Modified Whittaker plot ($1000m^2$), there is no longer an issue of pseudo-replication, and thus no need for the multilevel modeling framework. As a secondary analysis, I fit an ordinary linear regression model of pasture-level ***native species richness*** in R. Because the sample size is significantly reduced (n = 12 vs. n = 120), I only model the effects of ***pasture-level grazing intensity***. The coefficients of this model are similar to those of the plot-level model. Native species richness at the scale of the pasture is highest on pastures with moderate grazing pressure. These pastures have an average of 8 more species (se =2.5) than those with less grazing intensity ($p \geq |t| = 0.0103$). The model's adjusted R-square suggests that it explains 65% of the variance in native species richness at the pasture-level. A partial F-test between it and an intercept-only model shows that the amount of variance explained by the addition of the pasture-level grazing variable is significant ($p \geq |F| = 0.0035$). Additional explorations of the model assumptions are presented in Appendix 4.9.

DISCUSSION

The initial question motivating this analysis was whether or not there was evidence in support of over-grazing. This question led me to consider various scales over which to evaluate disturbance and response. The coefficients of the best plot and pasture-level models suggest that a long history of land use in the region has created a landscape adapted to grazing by resident agro-pastoralists. Although invasive species and indicators of over-grazing were encountered on the majority of pastures sampled, those pastures with historically moderate levels of use by resident herders have maintained the highest native species richness at both scales over which it was evaluated.

Many have studied the influence of abiotic and biotic disturbances on species diversity, but it is clear that there is no consensus on the form this relationship should take. A bell-shaped trend in response to a disturbance gradient (i.e., intensity, frequency, time since last disturbance) is predicted by the Intermediate Disturbance Hypothesis (Connell 1978, Huston 1979, Reice 1994). This relationship has been extensively explored in grazed landscapes (Adler, Raff, and Laurenroth 2001, Bakker, Blair, and Knapp 2003, Collins et al. 1998, Fensham, Holman, and Cox 1999, Milchunas et al. 1989, Olff and Ritchie 1998, Stohlgren, Schell, and Vanden Heuvel 1999). However, in a survey of published studies between the years of 1985–1996, Mackey and Currie (2001) show that this "peaked" trend is only statistically significant in approximately 16% of the studies of species diversity-disturbance relationships that they reviewed. While their findings clearly illustrate that other patterns are possible (e.g., monotonic negative, monotonic positive or insignificant effects), my findings, even if not generated from an explicit test of the IDH, contribute to this small percentage of studies that do find trends in support of it, including some from the Andes (Becerra 2006, Wilcox, Bryant, and Belaun 1987).

The significant positive effects of moderate grazing seen in Figure 4.13, contradict the notion that policies of exclusion and de-stocking are necessary to achieve the conservation goals of the HBR.[95] In fact, such policies may actually lead to losses in biodiversity if grazing were to be reduced below moderate levels or removed all together. It is often the case that protected area managers have removed various sorts of disturbances that, in turn, have had ripple effects that have actually done more harm than good (Chase 1987, Christansen 1988). Given the HBR's dual objectives of biodiversity conservation and poverty alleviation, removing or disallowing grazing seems equally problematic. I interpret this finding as support for local resource users, with two important qualifications.

The first involves an assumption that historic grazing practices will continue unchanged. The findings of Chapters 2 and 3 suggest that traditional grazing practices are changing. While distance from the community and distance to water are long-standing and ubiquitous costs for all household to consider, new constraints influencing household decision-making are emerging due to market involvement. One land use scenario suggested by the findings of previous chapters is what I call "absentee" herding. This is a strategy in which many non-native livestock such as cattle are left untended in the HNP for extended periods of time.[96] At present, Collón and Pashpa each have communal herds totaling approximately 179 animals that graze permanently in the

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[95] I chose to model native species richness, but abundance-weighted measures of diversity, such as *Evenness*, *Shannon-Wiener's H*, *and Simpson's D*, show similar patterns. See Appendix 4.3 for plots of these measures against the pasture-level grazing gradient.

[96] In addition to the changing grazing intensities implied by absentee herding, there is also the possibility that community pastures such as Hualcan, Canish Pampa, Chaucashina Ruri and Lachoc could be used more heavily if local NGOs begin pasture improvement projects in Pashpa similar to those of Collón. In particular, the existing policy of fencing improved pastures in the community and restricting access to paying households, could potentially shift grazing pressure to remaining pastures within the community that are open to all households.

Ishinca valley. In addition to these communally owned animals, 42% of the households in Collón and Pashpa reported using the Ishinca valley on occasion. As shown in Chapter 3, absentee herding in the national park may grow with involvement in tourism. In turn, this may intensify grazing pressure on pastures that have historically been used very little. It is possible that if grazing pressure increases beyond the historic range experienced at these sites, irreversible species extinctions could occur (Cingolani, Noy-Meir, and Diaz 2005, O'Connor 1991).

Not only does absentee herding stand to increase the use of these pastures, but also the possibility that grazing on them may be highly concentrated. Thus, the second qualification to my support for grazing requires considering the grazing practices within pastures. The coefficients of the plot-level grazing variable suggest that localized grazing intensities beyond a certain threshold results in significant negative losses in native species richness. The possibility of recovery for a patch that has been heavily grazed in the past is difficult to assess, but many authors studying grazing impacts in low productivity environments such as the Andes suggest that livestock concentrations can facilitate structural changes (e.g., compaction, erosion) that may hinder vegetation reestablishment (Jaeffret and Lavorel 2003, Perez 1993, Perez 1998, Reichman, Benedix Jr., and Seastedt 1993, Zeidler, Hanrahan, and Scholes 2002).

Generally speaking, state-transition models of rangeland dynamics suggest that changes in grazing intensity can shift vegetation communities to an alternative stable states, sometimes with little prospect of them returning to a previous one (Friedel 1991, Laycock 1991, Rietkerk and van de Koppel 1997). If plot-level recovery from grazing is difficult or impossible, the negative effects on diversity may actually be magnified in the long-term.[97] I interpret this as reason to suggest that the use of pastures in the Ishinca valley involve actively herding and dispersing animals throughout a pasture as opposed to leaving them untended and concentrated on sensitive sites.[98] Similar recommendations have been made by other researchers of grazing systems in the Andes (Molinillo and Monasterio 2006). The possibility of increased use of the high altitude pastures of the Ishinca valley by untended herds warrants further study. In the following section I provide a brief, statistical justification for using the plot-level model discussed above as a tool for predicting changes in native species richness associated with these possible land use scenarios.

Exploring the Model's Suitability for Prediction

The predictions of any model are to be interpreted with caution. All attempts to make predictions, and to attach meaning to them, must be tempered with the realization that many additional factors may alter actual outcomes, including existing conditions and historical contingencies not accounted for by the model (e.g., soil pH, exchangeable cations, recent climate history and future climate change). Fortunately, a number of statistical tools exist to explore the predictive uncertainty of the imperfect reality we do capture with a model. I utilize the posterior distributions generated by the Bayesian

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[97] Some studies suggest that *bofedales* may exhibit equilibrial dynamics and be more responsive than *gramadales* to management interventions affecting stocking rates and the timing of grazing (Alzerreca et al. 2006).

[98] This is only cautious speculation based on the assumption that animals left to their own devices would be less optimally dispersed than they would be if actively managed by a herder. Because I do not have strong support for this claim, I chose to predict changes in native species richness only with an increase in ***pasture-level grazing intensity*** in the projections that follow.

model output shown in Appendix 4.8b to generate simulated native species richness values that can then be compared to the actual data in order to explore the limitations of my best plot-level model. Given an *a priori* interest in exploring the effects of changing land use scenarios and increased grazing intensities for certain pastures, I explore how well the model predicts native species richness for plots of different pasture-level grazing intensities in Figure 4.16 below.

Figure 4.16: Comparison of actual and simulated vales of native species richness grouped by pasture-level grazing intensity. The left and right edges of the box represent the first and third quartiles (the middle 50% of the values) while the vertical bars represent medians and red asterisks represent means. Open circles are outliers. Note that the distribution of actual values (the first boxplot in each panel) appears similar to the distributions of simulated values (the following 10 boxplots in each panel) with the exception of a simulated negative value in the top left plot.

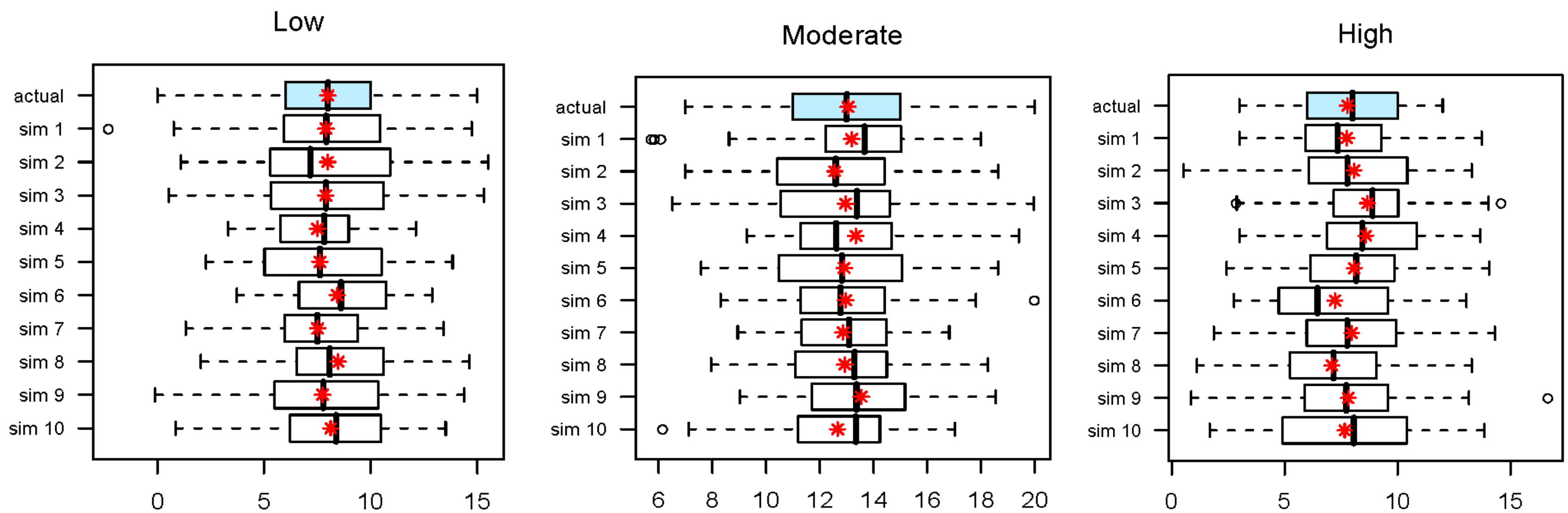


Figure 4.16 reveals that the actual data do not differ in obvious ways from the simulated data with one exception. The model is capable of producing negative values for native species richness, which in reality can never be less than zero. The first quartile, median, and second quartile of all simulations are similar to the actual data, and all are positive. However, one randomly sampled simulation out of the many run generates a negative outlier for pastures of low grazing intensity (see sim1 in the top left plot). In addition, the fences of several randomly sampled simulations approach 0 for pastures of both low and high grazing intensity extremes. These observations highlight a shortcoming of the model related to my earlier decision to treat the response as continuous and normally distributed. A closer look at the simulated values for pastures of low grazing intensity shows that the majority (62%) were positive. When simulations for these pastures did generate negative plot estimates, they were never more than 4 out of the 40 plot simulations that were run.[99] Predictions for pastures of high grazing intensity were better, with 92% producing positive values, and the remaining 8% producing no more than 1–2 negative estimates out of 40. Although these negative estimates are clearly poor reflections of reality, they are a minority. Nothing from this simulation suggests that the model is particularly unsuitable for making the predictions that follow.

[99] This number corresponds to the actual number of plots sampled within pastures of each level of grazing intensity.

Projecting Changes in Native Diversity

Below I summarize changes in native species richness predicted with the changing herding practices. This exercise was restricted to pastures that were actually sampled, and to those where current grazing is low or moderate, but may increase to high given the absentee and community-based herding strategies I discuss elsewhere (see Chapter 2). I consider these hypothetical projections to be worst-case scenarios that assume no response or modification to community rules governing the management of household herds. Table 4.6 summarizes the plot and pasture-level changes in native species richness predicted by the model.

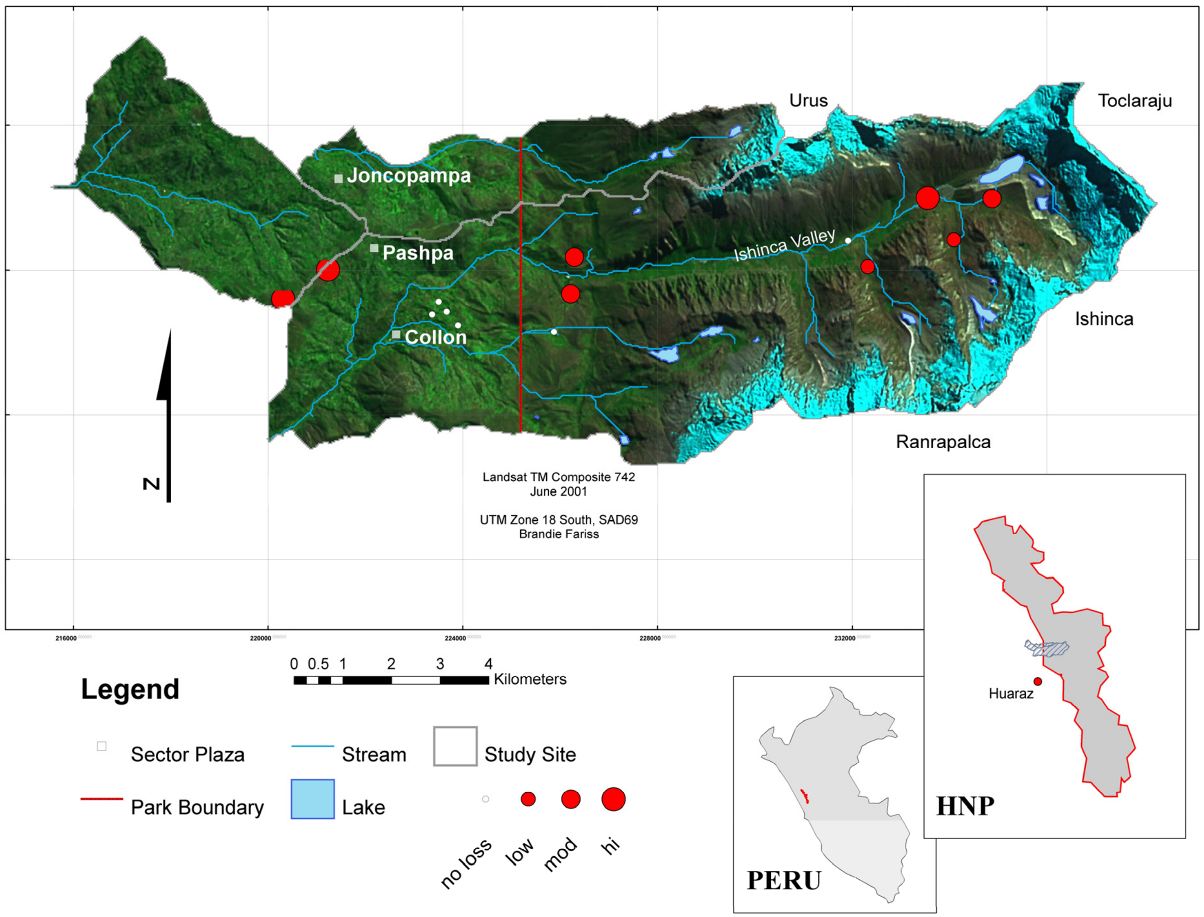
Table 4.6: Changes in native species diversity predicted with an increased use of community and park pastures associated with absentee and community-based herding strategies. Percentages represent the existing native diversity that may be lost. Parameter estimates for mean plot or pasture-level changes were obtained within a Bayesian framework using non-informative priors and are the estimated means of the posterior distributions of the parameters. The intervals following these estimates were derived from the estimated 0.025 and 0.975 quantiles of the corresponding posterior distributions. They can be treated as probability statements about the true values of the parameters given the data. Here I show 95% credibility intervals for the estimated changes.

PASTURE (# ON MAP)	LOCATION	MEAN PLOT-LEVEL CHANGE	% OF PLOT	MEAN PASTURE-LEVEL CHANGE	% OF PASTURE
Chacuashachina Ruri (2)	Collón	-0.9 $\pm$1.6	9.4%	-3.5 $\pm$2.5	5.5%
Chopi Uran (3)	Ishinca		11.7%		13.6%
Lachoc (7)	Pashpa		12.2%		13.6%
Pacha Pampa (10)	Collón		12.3%		8.3%
Canish Pampa (1)	Pashpa	-5.6 $\pm$3.8	38.9%	-11.5 $\pm$2.5	32.8%
Hualcan (6)	Pashpa		41.2%		32.8%
Miyu Pampa (8)	Ishinca		44.1%		26.2%
Pacclish (9)	Ishinca		43.7%		16.0%

Model predictions suggest a loss of native species on all pastures sampled. Chacuashachina Ruri (2), Chopi Uran (3), Lachoc (7), and Pacha Pampa (10) are predicted to experience an average loss of 1 ($\pm$1.6) native species at the plot-level. Pasture-level predictions for these four locales suggest a loss of as many as 3 ($\pm$2.5) native species overall, representing 8.3–13.6% of the existing native diversity found at these sites. Losses are greater for Canish Pampa (1), Hualcan (6), Miyu Pampa (8), and Pacclish (9). The first two pastures are in Pashpa. The latter two are within the HNP at the upper extent of the Ishinca valley. These pastures could experience an average loss of approximately 6 (± 3.8) native species per $1m^2$ with an increase in grazing pressure. This represents roughly 40% of the native diversity at this scale, which could result in losses of as many as 11 (± 2.5) unique native species at these sites, 16.0–32.8% of their overall native diversity.

Noteworthy is Miyu Pampa, which shows the highest percentage of plot-level loss (44.1%). This pasture also serves as base camp, and its poor condition is frequently noted by tourists, community members and reserve officials alike. In the dry season this locale not only has a fair number of cattle, but also a number of tents and climbers. With tourism in the region steadily increasing, the anthropogenic impacts on this pasture are clearly more than just those of local indigenous herders and their livestock. It will be the native species of this pasture, such as *Acaulamalva engleriana*, *Bartsia diffusa*, and *Belloa piptolepis* that could suffer as a result. Figure 4.17 below shows the locations and relative magnitudes of the pasture-level losses just summarized.

Figure 4.17: The relative loss of native species predicted for sampled pastures. Graduated symbols represent the proportion of extant species lost at the pasture-level. Larger symbols indicate greater proportional losses. Pastures not included in the projection exercise are those where grazing intensity is already high or were not part of the random sample (see Collón).



CONCLUSION

The results of this analysis do not provide unequivocal support for claims that indigenous herders are over-grazing the HBR. This finding is not altogether novel, considering that claims of deforestation were refuted with quantitative evidence from the same valley just a few years before (Tohan 2000). Although over-grazing indicators were observed throughout the fields and grasslands of the Ishinca valley, the outcomes for biodiversity require quantification rather than speculation. I have shown that the presence of species such as *Aciachne pulvinata, Astragalus garbancillo, Opuntia flocossa* and *Pennisetum clandestinum* are not adequate means of monitoring biodiversity conservation in the HBR. Moderate levels of grazing appear to have played an important role in maintaining the native diversity of this Andean ecosystem. It is assumed that if done sustainably, grazing can be complimentary to the reserve's conservation objectives, perhaps even critical to them.

Neither do the results advocate for leaving current management trends to play out as they may. Land use practices emerging in the reserve are problematic, as indicated by the projection exercise above. The very development encouraged by those most concerned with the conservation of the HBR (tourism), seems to have had the unintended consequence of increasing grazing pressure on pastures that could stand to lose native plant diversity if such uses can not be mitigated by the coordinated efforts of indigenous and conservation communities in the region. The results of Chapters 2 and 3 strongly support the possibility that a growing number of cattle will graze untended on the high altitude pastures within the park as more households enter into tourism work. A secondary possibility is one in which pasture improvement projects will be implemented in Pashpa similar to those of Collón. While I encourage NGOs working in the region to extend development assistance to this sector, I also encourage them to allow all *comuneros* access to these pastures, not just those that can pay for it. As it currently stands, restricting access to improved pastures to those that can afford it, is simply another means of enclosing the commons, which serves to increase grazing pressure elsewhere.

Projections of native species richness based on the intensified use of sampled pastures indicate a net loss of native plant diversity throughout the study area. The models on which these predictions are based are inherently simplified realities, and their predictions are qualified by a measure of uncertainty. Thus, my analysis was but a preliminary attempt at illustrating that the integration of statistical modeling and GIS can be a tool to assist in better land use and conservation planning for the region. Even so, a few cursory recommendations can be made in light of these predictions. Foremost is that reserve managers should recognize the importance of grazing to the maintenance of native plant diversity in the HBR. Any successful management policy in the HBR should not exclude, but rather assist local resource users in achieving sustainable levels of use. Specific suggestions for reserve managers include: (1) reevaluating the measures of 'over-grazing' frequently employed to evaluate the management practices of indigenous agro-pastoralists (2) providing development assistance for all sectors of a *common property* user-group, and all members of the community, (3) discouraging the increased use of the highest pastures of the park, and (4) encouraging herders to actively manage animals and not concentrate them within pastures. These outcomes could be achieved by:

(1) building on the existing relationship between community user-groups and park administration, which seems currently limited to community reporting of livestock totals, (2) developing a system of palatability monitoring by the user-group as a means of providing resident herders with useful information about the productivity of their pastures and reserve administrators with a measure more strongly correlated with biodiversity, (3) implementing pasture improvement projects in Pashpa (as well as Collón) that are free to all *comuneros,* which could perhaps be achieved by offsetting the cost with the sale of a proportion of the communal cattle herd, (4) establishing a herding cooperative for occasional use by households that experience labor shortages, with the herder's labor compensated by revenues generated from tourism entry fees to the Ishinca valley, and (5) assisting the community in developing and enforcing herd management strategies that are sensitive to conditions within as well as between pastures. These policies, which are directed at local resource users, should be considered in conjunction with policies directed at minimizing the impacts of visitors on sensitive grasslands in the park. This includes the possibility of establishing camping quotas, and the development of regulations governing activities with the greatest impacts on Miyu Pampa (base camp), including those of camp setup and waste disposal.

Given the park's dual objectives of biodiversity conservation and poverty alleviation, my findings suggest that more needs to be done to prevent a situation that would negatively affect both. I have shown that a multi-scaled modeling approach allows a more nuanced understanding of the interaction between populations (herders and livestock) and their environments. The treatment of grazing as a multi-scaled disturbance that is variable not only between but within pastures, allows for an exploration of the various ways that the actions of indigenous peoples affect biodiversity and how best to guide management and restoration efforts in the HBR. I have also shown that the integration of GIS and statistical modeling is a powerful tool that allows us to quantify the effects of humans on this landscape and to explore several “what-if” scenarios. Although models are at best, simplifications of reality, they provide important initial insights. GIS can be a useful tool for summarizing this analysis and communicating it to all stakeholders concerned with the protection of this unique highland environment. Insights gained from approaches such as these, offer us the best chance of achieving successful outcomes to “conservation with development.”

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CHAPTER 5

CONCLUSION: RECONSIDERING THE PROBLEM OF PEOPLE AND PARKS

Many landscapes over the globe have been wrested from local peoples through efforts to preserve the environment. In the process, an understandable desire to protect the natural world has disrupted local livelihoods and traditional institutions of resource management, fundamentally altering the very nature we seek to protect. To fully address the conservation challenges we confront today requires an admission that we have a role in creating them. This dissertation explores the scale and complexity of conservation in Peru's Huascarán Biosphere Reserve (HBR). I have shown that the problems of environmental degradation perceived in the region are over-simplified if viewed simply as a problem of unsustainable use by the region's indigenous agro-pastoralists. I have implicated government and non-government organizations and policies in creating realities for local resource users and communal institutions that undermine their ability to sustainably use the reserve. Beyond situating indigenous livelihoods in a broader political and economic context, I have also shown that indigenous livelihoods and institutions can be complimentary to, and even promote biodiversity conservation in the HBR.

Photo 5.1: Quechua woman and child in Collón (photograph by the author, 2002).

Having stated my support for indigenous peoples of the HBR, positive outcomes are not assured. Successful conservation is predicated on the support of communal institutions for resource management. This can only be achieved by an increased sensitivity to the conditions required for successful management of *common property*. I believe that the real problem with "people and parks" has not been the fact that they include people, but the fact that this inclusion continues to stop short of fully incorporating local institutions of resource management. This problem continues despite many years of recognition for the need to address it (e.g., Jodha 1992, Lynch and Maggio 2000, Prakash 1998).

Given the globalization of environmental conservation, the need to finally address this problem is nigh (Zimmerer 2006, Zimmerer, Galt, and Buck 2004). Conservation agendas have been especially prominent in South America, where an emphasis on biodiversity conservation has resulted in the establishment of protected areas in regions with long histories of human occupation (Rodriguez and Young 2000). In the HBR, approximately 226,000 indigenous Quechua agro-pastoralists along with a number of conservation interests ranging from national government agencies to international NGOs such as The Mountain Institute (TMI) are engaged in creating this contemporary Andean landscape. The paradigm adopted by the managers of the HBR has been one of including local peoples whose livelihoods are intimately tied to the land. This approach, often called the "third wave of conservation;" is a people-centered strategy that I believe is not only appropriate but necessary (Alcorn 1993, Oldfield and Alcorn 1991, Stevens 1997). People-centered conservation strategies often try to achieve sustainable use of protected areas by employing policies collectively referred to as "conservation with development."

Development is intended to encourage non-consumptive alternatives to the land-based economy for communities living in close proximity to protected areas. Tourism is often touted as an ideal means of achieving this outcome, and was a logical choice for the HBR. However, an adventure tourism industry has emerged in the region. Generally speaking, tourism offers a means of alleviating poverty while offsetting the reliance of the region's indigenous peoples on resources within the reserve, or at least reducing their reliance below a level of exploitation deleterious to biodiversity conservation. However, many empirical studies of tourism illustrate that it is neither inherently sustainable nor conducive to conservation despite the orthodoxy surrounding it (Lindberg, Enriquez, and Sproule 1996, McLaren 1998, Place 1995, Savage 1993). Similar realizations about the adventure tourism industry in the HBR were nascent when I began my research in 1999. Even today, problems with the tourism industry and problems with the land use practices of the region's indigenous community's are still primarily discussed as independent issues (INRENA 1990b, INRENA 2002). This research is one of the first efforts to explore their connection.

Like any form of development, the sustainability of tourism is intimately tied to how well it articulates with the existing subsistence economy (Grossman 1981). The particular brand of tourism that exists in the HBR (climbing and trekking) has done little to change the basic agro-pastoral focus of its indigenous communities. Quotes from members of Collón and Pashpa presented throughout the dissertation suggest that most continue to rely on mixed subsistence strategies of farming and herding, while opportunistically working in tourism to diversify the household economy. The dependence of the reserve's indigenous communities on agriculture and pastoralism is not a quaint anachronism; it is a contemporary reality. Adventure tourism is a seasonal and somewhat uncertain alternative to farming and herding dependent on a favorable political and biophysical climate. Beyond this, I have shown that tourism is not well-articulated with the traditional farming and herding activities on which residents of the HBR depend. This is primarily due to the increase in livestock it encourages, the labor shortages for herding it creates, and the potential for exacerbating problems of collective action. In sum, policies promoted by those concerned with the region's protection seem to have had the unintended consequence of undermining conservation efforts in the region. Such unintended outcomes have gained increasing attention in works such as *Globalization and New Geographies of Conservation* (Zimmerer 2006). It is this dilemma, inherent in "conservation with development," that motivated my dissertation research.

My research was specifically designed to explore the tensions and uncertainties in the outcomes created by the juxtaposition of adventure tourism and Andean livelihoods in the HBR. I explored these tensions as they were manifest among those with rights to graze in the Ishinca valley, which include the sectors of Collón and Pashpa in the indigenous community of Tupac Yupanqui. The conservation community considers the Ishinca valley to be degraded; to an extent, the community agrees (recall Figure 2.1). Contrary to the expectations of "conservation with development," this valley and its surrounding indigenous communities have received a steady stream of tourism traffic over the past decade without the apparent benefit of forestalling degradation. If the valley is truly degraded, it stands to reason that tourism has not fulfilled its promise for the HBR. This apparent contradiction led me to the following research objectives:

(1) Outline the current state of land use and land tenure among the Ishinca valley user-group, and explore how enclosure in a national park and the subsequent development of a tourism industry have affected household-level land use and the communal institutions for managing them;

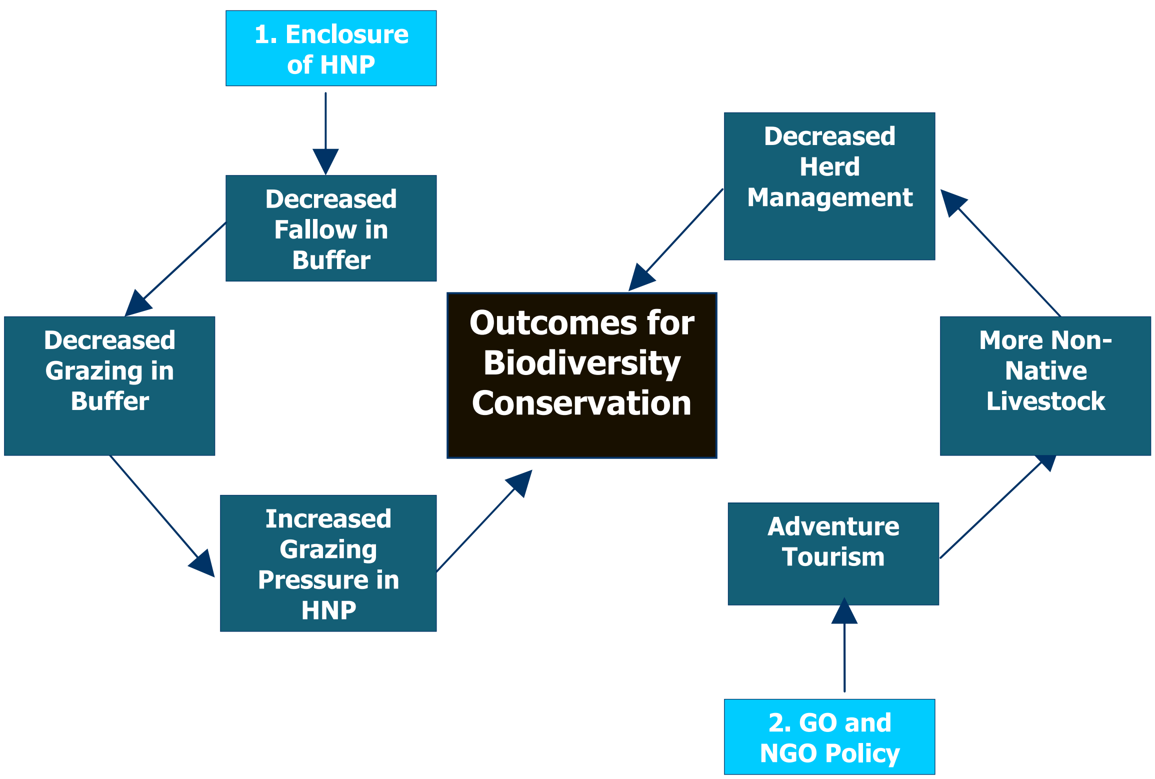
(2) Quantify how involvement in tourism affects household herding practices; and

(3) Describe the current condition of high elevation grasslands, the relative effects of grazing on native plant diversity, and the implications of changing herding practices for biodiversity conservation in the HBR.

SUMMARY OF FINDINGS

I conducted fieldwork over 18 months from July 2001 through December 2002 to pursue these objectives. Figure 5.1 summarizes the key findings of my work. I discuss each of them and their implications below.

Figure 5.1: Summary of the unintended consequences of (1) enclosure and (2) conservation policy in the HBR. Exogenous drivers of environmental change in the HBR are represented by the lighter blue boxes, while darker blue boxes indicate the changes in local resource management that have followed. First, I propose that enclosure of former agricultural lands has been partially responsible for reductions in fallow within the buffer zone. Continuous cropping in the buffer zone reduces the availability of grazing resources in the community, which in turn, increases the presence of livestock in the park. Secondly, policies of government and non-government organizations promoted adventure tourism in the HBR. I show that earnings from this industry have allowed some households to buy more non-native livestock. Tourism has simultaneously created labor shortages within the household, which I show have reduced the amount of time they spend herding. The outcomes for biodiversity conservation are mediated by communal institutions governing the rules of use and management of grazing resources in the HBR. This institution is represented by the black box. I argue that *common property* is an optimal institutional arrangement for managing the *puna* and can successfully mitigate environmental degradation, but I show that its ability to do so has been undermined. As a result, the future of biodiversity conservation in the HBR is uncertain, illustrating the importance of focusing efforts on developing policies sensitive to these institutions and the conditions required for their success.



Understanding Household Decision-Makers

In Chapter 2 I described the physical characteristics of land and resources in the Ishinca valley, their historical use and management, and the patterns of interaction that are emerging today in the farming and herding practices of variously scaled decision-makers (e.g., community, *empresa*, household). I specifically discuss how the creation of the park and the subsequent development of a tourism industry have affected the land use practices of households in the Ishinca valley user-group. The findings of this chapter suggest that the park boundary is substantially below the potential altitudinal range of cultivation for some communities. Although the boundary is often cited to occur at approximately 4000 meters (e.g., Byers 1999, Tohan 2000, Young and Lipton 2006), the actual boundary of the HNP is delineated by a series of survey markers digitized onto a 1:100,000 scale map, which places it at a lower elevation in Tupac Yupanqui. This boundary has effectively excluded from cultivation areas that were cultivated in the past (recall Figure 2.2). I argue that this may partially explain a number of ongoing conflicts between the reserve's indigenous communities and its administrators.

A simple inconsistency between the park boundary and the ecological reality of prior indigenous land use may partially explain the recent trend of fallow reduction on agricultural fields in the reserve's buffer zone, even though the size of the rural population has remained stable[100] Community informants from Collón and Pashpa reported that continuous cropping was on the rise, while household surveys showed that

[100] Population growth is another reason for fallow reduction (Boserup 1965), however rural population appears to be stable, as discussed in Chapter 2.

64% made fertilizer and pesticide purchases consistent with agricultural intensification. Given that agro-pastoral land use in the Andes is complimentary, changes in farming inevitably trigger changes in herding. Reductions in fallow have decreased the extent of grazing in the community. With less grazing land available in the buffer zone, there has been a spatial shift in the management of household herds that has increased their use of pastures within the boundary of the Huascarán National Park (HNP).

A year of ethnography and participant observation in the study community revealed two emerging herding strategies that support this claim. One of these, which I call "absentee" herding, is a strategy that involves leaving animals in the highest pastures of the HNP without supervision. This is a strategy typical of communal and *empresa* herds, but one of increasing importance for households with several cattle as well. Such practices are made more likely by the fact that permanent structures and residences from which herders might base themselves, are no longer permitted in the national park.[101] The distance between the herder's residence and the highest pastures grazed by his animals is now increased.[102] This spatial disjunction reduces the likelihood that livestock will be actively managed, especially in the dry season when sufficient forage is often found only at the highest altitudes.

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[101] The Italian Catholic Church is allowed to have a tourism lodge in the Ishinca valley, a double-standard frequently noted by some community members.

102 Winacpampa is an exception. It technically occurs within the national park boundary, but a dwelling attached to a corral located on this large pasture is used by families on a rotating schedule to manage the alpaca herds of EcoAgro. This applies only to the sector of Collón, Pashpa has no similar structures within the national park.

This picture is additionally complicated by policies of tourism development that have had similar unintended effects on household decision-makers. Changes unanticipated by conservation planners include an increase in herd size for tourism households, and a decrease in the odds of herding these additional animals. I explore these trends in Chapters 2 and 3. Household surveys revealed that nearly all households would like to own more livestock. Preferences were particularly strong for animals whose utility is enhanced by tourism. Tourism increases the utility of horses and mules, because they can readily carry the goods and gear of visiting mountaineers to base camp. Tourism also increases the utility of cattle, as they typically require less supervision and active herding than livestock such as sheep, llama and alpaca. The utility of animals requiring little supervision is more acute once households begin to experience labor shortages because of off-farm work. Tourism households had an average of 13 additional stock equivalents than their non-tourism counterparts ($p > |t| = 0.05$). This finding suggests that wage earning opportunities in tourism have provided the means to realize increases in herd size favoring larger non-native livestock, which in turn represent further investments in tourism.[103]

In Chapter 3 I attempt to address the implications of tourism involvement for the management of households herds in the HBR. The difference in herd size between tourism and non-tourism households translates to a moderate level of inequality in the community (Gini coefficient = 0.43). Roughly 20% of the households in the Ishinca

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[103] Stock equivalents are a measure of herd size that controls for diverse herd compositions based on relative forage requirements. See Appendix 2.2 for a list of values utilized.

valley user-group own more than half of all the livestock managed there (see Figure 2.11). While some suggest that inequality may exacerbate problems of collective action (Singleton 2001), game theory suggests that it can actually provide an incentive for users of a commons to cooperate (Dayton-Johnson and Bardhan 2002, Olson 1965). Specifically, the game theoretic model presented by Ruttan and Borgerhoff-Mulder (1999) assumes that wealthy herders are more vested in pasture resources; therefore they have more to gain from managing them sustainably. *Common property* theorists make supporting claims that dependence on a resource is a prerequisite for conservation of the commons (Bromley and Cernea 1989). In addition to showing how conservation can be "isomorphic with economic efficiency" for wealthy herders, the authors suggest that cooperation can be encouraged of relatively herd-poor households largely because wealthy households have an incentive to police their use and to coerce them into cooperation (Ruttan and Borgerhoff-Mulder 1999).

Chapter 3 tests for evidence of conservation by herd wealthy households, which I define as active management of the household herd. I consider this an optimistic hypothesis when recalling the previous discussion that households involved in tourism have a greater number of livestock. Yet other theoretical perspectives such as that provided by human behavioral ecology (HBE) provide less hopeful predictions about the effects of tourism on household decision-making. For example, there is an implicit correlation between tourism involvement and opportunity cost. Households with off-farm employment and wage-earning potential may have more animals, but they also have a number of competing uses of time which detract from their management. These opportunities include carrying gear to base camp in the dry season, a decision that earns a lucrative wage for the household, or going to school, a decision to invest in the social capital of the household.

The analysis in Chapter 3 was an attempt to test these competing hypotheses (herd wealth vs. opportunity cost). As well, it is an attempt to quantify how tourism affects the household decision-making process. Chapter 3 relies on the analysis of time allocation data and monthly household surveys. The time allocation data were analyzed using a multilevel modeling framework, which is extensively discussed in Chapter 3, as I believe that it has great potential for analysis of similar datasets which are common among ecological anthropologists, and human, cultural and political ecologists. Beyond the relative newness of the modeling approach to these data, I avoid null hypothesis testing in favor of testing competing hypotheses informed by the theories summarized above. To do so I use Akaike's Information Criterion (AIC) as a means of evaluating the relative strength of competing models. This paper illustrates the utility of time allocation data, multilevel modeling, and information theoretic methods of hypothesis testing for researchers of cognate disciplines and similar research questions.

The findings of Chapter 3 show that moderate levels of involvement in tourism exert a significant negative effect on household herding labor (see Figure 3.5). This finding is an unintended one for those promoting "conservation with development" in the HBR. The strength and direction of this effect suggests that the opportunity costs associated with a household's involvement in tourism are a realistic constraint to managing livestock. The prospect of earning 15 USD/day as an arriero, for example, appears to be enough to induce households with only moderate levels of involvement in tourism to spend less time herding in the dry season; similar trends are seen in the wet season. This finding supports the hypothesis of opportunity cost while simultaneously rejecting that of herd wealth (Ruttan and Borgerhoff-Mulder 1999). At the very least, the modeling effort of Chapter 3 suggests that the game theoretic model should be modified to include additional elements for subsistence households on the periphery of emerging markets. An initial attempt to modify theoretical predictions is presented in Appendix 3.8.

The practical implications of less herding effort by these households are numerous. Alternatives to actively managing the household herd include hiring or negotiating herding labor among social networks (e.g., neighbors and kin), buying fodder, or splitting the herd and managing only a part of them while leaving the remaining animals in the *puna*. I argue that many of these alternatives are unsustainable considering that reciprocity may be undermined by growing inequality and heterogeneity within the community (an issue for future fieldwork), and that the alternatives of concentrating livestock in the community or leaving them untended in the highest reaches of the national park are likely to have harmful environmental effects.

Up to this point, one might conclude that indigenous peoples (more specifically some that are involved in tourism) are inimical to the conservation objectives of the HBR. The objective of Chapter 4 is to describe the current condition of high elevation grasslands, the relative effects of traditional grazing strategies on native plant diversity, and the implications of newly emerging herding practices for biodiversity conservation in the HBR. I conducted vegetation samples using nested Modified Whittaker plots at 12 pastures throughout the grazing lands managed by the study. A preliminary analysis of the ecological data revealed that over-grazing indicators such as *Aciachne pulvinata*, *Astragalus garbancillo* and *Opuntia flocossa,* as well as the invasive *Pennisetum clandestinum,* occur throughout the study site (Poma 2002, Tovar and Oscanoa 2002). All but 1 of the 12 pastures sampled had an occurrence of at least one of these over-grazing indicators. This finding validates, to some degree, perceptions that the Ishinca valley is degraded. However, these species appear to be poor proxies for native biodiversity, the measure of degradation most important to the conservation objectives of the HBR (INRENA 1990a, INRENA 2003).

A quantitative model of native species richness suggests that biodiversity is not strongly correlated with the presence of these indicators; it is however, correlated with grazing pressure, but not in the way assumed by reserve administrators. Pastures with a long-standing history of moderate use by resident herders had the *highest* native diversity at both of the scales I measured (plot and pasture). This finding casts doubt on the prevailing assumption of over-grazing in the Ishinca valley, at least with respect to a measure of conservation important to reserve administrators. Rather than view the agro-pastoralists of the HBR as the problem, this analysis argues that they have been instrumental in creating the biodiversity we are attempting to protect. Any policies focused on excluding or destocking these grazers might result in reducing the biodiversity of the HBR (for similar arguments see Robbins et al. 2006).

However, this is not to say that the emerging herding practices among the region's agro-pastoralists are similarly conducive to high biodiversity. Indigenous decision-makers are embedded in an ever-changing social, political and economic context. This implies that herding practices will be equally dynamic.[104] An important methodological goal of Chapter 4 was to develop a suitable model of native species richness that could be used as a tool for predicting changes in biodiversity under a variety of potential land use scenarios. I was particularly interested in modeling native species richness in response to the emerging herding strategies suggested by the ethnographic observations I discuss in Chapter 2. To do so, I fit a number of models of native species richness with and without the effects of grazing. The best plot-level model emerging from information theoretic analyses included: the proportion of the plot in vegetative cover, the proportion of the plot dominated by a single species, aspect, wetness, pasture-level grazing intensity determined by herder decisions about which pastures to use, and plot-level grazing intensity determined by the grazing practices of livestock within pastures. The coefficients of this model suggest that moderate pasture-level grazing intensities have a positive effect on native diversity. Plots within pastures of moderate use had an average of 5 more native species (se = 1) than those of pastures with less use. Yet regardless of the level of utilization of a particular pasture, a high plot-level grazing intensity has a negative effect on native diversity. State-transition models of rangeland dynamics suggest that there is no guarantee that plots exposed to high localized grazing intensities will rebound once grazing pressure is removed (Laycock 1991, Rietkerk and van de Koppel 1997). This is especially likely under conditions of low productivity and short grazing history or relatively abrupt change in grazing regime (as reviewed in Cingolani,

[104] As discussed in Chapter 4, this model is based on "space for time" substitution, as there was no baseline from which to measure changes in grazing intensity for different locales. Furthermore, detailed climate and soils data were not available for this analysis, but will be incorporated in future work.

Noy-Meir, and Diaz 2005). I interpret these findings to suggest that grazing, when actively managed by the herder to avoid livestock concentrations on sensitive sites within the pasture, can be complimentary to biodiversity conservation in the HBR. Conversely, I conclude that any herd management favoring the concentration of livestock within pastures, as is probable with staking animals near the community or leaving them untended in the national park, may have deleterious and irreversible effects. Overall, plot and pasture-level grazing intensities explain 8% of the variance in native species richness within pastures, and 94% of the variance in native species richness between them. The best pasture-level model of native species richness attained similar explanatory power by including only pasture-level grazing intensity. This model explained 65% of the variance in native diversity on the pastures I sampled.

Given the decent predictive power of both models, I used each as tools for predicting the changes in plot and pasture-level diversity that might occur with the changing herding practices I observed. By doing so, I illustrate the dangers of assuming that indigenous livelihoods are static and somehow unaffected by larger social, political and economic realities. Emerging herding practices such as absentee and community-based herding (discussed in Chapter 2) imply that grazing intensities will increase at certain locales relative to their historic levels of use. These strategies additionally suggest that livestock will not be optimally managed or dispersed on these sites. Livestock concentrations in marginal environments such as these may lead to structural changes (i.e., compaction and erosion) with irreversible effects on plant communities (Van de Koppel, Rietkerk, and Weissing 1997). I modeled various hypothetical scenarios based on these emerging practices, assuming the worst-case scenario that communal institutions may not be able to adequately respond (see Chapter 2 for a discussion of reasons why this might be the case). The findings from this modeling effort indicate that native species richness could be adversely affected.[105]

Collectively, the results of Chapter 4 illustrate potential outcomes for biodiversity conservation in the HBR. These outcomes assume that the community might not adopt sufficient new operational rules to manage household herds. It *has* adapted historically and may continue to do so, thus I present these findings as a means to emphasize the critical importance of supporting communal institutions with appropriate policy. Throughout the dissertation I elaborate on the challenges of *common property* management experienced by the indigenous community, and show that they too have increased with enclosure and the subsequent development of tourism. In sum, I argue that successful biodiversity conservation in the HBR will require thoughtful consideration and acceptance of the responsibility which both local and supra-local actors have in shaping the future of the HBR.

Understanding Institutions of Common Property

The outcomes for biodiversity conservation in the Ishinca valley presented herein are speculative at this point. Future outcomes will depend largely on successful mediation by the *common property* institution formed by residents of Collón and Pashpa. I show throughout the dissertation that this mediation is currently hindered by the obstacles to cooperation outlined in Chapter 2 (e.g., inequality, tenure insecurity, and

[105] Although these model projections only test for changes in landscape-scale grazing intensity, both absentee herding and community-based herding implies that grazing may be concentrated within sites as well, enhancing the negative effects on native species richness.

disintegration of the moral economy). A substantial portion of this chapter was devoted to describing the important condition that resources in the reserve are managed implicitly as *common property*. Because of this arrangement, it is necessary to consider the conditions under which household cooperation is likely to be maintained, and when it is likely to be compromised. Empirical studies reveal that sustainable *common property* management is predicated on a number of factors that have historically existed in TY, but may now be faltering. Chapter 2 concludes by exploring the role of state intervention, market integration, and demographic change in affecting the likelihood of households continuing to cooperate in the management of commons in the HBR.

Common property theory suggests that insecurities of ownership and differential access may undermine the cooperation of *common property* users. The discussion in Chapter 2 highlights the ways in which the park's creation, the activities of NGOs, and involvement in tourism potentially exacerbate problems of ownership, access and equality. For example, although park decree permits indigenous communities continued access to grazing resources, lingering ambiguity over ownership and use rights has undermined cooperation within the community, as well as cooperation between it and the reserve. The premise of my research is that *common property* can be the optimal arrangement for managing a common pool resource such as the *puna*. Yet, its success will only be ensured when it is supported rather than undermined by exogenous actors (i.e., reserve administrators and NGOs).

A related issue presented in Chapter 2 involves the inequalities inadvertently introduced by NGO development projects. The outcomes of otherwise well-intentioned development projects illustrate that inequalities and conflict can emerge when projects benefit only a portion of the user-group. For example, pasture improvement projects in Collón were instrumental in providing its *comuneros* with additional grazing resources. By having improved and fenced pastures dedicated to grazing in this sector, Collón's households (at least those that paid for access to these pastures) were able to minimize labor conflicts and gain access to much needed grazing resources in the dry season. Yet many more still utilize pastures of the Ishinca valley, together with *comuneros* from Pashpa. In Pashpa I witnessed the negative consequences of not pursuing similar pasture improvement projects there. Frequent complaints, skepticism and feelings of being overlooked undermine a sense of fairness and equity that is critical to cooperation (McKean 2000). Recognition that they were not receiving similar benefits from the conservation community may undermine Pashpa's restraint in using resources for which households of both sectors compete. Unfortunately this applies directly to grazing resources in the conservation core of the HNP. Beyond the uncertainties created by the intervention of government and non-government conservation interests, it remains to be seen whether or not economic and social diversification emerging largely from market involvement, increased temporary migration to Huaraz, and frequent remittances of wage earnings from Huaraz, are problematic for collective action; a focus of future work by the author.

SYNTHESIS AND RECOMMENDATIONS

I have provided various forms of evidence to show that a number of contradictions exist between conservation, development and indigenous livelihoods in the HBR. This finding suggests that a "conservation catch-22," is a possibility in the highlands as well as in the lowlands of South America (Holt 2005). Such a dilemma begs the question, can we find common ground on which conservation interests and indigenous communities in the HBR can build? I believe that we can. In the foreword to "Communities and the Environment," Elinor Ostrom writes that, "the study of conservation policy has been filled with too many wild chases after the chimera of the ideal way to achieve conservation" (Agrawal 2001). Today's conservation policies range from strict protectionism that excludes local people to extractive reserves that permit sustainable use. Debates over the merits of each rage as the current frustrations of failed conservation efforts push the next generation of policy-makers toward the opposite pole (for a review of current perspectives among the largest conservation NGOs see Chapin 2004, Quammen 2006). By illustrating the imperfections of policy in the HBR, I do not intend to give weight to one conservation paradigm over another. I also do not intend to fall into the common criticism of cultural and political ecology research being too contextualized to offer more than idiosyncratic description of the challenges of a particular place and time (Robbins 2004, Walker 2006).

Ostrom continues by saying that "there simply are no ideal conservation policies" (*ibid* 2001). I agree, and argue that any effort spent on attempting to define a single ideal solution detracts from making real progress in today's complex conservation landscapes. The unintended consequences of the creation of this protected area or the policies of "conservation with development" that followed, do not suggest that these approaches should be abandoned in the HBR or elsewhere. What I have suggested is that efforts to protect biodiversity while promoting sustainable development through tourism are neither doomed to failure nor guaranteed success; in much the same way that "tragedies of the commons" are not inevitable. Positive outcomes are predicated on determining on how widely-adopted conservation paradigms can be adapted to a particular context (e.g., place, time, culture).; while negative outcomes are almost certain in the HBR if the actors involved are unwilling to critically examine long-standing orthodoxies surrounding what the Andean landscape is and how it should best be managed (Forsyth 2003). Orthodoxies to challenge in the HBR (as in many other protected areas) include notions that the physical environment is static, that indigenous livelihoods and institutions are ineffective or second-rate to scientific solutions, and that when they are compatible, they will remain that way. Critically assessing this received wisdom is an important initial step in assisting conservation actors, indigenous people and institutions in approaching conservation with an open-mind to the various ways in which it may be achieved.

Lessons of Scale

In reflecting on my findings, I wish to highlight one particular issue whose address by all is likely to be of great help in achieving positive conservation outcomes in the HBR. Each of the actors responsible for determining the outcome of this Andean landscape should pay increased attention to scale, a factor discussed extensively by other political ecology researchers (Zimmerer 2000, Zimmerer and Bassett 2003). Below I discuss the implications of scale with reference to reserve administrators, conservation NGOs and the indigenous community of Tupac Yupanqui.

Lessons for Reserve Administrators

Criticisms of the creation of conservation territories that ignore the ecological realities of local production systems are commonplace (e.g., Turner 2006). Although boundary drawing can have blatantly harmful effects such as forced resettlement, loss of access to critical resources and income streams, and violent conflict (Brockington 2001, Redford 2006), indirect but no less harmful effects can result from inappropriately scaling conservation territories to local livelihood strategies. The boundary of the HNP is only loosely coincident with the historical extent of agricultural production in the study community. Similar mismatches throughout the reserve may largely explain why the boundary is frequently challenged by the region's indigenous agro-pastoral communities today. As discussed above, the placement of the park boundary below the 4000 meter mark in TY may at least be partially responsible for causing the intensification of agricultural production in the buffer zone. In turn, this reduces the availability of grazing resources in the buffer and increases grazing pressure in the park.

The solutions to this dilemma are obvious, but not easily implemented. Assuming that reserve administrators would argue for hard boundaries that delineate the extent of their responsibility and their authority, adjustments may be in order. A critical evaluation of the idea of protecting static conditions or rigid territories is required (Holling 1978, Wiens 1989). I propose replacing the fixed boundary concept with a 'fuzzy' boundary that can account for social and ecological uncertainties. This form of adaptive management may better serve the concerns of reserve administrators and indigenous communities alike. Such a compromise may involve frequent negotiations, with reserve administrators granting permissions on a case-by-case basis to communities that petition to put selected fields within the limits of cultivation (up to roughly 4000 meters) under production. Allowing for this dynamism is admittedly a daunting task. The upside is that it could allow for greater social and economic resilience by protecting household producers (e.g., minimizing risk by field scattering and having sufficient area under cultivation), and allowing for flexibility in response to exogenous factors with already visible impacts on the region, such as climate change.

While this may seem to be a far-fetched solution, one could argue that occasionally permitting agriculture in these currently contested spaces would ultimately do more good than harm. Not only would this scenario help to maintain a mosaic of grazing resources within the community, which would discourage the current trend of increasing grazing pressure in the national park, but it would also reduce the need for the input of chemical fertilizers on remaining agricultural lands, an environmental concern in and of itself. Obvious benefits aside, the communication and coordination required of this sort of co-management would quite possibly foster a much needed realization among these parties that there is some common ground on which conservation and indigenous livelihoods are compatible.

Lessons for Conservation/Development NGOs

A similar criticism could be made that development projects are not scaled to the social realities of *common property* institutions. Thus, there are lessons to be learned for the region's conservation and development actors as well. During fieldwork I observed that the assistance provided by many organizations was often focused on a single community. This scale of development assistance is not inherently problematic, except when the intended goal of this assistance is to encourage conservation of a resource that, in fact, is used and managed by multiple communities. It is often the case that resource use in the HBR's steep inter-Andean valleys involves user-groups from multiple communities. For example, *comuneros* of Collón and Pashpa (different sectors of the indigenous community of Tupac Yupanqui) form the user-group for the Ishinca valley;

whereas *comuneros* of Pashpa and Joncopampa share joint responsibilities of use and management for the neighboring valley of Urus. Such community networks are common in the reserve and are largely ignored by the various development programs focused on assisting discrete villages.

I will use a single instance observed during my fieldwork to highlight the importance of considering 'community' as a scalar concept relative to the management of a given resource. As noted above, pasture improvement projects in Collón, which were intended to address the problem of over-grazing in the Ishinca valley, will only be partially effective in accomplishing their objective. While these projects substantially benefited Collón's residents, these benefits came at the cost of creating tension and resentment among residents of Pashpa, who had not received similar assistance. The fact that pasture improvement in Collón does not alter the use of the Ishinca valley by Pashpa's residents, and that it potentially discourages their cooperation and restraint when they do use it, suggests that inappropriately scaled development assistance may do more harm than good when communally managed resources are at stake.

This dilemma has an apparent solution, but one that is no less complicated than that which is facing reserve administrators. To the extent that development assistance has a conservation objective (e.g., reforestation, pasture improvement, irrigation), these programs should be scaled to try and include all decision-makers and others influencing the decision-making process affecting who has access to the resource and how it is managed. This requires careful study of livelihood strategies and the communal institutions that manage them so that locally appropriate and effective projects can be designed. In the case of the pasture improvement study outlined above, an appropriate scale would have minimally included Pashpa. Of course, this recommendation requires acknowledging that the appropriate scale in the HBR may in fact, be much larger. To illustrate why I make this claim, imagine that pasture improvement projects in Pashpa may alleviate problems among the user-group for the Ishinca valley, but will exacerbate them for the user-group of Urus valley, which includes Pashpa and Joncopampa. The only apparent solution, then, appears to be a holistic approach to conservation and development in the HBR. At the very least, development assistance should proceed only after careful consideration of the potential ripples created throughout *common property* networks with overlapping membership.

Lessons for Local Resource Managers

This discussion would not be complete without attempting to apply the logic of scale to the agro-pastoralists whose decisions are enacted on the landscape. The multi-scaled analysis of biodiversity and grazing in Chapter 4 reveals important lessons for how we evaluate grazing impacts, and how we can best mitigate them. Recall that my analysis showed that historic patterns of use for particular pastures have not had overly negative impacts on biodiversity in the HBR. At both plot and pasture scales, native species richness was significantly higher on pastures used moderately by resident herders. This finding contradicts fairly pervasive assumptions that local resource users are a detriment to biodiversity conservation in protected areas. However, a significant finding that does raise concern is that localized grazing intensities created by livestock concentrations within pastures can have negative effects on species richness at the plot-level. Researchers of state-transition models of rangeland dynamics suggest that these effects may be irreversible (Cingolani, Noy-Meir, and Diaz 2005). This finding implies that indigenous resource managers must address the scale of their impact in order to achieve sustainable management of their herds in the HBR. This means that sustainable herding practices will address not only issues of rotation and timing (e.g., which pastures to use and when), but also how livestock are managed on these pastures (e.g., whether to stake, disperse or set up moveable corrals within pastures). Community-based herding and absentee herding, to the extent that they increase the use of some pastures and encourage concentrations of livestock within them, are potentially unsustainable herding practices that will require the attention of indigenous communities, reserve administrators, government and non-government organizations. In my opinion, solutions for dealing with these issues will only emerge after consulting those actually practicing agro-pastoral livelihoods in the HBR.

CONCLUSION

The research presented is increasingly relevant at a time when people-centered approaches to conservation are being questioned as viable policies for protected areas (discussed by Chapin 2004). It is also relevant given the nearly decade-long call for increased attention to community-based property rights that still has not fully been answered (i.e., Jodha 1992, Lynch and Maggio 2000, Prakash 1998). In the tropical Andes, as a growing conservation focus has juxtaposed many of its rural and indigenous peoples with exogenous actors whose ideas of what the Andes are, or what they should be, increasingly affect local decisions regarding how they are managed. The situation in the HBR, for example, is a prime opportunity to learn valuable lessons for future conservation efforts. My research has shown that existing conservation policies have had many unintended consequences for indigenous peoples and institutions of resource management that could potentially result in negative outcomes.

However, I hope that my work succeeds in doing more than simply critiquing these policies, by showing how such outcomes can be avoided. Insufficient attention to *common property* and how its management is affected by "conservation with development" continues to contribute to failures of both. Developing successful conservation strategies that include people will require that we also include their institutions. This requires that we fully understand what makes highlands unique from other areas we seek to protect. To begin, optimal land use and land tenure strategies in the highlands often involve *common property* institutions. Such institutions are widespread in the highlands and many have functioned well historically (e.g., Gilles and Hahdi 1992, Gilles and Jamtgaard 1981, McKean 1992, Netting 1997, Orlove 1976). At the same time, these arrangements are prone failure when exposed to rapid socio-economic change. This presents an unusual irony for "conservation with development" as it is currently practiced in many of the world's protected landscapes.

In the HBR, the development of adventure tourism presents a challenge of adaptation for indigenous households of Collón and Pashpa, and the communal institution that governs them. Rather than unequivocally assume that tourism is sustainable, or that the inclusion of people in protected landscapes is impossible, we must seek to understand how the policies we promote affect how decisions are enacted upon the landscape. The first step is to accept responsibility for our mutual role in creating unsustainable outcomes. Turning such outcomes around will require a concerted effort on the part of local peoples and institutions, as well as those of government and non-government agencies. Most importantly, a commitment to collaboration and flexibility is a responsibility all concerned about the future of the HBR, and the future of conservation must find in common.

APPENDIX 2.1:

CROPS GROWN IN THE STUDY COMMUNITY

CULTIVAR		TYPE	ZONE
COMMON NAME	**SCIENTIFIC NAME**		
NATIVE:			
ChoCho	*Lupinus mutabilis*	Legume	Upper
Frijoles (Beans)	*Phaseolus vulgaris*	Legume	Lower
Habas (Broad Beans)	*Vicia fava*	Legume	Lower
Lino	*Linum unitatissimum*	Grain	Lower
Maiz (Corn)	*Zea mays*	Grain	Lower
Mashua	*Tropaeolum tuberosum*	Tuber	Upper
Oca	*Oxalis tuberosa*	Tuber	Upper
Papa (Potato)	*Solanum spp.**	Tuber	Upper
Qiwicha	*Amaranthus cau*	Grain	Upper
Quinua	*Chenopodium quinoa*	Grain	Upper
Trigo (Wheat)	*Triticum spp.*	Grain	Lower
Ulluco	*Ullucus tuberosom*	Tuber	Upper
INTRODUCED:			
Averja	*Pisum sativum*	Fodder plant	Lower
Avena (Oats)	*Avena fatua*	Grain	Lower
Cebada (Barley)	*Hordeum spp.*	Grain	Upper
Lentejes (Lentils)	*Lens culinaris*	Legume	Lower

Note: Approximately 16 different cultivars were regularly planted in the study community. The majority were native Andean grains and tubers. The chacra designation corresponds to Figure 2.4. Upper chacras are within the range of approximately 3500–4000 meters; lower chacras are below this elevation.

*18 different potato varieties are regularly cultivated in the community (TMI 2001a).

APPENDIX 2.2:

STOCK EQUIVALENTS

LIVESTOCK TYPE	FORAGE REQUIREMENT (HECTARES/YR)	STANDARDIZATION	STOCK EQUIVALENT
ALPACA	0.37	0.25	1.48
MULE	0.75*	0.25	3.00
COW	1.00	0.25	4.00
GOAT	0.50*	0.25	2.00
HORSE	0.75*	0.25	3.00
LLAMA	0.56	0.25	2.22
SHEEP	0.25	0.25	1.00

Note: stock equivalents are used throughout the dissertation as a means of standardizing the diverse herd compositions of Andean households. Forage requirements are based on the pasture area needed to support one adult animal per year, assuming good quality pasture (Sotomayor 2000).

*Indicates values estimated by the author.

APPENDIX 3.1:

TIME ALLOCATION FORM

Date: *January, 15th, 2002*

Time: *10:33 a.m.*

Observer: *Fariss*

Head of Household: Chinchay

Household #: 294

Neighborhood: Wilcashca

Community: Pashpa

Number of Members in Family: 7

Individual Identification			Level 1	Level 2	Level 3	Level 4	Notes
Relation to Head	Sex	Age					
1	**1**	**33**	*F*	*M*	*B*		*In Huaraz*
2	**2**	**31**	*C*	*F*	*PC*		*Preparing ollunco*
3	**2**	**11**	*F*	*G*	*RB*	*PA*	*Herding in Ishinca*
3	**2**	**9**	*F*	*G*	*RB*	*PA*	*Herding in Ishinca*
3	**1**	**5**	*C*	*I*	*RC*		
4	**2**	**61**	*CO*	*D*			*In community, activity unknown*

Codes for Relation to Head:	Codes for Sex:
Head of household.......................1 Spouse…......................................2 Son/Daughter............................. 3 Mother/Father............................ 4 Brother/Sister...............................5 Son/Daughter-in-law.…………..6 Grandson/granddaughter.……......7 Other Relative.............................8 Adopted Son/Daughter...............9 Non-relative...............................10	Male.……................... 1 Female 2

NOTES:

Activity of individual 1:1:33 reported by spouse

Location of individual 4:2:61 reported by spouse; activity could not be confirmed

Note: The form above is completed with hypothetical observations for a family of 7. Information in bold was pre-printed to facilitate rapid collection of observations for each family member. Information in italics was recorded by the observer using the 4-level coding scheme in Appendix 3.2.

APPENDIX 3.2:

TIME ALLOCATION CODES

LEVEL 1: LOCATION

AB = All members absent
CA = Individual at house and can be observed
CO = Individual in community, observed or reported
D E = Whereabouts of the individual unknown

LEVEL 2	LEVEL 3	LEVEL 4
I = Individual	BE = Drinking CB = Riding horse/mule CD = Running DE = Resting DU = Sleeping HG = Hygiene RC = Playing alone TA = Doing Homework OT = Other	Si = Sick
C = Care-giving	AL = Feeding or serving children/others AM = Breastfeeding CA = Caring for sick children/others LA = Washing children/others LL= Carrying children MN = Watching children OT = Other	
F = Domestic	AD = Making adobe bricks CR = Sewing or repairing clothes CU = Caring for household animals (e.g., chicken) ED = Repairing house/roof HR = Making/repairing tools HI = Spinning wool JA = Gardening LI = Cleaning house LP = Washing dishes LV = Washing clothes MR = Making merchandise PC = Preparing food RA = Collecting water TE = Weaving textiles/baskets WO = Cutting wood OT = Other	

S = Social	CE = In conversation with others FI = Attending a party IN = Activity influenced by observer JF = Playing organized sport JG = Playing with others OR = Attending meeting of outside group PM = Playing instrument in band RE = Attending a religious activity/lecture RU = Attending a meeting (community, other) VI = Visiting another household Sa = Visiting hospital/clinic OT = Other	
G = Herding	CN = Butchering/Processing Meat RB = Herd management	AM = Staking CH = Checking animals CL = Sheering wool LE = Milking PA = Herding VE = Inspecting/Vaccinating OT = Other
A = Farming	AB = Hand-fertilizing plants (maize) CS = Harvesting agricultural products MA = Maintenance, weeding, irrigating, etc. PO = Processing agricultural products PR = Preparing fields, planting, tilling OT = Other	YU = animal labor
H = Foraging	HA = Hunting PE = Fishing ME = Collecting medicinal plants RC = Collecting forage for household animals TM = Cutting timber OT = Other	CN = Construction CM = Combustion ME = For sale in market

T = Off-Farm	AS = Wage laor	EM = Permanent employee EN = Musician JP = Day laborer MI = Miner TU = Tourism-related FC = Government program OT = Other
	BU = Looking for wage labor	EM = Permanent employee EN = Musician JP = Day laborer MI = Miner TU = Tourism-related FC = Government program OT = Other
	EO = Performing duties of elected official	
	LC = Performing communal labor	A = Agriculture G = Herd management OT = Other
	LE = Labor for community group or business	A = Agriculture G = Herd management OT = Other
	LR = Labor for church MI = Minka/reciprocal labor exchange GE = Vocational training OT = Other	
M = Market	A = Buy, sale agricultural products B = Buy, sale household staples G = Buy, sale livestock VE = Buy, sale medicines/Vaccination for animals OT = Other	
E = Education		
OT = Other		

APPENDIX 3.3:

STATISTICAL JUSTIFICATION FOR A MODEL OF HOUSEHOLD HERDING LABOR WITH RANDOM EFFECTS

METHOD	LOG LIK	$-2\log\Lambda$ *	P-VAL**
SLR1: standard logistic regression model	-2253.53	X	X
LRRE1: logistic regression w/ 1 random effect (*hhnum*)	-2024.96	457.14	<0.0001
LRRE2 logistic regression w/ 2 random effects (*hhnum* and *dry season*)	-2019.40	11.11	0.0023

Note: the statistics above provide justification for the inclusion of random effects using the variance components model (VCM) of Chapter 3 as a baseline.

*The likelihood ratio statistic, -2log Λ, can be used to test for the significance of the predictors in the full model against the reduced model. Where L denotes the likelihood function:

$$-2\log\Lambda = -2\log\left[\frac{L(\text{reduced})}{L(\text{full})}\right] = -2[\log L(\text{reduced}) - \log L(\text{full})]\cdot$$

$-2\log\Lambda$ has an asymptotic chi-square distribution with degrees of freedom given by,
df (full model–df (reduced model).

**For this specific test of the need for random effects, the appropriate distribution of the likelihood ratio statistic is a mixture distribution of χ_0^2 and χ_1^2 (Verbeke and Molenberghs 2000), such that the p-value of the test is given by:

$$p = \frac{1}{2}P(\chi_0^2 > \Lambda) + \frac{1}{2}P(\chi_1^2 > \Lambda)$$

APPENDIX 3.4:

ADDITIONAL CONTROLS CONSIDERED FOR THE MULTILEVEL MODEL OF HOUSEHOLD HERDING LABOR

VARIABLE	DEFINTION	PREDICTED EFFECT
Household Demographic & Economic Variables:		
hh size	Number of people in household	+
labor age	Number of working age individuals in household	+
dep age	Number of dependents in household	-
dep ratio	Ratio of dependents to labor aged individuals in household	-
hh female	Total number of females in household	+
hh eduatt	Number of individuals in household reported to attend school at time of census	-
hh headedu	Level of education completed by household head	-
Herd Size & Composition Variables:		
total	Total number of animals in household herd	+
stock equivalents	Total stock equivalents in household herd	+
largeprop	Proportion of herd in large stock (cattle, horse and mule)	-
smallprop	Proportion of herd in small stock (sheep, goat, llama and alpaca)	+
Climate Variables:		
precip avg	Mean daily precipitation recorded at climate stations 1 and 2	Modifier
season 3dry	Categorical dummy for dry season including the months of June-August	Modifier
Time Variables:		
weekday	Day of week	X
time cat	Time of day of observation; am or pm	X
Other Variables:		
ecoag mem	Member of EcoAgro	+
barrio	Barrio (neighborhood) designation	X
sector	Sector designation/community	X

Note: Obvious control variables such as *hh size* and *labor age* did not produce significant effects or alter the coefficients or standard errors of the tourism variable in appreciable ways. I chose to exclude these control variables by comparing models with and without them using AIC (Burnham and Anderson 2002). Since this method of model selection penalizes for additional parameters, the models that included such controls were not ranked with the highest Akaike weight (i.e. a statement of the probability of being selected under repeated sampling).

APPENDIX 3.5:

ESTIMATES OBTAINED FOR THE MULTILEVEL MODEL OF HOUSEHOLD HERDING LABOR

A. The Frequentist version calculated in R:

LEVEL1	LEVEL2	PARAMETER	ESTIMATE	SE	P >|Z|
initial status β_{0i}	*intercept*	β_0	-0.3555	0.3023	0.2396
	head age	β_2	0.0558	0.0094	<0.0001
	head age^2	β_3	-0.0026	0.0004	<0.0001
	farming labor	β_4	2.0279	0.8537	0.0175
	moderate tourism involvement	β_5	-0.8572	0.3092	0.0056
	high tourism involvement	β_6	-0.1604	0.2968	0.5888
dry season β_{1i}	*intercept*	β_1	-0.6412	0.2532	0.0113
	farming labor	β_7	1.7943	0.7569	0.0178
	moderate tourism involvement	β_8	-0.5489	0.3258	0.0920
	high tourism involvement	β_9	-0.0399	0.2594	0.8777

VARIANCE COMPONENT	PARAMETER	ESTIMATE	SE
hhnum	β_0	X	0.3580

B. The Bayesian version calculated in WinBugs:

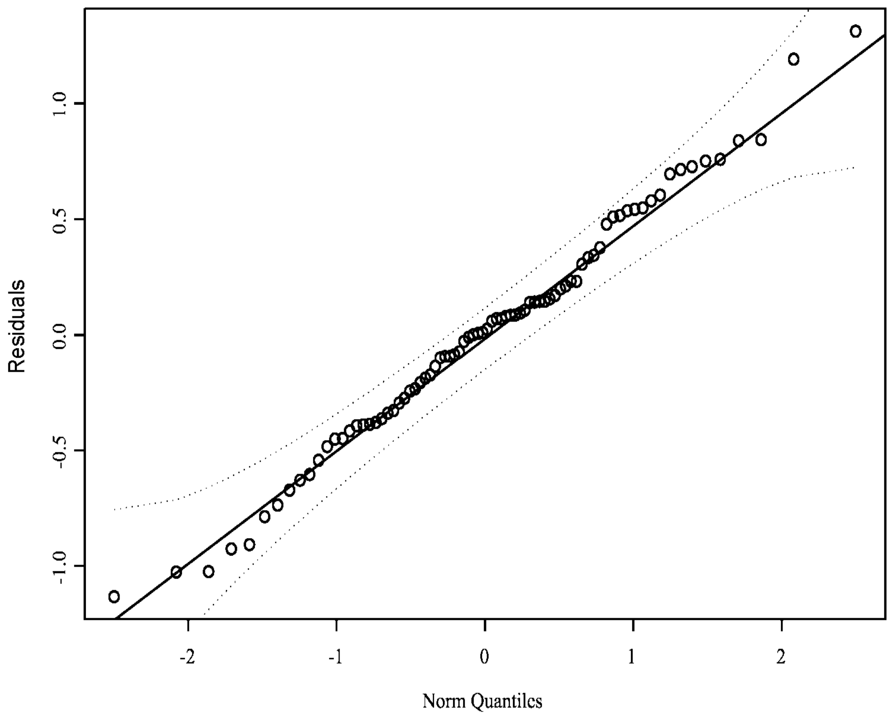
LEVEL1	LEVEL2	PARAMETER	ESTIMATE	STDEV	95% CI
initial status β_{0i}	*intercept*	β_0	-0.3696	0.3244	$\pm$ 1.2806
	head age	β_2	0.0574	0.0099	$\pm$ 0.0395
	head age^2	β_3	-0.0026	0.0005	$\pm$ 0.0019
	farming labor	β_4	2.0610	0.9131	$\pm$ 3.6220
	moderate tourism involvement	β_5	-0.8450	0.3286	$\pm$ 1.3114
	high tourism involvement	β_6	-0.1552	0.3186	$\pm$ 1.2613
dry season β_{1i}	*intercept*	β_1	-0.5684	0.2297	$\pm$ 0.9075
	farming labor	β_7	1.5850	0.6705	$\pm$ 2.7906
	moderate tourism involvement	β_8	-0.5594	0.3220	$\pm$ 1.3146
	high tourism involvement	β_9	-0.0648	0.2671	$\pm$ 1.0251

VARIANCE COMPONENT	PARAMETER	ESTIMATE	STDEV
hhnum	β_0	0.6566	0.0807

APPENDIX3.6:

AN EXPLORATION OF NORMALITY IN THE MULTILEVEL MODEL OF HOUSEHOLD HERDING LABOR

A. Level-2 Residuals:



Note: Dotted lines indicate 95% confidence bands. Since household residuals fall within these bands, there is no evidence that the normality assumption for the random effects was violated.

APPENDIX 3.7:

REGRESSORS OF HERD WEALTH AND THEIR EFFECTS ON HOUSEHOLD HERDING LABOR

REGRESSOR	DEFINITION	WET SEASON EFFECT	DRY SEASON EFFECT
herd wealth ratio	Continuous variable defined as the ratio of household to community herd size.	+	+
stock equivalent wealth ratio	Continuous variable defined as the ratio of household to community stock equivalents.	+	+
large stock wealth ratio	Continuous variable for herd wealthy households defined as the ratio of household to community large stock (cattle, horse and mule) holdings.	+	-
herd wealthy	Categorical dummy variable for herd wealthy households, defined as those that collectively own more than half of all livestock pastured on the commons.	+	-
stock equivalent wealthy	Categorical dummy variable for stock equivalent wealthy households, defined as those that collectively own more than half of all the stock equivalents pastured on the commons.	+	- $p > \|z\| = 0.0488$
large stock wealthy	Categorical dummy variable for large stock wealthy households. Defines wealthy households as those that collectively own more than half of all the large stock pastured on the commons.	+	- $p > \|z\| = 0.0809$

Note: Model runs were based on the CONTROL model plus the regressor and its interaction with *dry season*. Note that only two herd wealth regressors produced p-values indicative of statistical significance. In both cases, the sign of the effect was opposite to that predicted by the herd wealth hypothesis.

APPENDIX 3.8:

MODIFICATIONS TO THE GAME THEORETIC MODEL OF HERD WEALTH

The following discussion is an attempt to explore additional reasons why my results do not support game theoretic predictions of herding behavior relative to herd wealth. In the discussion below, I propose a different solution to the inequalities presented by Ruttan and Borgerhoff-Mulder (1999). My solution suggests that overall herd size, in addition to relative herd wealth is an important factor affecting household decision-makers. I begin by generating the indifference surfaces of Figures A, B and C from the equations presented in their article:

$$\alpha f D \frac{f}{f+m} > C \qquad \text{[eqn. 9]}$$

$$\alpha m D \frac{m}{f+m} > C \qquad \text{[eqn. 10]}$$

The authors solve the corresponding indifference equations incorrectly as follows.

Herder 1 cooperates when:

$$C = \alpha f D \frac{f}{f+m} = \alpha f D \frac{f}{m\left(\frac{f}{m}+1\right)} = \alpha f D \frac{\frac{f}{m}}{\left(\frac{f}{m}+1\right)} \overset{\text{invalid step}}{\neq} \alpha \frac{f}{m} D \frac{\frac{f}{m}}{\left(\frac{f}{m}+1\right)} = \alpha D \frac{\left(\frac{f}{m}\right)^2}{\left(\frac{f}{m}+1\right)}$$

Herder 2 cooperates when:

$$C = \alpha m D \frac{m}{f+m} = \alpha m D \frac{m}{f\left(\frac{m}{f}+1\right)} = \alpha m D \frac{\frac{m}{f}}{\left(\frac{m}{f}+1\right)} \overset{\text{invalid step}}{\neq} \alpha \frac{m}{f} D \frac{\frac{m}{f}}{\left(\frac{m}{f}+1\right)} = \alpha D \frac{\left(\frac{m}{f}\right)^2}{\left(\frac{m}{f}+1\right)}$$

A closer look at their solution reveals that there is an error which significantly affects the ESS regions they plot. A ratio was created in the multiplier of step 4 (see above) where none should exist. The correct solution to this inequality yields:

Herder 1 cooperates when:

$$C = \alpha f D \frac{f}{f+m} = \alpha D\left(\frac{f^2}{f+m}\right)$$

Herder 2 cooperates when:

$$C = \alpha m D \frac{m}{f+m} = \alpha D\left(\frac{m^2}{f+m}\right)$$

The result is a three, rather than two-dimensional solution as originally illustrated. This solution results in ESS regions for a two-person game, where cooperation on the commons is a function of herd size and asymmetries in herd wealth: I plot the correct 3-dimensional ESS regions in Figures A through G. Parameters were adjusted for the specifics of my case study, where α = forage intake (1), d = number of days that dry season reserves in the *puna* are available (123), f = number of animals owned by herder 1, m = number of animals owned by herder 2, and the total cost of travel to the *puna* = (α *D). These figures show how herd size, not just the herd wealth ratio, affects ESS regions.

Figure A:

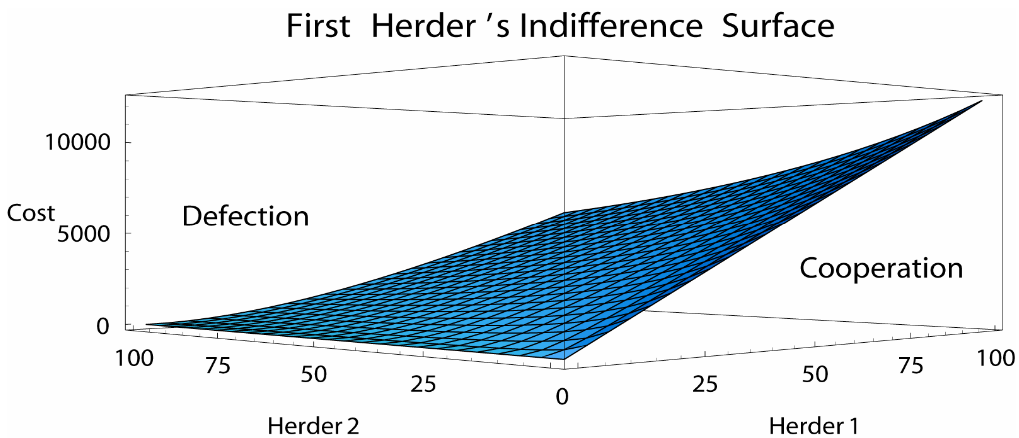


Figure B:

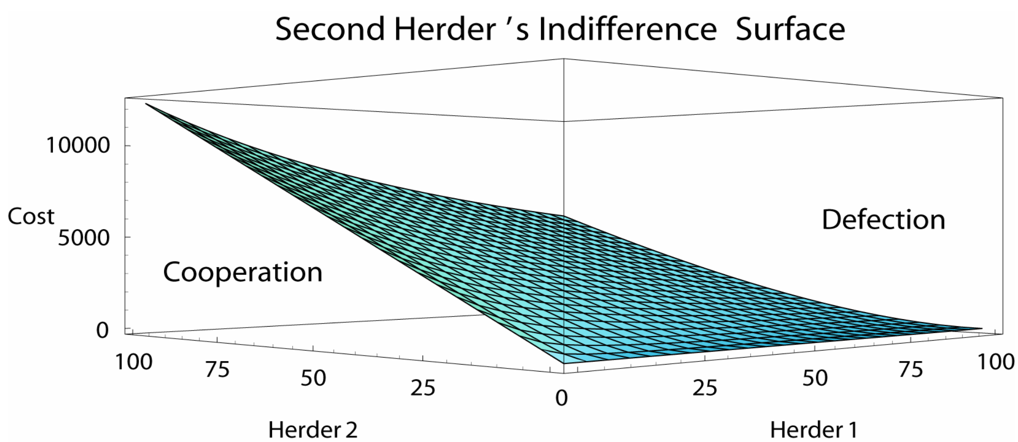
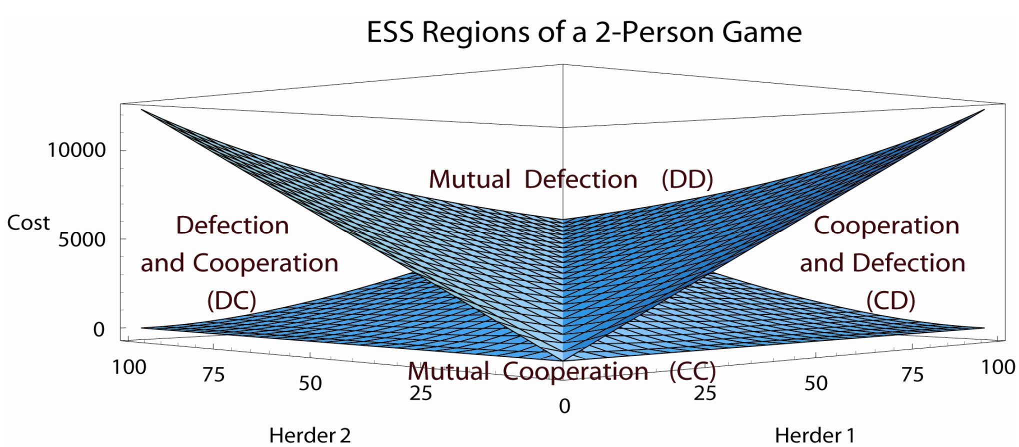


Figure C:



Additional information is provided by the figures above, which show that predicted behaviors for each herder are a function of their herd wealth relative to other herders, but also their absolute herd size. The authors plot indifference curves at a single intersection plane along these 3-dimensional surfaces. They derive their 2-dimensional plot by fixing herd size for one of the herders and allowing the other to vary to create the herd wealth ratio. My figures illustrate why this discrepancy matters. Figure D shows that different intersection planes can produce different ESS regions. Figure E plots the traces of indifference surfaces for the planes of intersection shown in the previous figure. What it illustrates is that the ESS regions reported by the authors shift depending on the herd size chosen for herder 2. Considering the ESS region created by a herd size of 50, we see that the threshold for mutual defection (i.e., the area above both red lines) is significantly lower than when its herd size is 100 (i.e. the area above both blue lines). This pattern is consistent across the entire range of possible herd wealth ratios. In other words, a herd wealth ratio of 0.5 can be created by a comparison of various herd pairings, for example, 50:100 and 25:50. The figures illustrate that smaller herd pairings result in stronger predictions of defection by wealthy and poor herders alike.

Figure D: Labels correspond to Figure C.

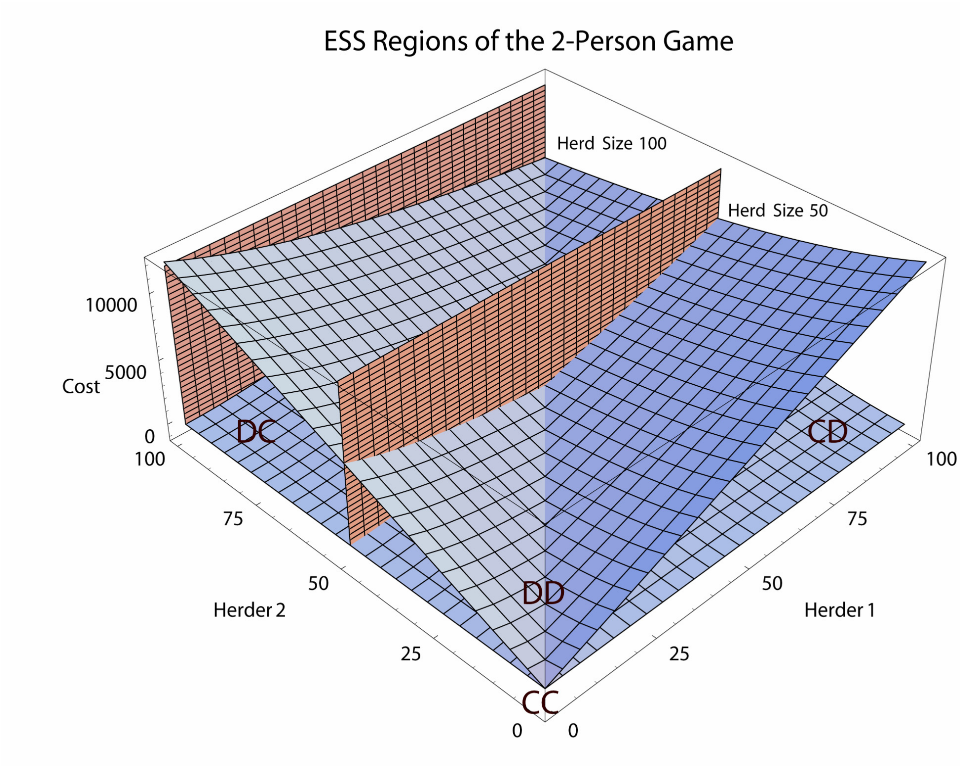
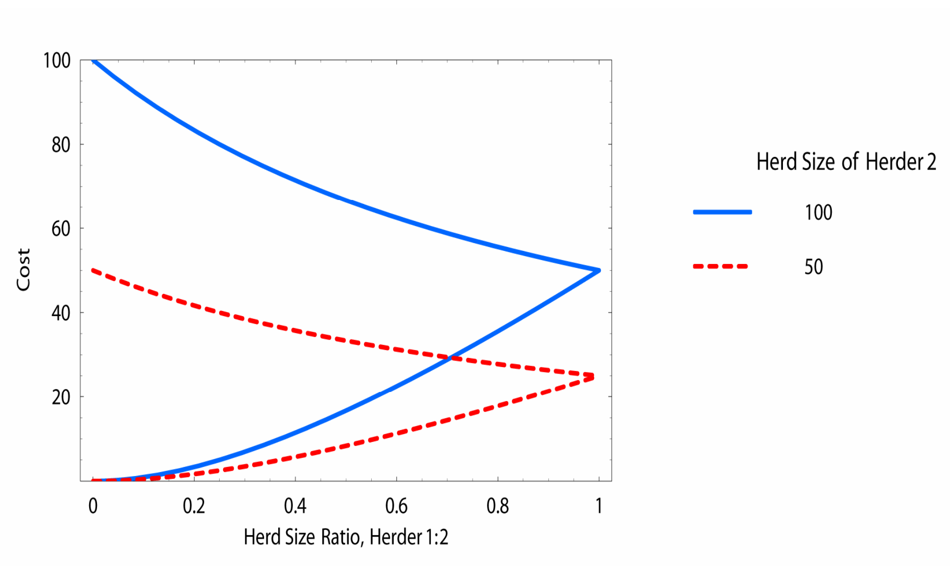


Figure E:



An evaluation of ESS regions created when varying the herd wealth ratio shows the general patterns discussed by the authors, but illustrates again, that the predictions change with regard to absolute herd size. Figure F below intersects herder indifference surfaces at various combinations of the same herd wealth ratio. For example, the intersection plane at a herd wealth ratio of 0.5 crosses a number of herd size pairings on the 3-dimensional graph. Note that although the herd wealth ratio is the same, ESS regions are not uniform as suggested by the authors. Figure G plots the traces of indifference surfaces created by the intersection planes of herd wealth ratios at 0.1 and 0.5. This plot shows that the likelihood of defection by both wealthy and poor (i.e., the area above both lines of the same type) decreases with increased herd size regardless of the herd wealth ratio. Note that at the highest levels of inequality (i.e., a herd wealth ratio = 0.1), mutual cooperation is virtually nonexistent regardless of the herd size. The region of unilateral cooperation by the wealthy (i.e., the area between the two solid lines) grows with increasing herd size, but shrinks as the wealth gap closes. To see this effect compare the area of unilateral cooperation by the wealthy when herd wealth ratios are pronounced (i.e., 0.1) to the area of unilateral cooperation by the wealthy when herd wealth ratios are more subtle (i.e., 0.5). Notice that the latter has a smaller area (see the area between both dotted lines.

Figure F: Labels correspond to Figure C.

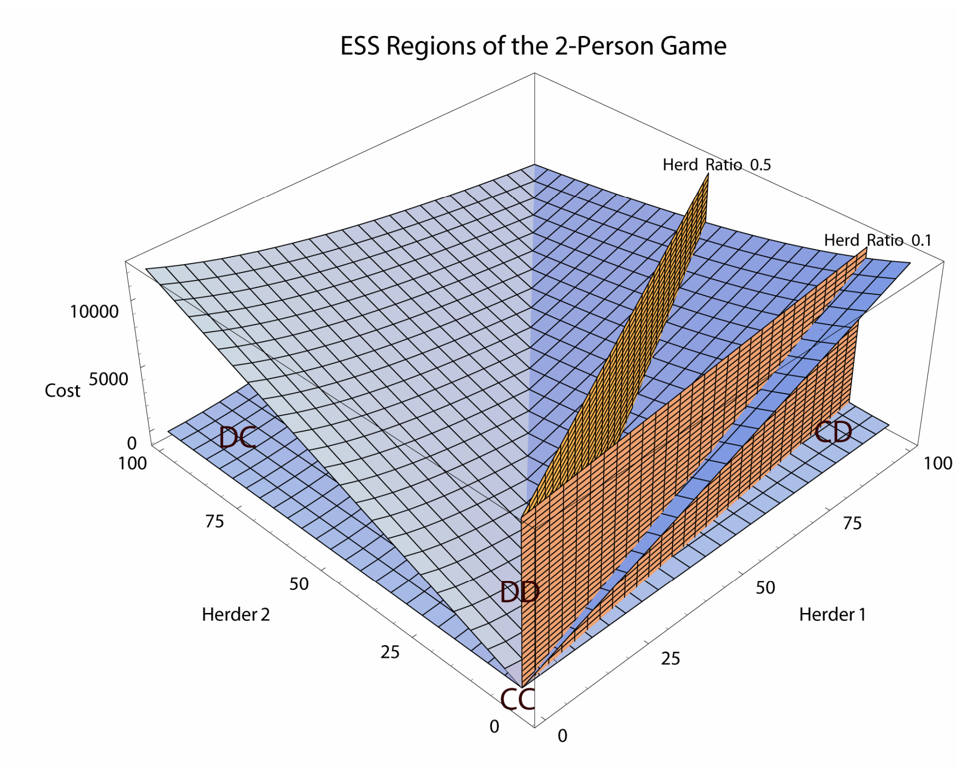
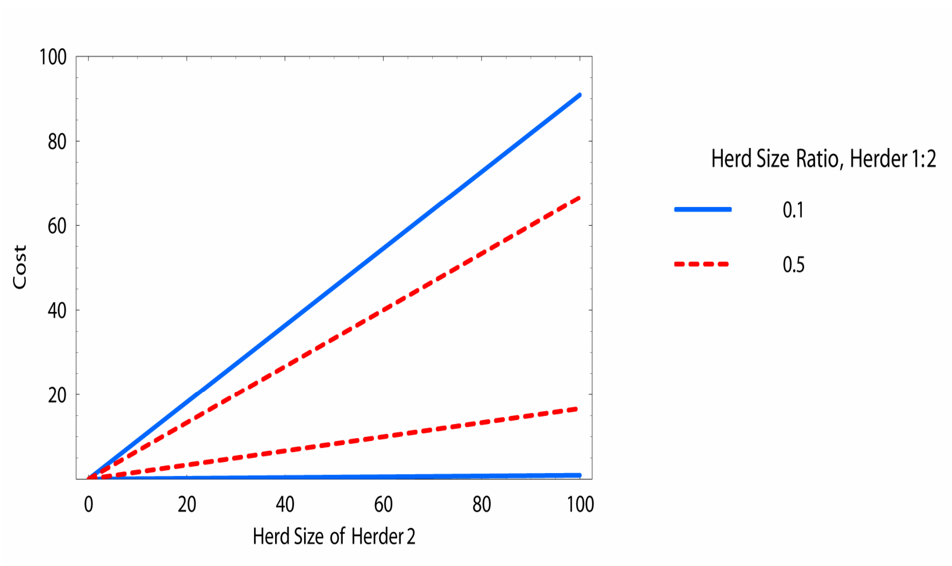


Figure G:



Collectively, what these observations contribute to the game theoretic model is that the effects of the herd wealth ratio on the herder's decision to cooperate or defect are mediated by herd size. Plotting the ESS regions in three dimensions yields the same regions of mutual cooperation, mutual defection and unilateral cooperation they predict, with the following important qualifications:

(1) Mutual defection occurs generally when costs are high and when inequalities are low as predicted. But, this scenario is most likely when everyone has only a few animals. In other words, **when herd sizes are small and inequality is subtle** nobody stands to gain much by cooperating, and nobody has much of an incentive to encourage it. The region where this occurs is above both surfaces, as shown in Figure F.

(2) Mutual cooperation occurs primarily when costs are low. This is more likely with increasing herd sizes so that everyone has enough animals to make the cost of cooperation worthwhile. In other words **when inequality is moderate, and herd sizes are large** everyone has something to gain by cooperating. The region where this occurs is difficult to see in Figure F, but its area is greatest in the back corner of the plot and under both surfaces.

(3) Lastly, unilateral cooperation by the wealthy occurs generally when costs are moderate to high and inequalities are moderate to high, as predicted. Given the increased cost of herding in the dry season, this is the scenario I test. However, as I show above, this unique and theoretically interesting result is **strongly dependent on herd size**. The region of unilateral cooperation by the wealthy is indicated by the placement of the label in Figure F.

In sum, I derive the same predictions as Ruttan and Borgerhoff-Mulder (1999) regarding the effects relative herd wealth on sustainable use of the commons. What is added by plotting their original function in three dimensions is the consideration of the variability of these predictions relative to absolute herd size. As herd size decreases, the condition that conservation is “isomorphic with economic efficiency” for the relatively wealthy herder is rarer than originally predicted. This begs the question as to whether the range of herd size in the community (min = 0, max =190) is large enough to allow for this condition.

APPENDIX 4.1:

UNSAMPLED PASTURES OF THE TMI PASTURE IMPROVEMENT PROJECT

A. Characteristics

PASTURE (# ON MAP)	LOCATION	BIOCLIMATIC ZONE	AREA (HECTARES)
Pucucu Pachan (11)	Collón	1	< 1
Tuspin Pampa (13)	Collón	1	2

Note: Pastures of the pasture improvement project of The Mountain Institute, not selected as part of the random sample discussed in Chapter 4. Summaries are presented here for the purpose of providing this organization with information on the condition of pastures following their efforts to improve them.

B. Summary of Conditions

PASTURE NAME (# ON MAP)	SECTOR	PLOT-LEVEL NATIVE SPECIES RICHNESS MEAN (STDEV)	PASTURE-LEVEL NATIVE SPECIES RICHNESS	PASTURE-LEVEL GRAZING INTENSITY	INVASIVE SPECIES		INDICATOR SPECIES	
					AP	PC	AG	OF
Pucucu Pachan (11)	Collón	7.2 (1.9)	39	3		++		
Tuspin Pampa (13)	Collón	3.9 (1.6)	37	1		++		

Note: This table focuses on native species richness and the presence of invasives and indicators of over-grazing. Species codes are as follows: PC = *Pennisetum clandestinium*, AP = *Aciachne pulvinata*, AG = *Astragalus garbancillo*, and OF = *Opuntia flocossa*. An (+) in the species column indicates that it was recorded on the pasture, while two (++) indicate that the species was present in the 1-m^2 where proportion cover was also recorded.

APPENDIX 4.2:

STATISTICAL JUSTIFICATION FOR A MODEL OF NATIVE SPECIES RICHNESS WITH RANDOM EFFECTS

METHOD	LOG LIK	$-2\log\Lambda$ *	P-VAL**
SLR1: standard logistic regression model	-302.95	X	X
LRRE1: logistic regression w/ 1 random effect (***sitenum***)	-281.79	42.32	<0.0001

Note: The statistics above provide justification for the inclusion of random effects using the random intercepts model (RIM) of Chapter 4 as a baseline.

*The likelihood ratio statistic, -2log Λ, can be used to test for the significance of the predictors in the full model against the reduced model. Where L denotes the likelihood function:

$$-2\log\Lambda = -2\log\left[\frac{L(\text{reduced})}{L(\text{full})}\right] = -2[\log L(\text{reduced}) - \log L(\text{full})]\cdot$$

$-2\log\Lambda$ has an asymptotic chi-square distribution with degrees of freedom given by,

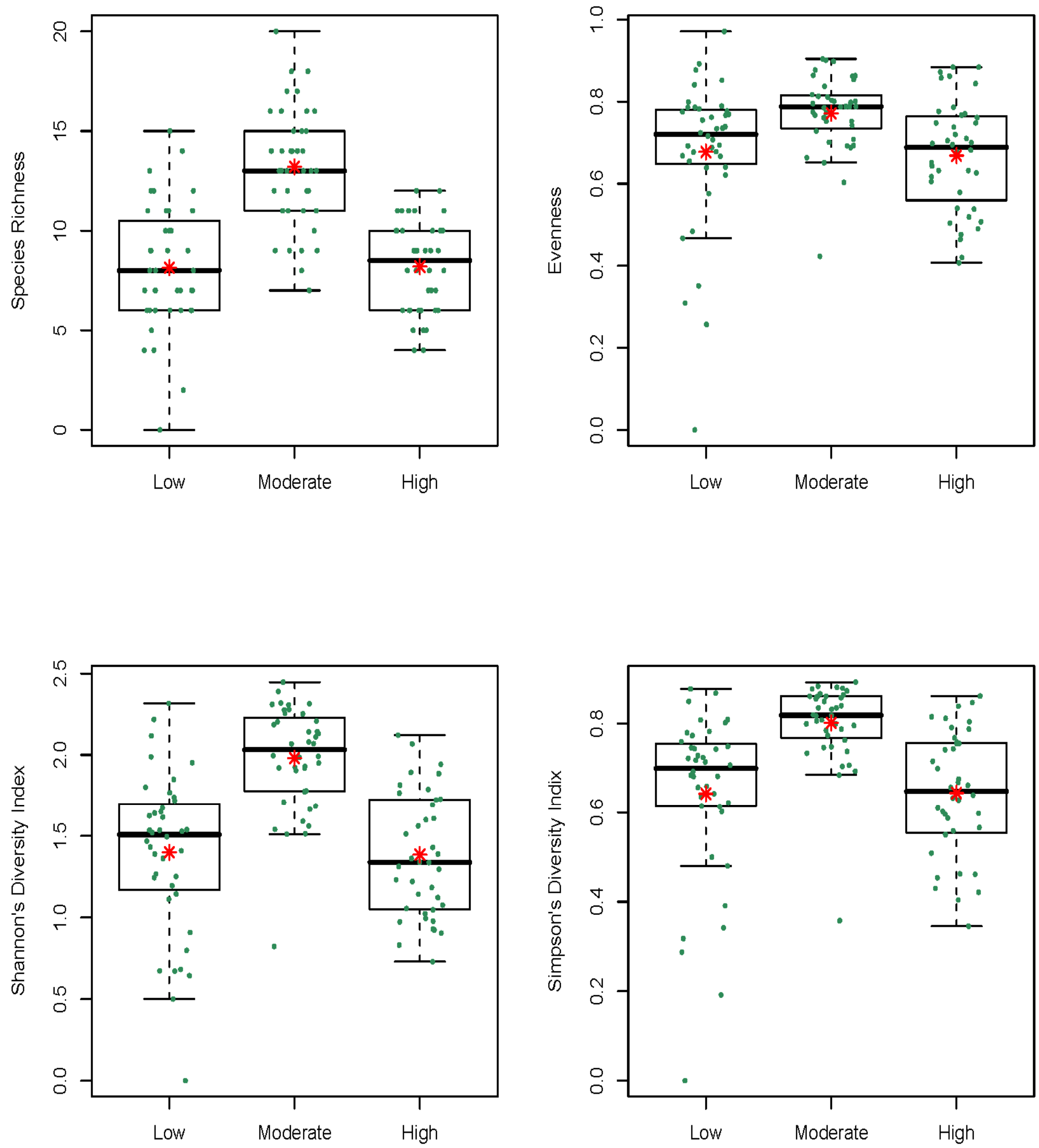
df (full model–df (reduced model).

**For this specific test of the need for random effects, the appropriate distribution of the likelihood ratio statistic is a mixture distribution of χ_0^2 and χ_1^2 (Verbeke and Molenberghs 2000), such that the p-value of the test is given by:

$$p = \frac{1}{2}P(\chi_0^2 > \Lambda) + \frac{1}{2}P(\chi_1^2 > \Lambda)$$

APPENDIX 4.3:

BOXPLOTS OF VARIOUS DIVERSITY MEASURES SUMMARIZED BY PASTURE-LEVEL GRAZING INTENSITY



Note: Each was calculated in the software package, PC-ORD (McCune and Mefford 1999) and plotted in R. Notice that these other measures of diversity show similar patterns to the response of native species richness chosen as the model response in Chapter 4.

APPENDIX 4.4:

PLANT SPECIES ENCOUNTERED DURING VEGETATION SAMPLING

FAMILY	NAME	AUTHORITY
Amaryllidaceae	*Bomarea dulcis*	(Hook.) Beauverd
	Bomarea sp.	Mirb.
Apiaceae	*Azorella sp.*	Lam.
	Hydrocotyle ranunculoides	L.f.
	Oreomyrrhis andicola	(Kunth) Hook.f.
Asteraceae	*Achyrocline alata*	(Kunth) DC.
	Ageratina azangaroensis	(Schultz-Bip. ex Wedd.) R. King & H. Robinson
	Alchemilla diplophylla	Diels
	Antennaria linearifolia	Wedd.
	Belloa piptolepis	(Wedd.) Cabrera
	Baccharis adnata	Humb. & Bonpl. ex Willd.
	Baccharis bogotensis	Kunth
	Baccharis caespitosa	(Ruiz & Pav.) Pers.
	Baccharis genistelloides	(Lam.) Pers.
	Baccharis sp.	L.
	Baccharis tricuneata	(L.f.) Pers.
	Baccharis uniflora	(Ruiz & Pav.) Pers.
	Bidens andicola	Kunth
	Bidens pilosa	L.
	Bidens sp.	L.
	Chersodoma ovopedata	(Cuatrec.) Cabrera
	Chuquiraga spinosa	Less. ssp. huamanpinta C. Ezcurra
	Cotula australis	(Sieb. ex Spreng.) Hook. f.
	Cotula Mexicana	(DC.) Cabrera
	Diplostephium foliosissimum	S.F. Blake
	Gamochaeta sp.	Wedd.

	Gamochaeta spicata	(Lam.) Cabrera
	Gynoxys sp.	Cass.
	Hypochaeris andina	(DC.) Reiche
	Hypochaeris sp.	L.
	Hypochaeris taraxacoides	(Walp.) Benth. & Hook. f.
	Inulese sp.	
	Loricaria ferruginea	(Ruiz López & Pavón) Wedd.
	Loricaria graveolens	(Sch.Bip.) Wedd.
	Lucilia tunariensis	(Kuntze) K. Schum.
	Oritrophium limnophilum	(Sch.Bip.) Cuatrec
	Paranephelius bullatus	A. Gray ex Wedd.
	Paranephelius sp.	Poepp.
	Perezia multiflora	(Humb. & Bonpl.) Less.
	Senecio canescens	(Kunth) Cuatrec.
	Senecio comosus	Schultz-Bip.
	Senecio culcitioides	Schultz-Bip.
	Senecio sp.	L.
	Senecio trephrosioides	Turez
	Siegesbeckia jorullensis	Kunth
	Sonchus oleraceus	L.
	Stevia sp.	Cav.
	Unknown sp1.	
	Unknown sp2.	
	Tegetes multiflora	L.
	Werneria caespitosa	Wedd.
	Werneria dactylophylla	Schultz-Bip.
	Werneria nubigena	Kunth
	Werneria villosa	A. Gray
Berberidaceae	*Berberis lutea*	Ruiz López & Pavón
Betulaceae	*Alnus jorullensis*	Kunth
Brassicaceae	*Lepidium sp.*	L.
Bromeliaceae	*Puya angusta*	Lyman B. Smith

	Puya cerrateana	L. B. Sm.
Cactaceae	*Matucana yanganucensis*	Rauh & Backeb.
	Opuntia floccose	Salm-Dyck
Caryophyllaceae	*Paronychia andina*	A. Gray subsp. boliviana Chaudhri
	Pycnophyllum sp.	Remy
Clusiaceae	*Hypericum laricifolium*	Juss.
Crassulaceae	*Crassula connata*	(Ruiz & Pav.) A. Berger
Cyperaceae	*Carex ancashensis*	D. N. Smith & Reznicek, ined.
	Carex boliviensis	Van Juerck & Műll.-Arg.
	Carex ecuadorica	Kűk.
	Carex sp.	L.
	Cyperus niger	Ruiz & Pav.
	Cyperus sp.	L.
	Eleocharis albibracteata	Nees & Meyen ex Kunth
	Scirpus rigidus	Boeck.
Ephedraceae	*Ephedra americana*	Humb. & Bonpl. ex Willd.
Equistaceae	*Equisetum bogotense*	Kunth
Ericaceae	*Pernettya prostrata*	(Cav.) DC.
Fabaceae	*Astragalus garbancillo*	Cav.
	Astragalus uniflorus	DC.
	Cassia hookeriana	Gillies ex Hook. & Arn.
	Lupinus mutabilis	Sweet
	Lupinus sp.	L.
	Medicago hispida	Gaertn.
	Trifolium amabile	Kunth
	Trifolium repens	L.
Gentianaceae	*Gentiana prostrata*	Haenke
	Gentiana sedifolia	Kunth
	Gentiana sp.	L.
	Gentianella nitida	Griseb.
	Gentianella sp.	Moench
	Gentianella thyrsoidea	(Hook.) Fabris

	Gentianella weberbauerii	(Gilg) Fabris
Geraniaceae	*Erodium cicutarium*	(L.) L'Her.
	Erodium sp.	Al'Her. ex Aiton
	Geranium sessiliflorum	Cav.
	Geranium sp.	L.
Iridaceae	*Orthrosanthus chimboracensis*	(Kunth) Baker var.
	Sisyrinchium junceum	E. Meyer ex Presl.
	Sisyrinchium sp.	L.
Juncaceae	*Distichia muscoides*	Nees & Meyen
	Juncus arcticus L. var. andicola	(Hook.) Balslev
	Juncus brunneus	Buchenau
	Juncus ebracteatus	E. Meyer
	Juncus sp.	L.
	Luzula peruviana	Desv.
	Luzula sp.	DC.
	Oreobolus obtusangulus	Gaudich.
Lamiaceae	*Lepechinia meyenii*	(Walp.) Epling
	Minthostachys sp.	(Benth.) Spach
	Salvia sp.	L.
Loganiaceae	*Buddleja incana*	Ruiz & Pav.
Lycopodiaceae	*Huperzia sp.*	
Malvaceae	*Acaulimalva engleriana*	(Ulbr.) Krapov.
	Nototriche acaulis	(Cav.) Krapov.
	Nototriche pinnata	(Cav.) Hill
	Nototriche sp.	Turez
Myrtaceae	*Eucalyptus globula*	Labill.
Orchidaceae	*A paleacea*	(Kunth) Reichb. F.
Oxalidaceae	*Hypsocharis pimpinellifolius*	Remy
	Oxalis sp.	L.
Pinaceae	*Pinus radiate*	D. Don
Plantaginaceae	*Plantago australis*	Lam. Subsp. hirtella (Kunth) Rahn
	Plantago extensa	Pilg.

	Plantago lamprophylla	Pilger
	Plantago rigida	Kunth
	Plantago sp.	L.
Plumbaginacea	*Plumbago sp.*	L.
Poaceae	*Aciachne pulvinata*	Benth
	Aegopogon cenchroides	Humb. & Bonpl. ex Willd.
	Agrostis breviculmis	Hitchc.
	Agrostis glomerata	(J. Presl) Kunth
	Agrostis haenkeana	Hitchc.
	Agrostis tolucencis	Kunth
	Andropogon bothriochloa	L.
	Aristida enodis	Hackel
	Bromus catharticus	Vahl
	Bromus lanatus	Kunth
	Calamagrostis curvula	(Wedd.) Pilger
	Calamagrostis heterophylla	(Wedd.) Hitchc.
	Calamagrostis ovata	(Presl) Steud.
	Calamagrostis recta	(Kunth) Trin.
	Calamagrostis rigescens	(C. Presl) Scribn.
	Calamagrostis rigida	(Kunth) Trin.
	Calamagrostis sp	Adams
	Calamagrostis vicunarum	Wedd.
	Cortaderia nitida	(Kunth) Pilger
	Dissanthelium macusaniense	(E. H. L. Krause) R. C. Foster & L. B. Smth
	Dissanthelium sp.	Trin.
	Festuca dolichophylla	Presl
	Festuca huamachucensis	Infanta, vel sp. aff.
	Festuca peruviana	Infantes
	Festuca rigescens	(J. Presl.) Kunth
	Festuca sp.	L.
	Hierochloa redolens	(Sol ex Vahl) Roem and Schult.

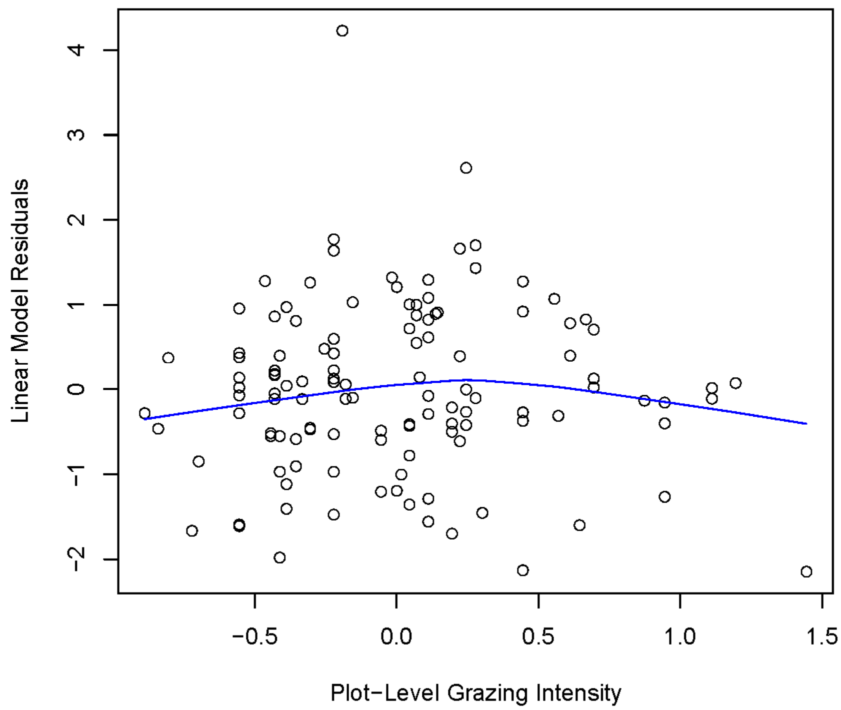
	Hordeum muticum	J. Presl.
	Lycurus phalaroides	Kunth
	Lycurus sp.	Kunth
	Muhlenbergia fastigiata	(Presl) Henrard
	Muhlenbergia ligularis	(Hackel) Hitchc.
	Muhlenbergia peruviana	(Beauv.) Steud.
	Paspalum pilgerianum	Chase
	Paspalum pygmaeum	Hackel
	Paspalum sp.	L.
	Paspalum tuberosum	Mez
	Pennisetum clandestinum	Hoshst. ex Chiov.
	Piptochaetium featherstonei	(Hitchc.) Tovar
	Piptochaetium indutum	Parodi
	Poa aequigluma	Tovar
	Poa annua	L.
	Poa candamoana	Pilger
	Poa gilgiana	Pilger
	Poa sp.	L.
	Polypogon elongatus	Kunth
	Polypogon interruptus	Kunth
	Setaria sp.	P. Beauv.
	Sporobolus indicus	(L.) R. Br.
	Stipa brachyphylla	Hitchc.
	Stipa depauperata	Pilg.
	Stipa hans-meyeri	Pilger
	Stipa ichu	(Ruiz López & Pavón) Kunth
	Stipa mucronata	Kunth
	Stipa sp.	L.
	Vulpia australis	(Nees ex Steud.) C. H. Blom
	Vulpia megalura	(Nutt.) Rydb.
Polygonaceae	*Muehlenbeckia volcanica*	(Benth.) Engl.
	Polygonum hydropiperoides	Michx.

	Rumex acetosella	L.
	Rumex crispus	L.
	Rumex sp.	L.
Portulacaceae	*Calandrinia sp.*	Kunth
Primulaceae	*Anagallis sp.*	L.
Ranunculaceae	*Krapfia weberbaueri*	(Ulbr.) Standley & J. F. Macbr.
	Ranunculus flagelliformis	Smith
	Ranunculus praemorsus	Kunth ex DC.
Rosaceae	*Alchemilla orbiculata*	Ruiz López & Pavón
	Alchemilla pinnata	Ruiz & Pav.
	Polylepis sp.	Ruiz & Pav.
	Polylepis weberbauerii	Pilger
Rubiaceae	*Arcytophyllum thymifolium*	(Ruiz López & Pavón) Standley
	Galium sp.	L.
Santalaceae	*Quinchamalium procumbens*	Ruiz and Pav.
Scrophulariaceae	*Alonsoa lineosis*	(Jacq.) Ruiz López & Pavón
	Bartsia canescens	Wedd.
	Bartsia diffusa	Benth.
	Bartsia sp.	L.
	Calceolaria incarum	Kränzlin
	Castilleja sp.	Mutis ex L.f.
Solanaceae	*Salpichroa hirsuta*	(Meyen) Miers
	Unknown sp1.	
Valerianaceae	*Stangea erikae*	Graebn.
	Valeriana globularis	A. Gray
	Valeriana rigida	Ruiz and Pav.
	Valeriana sp.	L.
Verbenaceae	*Verbena littoralis*	Kunth

Note: Authorities are included where possible as provided by Smith (1988). The TROPICOS database of the Missouri Botanical Gardens was utilized where authorities were not provided in the previous document (Luteyn et al. 1999).

APPENDIX 4.5:

RESIDUALS OF A MULTILEVEL MODEL OF NATIVE SPECIES RICHNESS ASSUMING A LINEAR VERSION OF THE PLOT-LEVEL GRAZING VARIABLE



APPENDIX 4.6:

ADDITIONAL CONTROLS CONSIDERED FOR THE MULTILEVEL MODEL OF NATIVE SPECIES RICHNESS

VARIABLE	DEFINTION	PREDICTED EFFECT
Plot-Level:		
biomass	Continuous measure of above ground net primary productivity (grams m^2) calculated from a digital photographic technique (Paruelo, Lauenroth, and Roset 2000).	+
soil texture	Ordinal ranking of soil texture in each plot, from fine (1) to coarse (5).	-
Pasture-Level:		
bofedale	Categorical dummy variable indicating a critical dry season reserve.	+
chacra	Categorical dummy variable indicating occassional cultivation by the community. No pasture sampled had been cultivated 5 years previous to vegetation sampling.	-
elevation	Continuous measure of meters above sea level, as derived from a 30m resolution digital elevation model of the site.	-
past improve	Categorical dummy variable indicating a history of pasture improvement efforts focused on seeding pasture with good forage species.	+
slope	The rate of change in elevation on the pasture; calculated as the average rise/run from a 30m DEM.	-
rsp	The average % distance from the bottom of a slope (0.0%) to the top (100.0%); a surrogate for the general thermal and hydrologic characteristics of a particular site.	-
trmi	A continuos measure based on the topogrpahic relative moisture index; a proxy for water retention potential as influenced by topographic factors like slope, curvature, and upslope contributing area (Parker 1982).	+
yprr	A continuous measure serving as a surrogate for the radiation and evaporative demand of each site; calculation based on average monthly iterations of the following equation in ArcGIS 9.1: $HS = 255[\cos(90-Z)\sin(s)\cos(\alpha - A) + \sin(90-Z)\cos(s)]$ Where the sun's altitude (Z) and azimuth (A) are taken from a solar calendar for the Julian days of every month of a particular year (see (Urban and Lookingbill 2005).	+

Note: The above variables were additional control variables considered for the modeling effort. Incomplete coverage or lack of variance in these measures precluded their use in the models presented in Chapter 4.

APPENDIX 4.7:

SPECIES OF RELATIVE ABUNDANCE ON SAMPLED PASTURES

SPECIES	% OF PLOTS WITH OCCURRENCE (n = 120)	TOTAL AREA (m^2)
Agrostis breviculmis	17.1	1.0
Carex boliviensis	6.4	1.1
Paspalum tuberosum	5.7	1.1
Carex sp.	20.0	1.2
Dissanthelium macusaniense	13.6	1.2
Quinchamalium procumbens	16.4	1.2
Muehlenbeckia volcanica	8.6	1.3
Oreobolus obtusangulus	7.9	1.3
Calamagrostis vicunarum	22.9	1.4
Rumex acetosella	21.4	1.7
Polypogon elongates	17.1	1.8
Festuca dolichophylla	22.1	1.8
Calamagrostis regescens	29.3	1.9
Inulese sp.	12.9	1.9
Contula australis	10.0	2.0
Muhlenbergia fastigiata	11.4	2.1
Muhlenbergia peruviana	15.7	2.1
Werneria caespitosa	14.3	2.4
Nototriche pinnata	17.9	2.5
Alchemilla pinnata	31.4	2.5
Stipa mucronata	13.6	2.8

Distichia muscoides	13.6	2.8
Werneria nubigena	27.9	3.4
Scirpus rigidus	34.3	4.7
Cyperus niger	17.1	6.3
Muhlenbergia ligularis	40.7	7.9
Juncus brunneus	22.1	8.5
Pennisetum clandestinum	27.9	10.0
Plantago rigida	28.6	10.4

Note: the list above is restricted to species with a total percent cover in the sample of at least $1m^2$.

APPENDIX 4.8:

ESTIMATES OBTAINED FOR THE MULTILEVEL MODEL OF NATIVE SPECIES RICHNESS

A. The Frequentist version calculated in R:

LEVEL	VARIABLE	PARAMETER	ESTIMATE	SE	P >|t|
Plot β_{0i}	*intercept*	β_0	8.4396	0.7347	0.0000
	cover	β_1	6.0526	1.2652	0.0000
	dominant cover	β_2	-7.6028	1.6260	0.0000
	plot-level grazing intensity	β_3	-1.2669	0.5787	0.0308
	plot-level grazing intensity^2	β_4	-2.1361	0.7756	0.0069
Pasture β_0	*aspect*	β_5	0.7098	0.3742	0.0996
	wetness	β_6	0.4345	0.1483	0.0220
	moderate pasture-level grazing intensity	β_7	4.7862	0.7599	0.0004
	high pasture-level grazing intensity	β_8	-0.9067	0.6911	0.2309

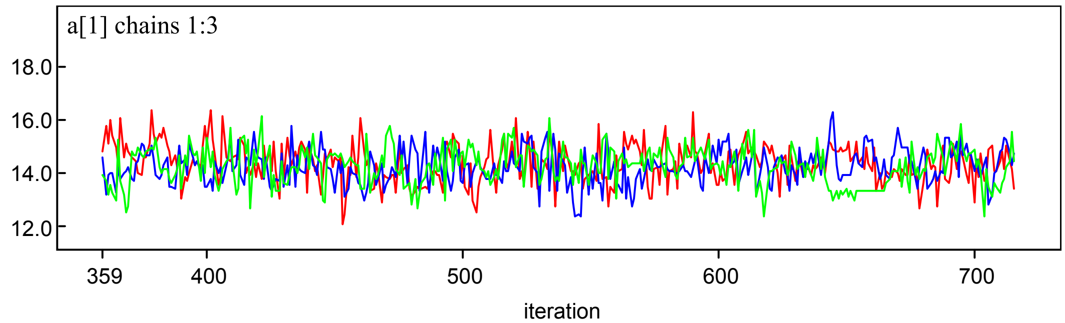
VAR COMPONENT	PARAMETER	ESTIMATE	SE
sitenum	β_0	X	0.0001

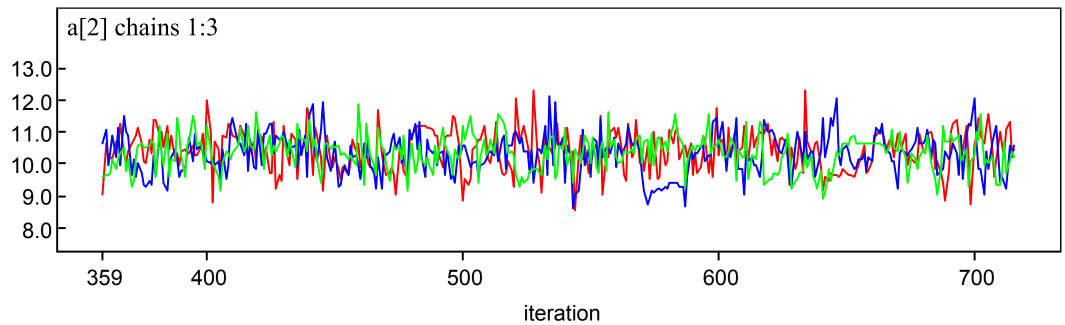
B. The Bayesian version calculated in WinBugs:

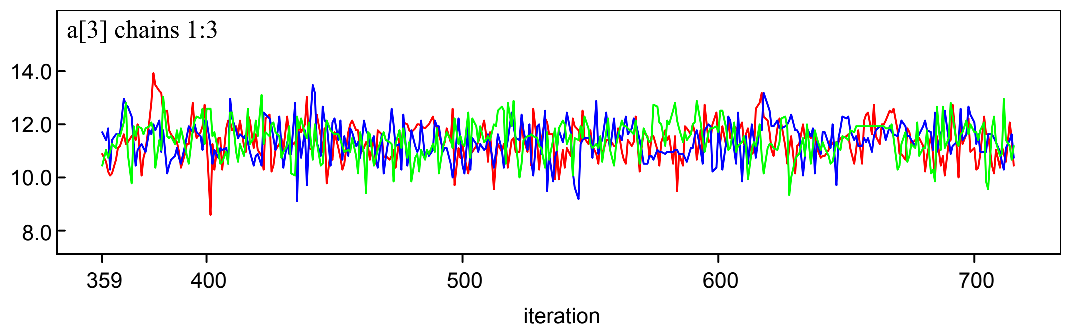
LEVEL	VARIABLE	PARAMETER	ESTIMATE	STDEV	95% CI
Plot β_{0i}	*intercept*	β_0	8.5	0.9	± 2.5
	cover	β_1	6.4	1.3	± 3.5
	dominant cover	β_2	-7.8	1.7	± 4.3
	plot-level grazing intensity	β_3	-1.1	0.7	± 1.9
	plot-level grazing intensity^2	β_4	-2.2	0.8	± 1.9
Pasture β_0	*aspect*	β_5	0.7	0.5	± 1.4
	wetness	β_6	0.4	0.2	± 0.5
	moderate pasture-level grazing intensity	β_7	4.9	1.0	± 2.8
	high pasture-level grazing intensity	β_8	-0.8	0.9	± 2.5

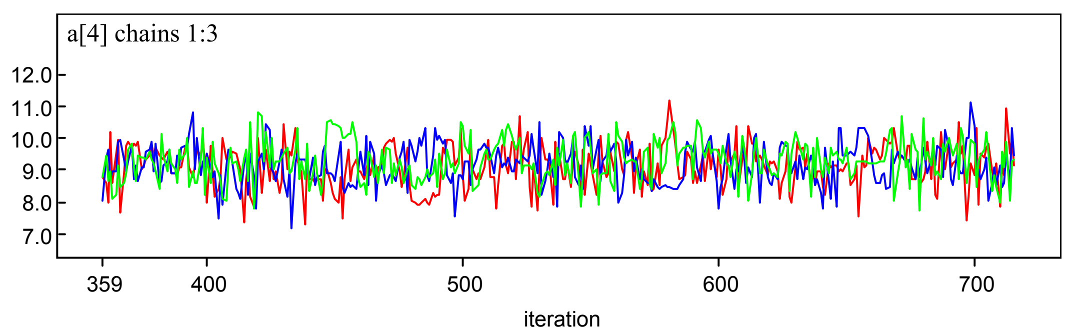
VAR COMPONENT	PARAMETER	ESTIMATE	STDEV
sitenum	β_0	0.5	0.4

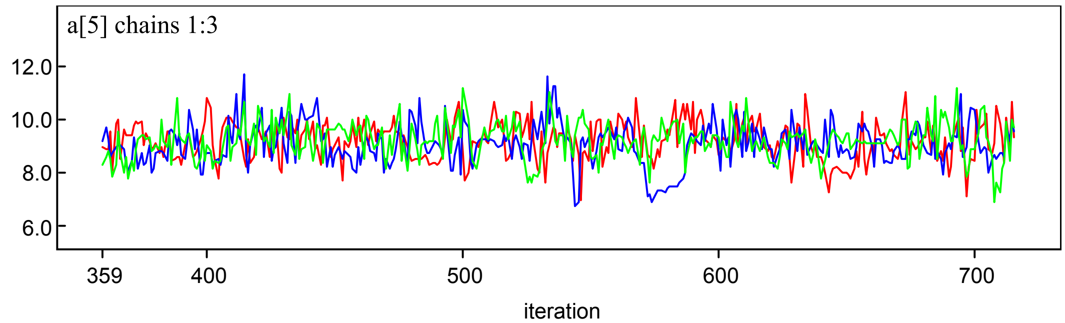
C. Simulation runs calculated in WinBugs. Rhat = 1 for all parameters, indicating model convergence. The following graphs show simulation results for model paramters, each of the 12 pastures, followed by all other predictors of the multilevel model of native species richness:







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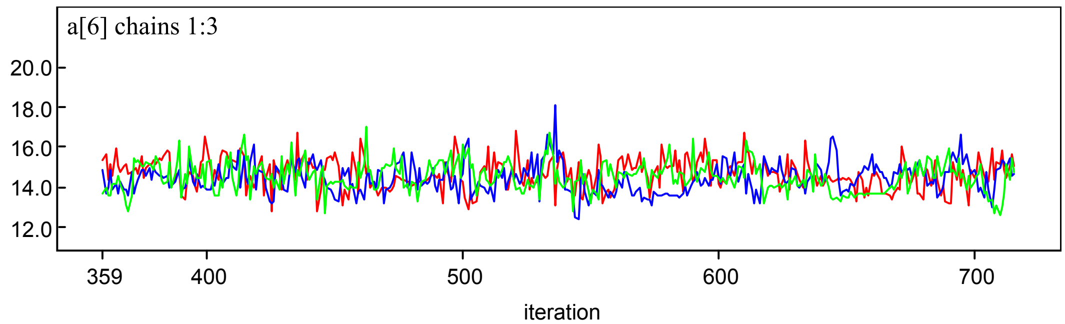
10.0
8.0
6.0
359
400
500
600
700
iteration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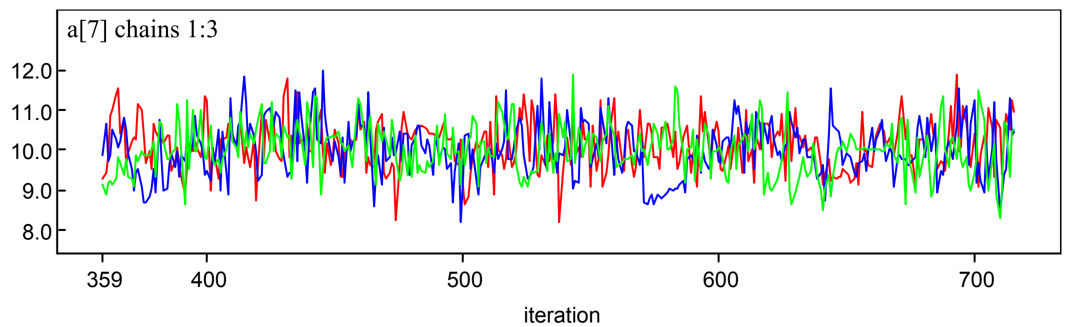
400
500
600
700
iteration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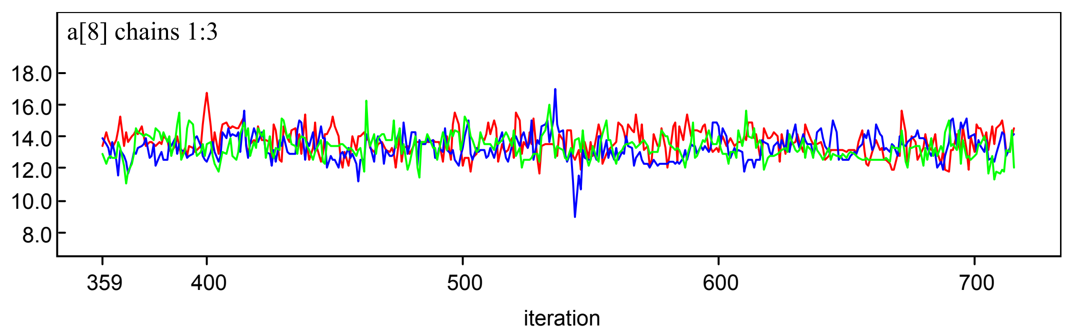
iteration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10.0
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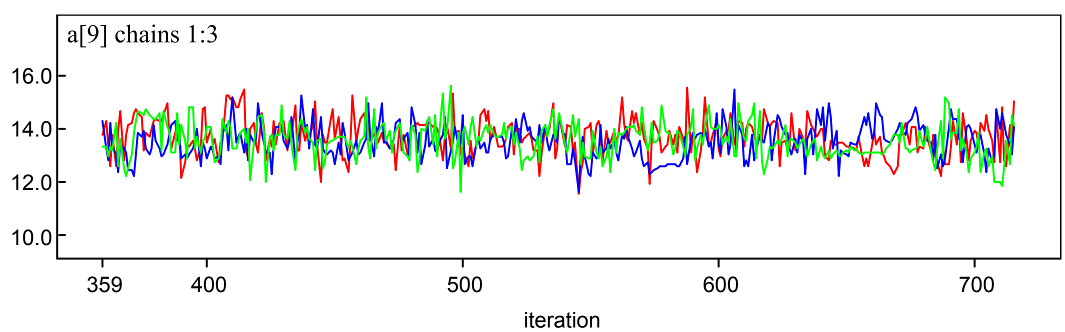
400
500
600
700
iteration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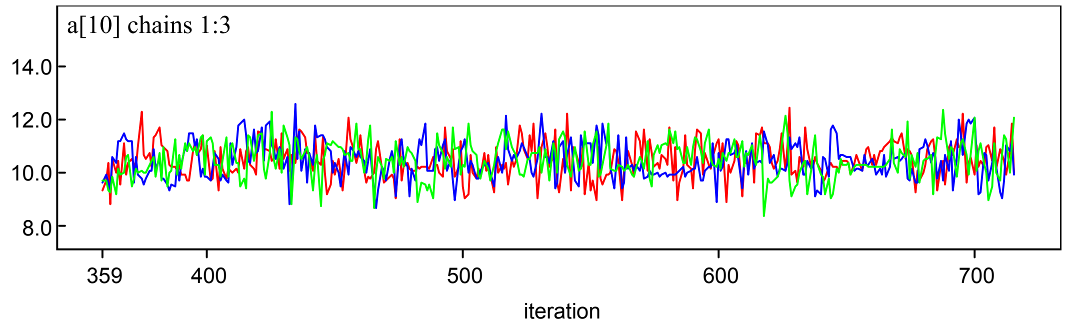
400
500
600
700
iteration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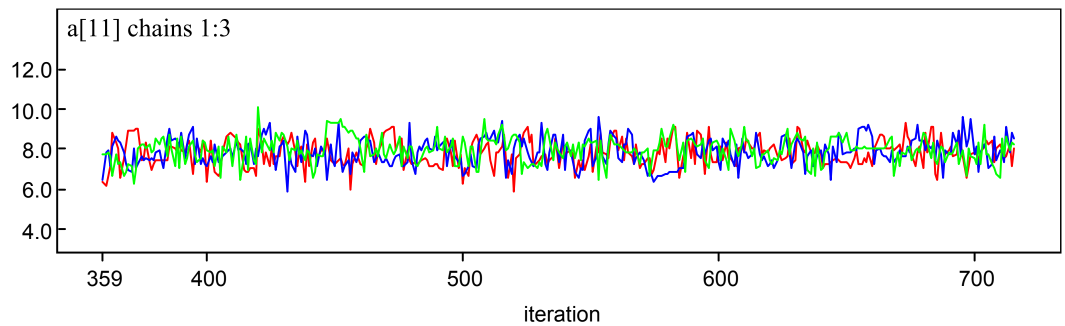
400
500
600
700
iteration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4.0
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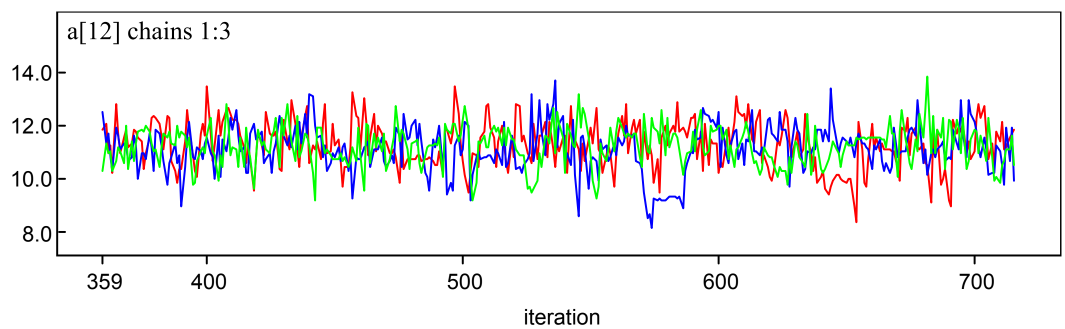
400
500
600
700
iteration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10.0
8.0
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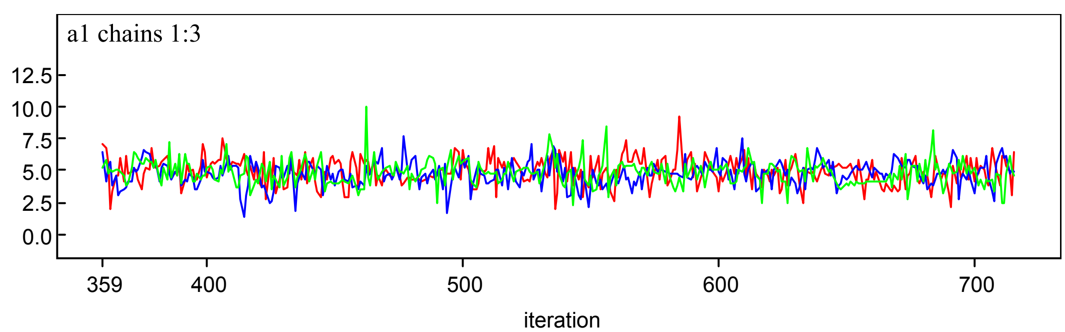
400
500
600
700
iteration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7.5
5.0
2.5
0.0
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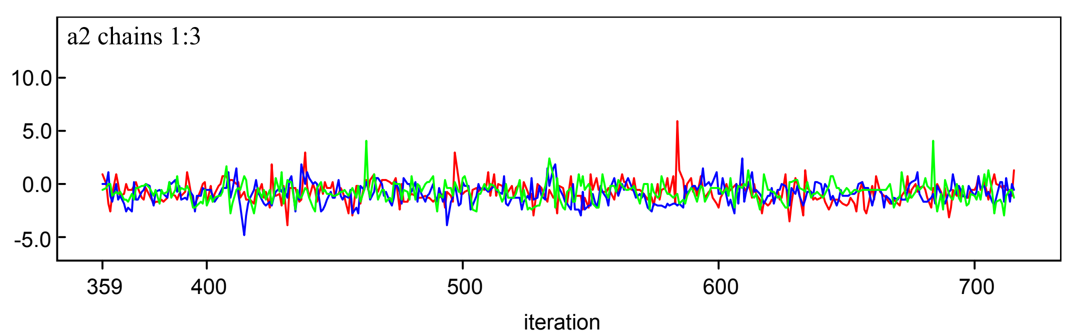
400
500
600
700
iteration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-5.0
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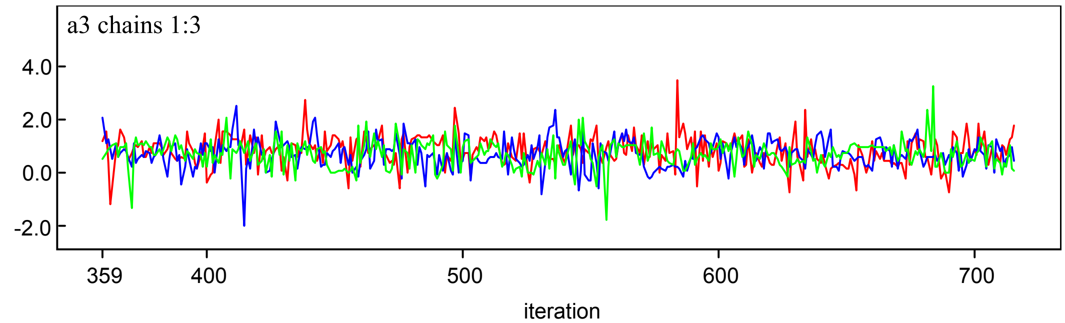
400
500
600
700
iteration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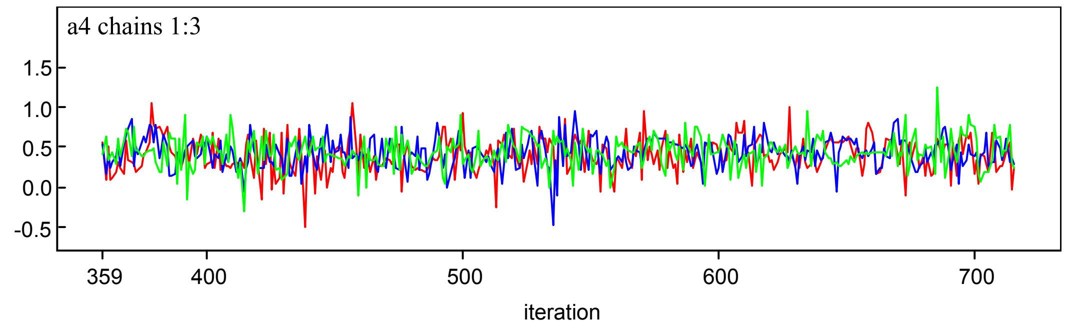
2.0
0.0
-2.0
359
400
500
600
700
iteration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1.5
1.0
0.5
0.0
-0.5
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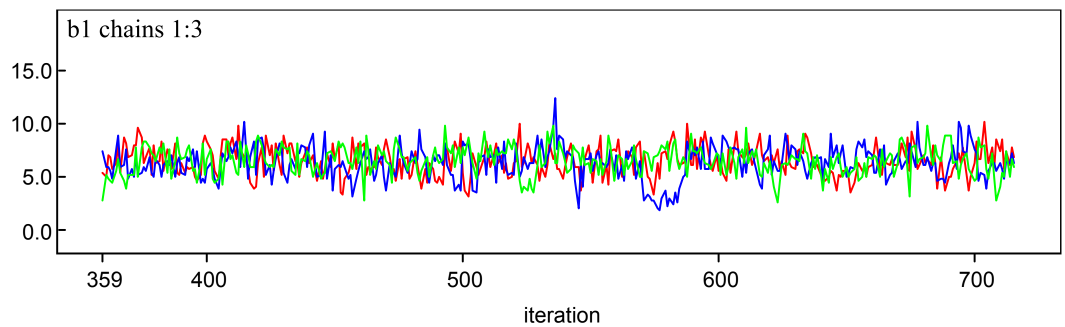
400
500
600
700
iteration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10.0
5.0
0.0
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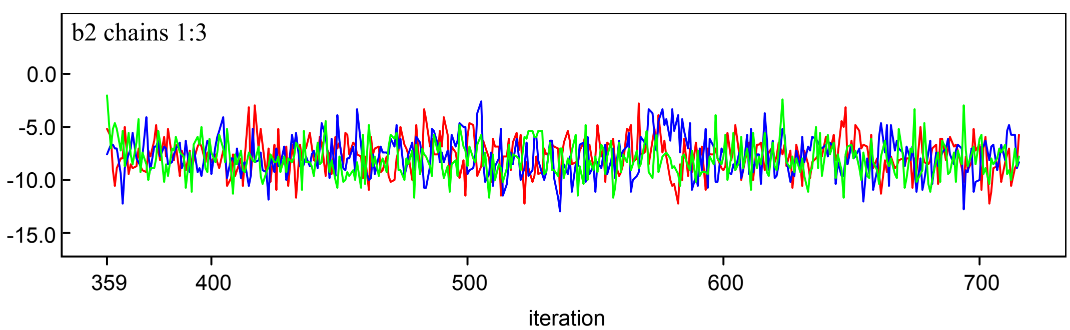
400
500
600
700
iteration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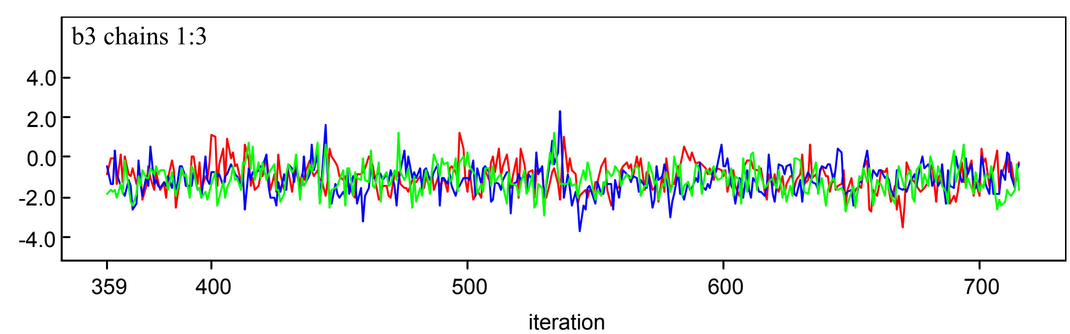
500
600
700
iteration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-4.0
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400
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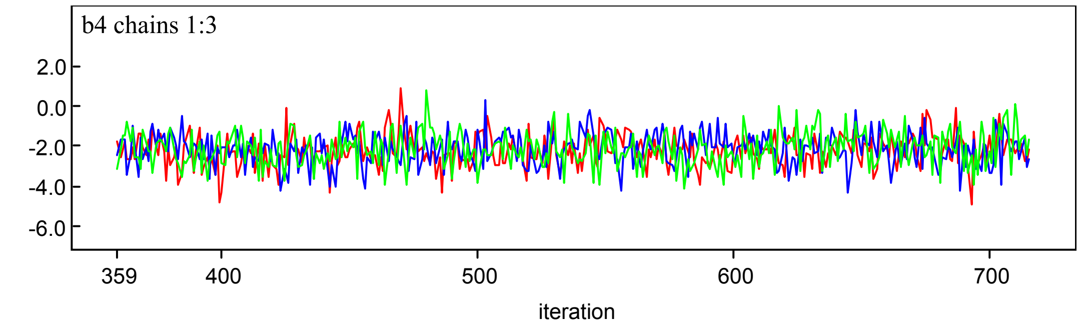
600
700
iteration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-4.0
-6.0
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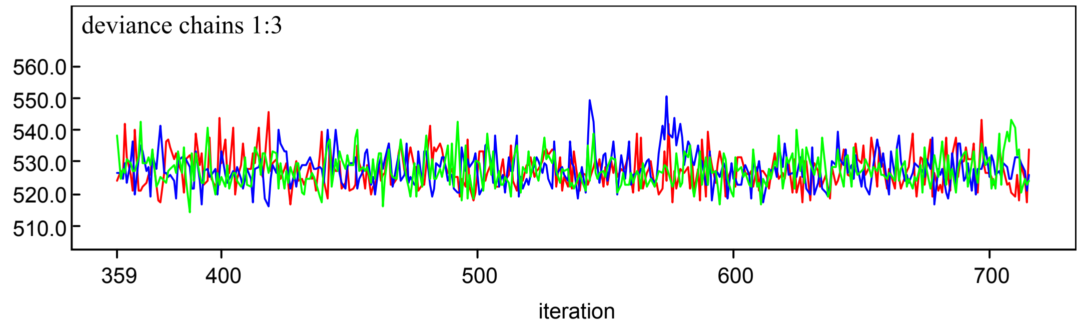
400
500
600
700
iteration


deviance chains 1:3
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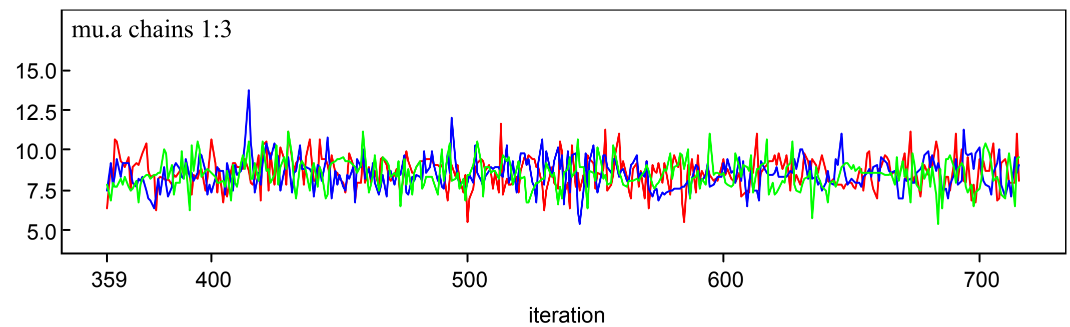
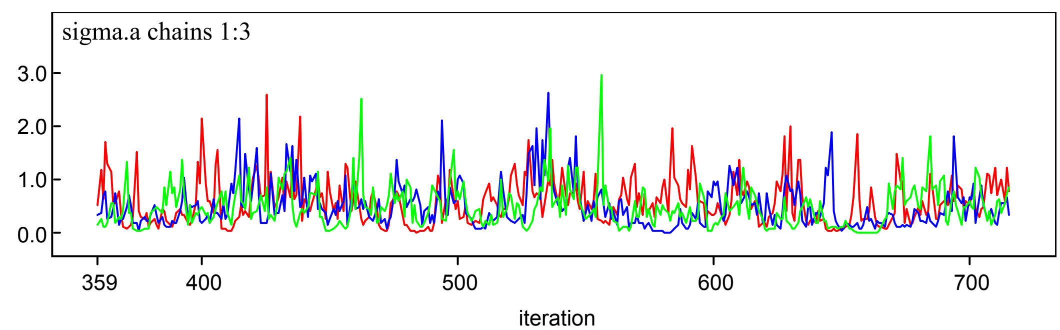
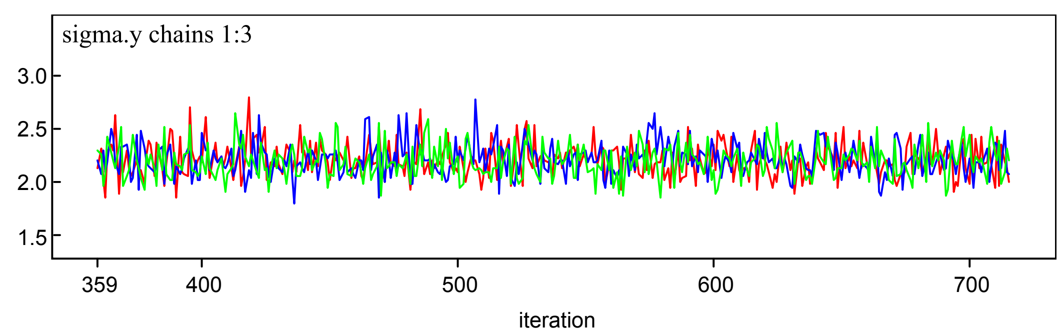
550.0
540.0
530.0
520.0
510.0
359
400
500
600
700
iteration


mu.a chains 1:3
15.0
12.5
10.0
7.5
5.0
359
400
500
600
700
iteration

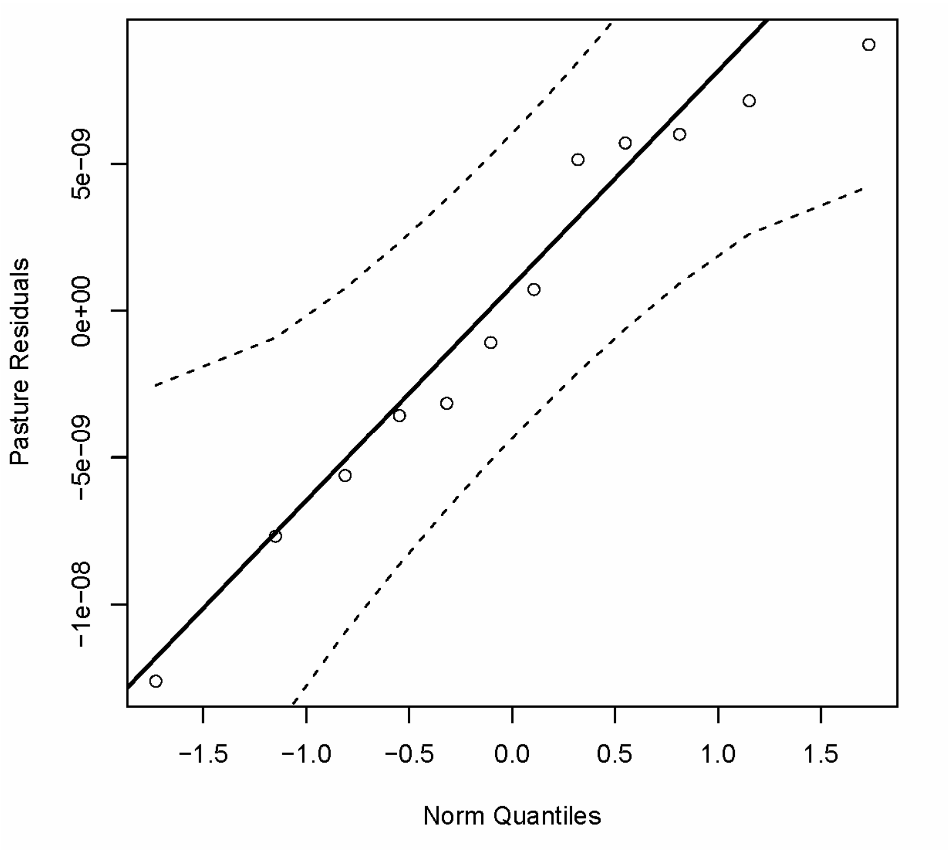

sigma.a chains 1:3
3.0
2.0
1.0
0.0
359
400
500
600
700
iteration


sigma.y chains 1:3
3.0
2.5
2.0
1.5
359
400
500
600
700
iteration

APPENDIX 4.9:

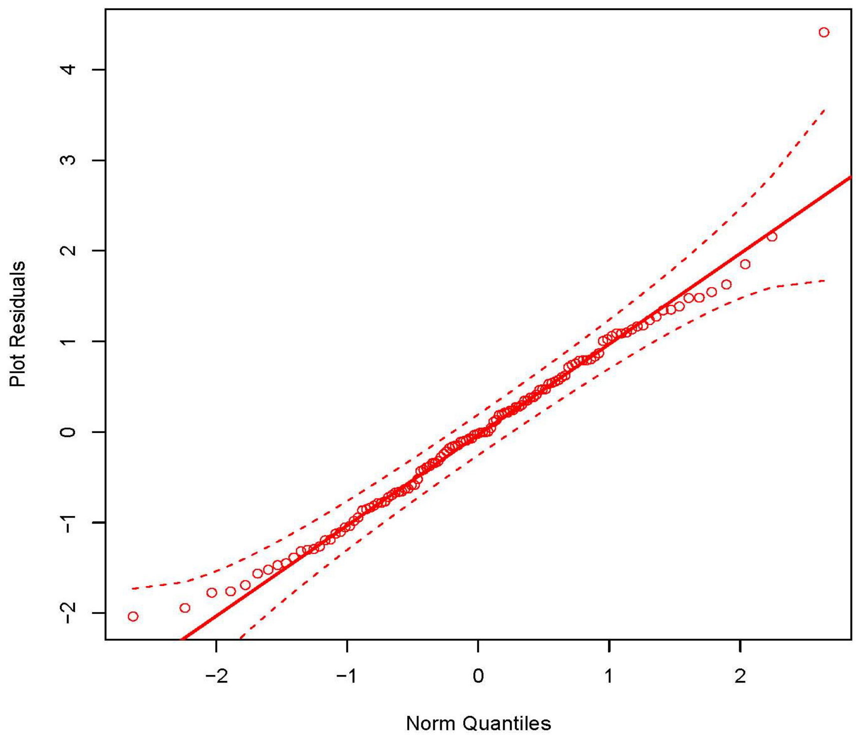
AN EXPLORATION OF NORMALITY IN THE MULTILEVEL MODEL OF NATIVE SPECIES RICHNESS

A. Level-2 Residuals:



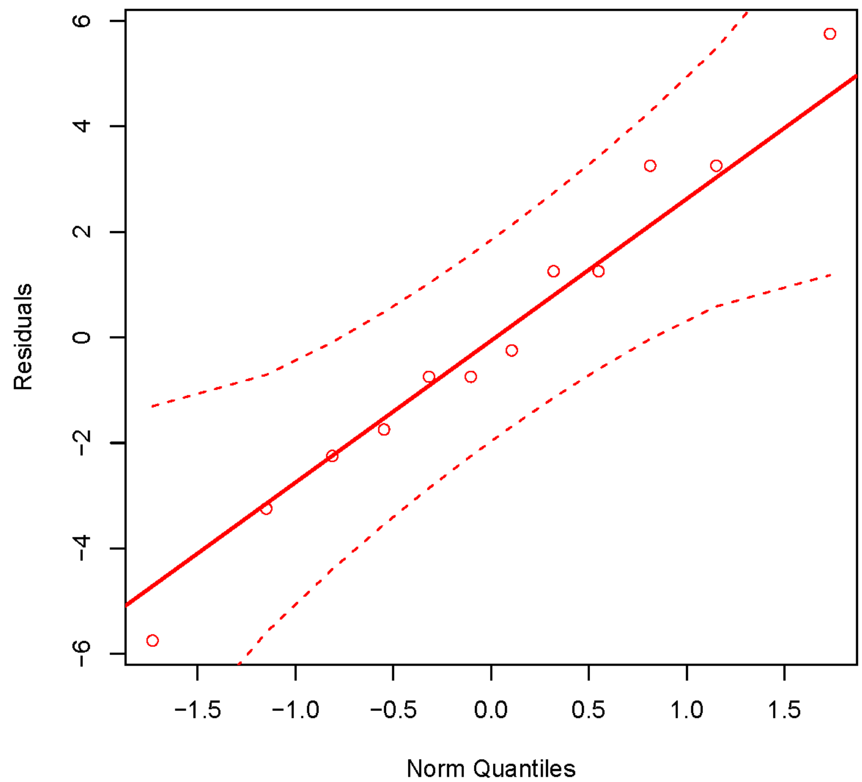
Note: Dotted lines indicate 95% confidence bands. All residuals fall within these bands, suggesting that the normality assumption was not violated.

B. Level-1 Residuals:



Note: Dotted lines indicate 95% confidence bands. Since all but one residual falls within these bands, there is no evidence that the normality assumption is violated.

C. Residuals of the pasture-level model of native species richness:



Note: Dotted lines indicate 95% confidence bands. All residuals fall within these bands, providing evidence that the normality assumption is not violated.

REFERENCES

Adams, W., D. Brockington, J. Dyson, and B. Vira. 2003. Managing tragedies: Understanding conflict over common pool resources. *Science* 302:1915-1916.

Adger, W., P. Kelly, A. Winkels, L. Huy, and C. Locke. 2002. Migration, remittances, livelihood trajectories, and social resilience. *Ambio* 31:358-366.

Adler, P. B., D. G. Milchunas, W. K. Lauenroth, O. E. Sala, and I. C. Burke. 2004. Functional traits of graminoids in semi-arid steppes: A test of grazing histories. *Journal of Applied Ecology* 41:653-663.

Adler, P. B., and J. M. Morales. 1999. Influence of environmental factors and sheep grazing on an Andean grassland. *Journal of Range Management* 52:471-481.

Adler, P. B., D. A. Raff, and W. K. Laurenroth. 2001. The effect of grazing on the spatial heterogeneity of vegetation. *Oceologia* 128:465-479.

Agrawal, A. 2001. Common property institutions and sustainable governance of resources. *World Development* 29:1649-1672.

Agrawal, A., and G. Yadama. 1997. How do local institutions mediate market and population pressures on resources? Forest Panchayats in Kumaon, India. *Development and Change* 28:435-465.

Alcorn, J. B. 1993. Indigenous peoples and conservation. *Conservation Biology* 7:424-426.

Allenby, R. J. 1987. Origin of the Bolivian Andean orocline: A geologic study utilizing landstat and shuttle imaging radar. *Tectonophysics* 142:137-154.

Alvard, M. S. 1993. Testing the Ecologically Noble Savage Hypothesis - Interspecific prey choice by Piro hunters of Amazonian Peru. *Human Ecology* 21:355-387.

Alzerreca, H., J. Laura, F. Loza, D. Luna, and J. Ortega. 2006. "Importance of carrying capacity in sustainable management of key high-Andean puna rangelands (bofedales) in Ulla Ulla, Bolivia," in *Land use change and mountain diversity*. Edited by E. Spehn, M. Liberman, and C. Körner. Boca Raton: CRC Press.

Anderson, D. R., K. P. Burnham, and W. L. Thompson. 2000. Null hypothesis testing: Problems, prevalence and an alternative. *Journal of Wildlife Management* 64:912-923.

Anderson, D. R., and K. R. Burnham. 2002. Avoiding pitfalls when using information-theoretic methods. *Journal of Wildlife Management* 66:912-918.

Andrew, M. H. 1988. Grazing impact in relation to livestock watering points. *Trends in Ecology and Evolution* 12:336-339.

Azhar-Hewitt, F. 1999. Women of the high pastures and the global economy: Reflections on the impacts of modernization in the Hushe Valley of the Karakorum, Northern Pakistan. *Mountain Research and Development* 19:141-151.

Bakker, C., J. Blair, and A. Knapp. 2003. Does resource availability, resource heterogeneity or species turnover mediate changes in plant species richness in grazed grasslands? *Oecologia* 137:385-391.

Banana, A. Y., and W. Gombya-Ssembajjwe. 2000. "Successful forest managment: The importance of security of tenure and rule enforcement in Ugandan forests," in *People and forests: Communities, institutions, and governance*. Edited by C. Gibson, M. McKean, and E. Ostrom. Cambridge: The MIT Press.

Bates, D., and D. Sarkar. 2007. "lme4: Linear mixed-effects models using S4 classes." R package: 0.99875-0.

Bebbington, A. 2000. Reencountering development: Livelihood transitions and place transformations in the Andes. *Annals of the Association of American Geographers* 90:495-520.

Becerra, J. A. B. 2006. "Grazing intensity, plant diversity, and rangeland conditions in the southeastern Andes of Peru (Palccoyo, Cusco)," in *Land use change and mountain diversity*. Edited by E. Spehn, M. Liberman, and C. Körner. Boca Raton: CRC Press.

Becker, C. D., and R. Leon. 2000. "Indigenous forest management in the Bolivian Amazon: Lesson from the Yuracare people," in *People and forests: communities, institutions, and governance*. Edited by C. Gibson, M.McKean, and E. Ostrom. Cambridge: MIT Press.

Beever, E., M. Huso, and D. Pyke. 2006. Multi-scale responses of soil stability and invasive plants to removal of non-native grazers from an arid conservation reserve. *Diversity and Distributions* 12:258-268.

Berkes, F. 1989. *Common property resources: Ecology and community-based sustainable resource development*. London: Belhaven Press.

Berkes, F., I. Davidson-Hunt, and K. Davidson-Hunt. 1998. Diversity of common property resource use and diversity of social interests in the western Indian Himalaya. *Mountain Research and Development* 18:19-33.

Berkes, F., D. Feeny, B. McCay, and J. Acheson. 1989. The benefits of the commons. *Nature* 340:91-93.

Beven, K. J., and M. J. Kirkby. 1979. A physical-based variable contributing area model of basin hydrology. *Hydrology Science Bulletin* 24:43-69.

Blaikie, P. M., and H. C. Brookfield. 1987. *Land degradation and society*. London; New York: Methuen.

Borgerhoff-Mulder, M. 1991. Datoga pastoralists of Tanzania. *Research & Exploration* 7:166+.

Borgerhoff-Mulder, M., and T. Caro. 1985. The use of quantitative observational techniques in Anthropology. *Current Anthropology* 26:323-335.

Boserup, E. 1965. *The conditions of agricultural growth: The economics of agrarian change under population pressure*. New York: Aldine Publishing Company.

Brandon, K., K. Redford, and S. Sanderson. 1998. *Parks in peril: People, politics, and protected areas*. Washington D.C: Island Press.

Bremner, J., and F. Lu. 2006. Common property among indigenous peoples of the Ecuadorian Amazon. *Conservation and Society* 4:499-521.

Brockington, D. 2001. Women's income and the livelihood strategies of dispossessed pastoralists near the Mkomazi Game Reserve, Tanzania. *Human Ecology* 29:307-338.

Bromley, D., D. Feeny, M. McKean, P. Peters, J. Gilles, R. Oakerson, C. Runge, and J. Thomson. Editors. 1992. *Making the commons work: Theory, practice, and policy*. San Francisco: Institute for Contemporary Studies Press.

Bromley, D. W., and M. M. Cernea. 1989. "The management of common property natural resources: Some conceptual and operational fallacies," in *World Bank Discussion Paper No. 57*. Washington D.C.: The World Bank.

Brower, B., and A. Dennis. 1998. "Grazing the forest, shaping the landscape? Continuing the debate about forest dynamics in Sagarmatha National Park," in *Nature's geography: New lessons for conservation in developing countries*. Edited by K. S. Zimmerer and K. R. Young pp. 184-208. Madison: University of Wisconsin Press.

Browman, D. L. 1983. Andean arid land pastoralism and development. *Mountain Research and Development* 3:241-252.

—. 1984. Pastoralism and development in high Andean arid lands. *Journal of Arid Environments* 7:313-328.

—. 1987. "Pastoralism in highland Peru and Bolivia," in *Arid land use strategies and risk management in the Andes: A regional anthropological perspective*. Edited by D. L. Browman, pp. 121-149. Boulder: Westview Press.

Bruner, A. G., R. E. Gullison, R. E. Rice, and G. A. da Fonseca. 2001. Effectiveness of parks in protecting tropical biodiversity. *Science* 291:125-128.

Brush, S. B. 1976. Mans use of an Andean ecosystem. *Human Ecology* 4:147-166.

—. 1977. *Mountain, field, and family: The economy and human ecology of an Andean valley*. Philadelphia: University of Pennsylvania Press.

—. 1980. "Peru's invisible migrants: A case study of inter-Andean migration," in *Land and power in Latin America: Agrarian economies and socical processes in the Andes*. Edited by B. Orlove and G. Custred. New York: Holmes and Meier Publishers, Inc.

—. 1982. The natural and human environment of the central Andes. *Mountain Research and Development* 2:19-38.

Brush, S. B., and D. W. Guillet. 1985. Small-scale agro-pastoral production in the central Andes. *Mountain Research and Development* 5:19-30.

Burnham, K. P., and D. R. Anderson. 2002. *Model selection and multimodel inference: A practical information-theoretic approach*. New York: Springer.

Byers, A. C. 1999. *Contemporary landscape change in the Huascaran National Park and buffer zone, Cordillera Blanca, Peru*. The Mountain Institute: Andes Program.

—. 2000. Contemporary landscape change in the Huascaran National Park and buffer zone, Cordillera Blanca, Peru. *Mountain Research and Development* 20:52-63.

Campbell, B., and R. Godoy. 1992. "Commonfield agriculture: The Andes and medieval England compared," in *Making the commons work: Theory, practice, and policy*. Edited by D. W. Bromley, pp. 99-127. San Francisco: Institute for Contemporary Studies.

Caro, T. 1998. "How do we refocus behavioral ecology to address conservation issues more directly?," in *Behavioral ecology and conservation biology*. Edited by T. Caro. New York: Oxford University Press.

Casimir, M., and A. Rao. 1998. Sustainable herd management and the tragedy of no man's land: An analysis of west Himalayan pastures using remote sensing techniques. *Human Ecology* 26:113-134.

Chakravarty-Kaul, M. 1998. Transhumance and customary pastoral rights in Himachal Pradesh: Claiming the high pastures for Gaddis. *Mountain Research and Development* 18:5-17.

Chamberlin, J. 1974. Provision of collective goods as a function of group size. *American Political Science Review* 68:707-716.

Chamberlin, T. C. 1995. Historical essay: The method of multiple working hypotheses (reprinted from Science, 1890). *Journal of Geology* 103:349-354.

Chapin, M. 2004. A challenge to conservationists. *World Watch* November/December:17-31.

Chase, A. 1987. *Playing God in Yellowstone: The destruction of America's first national park*. New York: HBJ Publishers.

Chepstow-Lusty, A., K. Bennett, J. Fjeldsa, A. Kendall, W. Galiano, and A. Herrera. 1998. Tracing 4,000 years of environmental history in the Cuzco area, Peru, from the pollen record. *Mountain Research and Development* 18:159-172.

Christansen, N. J. 1988. "Succession and natural disturbance: Paradigms, problems and the preservation of natural ecosystems," in *Ecosystem management for parks and wilderness*. Edited by J. K. Aggee and D. R. Johnson. Seattle: University of Washington Press.

Cingolani, A., M. Cabido, D. Renison, and V. Solis. 2003. Combined effects of environment and grazing on vegetation structure in Argentine granite grasslands. *Journal of Vegetation Science* 14:223-232.

Cingolani, A., I. Noy-Meir, and S. Diaz. 2005. Grazing effects on rangeland diversity: A synthesis of contemporary models. *Ecological Applications* 15:757-773.

Ciriacy-Wantrup, S. V., and R. C. Bishop. 1975. Common property as a concept in natural-resources policy. *Natural Resources Journal* 15:713-727.

Clapperton, C. M. 1983. The glaciation of the Andes. *Quaternary Science Review* 2:83-155.

Collins, S. L., and S. M. Glenn. 1997. Intermediate disturbance and its relationship to within and between-patch dynamics. *New Zealand Journal of Ecology* 21:103-110.

Collins, S. L., A. K. Knapp, J. M. Briggs, J. M. Blair, and E. M. Steinauer. 1998. Modulation of diversity by grazing and mowing in native tallgrass prairie. *Science* 280:745-747.

Connell, J. H. 1978. Diversity in tropical rain forests and coral reefs: High diversity of trees and corals is maintained only in a non-equilibrium state. *Science* 199:1302-1310.

Cronon, W. 1996. "The trouble with wilderness or, getting back to the wrong nature," in *Uncommon ground: Rethinking the human place in nature*. Edited by W. Cronon, pp. 69-90. New York: Norton.

Dayton-Johnson, J., and P. Bardhan. 2002. Inequality and conservation on the local commons: A theoretical exercise. *Economic Journal* 112:577-602.

de Ferreyra, E. C. 1979. Vegetacion del Valle De Chiquian, Thesis, Universidad Nacional Mayor de San Marcos.

del Castillo, L., M. Gallo, and C. Monge. 1995. "Ley 26505: "La nueva ley de tierras: Analisis y comentarios," 1st edition, pp. 30. Lima: Grafica Bellido S.R.L.

Denevan, W. M. 1992. The pristine myth: The landscape of the Americas in 1492. *Annals of the Association of American Geographers* 82:369-385.

Dewey, J. F., and S. H. Lamb. 1992. Active tectonics of the Andes. *Tectonophysics* 205:79-95.

Echavarria, F. R. 1998. "Monitoring forests in the Andes using remote sensing: An example from Southern Ecuador " in *Nature's geography: New lessons for conservation in developing countries*. Edited by K. S. Zimmerer and K. R. Young, pp. 100-120. Madison: University of Wisconsin

Eckholm, E. 1976. *Losing ground: Environmental stress and food problems*. New York: W.W. Norton.

Ellenberg, H. 1979. Man's influence on tropical mountain ecosystems in South America, 2nd Tansley Lecture. *Journal of Ecology* 67:401+.

Environment and Sustainable Development Programme. 2002. "Mountain momentum: Agenda for today and policy beyond the International Year of the Mountains 2002." United Nations University: http://www.unu.edu/mountains2002/policybrief.pdf, accessed on June 25, 2007.

Escobar, G., and C. Beall. 1982. Contemporary patterns of migration in the central Andes. *Mountain Research and Development* 2:63-80.

Fairhead, J., and M. Leach. 2000. "The nature lords: After desolation, conservation, and eviction," in *Times Literary Supplement*, pp. 3-4. New York.

Feeny, D., F. Berkes, B. McCay, and J. Acheson. 1990. The tragedy of the commons - 22 years later. *Human Ecology* 18:1-19.

—. 1998. "The tragedy of the commons - 22 years later," in *Managing the Commons*, 2nd. edition. Edited by J. A. Baden and D. S. Noonan. Bloomington: Indiana University Press.

Fensham, R. J., J. E. Holman, and M. J. Cox. 1999. Plant species responses along a grazing disturbance gradient in an Australian grassland. *Journal of Vegetation Science* 10:77-86.

Fjeldså, J. 1992. "Biogeography of the birds of the *Polylepis* woodlands of the Andes," in *Paramo: An ecosystem under human influence*. Edited by H. Balslev and J. Luteyn, pp. 31-44. London: Academic Press.

Fjeldså, J., E. Lambin, and B. Mertens. 1999. Correlation between endemism and local ecoclimatic stability documented by comparing Andean bird distributions and remotely sensed land surface data. *Ecography* 22:63-78.

Flores Ochoa, J. Editor. 1977. *Pastores de Puna: Uywamichiq punarunakuna*. Lima: Instituto de Estudios Peruanos.

Forsyth, T. 1998. Mountain myths revisited: Integrating natural and social environmental science. *Mountain Research and Development* 18:107-116.

—. 2003. *Critical political ecology: The politics of environmental science*. London: Routledge.

Friedel, M. H. 1991. Range condition management and the concept of thresholds: A viewpoint. *Journal of Range Management* 44:422-426.

Gade, D. W. 1992. Landscape, system, and identity in the postconquest Andes. *Annals of the Association of American Geographers* 82:460-477.

—. 1999. *Nature and culture in the Andes*. Madison: University of Wisconsin Press.

Ghez, G. R., and G. S. Becker. 1975. "The allocation of time and goods over the life cycle," in *National Bureau of Economic Research*. New York: Columbia University Press.

Gilles, J., and M. Hahdi. 1992. "Oukaimedene, Morocco: A high mountain Agdal," in *Making the commons work: theory, practice and policy*. Edited by D. W. Bromley, pp. 229-246. San Francisco: Institute for Contemporary Studies.

Gilles, J., and K. Jamtgaard. 1981. Overgrazing in pastoral areas - the commons reconsidered. *Sociologia Ruralis* 21:129-141.

Godoy, R. 2001. *Indians, markets, and rainforests: Theory, methods, analysis*. New York: Columbia University Press.

Godoy, R., N. Brokaw, and D. Wilkie. 1995. The effect of income on the extraction of non-timber tropical forest products: Model, hypotheses, and preliminary findings from the Sumu Indians of Nicaragua. *Human Ecology* 23:29-52.

Godoy, R. A. 2002. The life cycle, ecological, and economic determinants of spousal leisure sharing: Panel estimations from Tawahka Amerindians, Honduras. *Human Ecology* 30:317-337.

Goland, C. 1993. Field scattering as agricultural risk management: A case-study from Cuyo-Cuyo, Department of Puno, Peru. *Mountain Research and Development* 13:317-338.

Goodland, R., G. Ledec, and M. Webb. 1989. "Meeting environmental concerns caused by common property mismanagment in economic development projects," in *Common property resources: Ecologoy and community-based sustainable development*. Edited by F. Berkes, pp. 148-163. London: Bellhaven.

Grime, J. P. 1973. Competitive exclusion in herbaceous vegetation. *Nature* 242:344-347.

Gross, D. R. 1984. Time allocation - A tool for the study of cultural-behavior. *Annual Review of Anthropology* 13:519-558.

Grossman, L. 1981. The cultural ecology of economic development. *Annals of the Association of American Geographers* 71:220-236.

Grotzbach, E., and C. Stadel. 1997. "Mountain peoples and cultures," in *Mountains of the world: A global priority*. Edited by B. Messerli and J. Ives, pp. 17-38. New York: Parthenon Publishing Group.

Guillet, D. 1980. Reciprocal labor and peripheral capitalism in the central Andes. *Ethnology* 19:151-167.

—. 1981. Land tenure, ecological zone, and agricultural regime in the Central Andes. *American Ethnologist* 8:139-156.

—. 1983. Toward a cultural ecology of mountains - The central Andes and the Himalayas compared. *Current Anthropology* 24:561-574.

—. 1987. "On the potential for intensification of agro-pastoralism in the arid zones of the central Andes," in *Arid land use strategies and risk management in the Andes: A regional anthropological perspective*. Edited by D. Browman, pp. 81-98. Boulder: Westview Press.

Hammond, G. B., I. D. Fernandez, L. F. Villegas, and A. J. Vaisberg. 1998. A survey of traditional medicinal plants from the Callejón de Huaylas, Department of Ancash, Peru. *Journal of Ethnopharmacology* 61:17-30.

Hardin, G. 1968. Tragedy of the commons. *Science* 162:1243+.

Hernandez, Z., and M. Monasterio. 2006. "Functional diversity of wetland vegetation in the high-Andean paramo, Venezuela," in *Land Use Change and Mountain Diversity*. Edited by E. Spehn, M. Liberman, and C. Körner. Boca Raton: CRC Press.

Hewitt, K. 1988. "The study of mountain lands and peoples," in *Human impact on mountains*. Edited by N. Allan, G. Knapp, and C. Stadel, pp. 6-23. Totowa: Rowman & Littlefield Publishers.

Holdridge, R. L. 1967. *Life zone ecology*. San Jose: Tropical Science Center.

Holling, C. S. Editor. 1978. *Adaptive environmental assessment and management*. London: John Wiley and Sons.

Holt, F. L. 2005. The Catch-22 of conservation: Indigenous peoples, biologists, and cultural change. *Human Ecology* 33:199-215.

Hunter, C. 2002. "Aspects of the sustainable tourism debate from a natural resources perspective," in *Sustainable tourism: A global perspective*. Edited by R. Harris, T. Griffin, and P. Williams, pp. 3-23. Oxford: Elsevier Science Ltd.

Huston, M. 1979. General hypothesis of species-diversity. *American Naturalist* 113:81-101.

IDB. 2006. "Summary Demographic Data for Peru." U.S. Census Bureau, Population Division, International Programs Center, International Database (IDB): http://www.census.gov/ipc/www/idbprint.html, accessed on June 25, 2007.

INEI. 1994. "Directorio nacional de centros poblados segun codigo de ubicacion geografica." Lima, Peru: Instituto Nacional de Estadística e Informática (INEI).

— 2001. "Encuesta demografica y de salud familiar 2000." Edited by J. Reyes and L. H. Ochoa. Lima, Peru: Instituto Nacional de Estadística e Informática (INEI)/Programa DHS/Macro International Inc.

INRENA. 1990a. "Plan maestro del Parque Nacional Huascaran: Plan de conservacion para la investigacion y el desarrollo." Lima, Peru: Instituto Nacional de Recursos Nacionales (INRENA).

— 1990b. "Plan maestro del Parque Nacional Huascaran: Generalides y diagnosticos." Lima, Peru: Instituto Nacional de Recursos Nacionales (INRENA).

— 1996. "Plan de uso turistico y recreativo del Parque Nacional Huascaran." Lima, Peru: Instituto Nacional de Recursos Nacionales (INRENA).

— 2001. "Registro del ingreso de visitantes al Parque Nacional Huascaran." Huaraz, Peru: Instituto Nacional de Recursos Nacionales (INRENA), Sub Programa de Turismo,.

— 2002. "Plan de use turistico y recreativo del Parque Nacional Huascaran." Huaraz, Peru: Instituto Nacional de Recursos Nacionales (INRENA).

— 2003. "Plan maestro del Parque Nacional Huascaran, 2003-2007." Lima, Peru: Instituto Nacional de Recursos Nacionales (INRENA).

Insightful. 2005a. "S-PLUS Correlated Data Library, version 1.0, release 1," Insightful Corporation.

—. 2005b. "S-PLUS, Enterprise Developer Version: 7.0.6," Insightful Corporation.

Ives, J., B. Messerli, and E. Spiess. Editors. 1997. *Mountains of the world: A global priority*. New York: Parthenon Publishing Group.

Jaeffret, S., and S. Lavorel. 2003. Are plant functional types relevant to describe degradation in arid, Southern Tunisian steppes? *Journal of Vegetation Science* 14:399-408.

Jodha, N. S. 1987. "A case study of the degradation of common property resources in India," in *Land degradation and society*. Edited by P. Blaikie and H. Brookfield. London: Metheun.

—. 1992. "Common property resources: A missing dimension of development strategies," in *World Bank Discussion Papers*, vol. 169. Washington, D.C.: World Bank.

—. 1995. "The Nepal middle mountains," in *Regions at risk: Comparison of threatened environments*. Edited by K. A. Turner. Tokyo: United Nations University Press.

—. 1996. "Property rights and development," in *Rights to nature: Ecological, economic, cultural and political principles of institutions for the environment*. Edited by S. Hanna, C. Folke, and K.-G. Mäler, pp. 205-220. Washington, D.C.: Island Press.

Johnson, A., and R. Sackett. 1998. "Direct systematic observation of behavior," in *Handbook of methods in cultural anthropology*. Edited by H. R. Bernard, pp. 301-331. Walnut Creek: Alta Mira Press.

Kintz, D. B., K. R. Young, and K. A. Crews-Meyer. 2006. Implications of land use/land cover change in the buffer zone of a national park in the tropical Andes. *Environmental Management* 38:238-252.

Kleinbaum, D. G. 2002. *Logistic regression: A self-learning text*. New York: Springer.

Knapp, G. W. 1991. *Andean ecology: Adaptive dynamics in Ecuador*. Boulder: Westview Press.

Kolff, H., and K. Kolff. 1997. *Wildflowers of the Cordillera Blanca*. Huaraz, Peru: Instituto de Montaña.

Körner, C. 1999. *Alpine plant life: Functional plant ecology of high mountain ecosystems*. Berlin: Springer.

Körner, C., G. Nakhutsrishvili, and E. Spehn. 2006. "High-elevation land use, biodiversity, and ecosystem functioning," in *Land use change and mountain diversity*. Edited by E. Spehn, M. Liberman, and C. Körner. Boca Raton: CRC Press.

Kramer, R., C. van Schaik, and J. Johnson. 1997. "Preservation paradigms and tropical rain forests," in *Last stand: Protected areas and the defense of tropical* *biodiversity*. Edited by R. Kramer, C. van Schaik, and J. Johnson. New York: Oxford University Press.

Kuznar, L. 1991a. Mathematical models of pastoral production and herd composition in traditional Andean herds. *Journal of Quantitative Anthropology* 3:1-17.

Kuznar, L. A. 1991b. Herd composition in an Aymara community of the Peruvian altiplano: A linear-programming problem. *Human Ecology* 19:369-387.

Laird, N. M., and J. H. Ware. 1982. Random-effects models for longitudinal data. *Biometrics* 38:963-974.

Lambin, E., B. Turner, H. Geist, S. Agbola, A. Angelsen, J. Bruce, O. Coomes, R. Dirzo, G. Fischer, C. Folke, P. George, K. Homewood, J. Imbernon, R. Leemans, X. Li, E. Moran, M. Mortimore, P. Ramakrishnan, J. Richards, H. Skanes, W. Steffen, G. Stone, U. Svedin, T. Veldkamp, C. Vogel, and J. C. Xu. 2001. The causes of land-use and land-cover change: Moving beyond the myths. *Global Environmental Change-Human and Policy Dimensions* 11:261-269.

Lauer, W. 1993. Human-development and environment in the Andes: A geoecological overview. *Mountain Research and Development* 13:157-166.

Laycock, W. A. 1991. Stable states and thresholds of range condition on North-American rangelands - A viewpoint. *Journal of Range Management* 44:427-433.

Lindberg, K., J. Enriquez, and K. Sproule. 1996. Ecotourism questioned: Case studies from Belize. *Annals of Tourism Research* 23:543-562.

Lipton, J. 2007. Implications of land use and landcover change in Huascarán National Park, Peru. Dissertation draft, University of Texas at Austin.

Lu, F. 2001. The common property regime of the Huaorani indians of Ecuador: Implications and challenges to conservation. *Human Ecology* 29:425-447.

Luteyn, J., S. Churchill, D. Griffin III, S. Gradstein, H. Sipman, and M. Gavilanes. Editors. 1999. *Paramos: A checklist of plant diversity, geographical distribution, and botanical literature. Memoirs of the New York Botanical Garden*. Bronx: New York Botanical Garden Press.

Lynch, O. J., and G. F. Maggio. 2000. "A general overview of mountain laws and policies with insights from the Mountain Forum's electronic conference on mountain policy and law." *Mountain Laws and Peoples: Moving towards sustainable development and recognition of community-based property rights, electronic conference, Mountain Forum, 2000*.

MacArthur, R. H., and E. R. Pianka. 1966. On optimal use of a patchy environment. *American Naturalist* 100.

Mace, R., and A. Houston. 1989. Pastoralist strategies for survival in unpredictable environments: A model of herd composition that maximizes household viability. *Agricultural Systems* 31:185-204.

Mackey, R. L., and D. J. Currie. 2001. The diversity-disturbance relationship: Is it generally strong and peaked? *Ecology* 82:3479-3492.

Manne, L. L., T. M. Brooks, and S. L. Pimm. 1999. Relative risk of extinction of passerine birds on continents and islands. *Nature* 399:258-261.

Mauny, F., J. Viel, P. Handschumacher, and B. Sellin. 2004. Multilevel modelling and Malaria: A new method for an old disease. *International Journal of Epidemiology* 33:1337-1344.

Mayer, E. 1985. "Production zones " in *Andean ecology and civilization*. Edited by S. Masuda and C. Morris, pp. 45-84. Tokyo: University of Tokyo Press.

—. 2002. *The articulated peasant: Household economies in the Andes*. Boulder: Westview Press.

McArdle, B. H., and M. J. Anderson. 2004. Variance heterogeneity, transformations, and models of species abundance: A cautionary tale. *Canadian Journal of Fisheries and Aquatic Sciences* 61:1294-1302.

McCay, B., and S. Jentoft. 1998. Market or community failure? Critical perspectives on common property research. *Human Organization* 57:21-29.

McCay, B. J., and J. A. Acheson. 1996. "Human ecology of the commons," in *The Question of the Commons*. Edited by B. J. McCay and J. A. Acheson, pp. 1-34. Tucson: University of Arizona Press.

McCorkle, C. 1987. "Punas, pastures, and fields: Grazing strategies and the agropastoral dialect in an indigenous Andean community," in *Arid land use strategies and risk management in the Andes: A regional anthropological perspective*. Edited by D. Browman, pp. 57-79. Boulder: Westview Press.

McCune, B., and M. Mefford. 1999. "Multivariate analysis of ecological data, version 4." Gleneden Beach: MJM Software Design.

McKean, M. A. 1992. "Management of traditional common lands (Iriachi) in Japan " in *Making the commons work: Theory, practice, and policy*. Edited by D. W. Bromley, pp. 63-98. San Francisco: Institute for Contemporary Studies.

—. 2000. "Common property: What is it, what is it good for, and what makes it work?," in *People and Forests: Communities, Institutions and Governments*. Edited by C. C. Gibson, M. A. McKean, and E. Ostrom. Cambridge, Mass.: MIT Press.

McLaren, D. 1998. *Rethinking tourism and ecotravel: The paving of paradise and what you can do to stop it*. West Hartford: Kumarian Press.

Milchunas, D. G., W. K. Lauenroth, P. L. Chapman, and M. K. Kazempour. 1989. Effects of grazing, topography, and precipitation on the structure of a semiarid grassland. *Vegetatio* 80:11-23.

Mitchell, R., and P. Eagles. 2001. An integrative approach to tourism: Lessons from the Andes of Peru. *Journal of Sustainable Tourism* 9:4-28.

Molinillo, M., and M. Monasterio. 1997. Pastoralism in paramo environments: Practices, forage, and impact on vegetation in the Cordillera of Merida, Venezuela. *Mountain Research and Development* 17:197-211.

—. 2006. "Vegetation and grazing patterns in Andean environments: A comparison of pastoral systems in punas and paramos," in *Land use change and mountain diversity*. Edited by E. Spehn, M. Liberman, and C. Körner. Boca Raton: CRC Press.

Moseley, M. E. 1992. *The Incas and their ancestors :The archaeology of Peru*. New York: Thames and Hudson.

Murphy, V., and A. Dunne. 2005. Mixed effects versus fixed effects modelling of binary data with inter-subject variability. *Journal of Pharmacokinetics and Pharmacodynamics* 32:245-260.

Murra, J. V. 1972. El "control vertical" de un máximo de pisos ecológicos en la economía de las sociedades andinas. Visita de la Provincia de León de Huánuco en 1562, vol. 2:427-476.

Myers, N., R. Mittermeier, C. Mittermeier, G. da Fonseca, and J. Kent. 2000. Biodiversity hotspots for conservation priorities. *Nature* 403:853-858.

Netting, R. M. 1976. What alpine peasants have in common: Observations of communal tenure in a swiss village. *Human Ecology* 4:135-146.

—. 1981. *Balancing on an alp: Ecological change and continuity in a Swiss mountain community*. Cambridge: Cambridge University Press.

—. 1986. *Cultural ecology*, 2nd ed. edition. Prospect Heights: Waveland Press, Inc.

—. 1993. *Smallholders, householders: Farm families and the ecology of intensive, sustainable agriculture*. Stanford: Stanford University Press.

—. 1997. Unequal commoners and uncommon equity: Property and community among smallholder farmers. *The Ecologist* 27:28-33.

Neumann, R. P. 1998. *Imposing wilderness: Struggles over livelihood and nature preservation in Africa*. Berkeley: University of California Press.

O'Connor, T. G. 1991. Local extinction in perennial grasslands: A life-history approach. *American Naturalist* 137:753-773.

Oakerson, R. J. 1992. "Analyzing the commons: A framework," in *Making the commons work: Theory, practice, and policy*. Edited by D. W. Bromley, pp. 41-59. San Francisco: Institute for Contemporary Studies.

Oates, J. F. 1999. *Myth and reality in the rain forest: How conservation strategies are failing in West Africa*. Berkeley: University of California Press.

Oldfield, M. L., and J. B. Alcorn. 1991. *Biodiversity: Culture, conservation and ecodevelopment*. Boulder: Westview Press.

Olff, H., and M. Ritchie. 1998. Effects of herbivores on grassland plant diveristy. *Trends in Ecology and Evolution* 13:261-265.

Oliver, P., G. Marwell, and R. Teixeira. 1985. A theory of the critical mass. I. Interdependence, group heterogeneity, and the production of collective cction. *American Journal of Sociology* 91:522-556.

Olson, M. 1965. *The logic of collective action: Public goods and the theory of groups*. Cambridge: MIT Press.

Orlove, B. 1976. "The tragedy of the commons revisited: Land use and environmental quality in high-altitude Andean grasslands." *Hill Lands, Proceedings of International Symposium, Morgantown, 1976*, pp. 208-214.

—. 1980. "Pastoralism in the Southern Sierra," in *Andean peasant economics and pastoralism*, vol. 1. Edited by Small Ruminants Collaborative Research Support Program, pp. 87-111. Columbia: University of Missouri.

—. 1991. The evolution of common-field agricutlrue in the Andes: A hypothesis. *Comparative Studies in Society and History* 33:395-414.

Orlove, B., J. Chiang, and M. Cane. 2000. Forecasting Andean rainfall and crop yield from the influence of El Niño on Pleiades visibility. *Nature* 403:68-71.

Orlove, B. S. 1977. *Alpacas, sheep, and men: The wool export economy and regional society in southern Peru*. New York: Academic Press.

Ostrom, E. 1991. *Governing the commons: The evolution of institutions for collective action*. Cambridge[England]: New York: Cambridge University Press.

—. 1992. "The rudiments of a theory of the origins, survival, and performance of common-property institutions," in *Making the commons work: Theory, practice* *and policy*. Edited by D. W. Bromley, pp. 293-318. San Francisco: Institute for Contemporary Studies.

—. 1996. "Institutional arrangements for resolving the commons dilemma: Some contending approaches," in *The Question of the Commons: The culture and ecology of communal resources*. Edited by B. McCay and J. Acheson, pp. 250-265: University of Arizona Press.

—. 1998. "Reflections on the commons," in *Managing the commons*, 2nd edition. Edited by J. A. Baden and D. S. Noonan, pp. 95-116. Bloomington: Indiana University Press.

Ostrom, E., J. Burger, C. Field, R. Norgaard, and D. Policansky. 1999. Revisiting the commons: Local lessons, global challenges. *Science* 284:278-282.

Palomino, M. Z. A. 2000. Estudio de las plantas medicinales en la Reserve de Biosfera Huascaran (Carpa-Pastoruri & Yupa-Pitec). Thesis, Universidad Nacional Mayor de San Marcos.

Panayotou, T. 1992. *Environmental Kutnez Curve: Empirical tests and policy implications*: Harvard Institute for International Development.

Parker, A. J. 1982. The topographic relative moisture index: An approach to soil-moisture assessment in mountain terrain. *Physical Geography* 3:160-168.

Paruelo, J. M., W. K. Lauenroth, and P. A. Roset. 2000. Technical note: Estimating aboveground plant biomass using a photographic technique. *Journal of Range Management* 53:190-193.

Perez, F. L. 1987. Needle-ice activity and the distribution of stem-rosette species in a Venezuelan Paramo. *Arctic and Alpine Research* 19:135-153.

—. 1993. Turf destruction by cattle in the high equatorial Andes. *Mountain Research and Development* 13:107-110.

—. 1998. "Human impact on the high paramo landscape of the Venezuelan Andes," in *Nature's Geography: New lessons for conservation in developing countries*. Edited by K. Zimmerer and K. Young, pp. 147-183. Madison: The University of Wisconsin Press.

Petford, N., and M. Atherton. 1992. Granitoid emplacement and deformation along a major crustal lineament: The Cordillera Blanca, Peru. *Tectonophysics* 205:171-185.

Pinedo, G. D. 1999. Rebanos privados, pastos colectivos: Equidad y eficiencia en una comunidad de la Cordillera Huayhuash, Andes del norte del Peru. Thesis, Universidad Nacional Mayor de San Marcos.

Piperno, D. R., and D. M. Pearsall. 1998. *The origins of agriculture in the lowland neotropics*. San Diego: Academic Press.

Place, S. E. 1995. Ecotourism for sustainable development: Oxymoron or plausible strategy? *Geoforum* 35:161-173.

Poma, M. L. S. 2002. *Manual de pastos nativos del Parque Nacional Huascaran*. Huaraz, Peru: Instituto Nacional de Recursos Nacionales (INRENA).

Pouyaud, B., B. Francou, P. Chavallier, and P. Ribstein. 1997. "Contribucion del programa "Nieves y Glaciares Tropicales" (NGT) al conocimiento de la variabilidad climatica en los Andes." Edited by http://www.unesco.org.uy/phi/libros/enso/pouyaud.html.

Prakash, S. 1998. Social institutions and common property resources in the mountains. *Mountain Research and Development* 18:1-3.

Preston, D. 1998. Post-peasant capitalist graziers: The 21st century in southern Bolivia. *Mountain Research and Development* 18:151-158.

Pyke, G. H., H. R. Pulliam, and E. L. Charnov. 1977. Optimal foraging: A selective review of theory and tests. *Quarterly Review of Biology*:137-154.

Quammen, D. 2006. An endangered idea. *National Geographic* October:63-67.

R Development Core Team. 2007. "R: A language and environment for statistical computing." Vienna, Austria: R Foundation for Statistical Computing.

Ramirez, V. R. 2002. "Plan de conservacion de quebradas: Manejo turistico local." Huaraz, Peru: The Mountain Institute: Andes Program.

Redford, K. H. 2006. Parks as Shibboleths. *Conservation Biology* 20:1-2.

Redford, K. H., and A. M. Stearman. 1993. On common ground - Response to Alcorn. *Conservation Biology* 7:427-428.

Reice, S. R. 1994. Nonequilibrium determinants of biological community structure. *American Scientist* 82:424-435.

Reichman, O. J., J. H. Benedix Jr., and T. R. Seastedt. 1993. Distinct animal-generated edge effects in a tallgrass prairie community. *Ecology* 74:1281-1285.

REPAAN. 1993. "Produccion y utilizacion de los pastizales de la zona alto-Andina." Edited by C. L. Velarde and F. I. Cadena. Quito, Ecuador: Red de Pastizales Andinos (REPAAN), Centro Internacional de Investigacion para el Desarrollo (CIID-Canada).

RERUMEN. 1992. "Manual de forrajes para zonas aridas y semi-aridas Andinas." Edited by A. M. Florez. Lima, Peru: Red de Ruminantes Menores (RERUMEN), Programa de Apoyo a la Investigacion Colaborativa en Rumiantes Menores (SR-CRSP).

Rietkerk, M., and J. van de Koppel. 1997. Alternate stable states and threshold effects in semi-arid grazing systems. *Oikos* 79.

Robbins, P. 2004. *Political ecology: A critical introduction*. Malden: Blackwell Publishing.

Robbins, P., K. McSweeney, T. Waite, and J. Rice. 2006. Even conservation rules are made to be broken: Implications for biodiversity. *Environmental Management* 37:162-169.

Rodriguez, A., S. Andelman, M. Bakarr, L. Boitani, T. Brooks, R. Cowling, L. Fishpool, G. da Fonseca, K. Gaston, M. Hoffmann, J. Long, P. Marquet, J. Pilgrim, R. Pressey, J. Schipper, W. Sechrest, S. Stuart, L. Underhill, R. Waller, M. Watts, and X. Yan. 2004. Effectiveness of the global protected area network in representing species diversity. *Nature* 428:640-643.

Rodriguez, L. O., and K. R. Young. 2000. Biological diversity of Peru: Determining priority areas for conservation. *Ambio* 29:329-337.

Runge, C. F. 1981. Common property externalities - isolation, assurance, and resource depletion in a traditional grazing context. *American Journal of Agricultural Economics* 63:595-606.

—. 1984. Institutions and the free rider: The assurance problem in collective action. *Journal of Politics* 46:154-181.

—. 1986. Comon property and collective action in economic-development. *World Development* 14:623-635.

Ruttan, L. M., and M. Borgerhoff-Mulder. 1999. Are east African pastoralists truly conservationists? *Current Anthropology* 40:621-652.

Savage, M. 1993. Ecological disturbance and nature tourism. *Geographical Review* 83:290-300.

Schlager, E., and E. Ostrom. 1992. Property-rights regimes and natural-resources: A conceptual analysis. *Land Economics* 68:249-262.

Sen, A. 1973. *On economic inequality*. Oxford: Clarendon.

Sharman, D. 1995. *Climbs of the Cordillera Blanca of Peru*. Aberdeen: Whizzo Climbs.

Singer, J. D., and J. B. Willett. 2003. *Applied longitudinal data analysis: Modeling change and event occurrence*. New York: Oxford University Press.

Singleton, S. 2001. "Communities, states, and the governance of Pacific northwest salmon fisheries," in *Communities and the Environment: Ethnicity, gender and the state in community-based conservation*. Edited by A. Agrawal and C. Gibson, pp. 138-156. New Brunswick: Rutgers University Press.

Smith, D. N. 1988. Flora and vegetation of the Huascaran National Park, Ancash, Peru, with preliminary taxonomic studies for a manual of the flora. Dissertation., Iowa State University.

Smith, E., and M. Wishnie. 2000. Conservation and subsistence in small-scale societies. *Annual Review of Anthropology* 29:493-524.

Smith, J. M. B. 1977. Vegetation and microclimate of east and west-facing slopes in the grasslands of Mt. Wilhelm, Papua New Guinea. *Journal of Ecology* 65:39-53.

Snijders, T. A., and R. J. Bosker. 1999. *Multilevel Analysis: An introduction to advanced and multilevel modeling*. London: Sage Publications.

Sotomayor, G. L. 2000. "Como saber la calidad o condicion del pasto nativo." Huaraz, Peru: The Mountain Institute: Andes Program.

Spehn, E., M. Liberman, and C. Korner. 2006. *Land use change and mountain biodiversity*. Boca Raton: CRC Press.

Spiegelhalter, D., A. Thomas, N. Best, and D. Lunn. 2007. "WinBUGS version 1.4.2." Cambridge.

Stevens, S. 1993. *Claiming the high ground: Sherpas, subsistence and environmental change in the highest Himalaya*. Berkeley: University of California Press.

—. 1997. *Conservation through cultural survival: Indigenous peoples and protected areas*. Washington D.C.: Island Press.

Stevenson, G. 1991. *Common property economics: A general theory and land use applications*. Cambridge: Cambridge University Press.

Stohlgren, T. J., M. B. Falkner, and L. D. Schell. 1995. A modified-Whittaker nested vegetation sampling method. *Vegetatio* 117:113-121.

Stohlgren, T. J., L. D. Schell, and B. Vanden Heuvel. 1999. How grazing and soil quality affect native and exotic plant diversity in Rocky Mountain Grasslands. *Ecological Applications* 9:45-64.

Talle, A. 1988. *Women at a loss: Change in Maasai pastoralism and their effects on gender relations*: University of Stockholm.

Terborgh, J., and C. van Schaik. 2002. "Why the world needs parks," in *Making parks work: Strategies for preserving tropical nature*. Edited by J. Terborgh, C. van Schaik, L. Davenport, and M. Rao. Washington, D.C.: Island Press.

Terborgh, J., C. van Schaik, L. Davenport, and M. Rao. Editors. 2002. *Making parks work: Strategies for preserving tropical nature*. Washington, D.C.: Island Press.

Thompson, L., E. Mosley-Thompson, H. Brecher, M. Davis, B. Leon, D. Les, P. Lin, T. Mashiotta, and K. Mountain. 2006. Abrupt tropical climate change: Past and present. *Proceedings of the National Academy of Sciences of the United States of America* 103:10536-10543.

TMI. 1996a. "Evaluacion rapida ecologica del Cordillera Huayhuash." Huaraz, Peru: The Mountain Institute (TMI): Andes Program,.

— 1996b. "Plan de uso y recreativo del Parque Nacional Huascaran." Lima, Peru: The Mountain Institute (TMI): Andes Program,.

— 1997. "Plan estrategico: El Programa Andino del Instituto de Montana 1997-2000." Huaraz, Peru: The Mountain Institute (TMI): Andes Program,.

— 2001a. "Fortalecimiento de la capacidad local para el desarrollo y conservacion integrada de las montañas de la Reserve del Biosfera Huascaran." Huaraz, Peru: The Mountain Institute (TMI): Andes Program,.

— 2001b. "PICD "Ke shumaq pasto kuyaquipaq": Comunidad campesina Tupac Yupanqui, sector Collón." Huaraz, Peru: The Mountain Institute (TMI): Andes Program,.

Tohan, A. 2000. Polylepis forests of the Andes: A socio-environmental case study from the Huascaran Biosphere Reserve, Ancash, Peru.Thesis, University of Wisconsin.

Tovar, O., and L. Oscanoa. 2002. *Guia para la identificacion de pastos naturales alto andinos de mayor importancia ganaderia*. Huaraz, Peru: The Mountain Institute (TMI).

Trawick, P. B. 2001. Successfully governing the commons: Principles of social organization in an Andean irrigation system. *Human Ecology* 29:1-25.

Troll, C. 1968. "The Cordilleras of the tropical Americas: Aspects of climatic, phytogeographical, and agrarian ecology." *1966 UNESCO (Mexico) Symposium: Geo-ecology of the mountainous regions of the tropical Americas, Bonn, 1968*, pp. 15-56.

Tucker, B. 2007. Applying behavioral ecology and behavioral economics to conservation and development planning: Example from the Mikea Forest, Madagascar. *Human Nature* 18.

Tucker, B., and A. Young. 2005. "Growing up Mikea: Children's time allocation to tuber foraging in southwestern Madagascar," in *Hunter-Gatherer Childhoods*. Edited by B. Hewlett and M. Somerset, pp. 147-171. New Jersey: Transaction Publishers.

Turin, F. C. 2001. Estudio de las Plantas Medicinales en la Reserve de Biosfera Huascaran (Quebrada Llaca-Quebrada Ishinca). Thesis, Universidad Nacional Mayor de San Marcos.

Turner, F. C. 1976. The rush to the cities in Latin America. *Science* 192:955-962.

Turner, M. D. 1998. "The interaction of grazing history with rainfall and its influence on annual rangeland dynamics in the Sahel," in *Nature's geography: New lessons for conservation in developing countries*. Edited by K. Zimmerer and K. Young, pp. 237-261. Madison: University of Wisconsin Press.

—. 1999. Labor process and the environment: The effects of labor availability and compensation on the quality of herding in the Sahel. *Human Ecology* 27:267-296.

—. 2006. "Shifting scales, lines, and lives: The politics of conservation science and development in the Sahel," in *Globalization and the new geographies of conservation*. Edited by K. Zimmerer, pp. 166-185. Chicago: University of Chicago Press.

Urban, D., and T. Lookingbill. 2005. A simple method for estimating potential relative radiation (PRR) for landscape-scale vegetation. *Landcape Ecology* 20:137-147.

Urban, D., C. Miller, N. Stephenson, and P. Halpin. 2000. Forest pattern in Sierran landscapes: The physical template. *Landcape Ecology* 15:603-620.

Valdivia, C., E. Dunn, and C. Jette. 1996. Diversification as a risk management strategy in an Andean agropastoral community. *American Journal of Agricultural Economics* 78:1329-1334.

Van de Koppel, J., M. Rietkerk, and F. Weissing. 1997. Catastrophic vegetation shifts and soil degradation in terrestrial grazing systems. *Trends in Ecology and Evolution* 12:352-356.

van Es, G. 2002. "The Destination Analysis: Callejon de Huaylas." Huaraz, Peru: The Mountain Institute: Andes Program.

van Schaik, C., and H. Rijksen. 2002. "Integrated conservation and development projects: Problems and potential," in *Making parks work: Strategies for preserving tropical nature*. Edited by J. Terborgh, C. van Schaik, L. Davenport, and M. Rao. Washington: Island Press.

Velasco, J. 2005. "The master will no longer feed off your poverty," in *The Peru Reader: History, culture, politics*, 2nd edition, *The Latin American Readers*. Edited by O. Starn, C. Degragori, and R. Kirk. Durham: Duke University Press.

Verbeke, G., and G. Molenberghs. 2000. *Linear mixed models for longitudinal data*. New York: Springer.

Walker, P. A. 2006. Political ecology: Where is the policy? *Progress in Human Geography* 30:382-395.

Whipple, G. M. 1907. A quick method for determining the index of correlation. *The American Journal of Psychology* 18:322-325.

Whittaker, R. H. 1960. Vegetation of the Siskiyou Mountains, Oregon and California. *Ecological Monographs* 30:279-338.

Wiens, J. A. 1989. Spatial scaling in ecology. *Functional Ecology* 3:385-397.

Wilcox, B. P., F. C. Bryant, and V. F. Belaun. 1987. An evaluation of range condition on one range site in the Andes of Central Peru. *Journal of Range Management* 40:41-45.

Wilshusen, P. R., S. R. Brechin, C. L. Fortwangler, and P. C. West. 2002. Reinventing a square wheel: Critique of a resurgent "protection paradigm" in international biodiversity conservation. *Society & Natural Resources* 15:17-40.

Winterhalder, B. 1983. Opportunity-cost foraging models for stationary and mobile predators. *American Naturalist* 122:73-84.

—. 1990. "Open field, common pot: Harvest variability and risk avoidance in agricultural and foraging societies," in *Risk and Uncertainty in Tribal and Peasant Economics*. Edited by E. Cashdan, pp. 67-87. Boulder: Westview Press.

—. 2002a. Behavioral and other human ecologies: Critique, response and progress through criticism. *Journal of Ecological Anthropology* 6:4-23.

—. 2002b. "Models," in *Darwin and Archaeology: A handbook of key concepts*. Edited by J. P. Hart and J. E. Terrell, pp. 201-223. Westport: Bergin and Garvey.

Winterhalder, B., C. Goland, M. Graham, A. Larme, and J. Recharte. 2007. "Time allocation and activity patterns in two communities on the Andean escarpment (Dept. of Puno, Peru)." Davis: University of California.

Winterhalder, B., R. Larsen, and R. B. Thomas. 1974. Dung as an essential resource in a highland Peruvian community. *Human Ecology* 2:89-104.

Winterhalder, B., and F. Lu. 1997. A forager resource population ecology model and implications for indigenous conservation. *Conservation Biology* 11:1354-1364.

Winterhalder, B., and E. Smith. 2000. Analyzing adaptive strategies: Human behavioral ecology at twenty-five. *Evolutionary Anthropology* 9:51-72.

Winterhalder, B., and R. B. Thomas. 1978. "Geoecology of southern highland Peru: A human adaptation perspective " in *Occasional Paper No. 27*. University of Colorado, Boulder: Institute of Arctic and Alpine Research.

Wolock, D. M., and G. J. McCabe. 1995. Comparison of single and multiple flow direction algorithms for computing topographic parameters in topmodel. *Water Resources Research* 31:1315-1324.

Young, E. H. 2001. State intervention and retreat in the abuse of the commons: The case of Mexico's fisheries in Baja California Sur. *Annals of the Association of American Geographers* 91:283-306.

Young, K. R. 1993. National-park protection in relation to the ecological zonation of a neighboring human community - An example from Northern Peru. *Mountain Research and Development* 13:267-280.

—. 1998. "Deforestation in landscapes with humid forests in the Central Andes: Patterns and process," in *Nature's geography: New lessons for conservation in developing countries*. Edited by K. Zimmerer and K. Young, pp. 75-99. Madison: University of Wisconsin.

Young, K. R., and B. León. 1993. "Connectivity, social actors, and conservation policies in the Central Andes: The case of Peru's montane forests," in *Biodiversity and Conservation of Neotropical Montane Forests*. Edited by S. P. Churchill, E. H. Balslev, and J. L. Luteyn. Bronx, NY: New York Botanical Garden.

Young, K. R., and J. K. Lipton. 2006. Adaptive governance and climate change in the tropical highlands of western South America. *Climatic Change* 78:63-102.

Zeidler, J., S. Hanrahan, and M. Scholes. 2002. Land-use intensity affects range condition in arid to semi-arid Namibia. *Journal of Arid Environments* 52:389-403.

Zimmerer, K. S. 1991. Wetland production and smallholder persistence: Agricultural change in a highland Peruvian region. *Annals of the Association of American Geographers* 81:443-463.

—. 1993. Soil-erosion and labor shortages in the Andes with special reference to Bolivia, 1953-91: Implications for conservation-with-development. *World Development* 21:1659-1675.

—. 1996. *Changing fortunes: Biodiversity and peasant livelihood in the Peruvian Andes*. Berkeley: University of California Press.

—. 2000. The reworking of conservation geographies: Nonequilibrium landscapes and nature-society hybrids. *Annals of the Association of American Geographers* 90:356-369.

—. 2002. Common field agriculture as a cultural landscape of Latin America: Development and history in the geographical customs of land use. *Journal of Cultural Geography* 19:37-63.

—. 2003. "Environmental zonation and mountain agriculture in Peru and Bolivia," in *Political Ecology: An integrative approach to geography and environment-development studies*. Edited by K. S. Zimmerer and T. J. Bassett, pp. 137-158. New York: The Guilford Press.

—. 2004. Cultural ecology: Placing households in human-environment studies: The case of tropical forest transitions and agrobiodiversity change. *Progress in Human Geography* 28:795-806.

—. Editor. 2006. *Globalization and new geographies of conservation*. Chicago: University of Chicago Press.

Zimmerer, K. S., and T. J. Bassett. 2003. "Approaching political ecology: Society, nature, and scale in human-environment studies," in *Political Ecology: An Integrative Approach to Geography and Environment-Development Studies*. Edited by K. S. Zimmerer, and T. J. Bassett, editors, pp. 1-28. New York: Guilford Press.

Zimmerer, K. S., R. E. Galt, and M. V. Buck. 2004. Globalization and multi-spatial trends in the coverage of protected-area conservation (1980-2000). *Ambio* 33:520-529.

Zimmerer, K. S., and K. R. Young. Editors. 1998. *Nature's geography: New lessons for conservation in developing countries*. Madison: University of Wisconsin Press.