UNPACKING THE EFFICACY OF COMMUNITY-BASED WILDLIFE GOVERNANCE: THE INFLUENCE OF ECONOMIC BENEFIT TYPES, RISK, AND HETEROGENEITY ON COLLECTIVE ACTION

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December 17, 2019
This dissertation is dedicated to my wife, Roopa, my kids, Anand and Eashan, my parents, Joyce and Bill, and my mother-in-law, Mala. Your support has made all the difference in the world. I could not have done this without you.
ACKNOWLEDGEMENTS

I thank all the individuals who provided help with and feedback on the various portions of this dissertation, including but not limited to Drs. Lauren MacLean, Bill Blomquist, Jennifer Brass, David Konisky, and Vicky Meretsky. I also thank the Wildize Foundation, Ostrom Workshop, and Indiana University’s Office of Sustainability for providing financial support for my research.
PREFACE

Six years ago, I left a partner-track position at a law school to tilt at windmills as a Ph.D.
student. From a financial standpoint, moving from a lawyer’s salary to a graduate stipend was
indefensible, particularly given that I was, at the time, the sole breadwinner for my family of
four. My mental state, however, demanded the change.

I entered the practice of law with the goal of the saving the world (or, more realistically,
at least contributing to its salvation) by practicing as an environmental lawyer “for the side of
good.” The crushing realities of law school debt, however, instead required that I first focus on
paying off my loans. So, I entered into the world of “Big Law,” ensconced in a swanky office
that looked over Washington D.C.’s K Street and, if I contorted my neck just right, a view of the
National Cathedral. It was in this office that I worked endless hours and ate almost all of my
lunches, many of my dinners, and oftentimes my breakfasts.

I had discovered early on, as a second-year summer associate, that I couldn’t countenance
working for the firm’s environmental law clients, many of whom were challenging provisions
and/or governing interpretations of core environmental protection statutes. I would be obliged as
their counsel to zealously advocate on their behalf, all the while secretly hoping that their
arguments would fail – a position that would be fair to neither the client nor myself.

Consequently, I chose to work as commercial litigator, with the plan that I would hone my
litigation chops and then jump to the world of nonprofit environmental advocacy after my
student loans were paid.

My first full year as a practicing attorney saw three key developments. The most
significant development was my marrying my wife, Roopa. She is an amazing person and
marrying her is far and away one of the best things that has ever happened to me. The other two
developments were less enjoyable and consisted of (a) slaving under a petty tyrant of a law partner, and (b) the economic crash of 2007. The former development sucked nearly all the joy out of my life and caused me to flee to warmer climes in Tampa, where I worked for two other law firms. Each of those was better than the last, but none provided me with the intrinsic satisfaction necessary to put up with the near constant drudgery, conflict, and panic associated with the world of litigation. The latter development meant that, when I was financially able to make the jump to environmental litigation, most of the nonprofit environmental law firms had gone belly-up, and paying litigation jobs on behalf of the environment were virtually nonexistent.

Faced with little prospect of career satisfaction in the arena of law, I made the fateful decision to, once again, return to school. However, I did not make this decision in a vacuum, and the switch would not have been possible without the support of my family. Roopa, herself a former practicing attorney, understood why I needed the change and, to her immeasurable credit, gave up a rewarding life as a full-time mom to take up the mantle as the family’s primary breadwinner. She completely reinvented herself professionally, eschewing the practice of law and instead focusing on optimizing the user experience for a range of e-commerce businesses. She possesses an uncanny ability to assess and to innovate to meet the needs of her company and its clients. Roopa has made remarkable career progress in a short period of time, and I stand in awe of her ability and work ethic. I could not do what she does. She is also a wonderful mother, a loving partner, and the bedrock of my family. I am incredibly fortunate to be with her.

My parents, having seen me go through the highs and lows (mostly the latter) of life as an attorney, were also fully supportive of my career switch, as they were of my decision a decade earlier to give up being a track and field coach to go study law. Over the years, they have
provided critical emotional and logistical support. They have spent a multitude of hours helping with two long moves (Florida to Indiana and back again) and visiting Bloomington to help cook, do laundry, and look after my kids. Throughout my various career paths, they have been my consistent cheerleaders, never wavering in their belief in my abilities. They are kind and generous and role models for how good parents should be.

My kids, Anand and Eashan, have also shown an unwavering faith in me. They are too young to fully understand the import of a Ph.D., but that does not stop them from expressing their consistent optimism regarding my academic and career prospects. They serve as an antidote against my tendency to downplay my achievements, constantly reminding that self-effacing comments should be avoided, even if they are presented in the form of a joke. While kids (along with dogs) are not necessarily conducive to the timely completion of a dissertation, they do serve as reminder that even more important priorities exist. Spending time with them provides a much-needed counterbalance in my life.

I have heard horror stories about mothers-in-law, but I am fortunate to have never experienced it firsthand. My mother-in-law has welcomed me from day one, and constantly tells me how lucky she is to have me in the family, and I feel the same way about her. She has always been willing to help out whenever and however she can, including dropping everything else in her life to come out and stay with us when I had a bizarre muscular reaction to medication and spent a week without the strength to even lift a plate. Other people might have questioned the sanity of their son-in-law giving up a promising law career to go back to school. She did not, and her approval made the experience that much easier.

Finally, my sisters, Erica and Claire, have also shown a high degree of interest in and support for my academic pursuits. Erica, in particular, has stated several times that I am an
inspiration – a statement that I find equal parts baffling and validating. Both are more than happy to ask about my research and to listen to my ramblings about it. Their interest is always appreciated when I am dealing with a world (academia) that seems peculiarly well designed to generate self-doubt.

Overall, my life as a doctoral student has been as rewarding as I had hoped, albeit more stressful than anticipated. I have met many interesting students that are working on truly fascinating projects. I have also been fortunate enough to have had a cadre of brilliant and caring professors, some of whom I want to mention here in addition to their brief, formal recognition in the preceding Acknowledgement section.

I first met Lauren MacLean when I showed up unannounced at her office at the Ostrom Workshop. I presume that she was using that office because she specifically wanted to avoid being disturbed by students, and I know for a fact that she was already stretched thin in terms of the number of people that she mentored. Nevertheless, Lauren graciously took the time to talk with me then, and she spent more time over the subsequent years serving as an informal sage. Among other notable investments of energy, she accommodated my need for an additional comparative course by agreeing to serve the professor of record for an independent reading course (her first and, I expect, last such course), and to chair my dissertation committee. Lauren has also gone out of her way to provide me with research opportunities and economic support as I remotely worked on my dissertation. She prods and pokes at my proposed research approaches and work product while, at the same time, providing moral support for my efforts. Lauren is a caring and thoughtful person who can sharpen her knives with the best of them, and she has made me a better scholar.
I came across Bill Blomquist in his polycentricity course at the Ostrom Workshop. As an Elinor Ostrom acolyte, Bill was exactly the type of professor to I traveled to Bloomington to meet (his residency at IUPUI notwithstanding). Despite having been steeped in the topic of polycentricity for decades, he was nevertheless still receptive to the borderline inane musings of a second year Ph.D. student. I have never seen him in anything other than good spirits, with a ready laugh and a willingness to chat, even when facing a long drive back to Indianapolis. He has also always been willing to help or provide feedback whenever needed.

Saying that I dropped into Jen Brass’ lap would be an understatement – I flung myself. Prior to asking her to be on my committee, I am not sure if I had more than 10 total minutes of interaction with her. Yet, despite our conspicuous lack of shared history, and armed only with the knowledge that I was interested in conducting field work in Africa and that Lauren was chairing my committee, Jen agreed. Since then, I have found her to be an invaluable source of help (with a great sense of humor). She led a discussion group where her graduate students could submit their work product to friendly fire before sending it out to be seen by less forgiving audiences. She is quick to offer logistical and emotional support (on a couple of occasions even from a hospital bed) despite a multitude of other time commitments arising from her candidacy for tenure requirements and her duties as a mom.

David Konisky has a wonderful view from the deck of his vacation home. I know this because he interrupted his enjoyment of that view to Skype with me in order to calm me down after I had received particularly vituperative comments from a reviewer. Nor was that the only time that David has responded to a frantic email from me as I wandered through the wilderness of developing my dissertation. Each time, he has thoughtfully and patiently talked things through while I grapple with how little even Ph.D. level statistics courses prepared me to handle the sort
of real-world data that I encountered. David was also considerate enough to bring me into an unexpected research project on the killing of Cecil the Lion. He became involved in that project by pure happenstance, and I am sure it represents an inconsequential sidetrack from his regular, well-regarded body of research. To me, however, it afforded me the critical opportunity to engage with other scholars examining wildlife conservation and human-wildlife interactions.

The approximately ten minutes of time that I spent with Jen Brass prior to asking her to serve on my committee is a lifetime compared to my prior interactions with Vicky Meretsky. If my memory serves correctly, I simply stepped into her office to ask her if she would be interested in joining my committee. My only prior interaction with Vicky was during the orientation week of my first year at IU, where she led a one-time class designed to prevent incoming doctoral students from accidentally committing plagiarism. That class didn’t start well, given that the entire group of us erroneously showed up outside a different classroom, apparently giving Vicky the impression that we had all decided to blow it off. I have since learned that Vicky has a wonderful intensity and keen sense of humor, but I suspect the misunderstanding that day led to only the former being on display. I found her intimidating enough that day that I nearly did not ask her to be on my committee, and I was persuaded otherwise only by her interdisciplinary interest in law, policy, and wildlife conservation. To my very pleasant surprise, Vicky readily agreed to be on my committee and turned out to anything but the daunting individual that I remembered from my first week. She is warm, open, and supportive and has readily used substantial chunks of her own time to help.

Dan Cole was a regular at the Ostrom Workshop during my first three years at IU. While Elinor Ostrom unfortunately passed away prior to my arrival, Dan was one of the institution’s true believers, looking to ensure the survival of its research and unique ethos. Dan was more than
willing to invest time in students who showed a genuine interest in the Workshop’s teachings, and our shared interest was augmented by the fact that we were both attorneys (although Dan had the good sense to never enter private practice). A logjam of professors prevented me from asking him to serve on my committee, as a committee of six would have been untenable. Nevertheless, Dan has consistently gone out of his way to support me when and how he can. He approached me about writing a research paper on Kenyan water law, allowing me to generate an early academic publication, and has readily offered his help in scouting the academic job market.

To my professors, I am profoundly grateful for all your support. To my family, I love you the moon and back, or to my receipt of a Ph.D., whichever proves to be the longer journey. Throughout this journey, I have never walked alone. I still harbor aspirations of saving the world, and I look forward to what comes next.
UNPACKING THE EFFICACY OF COMMUNITY-BASED WILDLIFE GOVERNANCE: THE INFLUENCE OF ECONOMIC BENEFIT TYPES, RISK, AND HETEROGENEITY ON COLLECTIVE ACTION

The world is in the midst of an extinction crisis, and policymakers grapple with how to best facilitate wildlife conservation. Traditionally, wildlife conservation has used a “fortress” approach, consisting of the nationalization of wildlife and the creation of strict formal protected areas. However, over the past several decades, concerns over the efficacy, cost, and fairness of the fortress model approach have led to the adoption, particularly in developing countries, of community-based natural resource management (CBNRM). In this dissertation, I examine two core issues: (1) the efficacy of the CBNRM approach, and (2) the factors impacting the approach’s effectiveness.

From these two core issues, I use a mixed-methods research design to develop and examine three specific, related inquiries. I use a large-N analysis to compare the effectiveness of CBNRM and governmental approaches at protecting the African elephant. I utilize data from a field survey in four CBNRM areas in Namibia’s northwest Kunene region to measure the relationship between different types of wildlife-generated benefits received by residents and their opinions of whether wildlife has improved their lives. Finally, together with a coauthor, I conduct and use a behavioral laboratory experiment to analyze the impact of economic and risk heterogeneities, and their interaction, on collective action where users face a risk of loss resulting from human-wildlife conflict.

I determine that both the national and CBNRM governance approaches are associated with a range of elephant protection outcomes, and that neither is clearly more efficacious. Based
on analyzing data from my field survey, I find that different types of wildlife-generated economic benefits have markedly different associations with whether CBNRM residents feel that wildlife has benefitted their lives. My behavioral experiment data suggest that risk and economic heterogeneities both undermine collective action, elevate risk levels significantly decrease individual cooperation, and that relative wealth levels can markedly influences individual risk tolerance. Tying together these three strands of research, I contend that CBNRM participants should be viewed as dynamic entities, with needs and preferences that shift in response to psychological and external factors.

Lauren MacLean, Ph.D.

William Blomquist, Ph.D.

Jennifer Brass, Ph.D.

David Konisky, Ph.D.

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# TABLE OF CONTENTS

## INTRODUCTION ........................................................................................................................................................................1
- The Intractable Problem of Wildlife Conservation ..........................................................................................................................1
- Common Pool Resource Governance as a “Wicked Problem” .............................................................................................................3
- Wildlife Conservation as a “Wickeder” Problem? .............................................................................................................................5
- Hardin’s Dichotomous Solution to the CPR Problem and the Rise of Community-Based Wildlife Governance ........................................8
- The Debate over Community-Based Governance of Wildlife ........................................................................................................11
- The Focus of this Dissertation: Evaluating and Understanding the Efficacy of CBNRM as a Wildlife Conservation Tool ..............12

## Research Design and Study Sites .................................................................................................................................................13

## Organization of Dissertation ...........................................................................................................................................................17

## Chapter Findings and Synthesis .....................................................................................................................................................18
- The Overall Effectiveness of the CBNRM Approach ..........................................................................................................................18
- The Role of Different Benefit Types on Residents’ Perceptions of the Impact of Wildlife on their Lives ............................................19
- The Influence of Risk and Economic Heterogeneities on Collective Action ..................................................................................20
- Synthesis of Findings ........................................................................................................................................................................21

## CHAPTER 1. The Rise of CBNRM in Namibia: Exploring the Approach’s Theory, Development in Namibia, and Unresolved Questions ..................................................................................................................24

- CBNRM: Devolution of Control and Realization of Benefits ............................................................................................................25
- The Development of Namibia’s CBNRM Program: From Tribes to Conservancies ........................................................................26

## Pre-Colonial Governance in Namibia and Germany ..........................................................................................................................28
- Namibia: A patchwork system of governance and the impact of rinderpest .......................................................................................28
- Germany: From royal hunting to the conservation of nature ..............................................................................................................31

## 1884-1915: The Creation of a German Colony in Namibia and the Systematic Disenfranchisement of Native Namibians ..................33
- Land policy: Confiscation for white settlers ......................................................................................................................................35
- Environmental policy: Justifying indigenous disenfranchisement ..................................................................................................38

Land policy: A continuation of Germany’s policy of dispossession and relocation, and the development of traditional authorities ........................................41

Environmental policy: The increased consolidation of governmental control over wildlife and the beginnings of CBNRM ...............................................44

Namibian Independence (1990-present): The Adoption of Community-Based Conservation as a Conservation Tool and Redress for Inequality ........................................48

Land policy: The continuation of communally owned property and the codification of land allocation rights of traditional authorities ........................................48

Environmental policy: The statutory creation of conservancies and the involvement of traditional authorities in conservancy functions .....................................51

Structural Challenges to the Success of the Conservancy Program ........................................54

Common Criticisms of the CBNRM Approach and Formation of Specific Research Questions ............................................................................................................58


Background ........................................................................................................................................61

A Brief Review of the Protectionism versus Sustainable Use Debate .............................................61

Existing Comparisons of the Efficacy of the Two Approaches ......................................................63

Design and Methods ..........................................................................................................................65

Variable Identification and Selection ................................................................................................65

Proportion of illegal elephant kills as the outcome measure ..........................................................65

Governance approach as the primary explanatory variable of interest ................................................68

Identification and selection of other explanatory variables ...............................................................69

Site Selection ......................................................................................................................................70

Method of Analysis ............................................................................................................................72

Results .............................................................................................................................................72

Discussion .........................................................................................................................................75

Conclusion ..........................................................................................................................................79
Price of ivory .................................................................................................................... 211

APPENDIX 2 (CHAPTER 3) .................................................................................................... 213
Supplemental Field Research Information ................................................................. 213
Full Survey ....................................................................................................................... 214

APPENDIX 3 (CHAPTER 4) .................................................................................................... 231
Experiment Instructions ................................................................................................. 231

CURRICULUM VITAE .................................................................................................................
INTRODUCTION

1. The Intractable Problem of Wildlife Conservation

Multiple recent analyses highlight the deleterious effects that humanity has had on wildlife species. Ceballos et al. (2015:5) determine that, even using a conservative estimate of current extinction rates, humans are causing a “mass extinction episode unparalleled for 65 million years.” A 2016 report finds that worldwide vertebrate populations declined by 58% between 1970 and 2012 and the level of decline may reach 67% by 2020 (WWF 2016:44). Habitat loss and unsustainable harvesting are two of the primary drivers of this loss. Poaching and encroachment, for example, have reduced global elephant numbers from approximately 26 million in the 1500s to around 415,000 today (GEC 2016). Poaching alone is thought to have reduced the population of African savanna elephants by 30 percent between 2007 and 2014, and the population is estimated to be decreasing by a further 8% per year (ibid).

Why should we care about species loss? As an initial matter, there are moral implications associated with anthropogenic extinctions. In his seminal article introducing the field of conservation biology, Soulé (1985:731) argues that “species have value in themselves, a value neither conferred nor revocable, but springing from a species' long evolutionary heritage and potential or even from the mere fact of its existence.” Cafaro and Primack (2014:2) echo this ethos, writing “fairly sharing the lands and waters of Earth with other species is most importantly a matter of justice, not economic convenience.”

Setting aside moral concerns, species loss can also have significant local and global implications (Ehrlich 1990). Megaherbivores (such as elephants and rhinos) play an important role in altering habitat and, in turn, providing a diversity of ecosystems (Owen-Smith 1987, Owen-Smith 1989, Ripple et al. 2016). The loss of such “keystone” species, along with resultant ecosystem homogenization, may result in the subsequent extinction of multiple smaller species
(Owen-Smith 1987, Owen-Smith 1989). Indeed, Owen-Smith (1987 and 1989) theorizes that the extinction of megaherbivores throughout North and South America, Europe, Asia and Australia approximately 11,000-15,000 years ago resulted in a 60% reduction (40% in Australia) in smaller large mammal species.

These sorts of widespread extinctions can have significant economic repercussions. May (2011:1) notes that biodiversity “underpins ecosystem services that – although not counted in conventional GDP – humanity is dependent upon.” Plant and insect species confer wide ranging economic benefits, including carbon sequestration and water filtration and crop fertilization (Losey and Vaughan 2006). Additionally, approximately 1 billion people rely on wild meat for subsistence, and more than 3 billion get at least 20% of their animal protein from fish species (Ripple et al. 2015:7, WWF 2016:55).

Finally, individual wildlife species can themselves have significant economic value. For instance, elephant poaching costs African countries a combined annual total of approximately $25 million in tourism income (Naidoo et al. 2016:3). In 2008, dolphin and whale watching contributed $2.1 billion to the global economy and supported 19,000 jobs (Mustika et al. 2012:11). In addition to their role in insect suppression, Mexican free-tailed bats generate more than $6.5 million in tourism income across the Southwestern US (Bagstad and Wiederholt 2013:309). And, a 2006 study estimated the total tourism value of Botswana’s Okavango Delta (consumptive, non-consumptive, and willingness of tourists to pay additional money for conservation) was nearly $17.5 million (Mmopelwa and Blignaut 2006:120-124).

Despite its moral and economic justifications, wildlife conservation has proven to be a difficult undertaking in practice. As discussed in greater length below, wildlife is a common-pool resource which (as with all other types of common-pool resources) is inherently difficult to
govern. In addition to that challenge, wildlife users must also contend with the presence of HWC, which compounds the challenges associated with conserving wildlife.

2. **Common-Pool Resource Governance as a “Wicked Problem”**

   Natural resources are often described as presenting particularly “wicked” governance challenges. They have a limited capacity for regeneration, meaning that extraction of the resource above a certain level will cause a decline in its future availability.\(^1\) Yet, stakeholders often exceed this level of harvesting, continuing to over-extract resources even in the face of their obvious decline, and the ramifications for the resource’s potential collapse on their own well-being. Recent examples of this sort of mismanagement of natural resources include the commercial extinction of the Northwest Atlantic cod (Steele, et al. 1992), the functional extinction of the northern white rhino\(^2\) and, of course, global climate change.

   The difficulty of governing these sorts of natural resources, generally described as common pool resources (CPRs), is widely attributed to two fundamental characteristics: they are rival and non-excludable (Ostrom, et al. 1994; Ostrom and Ostrom 1999; Epstein, et al. 2014). CPRs are rival in that the use by an individual of one unit of the resource precludes that unit’s use by anyone else. So, for example, when a fisherman catches a fish or a lumberjack fells a tree, that fish or tree is no longer available to be harvested by another user of the resource.

   CPRs are described as non-excludable because it can be impossible, or at least impracticable, to prevent harvest by individuals that have access to the resource (ibid). To return to the previous example of resource harvesting by fishermen and lumberjacks, imagine a bay or a

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\(^1\) These resources can include harvestable units (such as wildlife, fish, forests, or water for irrigation or drinking) or the capacity of ecosystems to absorb anthropogenic externalities without degrading (for example, the capacity of rivers to absorb pollution without becoming unpotable or the atmosphere and oceans to absorb carbon dioxide without the world experiencing a significant rise in temperatures).

\(^2\) At the time of this writing, the only two known living animals are both female, although stored DNA has been used to create embryos that are currently frozen pending implantation in a southern rhino host mother (Gilliland 2019).
forest that has people living on its boundaries. Each of the individuals bordering the resource has the capacity to enter and harvest the resource, and it would be exceedingly difficult to stop them (even if it was possible to continuously and effectively patrol the boundaries of the bay/forest, doing so would be extremely resource intensive). Of course, the larger the boundaries of the CPR, the greater the difficulty associated with excluding potential resource users. The governance of global climate presents the thorniest non-excludability problem of them all given that the entire world’s population is able to access it to produce greenhouse gases (i.e., “harvesting” its capacity to absorb those gases without changing) (Stern 2011).

Because CPRs are reliant on the cooperative governance of multiple resource users (whether individuals or jurisdictional units such as states or countries), each user must make the individual choice whether to cooperate in the sustainable use of the resource or to act selfishly in a way that undermines collective action. Hardin (1968) famously described the motivation for CPR users to act selfishly. He writes that the resource user knows that overharvesting of the resource will result in loss of that resource’s productivity. However, the harm associated with that productivity loss is shared among all of the resource users, whereas the user individually gains all of the benefits associated with that activity (i.e., the economic value associated with harvesting a unit of the wildlife, fish, forest, or climate resource). Each time the user considers whether to an additional unit, she is faced with this same set of incentives: she receives all of the benefits of that action and incurs only a portion of the resulting harm (ibid). Of course, once the maximum sustainable level of harvesting is reached, each of the other resource users face the same choice, with each facing compelling incentives to harvest additional units of the resource.

Compounding these users’ decision-making is their knowledge that each of the other users face the same incentive to ask selfishly, and presumably will make the rational decision to
continue harvesting (Hardin 1968). This knowledge further incentivizes the resource users to continue harvesting because, if they do not, they will have shared in all of the collective harm inflicted by the selfish decisions of the other resource users without themselves experiencing any of the individual benefits associated with overharvesting (ibid).

3. **Wildlife Conservation as a “Wickeder” Problem?**

CPR studies have analyzed a wide range of resources. For instance, in her seminal book *Governing the Commons*, Ostrom (1990) analyzed case studies involving alpine meadow livestock grazing, inshore fisheries, California groundwater basins, Japanese communal lands, and Spanish and Philippine irrigation systems. Other resources studied include community forests (Varughese and Ostrom 2001; Agrawal 2012), lobster, salmon, herring, and cod fisheries (Schlager and Ostrom 1992), information (Hess and Ostrom 2003), renewable energy (Wolsink 2012), and global environmental issues (Stern 2011).

Yet, despite the heterogeneity of those resources, they all have one key commonality: the governed resource is understood to confer an unequivocal benefit upon its users. Forests provide fuel for households, forage for livestock, and wood products for market. Fisheries of all types provide food and saleable commodities. Groundwater basins and irrigation systems provide water for personal, industrial, pastoral, and agricultural consumption. Renewable energy and information provide the basis for economic innovation and advancement. And, the prevention or mitigation of global environmental problems, such as global warming, can ward off disastrous impacts on global and local economies.

Because they provide an unambiguous benefit, an increase in available amount of these resources (referred to in this paper as “traditional CPRs”) will result in a corresponding increase in the benefits they confer on their users. Indeed, it is difficult to imagine a realistic scenario
where, for example, cod fishermen, forest users, or renewable energy consumers could be
harmed by a dramatic improvement in their respective resource stocks. Thus, users of traditional
CPRs will presumably desire as robust a resource pool as possible so that they can achieve the
largest possible harvest.

The corollary to the demand by users for a healthy stock of resource is that the users of
traditional CPRs are generally harmed by a significant decrease in the resource pool, because
that decrease will result in a lower (or perhaps nonexistent) present and/or future harvest.
Scarcity thus requires either that the users (a) ration and monitor consumption to preserve or
restore the resource, such as in the CPRs studied by Ostrom (1990) and others, or (b) risk the sort
of catastrophic collapse of the CPR prophesied by Hardin (1968).

The relationship between resource and users is much less straightforward when the
resource is wildlife. Wildlife, especially smaller game species such as deer, bushpig, rabbit, and
the like, can provide supplemental income in the form of meat for consumption or trade. On the
other hand, wildlife of all types can inflict substantial personal or economic harm on rural
populations.

Most rural populations rely on some form of agriculture or pastoralism for their primary
impact such populations in a multitude of ways, including livestock predation, crop-raiding,
destruction of food stores, attacks on humans, disease transmission (to both crops and humans),
and foregone economic or lifestyle choices due to either the presence of wildlife or restrictions

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3 For the harvesting of some living natural resources, such as forests or fisheries, the optimum level of that resource
may sometimes be below the resource’s naturally occurring equilibrium point. In those circumstances, the reduction
in population may decrease intra-species competition for resources, resulting in a more rapid regeneration of the
resource. For example, research suggests that olive ridley turtle recruitment may increase when nesting occurs at
lower densities (Honarvar et al. 2008).
related to conservation areas. Large carnivores are particularly prone to be involved in HWC because of their expansive home ranges and their dietary requirements (Kabir, et al. 2014).

For subsistence and small-scale farmers and pastoralists, the economic losses resulting from HWC can be devastating. Lamarque, et al. (2009) write that, for some of these residents, “losses to wildlife can mean the difference between economic independence and dire poverty.” And, given that the official records of HWC reflect only overt human-wildlife conflict, and not more passive conflict resulting from disease transmission (such as rabies, anthrax, foot-and-mouth disease) or impact on livelihood choices (such as the loss of grazing or foraging territory resulting from the threat of overt wildlife conflict or the loss of schooling opportunities by children tasked with guarding crops), the true economic impact of human-wildlife conflict is likely even greater (Barua, et al. 2013).

The presence of HWC in these rural areas means that, while residents face the same core set of incentives in managing their wildlife as would be found in traditional CPRs (i.e., they must choose whether to harvest an additional animal for their own personal benefit despite the resulting harm to the resource), they also risk suffering a loss if they do not overharvest the resource. Even in areas where wildlife has the potential to generate significant income, such as through the ecotourism and/or trophy-hunting industries, the threat that wildlife poses to their lives and livelihoods means that residents can have a strong incentive to engage in the preventative killing of wildlife (Pienaar, et al. 2013).

In short, the risk of harm places an additional “thumb” on the proverbial scale when an individual is determining whether to overharvest a wildlife resource. In addition to the economic gains a user might realize in harvesting an additional animal, that user must also consider the increased likelihood of harm that she might suffer if she does not. It is widely accepted in
psychology and social sciences literature that people tend to be risk adverse, reacting much more strongly to the prospect of a loss than they do to the potential for a gain (Kahneman & Taversky 1979; Zhang, et al. 2014; Walasek & Stewart 2019). The threat of loss (in addition to the standard CPR motivation of harvesting wildlife for personal gain) can provide rural residents a strong incentive to kill wildlife above a sustainable level. This additional incentive makes a “wicked” governance challenge even “wickeder.”

4. Hardin’s Dichotomous Solution to the CPR Problem and the Rise of Community-Based Wildlife Governance

In his 1968 article, *Tragedy of the Commons*, Garrett Hardin prophesied the inevitable collapse of CPRs. Because CPR users were unable to self-regulate their own antisocial behavior, Hardin proclaimed that the resource could only be effectively governed under either of two institutional arrangements. The first circumstance under which CPRs could be effectively governed is through the conversion of those CPRs into private property. In this scenario, Hardin’s hypothetical herders would no longer communally graze their livestock across the entirety of the town commons. Rather, the commons would be divvied up among the townspeople, with each herder receiving her own allocated grazing area. Under this scenario, a herder would realize both the full benefit and full harm associated with adding a surplus animal to her own private grazing area, and would presumably make the rational decision not to exceed the carrying capacity of her grazing grounds. By privatizing the commons, the same rational self-interested behavior that inevitably leads to the resource’s decline could instead be harnessed to ensure its sustainable management (ibid).

The second circumstance proposed by Hardin to save his allegorical commons is to bring in a leviathan – the state – to require that livestock herders ignore their fully rational, yet antisocial, impulses (Harden 1968). In this scenario, the town herders would continue to
collectively utilize the commons, and each would continue to have the economic incentive to add livestock in excess of the common’s carrying capacity. The state, however, would impose additional costs associated with adding the additional animal in form of taxes on that animal, fines or criminal sanctions for exceeding a set quota, or some other type of formalized incentive. Through its rulemaking and enforcement powers, the involvement of the state would change the individual calculations of the self-rational herders, incentivizing them to make the prosocial decision to keep their individual livestock numbers at the carrying capacity of the commons (ibid).

Hardin’s dichotomous solution to CPR governance – privatization or state regulation – has been challenged over the last several decades by researchers such as Elinor Ostrom and Marshall Murphree. What these researchers have found is that CPRs often collapse under the weight of communal mismanagement, but some communities have adopted collective governance mechanisms that allow for prolonged sustainable governance of their respective natural resources (see, e.g., Ostrom 1990). The collapse of CPRs is not inevitable. And, the converse is also true: successful management of privatized and government-led CPRs is not guaranteed, as history is littered with examples of failed privatize and nationalized resources (Ostrom 2008).

Wildlife conservation efforts have traditionally relied on the intervention of the state to solve its governance challenge, using a “fortress model” approach consisting of a combination of trade restrictions, demarcation of formal protected areas, and the nationalization of wildlife (Spiteri and Nepalz 2006). This approach has deep historical roots, beginning with the setting aside of certain hunting ground for elites of the Roman Empire (Fischer 2011), and continuing through the assertion of sovereign ownership over wildlife within Europe’s Medieval and
Renaissance kingdoms (Wolfe 1970; Fletcher 2015). The modern form of the approach involves the use of national parks (along with wildlife refuges and other governmental protected areas) to protect wildlife and this model, starting with the creation of Yellowstone National Park in 1872, was adopted worldwide as the default form of wildlife protection throughout most of the 20th Century.

Beginning in the 1980s, countries in southern Africa began experimenting with a community-level approach to species conservation, often referred to as community-based natural resource management (CBNRM). As with other concepts (democracy being one such example), the term has been applied to a wide range of approaches that can vary significantly in conceptualization and design (Adams and Hulme 2001; Bandyopadhyay, et al. 2009; Child 2009). However, for purposes of this document, I adopt the definition used by Child (2009), which broadly describes the term referring to a collection of economic, political, and organizational principles that rely on the devolution of rights (see also Adams and Hulme 2001; Child and Barnes 2010).

The CBNRM approach is premised on the concept that local populations will be incentivized to actively participate in wildlife conservation efforts when they can manage the resource and share in the resulting profits (de Beer 2012). It relies on the empowerment of local communities to manage the wildlife populations located within their own territories, often with any or all of the goals of (a) empowering and developing local communities, (b) recognizing traditional knowledge or culture, and (c) encouraging the conservation or sustainable use of wildlife resources (Armitage 2005; Tsing, et al. 2005; Child 2009; Gruber 2010).
5. The Debate over Community-Based Governance of Wildlife

While the CBNRM approach holds much theoretical promise, CBNRM programs have had mixed results in practice (Swatuk 2005). In some places, such as Namibia and certain communities in Nicaragua and Zimbabwe, CBNRM is often viewed as a significant improvement over earlier conservation efforts (Dressler, et al. 2010). In other places, such as Zambia and Madagascar, the introduction of CBNRM seems to have resulted in little or no improvement in wildlife conservation (ibid). Case studies of both successful and unsuccessful CBNRM programs provide ammunition for proponents and critics of CBNRM alike.

At times, the adoption of CBNRM has coincided with a marked recovery in wildlife species and the receipt by local populations of wildlife-generated benefits. For instance, Jones (2009), for example, highlights the income generated within CBNRM areas in southern Africa, along with the relative value to residents of even small direct benefits. Among other things, he notes that members of the Torra Conservancy in Namibia received cash payments in 2003 that equaled around 14% of annual average individual income for those residents; the traditionally marginalized San community in the Nyae Nyae Conservancy received 28% of its jobs and around 35% of its cash income directly from the conservancy; and, between 1997 and 2005, the Sankuyo Tshwaragano Management Trust in Botswana created between 22 and 104 jobs per year, and generated a range of community benefits (residents voted each year to determine exactly which benefits the Trust would produce) (ibid). At the same time, some areas have observed a marked increase in wildlife compared to before the implementation of CBNRM. For example, wildlife in the Nyae Nyae Conservancy increased sixfold between 1995 and 2004 (WRI 2005), elephant, springbok and oryx populations in the Torra Conservancy increased by 280%, 11,385%, and 3,825%, respectively, between the early 1980s and 2012 (UNDP 2012).
On the other hand, critics of the CBNRM approach point out that the approach has not realized consistent success or, at least, has failed to consistently realize its potential (Hutton, et al. 2005; Blaikie 2006; Nunan 2006). Criticisms are also based on a number of other grounds, including that the approach relies on overly simplistic and romantic notions of “traditional” communities as homogenous units, that it exacerbates problems with corruption and local elite capture and inequity, and that it lacks a scientific foundation (Child and Lyman 2005; Hutton, et al. 2005; Murphree 2009).

6. The Focus of this Dissertation: Evaluating and Understanding the Efficacy of CBNRM as a Wildlife Conservation Approach

This dissertation ties together two important issues relating to the implementation of CBNRM. The first issue involves the desirability of the CBNRM approach as a wildlife conservation tool. An unresolved dispute exists over whether conservation should focus on sustainable development or on the creation of formal protected areas, and much of this debate centers on whether sustainable development approaches are as effective at species and biodiversity conservation as are demarcated protected areas. Given the intensity and duration of this debate, there is surprisingly little existing research that systematically examines the two approaches’ respective effectiveness at facilitating conservation. The few studies that have engaged in multi-site analyses have either been inconclusive or have produced conflicting results (Stoner, et al. 2007; Ihwagi, et al. 2015; Gray, et al. 2016; Oldekop, et al. 2016; Lee and Bond 2018).

The second issue involves the identification of factors potentially impacting the success of CBNRM programs. Community-based programs are implemented in a complex mélange of climactic, socioeconomic, cultural, historical, and political conditions, and these conditions can vary both between and within communities. The presence of, and interactions between, those
conditions can influence the ability of CBNRM programs to achieve their development and conservation goals. A body of literature examines how the presence of commonly observed variables potentially affects the outcome of CBNRM initiatives, but the list of potentially important variables is lengthy, and much remains to be learned about they might contribute to or undermine the sustainable governance of wildlife.

From these two issues – the relative effectiveness of the CBNRM approach and the variables that influence its success – this dissertation identifies and examines three specific questions. The first question asks how the conservation outcomes of CBNRM areas compare to those of formally demarcated protected areas. The second question explores whether different types of economic benefits (a core element of most CBNRM programs) vary in their impact on residents’ perceptions of whether they gain from wildlife. The third question investigates the potential impact of economic and risk heterogeneities on the effectiveness of CBNRM efforts, where participants face a risk of suffering harm from the very wildlife they are tasked with conserving. This sort of scenario is frequently the case in CBNRM areas, as interactions between people and wildlife can cause people to experience economic and emotional losses, often in the form of livestock and/or crop predation, infrastructure damage, or personal injury or death.

7. Research Design and Study Sites

The questions explored in this dissertation are complex, and demand investigation across multiple scales. An inquiry into the overall effectiveness of a policy approach, for example, is best served by an analysis of outcomes across a range of sites and situations. On the other hand, an exploration into the potential psychological drivers of collective action participants likely requires a more focused approach. As such, rather than relying on a single method, this
dissertation utilizes a mixed-methods approach to better match the scale of the analyses to the demands of the particular inquiries.

First, in order to examine the efficacy of the CBNRM approach, I employ a multi-site, longitudinal study that assesses performance of both CBNRM and governmental conservation areas across a range of settings on a single measure: the prevention of illegal elephant kills. Specifically, I use an elephant kill database, compiled by the Convention on International Trade of Endangered Species (CITES), to identify the annual proportion of illegally killed elephants (out of the total number of observed elephant deaths) within 39 designated sites, spread across 19 countries in southern, central, and eastern Africa. I also identify the percentage of the different observation sites falling under each of the CBNRM and governmental governance approaches. Controlling for variables previously identified as potentially contributing to the risk of elephant poaching, I compare the relationship between (a) the percentage of observation sites falling under a particular governance approach and (b) the likelihood of an observed elephant death being the result of an illegal kill.

As noted above, much of the research on CBNRM effectiveness is in the form of either case studies or small-scale comparisons, often involving observations from a single point in time. Yet, numerous countries – generally differing in their wildlife populations, tourism potential, policy goals, and economic and institutional factors – utilize both CBNRM and governmental approaches. Therefore, the success or failure of a particular approach in one site or country, at a particular moment in history, arguably provides limited insight into the efficacy of the approach as a whole. The use of a large-N quantitative analysis allows for a more broadly applicable analysis of the relative effectiveness of CBNRM.
Second, in order to assess the potential impact of different benefit types on CBNRM residents’ perceptions of the impact of wildlife on their lives, I utilize data collected from a field study in northwest Namibia, Africa. In that study, I administered surveys to households located within four conservancies (formally delineated CBNRM areas) in Namibia’s Kunene region. Among other things, the surveys asked respondents about (a) the types of benefits their households had received from their conservancy (such as conservancy employment, cash distributions, or meat from trophy hunts), and (b) their opinions regarding the impact of wildlife on their lives. Controlling for other factors identified in the literature as potentially impacting attitudes toward wildlife, I compare the relationship between residents’ receipt of different wildlife-generated economic benefits and the likelihood that they feel that wildlife has made their lives better.

Little data currently exist about either the type or respective impact of benefit types received by CBNRM participants, and this paucity necessitates the gathering of data in the field. The Kunene is a particularly apt place to conduct this sort of investigation. Namibia is frequently heralded as having one of the world’s most successful CBNRM programs (Massyn 2007; Boudreaux and Nelson 2011) and its Kunene region has a well-developed tourism industry centered around both wildlife photography and trophy hunting. The four conservancies selected for this study were among the first conservancies established in Namibia, and they all participate in a profit sharing arrangement under which they receive funds from black rhino tourism in the adjacent national park that are earmarked for helping families (selected by the conservancies) to cover school expenses.

In addition to these school payments, the conservancies also receive income from tourism and meat from the sanctioned utilization of their wildlife populations, which they can distribute
to their residents how and to whom they choose. Consequently, the four conservancies have, over the course of their existence, distributed a variety of benefits to their residents, with residents differing in both the number and types of benefits received. The longevity of these four conservancies, their relatively high income from wildlife, and the discretion that they have in distributing benefits to their residents, means that respondents (a) vary regarding the benefits they have received from the conservancies, and (b) have had the opportunity to form an opinion about the impact of wildlife on their lives in a CBNRM setting.

Third, to assess the impact of risk and economic heterogeneities on collective action in the presence of risk, I use data from a behavioral laboratory experiment. In that experiment, I and a coauthor asked participants to play prisoner’s dilemma game variant in which they risked suffering a loss if they chose to contribute to a collective pool. The game we used was designed to simulate the following stylized scenario: CBNRM residents must decide whether to participate in the conservation of wildlife by refraining from killing wildlife that might inflict on them physical or economic harm. When residents choose to participate in wildlife conservation, the community receives wildlife-generated income, and that income is distributed evenly to the community members. The community income increases or decreases in relationship to the concentration of wildlife in the CBNRM area. However, if residents choose to refrain from the anticipatory killing of wildlife, they risk suffering economic losses from human-wildlife conflict. The experiment investigated the impact of risk and economic heterogeneity by altering the individual participants’ respective risk of loss and by varying the level of endowments they received before each round of the game.

Controlled behavioral experiments are useful in an initial exploration of the impact of a particular variable of interest, as they allow the investigator to isolate its relationship with the
outcome measure by stripping away many of the potentially confounding variables found in the field (McDermott 2002). Consequently, behavioral experiments yield generalizable hypotheses that can then be tested across a range of conditions in the real world.

8. Organization of Dissertation

This dissertation begins with an introduction to the theoretical underpinnings of the CBNRM approach and then explores the development of CBNRM in Namibia. I trace the history of wildlife governance in Namibia, from decentralized, tribe-based pre-colonial approaches, through increasingly rigid and nationalized colonial governance, to its post-colonial adoption of the community-based Conservancy program. I then discuss structural issues with the conservancy program that have the potential to undermine the success of this particular implementation of the CBNRM approach. I conclude the first chapter by reviewing some of the most common criticisms of the CBNRM approach, which I use to refine the questions guiding this research.

In Chapter 2, I turn to my evaluation of CBNRM’s overall efficacy. I present this analysis first for the simple reason that the relevance of the remainder of my research hinges, at least to some degree, on the findings of this chapter. In short, the need for further inquiry into how different variables may impact wildlife outcomes in CBNRM areas could arguably become less pressing upon a determination that it is the clearly inferior conservation approach.

In Chapter 3, having tackled the foundational question of the potential viability of CBNRM, I turn my attention to evaluating the central tenet of most CBNRM efforts – the assumption that participants’ receipt of wildlife-generated benefits will cause them to view wildlife as “paying its way.” Given the many different forms that benefits can take, and the potential role of institutions, cultures, and past experiences in shaping individuals’ perceptions of
those benefits, this evaluation required that I move from a large-N quantitative analysis to one driven by “boots on the ground” data collection.

Finally, in Chapter 4, I focus on the generation of theoretically based hypotheses regarding the potential impact of risk and economic heterogeneity on collective action by CBNRM participants. Unlike the study of benefit types, which required an examination of the particularized experiences and thoughts of CBNRM residents, the depth of existing experimental research on the effects of heterogeneity means that the research in this chapter was particularly suited for the laboratory. The data generated by my field research was, unfortunately, not able to provide any basis for evaluating the applicability of my experimental findings, as the survey participants evidenced almost no variation regarding their risk of predation. In their responses, virtually all participants that owned livestock stated that they had been frequent victims of recent depredation by wildlife, and I was not able to reliably quantify the number of incidents of predation. Consequently, the findings of this chapter are presented last, as a generalizable hypothesis that warrants future evaluation in the field.

9. Chapter Findings and Synthesis

What does this dissertation ultimately tell us about CBNRM? Individually, each chapter individually expands our understanding of whether, and under what circumstances, the CBNRM approach works. But, when considered as a whole, the findings presented in this dissertation lend even greater insight into the efficacy of CBNRM. Here, I will present a brief summary of the findings of each of Chapters 2-4 before synthesizing their findings.

a. The Overall Effectiveness of the CBNRM Approach

In Chapter 2, in which I compare the likelihood of illegal elephant kills across areas falling under CBNRM and governmental governance, I find that an increase in either form of
governance is associated with an increase in the likelihood that an observed elephant carcass resulted from an illegal kill. The increase in likelihood was greater for CBNRM governance than for governmental governance, but the available data does not lend insight into why CBNRM areas are associated with a larger increase. Given this uncertainty, and the fact that increases in both forms of governance were associated with an increase in the likelihood of illegal kills, I conclude that it is premature to declare either governance approach as clearly superior at protecting wildlife. Rather than arguing for adherence to a particular approach, I contend that conservation stakeholders should focus on the adoption of approaches that are tailored to a site’s particular mix of natural, institutional, and socio-economic characteristics.

b. The Role of Different Benefit Types on Residents’ Perceptions of the Impact of Wildlife on their Lives

In Chapter 3, where I used survey data to examine the impact of different benefit types on recipients’ perceptions of whether wildlife improved their lives, I discovered that the total number of benefits received was positively and significantly correlated with the likelihood of participants having a more positive view of whether wildlife improved their lives. But, surprisingly, more fine-grained analysis suggests that the individual benefit types vary greatly in the direction and magnitude of their impact on participants’ perceptions. Certain benefit types (employment, meat, tuition money, and funeral assistance) consistently had positive associations with the likelihood of participants having a more favorable perception of the impact of wildlife. On the other hand, two seemingly desirable benefits (cash distributions and business loans) were consistently negatively correlated with the outcome measure (albeit not significantly for most of the models used). Additionally, I found that, once recipients received both meat and funeral benefits, the receipt of any additional benefit (other than conservancy employment) had no, or
perhaps even a negative, relationship with residents’ perceptions of the impact of wildlife on their lives.

I hypothesize that the surprising difference in the association of the benefit types with the outcome measure is most likely explained by a psychological phenomenon in which people do not objectively assess their experiences, but instead compare them to subjective standards that are based on prior experiences or expectations. Based on comments made by survey participants, and by traditional authorities and conservancy representatives I interviewed prior to the administration of the survey, I hypothesize that recipients of rarer benefits (with the exception of conservancy employment) are more likely to have heightened subjective benchmarks than are others in the community. These heightened benchmarks might exist because either (a) the earlier receipt of infrequently distributed benefits (such as cash) caused the recipients to readjust their expectations upwards, or (b) the recipients of rare benefits were more likely to be elite or have connections to local elites, and so they had heightened initial expectations about the magnitude of the benefits they would receive from the conservancies.

c. The Influence of Risk and Economic Heterogeneities on Collective Action

In Chapter 4, I and a coauthor used a behavioral laboratory experiment to explore the impact of risk and/or economic heterogeneity in CBNRM when participants are faced with the prospect of human-wildlife conflict. We found that unidimensional heterogeneity – i.e., where only risk or economic heterogeneity was present – decreased overall group cooperation, but the two forms of heterogeneity differed in their impacts on individual behavior. The presence of wealth heterogeneity decreased cooperation by all participants, regardless of their individual economic status in the group. On the other hand, the presence of risk heterogeneity reduced

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4 As further discussed in Chapter 4, human-wildlife conflict refers to harm caused by wildlife to human lives, livestock, crops, or infrastructure.
cooperation by high-risk participants while increasing cooperation by participants facing low risk.

Where risk and economic heterogeneity were both present, cooperation levels were markedly higher than in the unidimensional environments. For groups in which the heterogeneity was balanced (participants had both high risk and high endowments, and vice versa), overall cooperation was lower than that found in homogenous settings. However, groups that had unbalanced heterogeneity (where participants with low endowments faced high risk, and vice versa) achieved higher levels of cooperation than in the homogenous treatments, although that increase was not statistically significant.

As in Chapter 3, we believe that this difference may results from participants’ reliance on reference points. Specifically, we hypothesize that the participants’ tolerance of risk was impacted by whether the endowment they received was below or above that of a social comparator – in this case, others in the participants’ experimental groups. We propose that participants who received lower endowments were more likely to tolerate risk as they attempted to “catch up” to the value of the higher endowment distributed to their peers. Meanwhile, recipients of higher endowments were less likely to tolerate risk out of a desire not to “fall back” to the level of the lower earners in their group.

d. Synthesis of Findings

Read together, the finding presented above confirm the importance of historical institutions and the fact that, similar to that of their governmental counterparts, CBNRM efforts have yielded a wide range of outcomes (both of these statements are addressed in greater length in the conclusion). They also suggest that participants in CBNRM regimes are not static entities. Policymakers often have a tendency to “wish away” the overall heterogeneity of CBNRM
communities by erroneously treating them as cookie-cutter, homogenous groups (Richard and Ratsirarson 2013). My research indicates that, in addition to accounting for this community-level heterogeneity, policymakers should also view individuals within those communities as dynamic entities with perceptions and preferences that shift over time. These shifts can result from both (a) the recalibration of participants’ goals and expectations (i.e., their perceived needs) based on their prior experiences and shifting reference points (Chapter 3), and (b) changes in exogenous factors such as risk exposure (Chapter 4). If CBNRM programs are unable to adapt to the changing perceptions and goals of their residents, we might expect a gradual erosion of support for the approach.

Perhaps because most CBNRM studies are not longitudinal, I identified surprisingly few articles that examined changes in participant support for CBNRM over time. Nevertheless, a few practitioners and CBNRM scholars have produced works that suggest that the long-term decay of support for CBNRM resulting from shifting perceptions may be a very real risk. Snyman (2014) found that respondents living near long-established tourism camps and lodges had more negative attitudes towards conservation than did those living near more recently established tourist lodging. Jones (2007) and Jones and Weaver (2009: 240) warn that initial strong community support in Southern Africa for CBNRM efforts can be threatened if typically modest initial benefits from CBNRM are not increased over time. Chikozho and Magweregwede (2006) note that residents’ initial enthusiasm for their CBNRM programs can wane over time when those programs fail to meet their poverty reduction needs. And, MacKenzie et al. (2017) found that an increased risk of human-wildlife conflict led to a deterioration of residents’ perceptions of the benefits of adjacent protected areas, even though those same individuals also tended to report an increase in the conservation-based benefits they derived from those protected areas. While this
last study did not directly address CBNRM, it nevertheless shines light on the potential for resident perceptions to shift in response to changes in both benefits and risk.

Most CBNRM areas remain underdeveloped, with residents facing few local economic opportunities. The initial excitement over the receipt benefits (or even the anticipation of future benefits) is likely to turn to frustration if residents feel that this early progress has stagnated. To be fair, the development of tourism and trophy hunting industries in CBNRM areas provides invaluable employment opportunities, but these jobs are scarce, and both my survey data and other literature (e.g., Naidoo, et al. 2016; Lubilo and Hebinck 2019) suggest that such employment is more often given to individuals with connections to local leaders. Consequently, CBNRM residents can develop the perception that CBNRM efforts are ineffectual (or worse), even if an objective economic analysis might show otherwise. It is this perception that ultimately determines whether CBNRM residents support collective efforts to achieve sustainable wildlife governance.

There is, of course, no simple solution for preventing CBNRM residents from growing impatient over time. CBNRM programs and their constituent communities vary widely across a spectrum of legal, institutional, socioeconomic, and cultural variables, and any proposed solutions will ultimately need to be tailored to match local conditions. Nevertheless, in the conclusion section of this paper, I propose two policy approaches that may have broad applicability in facilitating CBNRM programs in addressing the evolving needs of their constituents.
CHAPTER 1

THE RISE OF CBNRM IN NAMIBIA: EXPLORING THE APPROACH'S THEORY, DEVELOPMENT IN NAMIBIA, AND UNRESOLVED QUESTIONS

Throughout much of the 20th Century, wildlife management found in sub-Saharan Africa mirrored that found across the globe – primarily relying on a “fortress model” approach of demarcated protected areas, nationalization of wildlife, and bans on the hunting or utilization of protected species (Martin 2000; Carpenter 2011). However, beginning in the 1960’s and 1970’s, countries in southern Africa began experimenting with devolving control over wildlife resources – first to private landholders, and subsequently to communities located in communally-owned lands (Roe, et al. 2009; Child and Barnes 2010).

These experiments coincided with the rise of the “Washington Consensus” in 1980s – a neoliberal development policy that emphasized the free market and limited involvement by the state (Roe, et al. 2009). The resulting devolution of wildlife management rights to rural communities – often referred to as Community-Based Natural Resource Management (CBNRM) – marked a notable shift from the paternalistic and increasingly rigid wildlife management policy favored by the countries’ prior colonial rulers.

The CBNRM approach has had both notable successes and failures, and it has been criticized on both theoretical and practical grounds (Adams and Hulme 2001; Child 2009; Murphree 2009; Dressler, et al. 2010; Gruber 2010). Among other things, states, NGOs, and local officials are often reluctant to fully devolve control over wildlife resources to the local communities (Gibson 1999; Roe, et al. 2009). Nevertheless, it remains a popular and widely utilized wildlife governance approach, particularly in the global south (Gruber 2010).

In this chapter, I will introduce the theoretical foundation of the CBNRM approach. I will then use Namibia as a case study to both (a) illustrate the sort of historical, political and
economic drivers that can motivate the development of the approach, and (b) highlight the existence of potential structural weaknesses present in even the most lauded CBNRM programs. Finally, I will present some of the common theoretical and results-based criticisms of CBNRM, emphasizing those which give rise to the questions motivating the empirical chapters of this dissertation.

1. **CBNRM: Devolution of Control and Realization of Benefits**

Child and Barnes (2010) observe that the term ‘CBNRM’ refers to both an aspirational ideal and real-world implementation of the approach. However, much as is the case with other aspirational terms such as ‘democracy,’ there is not a universal agreement about the range of governance approaches that should fall under the label of ‘CBNRM’ (Child 2009; Murphree 2009; Child and Barnes 2010; Gruber 2010; Measham and Lumbasi 2013). The term is used in different regions of the globe to refer to a range of approaches that can vary significantly in conceptualization and design (Adams and Hulme 2001; Bandyopadhyay, et al. 2009; Child 2009). In this dissertation, I examine the form of CBNRM that evolved out of southern Africa, which primarily focuses on wildlife governance and which Child (2009) describes as a “shorthand for a set of economic, political, and organizational principles within a strongly devolutionary rights-based approach” (see also Adams and Hulme 2001; Child and Barnes 2010).

At its core, the CBNRM approach calls for a degree of empowerment of local communities to manage wildlife stocks found within their own territories (Armitage 2005; Gruber 2010). The aspirational goals of the approach represent the interests of myriad different stakeholders. The approach is alternately (and often simultaneously) viewed as a mechanism for (a) the empowerment and economic development of rural communities, (b) the formal
The CBNRM approach rests on two core assumptions. First, it assumes that local populations have a greater potential for the sustainable use of resources than does the state (Tsing, et al. 2005; Dressler, et al. 2010). Part of this potential lies in the fact that, in theory, local communities are more aware of local environmental and ecological conditions (Tsing, et al. 2005; Blaikie 2006; Cox, et al. 2010). Local communities may also have their own institutions – formal or informal – that are better suited than equivalent national institutions at facilitating the sustainable use of wildlife resources (Tsing, et al. 2005). The second assumption is that, if they receive an enduring interest in and are able control and profit from wildlife resources, communities will govern those resources in a sustainable manner in order to ensure the availability of future benefits (Songorwa, et al. 2000; Silva and Mosimane 2012).

The CBNRM approach is, in many respects, a market-based one (Lyons 2013). Simply put, the approach holds that people will sustainably manage their resources when the perceived benefits of doing so outweigh the perceived costs (Murphree 2009). Formal policies are often designed to provide communities with economic benefits (Child 2009; Murphree 2009), but participants can also have non-economic incentives, such as a sense of pride associated with the presence of wildlife (Jones and Weaver 2009). Nevertheless, increasing the value of wildlife to communities does not alone qualify as CBNRM – the approach requires the presence of both benefits and of the devolved control (Child and Lyman 2005).

2. The Development of Namibia’s CBNRM Program: From Tribes to Conservancies

Thirty-nine percent of Namibia’s total land area (home to approximately two-thirds of the country’s population) falls under communal ownership, where ownership is legally recognized as
collectively belonging to the “traditional communities” of the area in which it is located and permanent individual ownership rights are generally prohibited (USAID 2019; Communal Land Reform Act 2002). Namibia has embraced the implementation of CBNRM in these communal areas, and its program is frequently held up as one of the world’s flagship examples of CBNRM success in promoting wildlife conservation. Communal-based conservation features prominently in tourism marketing materials for the country. For instance, the official website of the Namibia Tourism Board describes Namibia’s implementation of CBNRM as “facilitating a remarkable recovery of wildlife,” and notes that this recovery “has led some to call Namibia’s conservation effort the greatest African wildlife recovery story ever told” (NTB 2019). Similarly, Wilderness Safaris (a high end tourism company with numerous tourist lodges spread across seven African countries) writes that its Damaraland Camp is the “successful result” of a partnership with the Torra Conservancy (located in Namibia’s northwest Kunene region) that “has become an inspiration for communities and conservationists throughout Africa” (WS 2019).

CBNRM in Namibia, however, is not merely an alternate approach to effectuating wildlife governance, but also represents an attempt to combat the grossly inequitable legacy of Namibia’s colonial and apartheid past. Through the creation of its conservancy program, the post-colonial Namibian government expressly sought to empower rural communities that had long been denied economic and administrative opportunities under German and South African rule.

5 In one example of such praise, the WWF wrote a 2011 article titled *Namibia: how communities led a conservation success story* (WWF 2011).
a. Pre-Colonial Governance in Namibia and Germany

i. Namibia: A patchwork system of governance and the impact of rinderpest

Little is written on pre-colonial environmental management in Namibia. Nevertheless, a limited number of writings provides an idea of what environmental governance in Namibia may have looked like prior to European colonization. Across sub-Saharan African generally, societies used social, rather than ecological, management systems (DeGeorges and Reilly 2009). These systems restricted access to natural resources through hierarchical social structures, often based on clanship, families, or religious authority, and sometimes imposed hunting limits on certain types of game species or young or pregnant animals (Hitchcock 2000; DeGeorges and Reilly 2009).

Hunting limitations were not driven by ecological concerns, but rather considerations such as the need to conserve game resources for future hunting, personal or societal preferences or taboos, or limits in hunting technology (Khan 1994; DeGeorges and Reilly 2009; Manyanga and Pangeti 2017). In highly centralized societies, ruling families sometimes imposed hunting restrictions through the demarcation of their own personal hunting grounds (Manyanga and Pangeti 2017). On the other hand, hunting was also a tool used by pastoral and agricultural communities to protect their livestock and crops by limiting populations of certain wildlife species (Manyanga and Pangeti 2017).

It appears that the governance approaches used throughout sub-Saharan Africa were also present in at least some of the pre-colonial areas of Namibia. The Ovambos, located in northern Namibia and present-day south Angola, were organized into chiefdoms in which royal families held a degree of ownership over land and resources (Hahn, et al. 1928; Eirola 1992). As such, the Ovambo chiefs had the capacity to implement some degree of limitations on wildlife utilization.
For instance, an Ovambo chiefdom established a traditional royal hunting ground that encompassed parts of the current Etosha National Park (Lindeque 1991). In many Ovambo kingdoms, subjects were forbidden from hunting until after the conclusion of the king’s ceremonial hunt at the beginning of the dry season, at which point the wildlife had already given birth (Siiskonen 1990; Hinz 2003). Large game such as elephants were often regarded as belonging to the Ovambo kings, who hunted the species in order to elevate their status (Siiskonen 1990).

The Nama and the San, lacking the strong chiefdoms of the Ovambo, instead appear to have had more informal restrictions on hunting in the form of taboos and norms regarding the consumption of certain wildlife species. In a 1928 account of the Nama, the German missionary Heinrich Vedder noted, “a real Nama never eats the flesh of a wild dog, a monkey, a hyena, jackal, a lion or a hare. He believes the meat of these animals to be impure and injurious to health. But this custom is also slowly disappearing” (Hahn, et al. 1928:129). Similarly, in an ethnographic study conducted over a series of expeditions between 1950 and 1961, Marshall (1976: 126-127) observed that the San people in Namibia’s Nyae Nyae region did not eat many species of wildlife found in the area. Some (such as foxes, shrews, genets, some small wildcats, gerbils, bats and a variety of snakes, birds and insects) were apparently not ordinarily considered to be food. Other animals were viewed as repugnant, and the people whom Marshall interviewed stated that they would rather die of starvation than to eat them. The latter category of animals consisted primarily of predators and scavengers (lion, leopard, cheetah, hyena, aardwolf, wild dog, and vulture) but also included flamingo, chameleon, mongoose, meerkat, and squirrel (ibid).
As a result of human activity, including hunting and livestock grazing, Africa was not generally dominated by wildlife as is often depicted in popular imagery today. Rather, the landscape in many places was dominated by cattle and goats, with minimal bush cover (Nelson 2003; DeGeorges and Reilly 2009). The arrival of rinderpest in Africa drastically reduced the impact of human activity on wildlife populations.

In 1891, an outbreak of rinderpest occurred among cattle on Kilimanjaro (Ofcansky 1981:31). Six years later, in 1897, the disease reached Namibia, spreading across the area within a matter of weeks (ibid:36). Rinderpest had a devastating impact on native African populations, including those in Namibia, and a profound impact on its landscape. Namibia’s Herero people lost as many as 95% of their cattle – a number that mirrors the percentage of cattle losses across Africa as a whole (Ofcansky 1981:36; DeGeorges and Reilly 2009:737). It also decimated other types of livestock, as well as wildlife populations of buffalo, giraffe, eland, small antelopes, and warthogs (Nelson 2003).

Whereas the presence livestock and herbivorous wildlife had previously encouraged the growth of grassland, the lack of grazing in the aftermath of rinderpest allowed for the encroachment of bush which, in turn, provided habitat for the tsetse fly (Nelson 2003). The resulting outbreaks of sleeping sickness limited the recovery of livestock but not of wildlife, which were immune to the disease. Absent competition from livestock, wildlife populations quickly rebounded (ibid). A countryside predominated by bush and wildlife was a marked departure from the historical status quo but, “[f]or European conservationists, typically ignorant of the recent ecological history of the continent, this landscape appeared to be the ‘true Africa’ of wild game” (Nelson 2003:72; see also Adams and McShane 1992).
ii. Germany: From royal hunting to the conservation of nature

During the time of the Roman Empire, no known wide-spread formal legal restrictions on hunting existed within the territories that comprise current-day Germany, with hunting being allowed even on private lands (Fischer 2011). However, the hunting of certain species was considered an elite pastime, and there is evidence that imperial hunting grounds were created in which hunting general hunting was prohibited. For example, the city of Trier is believed to have had an imperial hunting grounds that encompassed an area of approximately 220 km² (ibid: 262).

The Germanic tribes gained hegemony in western Europe in the late 4th century. Wolfe (1970) provides a detailed account of the development of wildlife governance among the Germanic people. Initially, the tribes likely lacked any restrictions on wildlife use other than those related to religious taboos. The Germanic Codes from the sixth century contained some limited restrictions on wildlife capture, but the Codes largely addressed the methods used for hunting rather than imposing any seasonal restrictions or species protections (Wolfe 1970; Schaller 2007). Further, under the Codes, wildlife was considered ownerless (regardless of where it was found) until it was wounded, at which point it belonged to the hunter that inflicted the wound (Wolfe 1970).

The seventh century saw the first establishment of bannforste, or royal forest preserves (Wolfe 1970). Within these preserves, kings employed a practice of “inforestation” that reserved to them the exclusive right to hunt – a marked departure from the right of free chase codified in the Germanic Codes. Over the course of the next 600 years, German sovereigns extended their control over wildlife to include the hunting of game on vacant and captured lands. Additionally, landowners seeking the protection of Germanic sovereigns often had to cede to them the hunting rights on their lands (ibid). Many of the bannforste created during this period now form the core
of national parks and other formal protected areas (Mose and Weixlbaumer 2007). In addition to the bannforste, German kings also established private royal hunting preserves around their various seats of power (Giese 2011). In 1640, for instance, Frederick III, Duke of Holstein-Gottorp, created a new game park next to Gottorf castle in Schleswig, complementing a pre-existing smaller one (Radtke 2011:429-431).

In 1232, the Sicily-based Holy Roman Emperor Frederick II responded to unrest in the German principalities by confirming the Statutum in favorem principum (Statutes in favor of the princes), which had the effect of conferring much of the Emperor’s rights and privileges, including the right of inforestation, to the individual German sovereigns (Wolfe 1970). These sovereignties eventually used their inforestation powers to preempt all hunting rights throughout the entirety of their territories, preventing hunting without express permission (ibid). This expansion of hunting restrictions led to the establishment of the Jagdregal, a legal institution consisting of two executive powers: (1) the right of the provincial sovereigns to make any laws “necessary for the welfare of the state” with regard to wild game and hunting; and (2) “the jus venandi, the exclusive right to hunt anywhere within the province” (Wolfe 1970:10).

Individual German sovereignties began imposing seasonal hunting restrictions starting at the beginning of the 16th Century (Wolfe 1970; Schaller 2007). Even in season, hunting was considered a noble pastime, with peasants forbidden from participating or owning firearms (Wolfe 1970). The sovereignties also continued the tradition of creating royal forests, one example of which is a 4,633-hectare private hunting ground created by Maximilian II Emanuel, the elector of Bavaria from 1679-1726 (Knoll 2004:13).

The Jagdregal period ended after Germany’s 1848 Revolution, at which time all landowners were granted the express right to hunt game on their own property (Wolfe 1970).
The resultant overhunting, however, led the German states to later re-institute restrictions on hunting, a process that accelerated after the creation of the German Empire in 1871. Among other things, these restrictions limited who had the right to hunt, instituted seasonal bans on hunting, and created hunting districts (ibid).

The 1800s witnessed a growing environmental concern within Germany. In 1819, Alexander von Humboldt, having previously observed British informal landscape design, coined the term *naturdenkmal*, or “nature monument” (Jones-Walters and Čivić 2013:122). The concepts of ecology and nature conservation later appeared in Germany in 1866 and the 1880s, respectively (Brüggemann 1997). However, despite the growing interest in nature conservation, in 1898 the German government rejected the idea of creating Yellowstone-style national parks within Germany because of the perception that the country lacked sufficiently large remaining wild areas and concerns over the removal of large areas of land from productive use (ibid).

b. **1884–1915: The Creation of a German Colony in Namibia and the Systematic Disenfranchisement of Native Namibians**

Lacking viable natural harbors for the much of its coastline, and further insulated by the inhospitable coastal Namib desert, large swaths of Namibia remained largely untouched by European exploration until the mid-19th Century (Vedder 1966; Bollig and Heinemann 2002; Siiskonen 2015). England displayed an early interest in the region, annexing Walvis Bay (the primary port on Namibia’s coastline) in 1878 (O'Callaghan 1977; Siiskonen 2015). However, in 1883, Germany began aggressively pursuing a colonization effort, beginning with the purchase of approximately 345 km² of land around Angra Pequena in southern Namibia (subsequently renamed by the Germans as Lüderitzbucht, or Lüderitz Bay) (O'Callaghan 1977:17). By the next year, Germany had declared as its protectorate the coastline of Namibia extending nearly 1,000 km northward from Angra Pequena to Cape Fria near the Angolan border, excluding only the
English territory in Walvis Bay. In 1885, Germany also claimed hegemony over inland areas occupied by the Herrero, Nama, and Baster tribes (ibid.).

South Africa’s Cape Parliament also claimed control over the region during this period, voting unanimously in 1884 to annex the lands south of those controlled by Portuguese in current-day Angola (O’Callaghan 1977:17). To resolve its territorial disputes with the region’s other colonizers, Germany entered into treaties with Portugal in 1866 and, via the Anglo-German Agreement, with England and South Africa in 1890 which, together, established Namibia’s present boundaries, including a 20 mile (32.2 km) wide strip that granted Namibia access to the Zambezi River (ibid.).

Germany sought to establish control over Namibia’s native populations by entering into agreements with tribal leaders in which Germany offered protection in exchange for promises of allegiance (Miescher 2012). While these agreements were technically treaties between individual sovereigns, international law at that time permitted European Powers to acquire full sovereignty over any indigenous lands the Powers occupied, as it was assumed that native Africans were incapable of fashioning a government that could adequately protect or meet the needs of Europeans (O’Callaghan 1977). As such, after the 1890 treaty, Germany effectively had the de jure capacity under international law to claim sovereignty over the whole of Namibia (ibid.).

For purposes of this chapter, Germany’s colonial interactions with native Namibians were shaped by two central policies: land allocation and environmental governance. As discussed below, the former policy focused on the removal of black Namibians from much of the country’s commercially viable land and the creation of a subjugated labor pool. The latter policy focused on warding off a perceived environmental crisis while providing Germans with a conceptual justification for their takeover of lands utilized by black Namibians.
i. **Land policy: Confiscation for white settlers**

German colonial land policy was guided largely by the singular idea that indigenous populations should relinquish their grazing and farmlands for use by white settlers (Parker 1991). In 1890, a member of the German Colonial Office made clear the colonizing country’s intent: “[t]he decision to colonise in South-West Africa could after all mean nothing else but this, namely, that the native tribes would have to give up their lands on which they had previously grazed their stock in order that the white man might have the land for the grazing of his stock” (Scott 1958:321). The German desire for land was particularly acute given that Theodor Leutwein (who served as colonial administrator of Namibia from 1894-1904) envisioned the development of the country as a global beef exporter (Siiskonen 2015:290). In order to accomplish this goal, Leutwein called for the development of livestock ranches, 5,000 to 10,000 hectares in size, managed exclusively by European settlers (ibid.).

Germany initially acquired land in Namibia using a divide-and-rule approach, employing a combination of force, trade, trickery, and exploitation (for example, during the rinderpest outbreak in 1897, German authorities provided vaccinations for the livestock of white settlers, but only provided vaccinations for indigenous livestock in return for payment, often in the form of livestock or land) (Rosenberg 2008:56; Cooper 1991:20-21). In 1898, to facilitate the provision of land to the continuing influx of white settlers into southern Namibia, the German Government issued a decree establishing “reserves” for the resettlement of the country’s black population (Cooper 1991:21; Parker 1991:94). By 1903, over 30% of Namibia’s surface was controlled by white settlers (Parker 1991:94).

The arrival of rinderpest south of the Zambezi River in 1896 led to a conference of the colonial powers in the British Cape Colony, the outcome of which was an agreement to prevent
livestock and wildlife from crossing colonial borders and to strictly control the movement of native Africans (Miescher 2012:20). Germany, however, lacked the resources to seal off the entirety of Namibia’s boundaries, and instead focused on cordonning off only the colony’s north and east (ibid.:23). The largest of the north’s ethnic groups, the Ovambo, were armed, organized, and outnumbered the entire population of the area of the country under German control (Soggot 1986). Consequently, the colonial government chose to treat the area outside of its de facto control as a “foreign territory” for purposes of establishing the rinderpest fence, a decision that resulted in one half of Namibia’s population living north of the cordon (Miescher 2012). The fence ultimately proved ineffective at preventing the spread of rinderpest, but its construction nevertheless established a physical demarcation of the edge of Germany’s sphere of influence within the Namibian colony (ibid).

At the turn of the century, Germany faced a series of armed rebellions by its black ethnic groups. These uprisings occurred, in part, in response to Germany’s repressive colonial policies, such as the taxing and confiscation of livestock and horses, and its generally inequitable treatment of indigenous populations and leaders (Rosenberg 2008). However, the uprisings were also the result of increasing resource degradation and scarcity on the native reserves that served to exacerbate the poverty of their residents (Siiskonen 2015).

The Grootfontein launched a short-lived rebellion in 1901 (Rosenberg 2008:56). Upon their defeat, Germany relocated Grootfontein prisoners to Windhoek (the capital city) to serve as laborers and confiscated their land, horses, and livestock. The Bondelwart subsequently rose up in 1903, and their early success caused Germany to transfer its Namibian-based troops southward to combat the rebellion. The movement of German troops from their lands in the north spurred the Herero to start their own uprising in January of 1904 (ibid.).
Unprepared for the Herero rebellion (the Germans had considered the chief that initiated the rebellion to be loyal and had openly favored him over chiefs of other tribes as a result), the German military suffered multiple defeats in the first half of 1904 (Rosenberg 2008:57). Germany responded, however, by sending additional troops to Namibia under the command of General Lothar von Trotha (ibid.). On October 2, 1904, von Trotha issued an extermination order, which stated that any Herero, including women or children, found in German territory “with or without a gun, with or without cattle, [would] be shot” (Kaumbi 2006:47). By the conclusion of the Herero rebellion, Germany had slaughtered approximately 80% of the Herero population, leaving only 15,000-16,000 survivors from an estimated pre-rebellion population of 80,000 (Katjavivi 1988:10; Cooper 1991:24). Of those survivors, 14,000 were sent to concentration camps where they worked in hard labor (Katjavivi 1988:10).

A number of Nama tribes started their own rebellions in October 1904, each being met with a similarly ruthless response (Rosenberg 2008:57). By the end of the hostilities, the Nama’s numbers were reduced by half, from approximately 20,000 to 10,000, with thousands sent to concentration camps (Soggot 1986:11). Caught in the middle of the fighting, the Damara also lost 17,000 individuals (Cooper 1991:24), roughly a third of their population (Emmett 1999:59).

In 1907, the lands occupied by the various rebelling tribes were officially confiscated as German crown lands (Kössler 2008, Katjavivi 1988, Soggot 1986). The result of that confiscation, combined with Germany’s prior land acquisitions, was that, by the end of German rule, south and central Namibia (the area under German control) was “almost devoid of visible settlements, ordered into neatly fenced-in farms” (Kössler 2008:315).

Because of the costs associated with suppressing the rebellions, in 1905, Germany instructed Namibia’s colonial government to restrict its police protection to the smallest possible
area (Miescher 2012). In response to this order, the colonial government established a “police zone,” the northern boundary of which generally tracked the rinderpest cordon, subsequently referred to as the “red line” (Parker 1991; Miescher 2012). The northern 30% of current-day Namibia remained largely free of German control, with the colonial governor issuing a 1906 order restricting travel and trade in the Ovambo region by anyone other than “indigenous” tribes (Siiskonen 1990; Miescher 2012). Despite their subsequent attempts to do so, the German colonial authorities simply lacked the necessary military might to subjugate the Ovambo, who retained a level of autonomy throughout the remainder of German rule (Soggot 1986).

ii. **Environmental policy: Justifying indigenous disenfranchisement**

Compared to the more capitalist approach found in Germany policymaking, colonial decision-making was highly influenced by a “state and science” collaboration between administrators and scientists, permitting the development of an autocratic, scientific approach to policy development (Rollins 1999). Beginning in the late 1800s, policymakers in Namibia focused on two perceived environmental problems: desiccation and the overharvesting of wildlife (Siiskonen 2015). Regarding the former problem, policymakers were concerned that the misuse of land would lead to altered rainfall and the drying up of surface water (ibid). Regarding the latter, colonial administrators, along with interested parties in Germany such as hunters and members of the *Deutsche Kolonialgesellschaft* (German Colonial Society), were worried that Namibia’s wildlife (or, at least, those species of wildlife the Germans considered worth preserving) were at risk of extinction from overhunting (Rollins 1999). Wildlife was viewed as an important economic resource within the colony, particularly given Namibia’s limited capacity for agricultural development (Wenning 2008; Miescher 2012).
Policymakers placed the onus for both issues primarily on native Namibians. Colonial authorities blamed, at least in part, variations in rainfall and water availability on the lack of any environmental ethos among the Herero and Nama (Siiskonen 2015). Native Namibians were also viewed as endangering wildlife by hunting in a manner that was profligate, ignorant, and unsporting (DeGeorges and Reilly 2009; Rollins 1999; Miescher 2012). Engaging in what Rollins (1999) describes as “environmental chauvinism,” the perceived excesses of native Africans were seen as a stark contrast with the environmental ethos and unique scientific and economic capacity of the German settlers (Rollins 1999, Botha 2005). This disparity provided a reason that “the land itself would be turned against its native inhabitants: it would cry out in the language of environmentalism to be relieved from their incompetence, and thereafter would gratefully bear witness to the care of the colonial masters” (Rollins 1999:196).

The German colonial government sought to control the hunting of Namibia’s wildlife through the introduction of a regulatory system of hunting bans and licenses (Berry 1997; Cioc 2009; Wenning 2008). Hunting regulations first appeared in 1892, requiring would-be hunters to first seek the permission of the colonial governor, and imposing a total ban on the hunting of elephant cows and calves and a seasonal ban on the hunting of ostriches (Joubert 1974:35). The first hunting ordinance entered into force in 1902, prohibiting the use of traps or snares (two hunting approaches traditionally favored by native Namibians), closing large areas off to hunting altogether, and requiring a permit by all hunters (both African and European) on lands under government control (Joubert 1974:35; Miescher 2012:52). The German-controlled area of Namibia was divided into districts, each under the authority of its own district chief. The district chiefs were granted the authority to establish and enforce seasonal hunting restrictions for particular game species (Joubert 1974).
In 1907, the German government established three expansive game reserves incorporating land protected under the 1902 ordinance (Joubert 1974:35). The largest of those reserves, which included the modern day Etosha National Park and was known simply as “Wildschutzgebiet Nr. 2” (Game Reserve No. 2), covered approximately 80,000 square kilometers – an area roughly the size of modern-day Austria (Berry 1997:4). At the time of its creation, this reserve was the largest game reserve in the world, although it was subsequently reduced in size by over 70% (ibid.). The other game reserves encompassed 10,000 square kilometers of the Namib Desert, and an area in the northeast of the country, respectively (Wenning 2008:7).

Similar nature reserves in East Africa were purposefully designed to harken back to Jagdregal hunting estates, conveying “the idealized picture of . . . an untamed wilderness” where the wildlife was referred to as “imperial game” (Gißibl 2006:126). It is unclear whether the Namibian reserves were generated with the same nostalgia in mind but, nevertheless, they instituted the same sorts of restrictions as found on Germany’s historical royal reserves by banning all hunting by German and Bantu-speaking Africans. Only groups such as the Hai//om San were allowed to remain and hunt within the reserves, as they were viewed as being part of the natural environment (Suzman 2004).

In 1909, the German government adopted a hunting statute that prohibited hunting without a permit anywhere other than in enclosed, privately owned lands, and this statute included hunting on native reserves by black Namibians (Hinz 2003:21). The statute also prohibited many forms of hunting traditionally favored by native Namibians: snares, traps, and pits (ibid). Thus, by the end of its rule, Germany effectively removed the capacity of black Namibians within the police zone to engage in any form of legal wildlife governance through
land dispossession and the enactment of restrictions on hunting in the areas within which they lived.


After the First World War, South Africa governed Namibia as a League of Nations mandate. However, while the formal colonial ruler changed, little else did from the standpoint of most black Namibians. For most of its rule, South Africa continued and expanded the German policy of land disenfranchisement and exclusionary wildlife protection.

i. Land policy: A continuation of Germany’s policy of dispossession and relocation, and the development of traditional authorities

Like Germany before it, South Africa removed indigenous residents from their land and allocated it to white settlers (UNESCO 1974). Further, South Africa continued Germany’s policy of relocating ethnic groups within Namibia to reservations. In 1923, South Africa allocated 2 million hectares in southern and central Namibia as native reserves (Katjavivi 1988:14) and forcibly relocated black residents found in crown lands to those reserves (Silvester, et al. 1998:19). In addition to representing only 3.5% of the available land (allocated to 90% of the population), the native reserves were located in the region’s most barren and least productive locations (Katjavivi 1988:14). By 1937, nearly the entire black population of the police zone was confined to the reserves, and the movement of black individuals was strictly controlled by a combination of curfews and travel, vagrancy, labor, and identification laws (Soggot 1986; Katjavivi 1988).

South Africa pushed the German-established red line further north and, in 1915, gained a degree of control over the Ovambo kingdoms in northern Namibia (Katjavivi 1988; Silvester et. al 1998). Unlike with the Germans, however, the focus of South Africa within the northern region was not land expropriation; instead, its residents were viewed as a source of labor for the
white population to the south (Katjavivi 1988; Silvester, et al. 1998; Emmett 1999; Sarkin 2009). Consequently, South Africa never opened the land above the red line to white settlers, and land governance in that region continued largely in accordance with traditional rules (Parker 1991).  

Meanwhile, forced removals of black Namibians from land in the police zone continued. In 1958, for example, South Africa relocated 400 Damara living near Windhoek to a reserve further north, with part of the newly vacated land designated as a game reserve and the remainder allocated to white settlers (Katjavivi 1988:47). In the 1960’s, South Africa began implementing its Odendaal Plan, which envisioned the creation of separate, semi-self-governing “homelands” for each of the black ethnic groups within Namibia (Kaela 1996:82). The conceptualization of ethnic homelands marked a departure from South Africa’s previous stance, which viewed the reserves as being open to black Namibians of all ethnicities (Kössler 2008). South Africa never fully implemented the Odendaal Plan, and it appears unsettled how many people were ultimately relocated as a result of the plan.

These policies (representing the completion of the dispossession efforts begun by Germany) resulted in a grossly inequitable land distribution. In the middle of the twentieth century, white farmers in Namibia possessed roughly 50% of the country’s agricultural land, while black farmers (who made up the vast majority of Namibia’s population) were formally allotted only 25%, mostly above the red line in the north (Sachikonye 2004:65). At the time of Namibia’s independence in 1990, the percentage of farmland owned by whites – who made up 6% of Namibia’s population at that time – had increased slightly to 52% of all agricultural farmland, while black residents had access to the remaining 48% (Hunter 2004:1). In total, an

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6 Emmett (1999) observes that the system of land allocation by tribal kings and headmen evolved over the course of the early 20th century to become more of a rent-based system.

7 Silvester et. al (1988) notes that the Odendaal Plan “remains to be researched in depth,” and I am unable to identify any work that discussed in any depth South Africa’s attempts to implement the plan.
estimated 4,205 mostly white-owned southern freehold estates held 44% of all available land in Namibia (regardless of suitability for agriculture) while roughly 160,000 northern black households occupied 43% of available land (Sachikonye 2004:66).

Much of the land occupied by black households fell under common-property ownership, where the allocation of usufruct rights often occurred outside of any formally sanctioned legal procedure. South Africa employed a system of indirect administration that relied on local tribal authorities, variously referred to as “leaders,” “headmen,” “chiefs,” and “councilors” (depending, in part, on the esteem in which South African officials held the particular tribe or group) (Friedman 2005). While there may be some question as to how “traditional” the chosen headmen were within Namibia’s various tribes, their role as subordinate officials in the political process was clearly entrenched by the time the country gained its independence (Corbett and Daniels 1996, Friedman 2005).

In addition to their limited formal *de jure* powers, the traditional authorities frequently had much more expansive *de facto* power within their communities. For instance, despite the law expressly prohibiting headmen from allocating land, reserving that power solely for magistrates and superintendents within the colonial administration, the traditional authorities effectively did just that (Corbett and Daniels 1996, Werner 2011). Thus, when a resident in a communal area applied for permission to occupy communal land, the magistrate or superintendent would seek out and defer to the advice of the relevant traditional authority (Corbett and Daniels 1996, Werner 2011). Some traditional authorities also had their own tribal court systems that resolved disputes separately from the formal South African courts (Jones and Luipert 2002).
ii. Environmental policy: The increased consolidation of governmental control over wildlife and the beginnings of CBNRM

South Africa continued to apply the environmental laws enacted by Germany, including keeping the nature reserves enacted under the previous regime in 1902 (Joubert 1974). However, in place of an absolute ban on hunting, South Africa allowed the purchase of special hunting licenses within the reserves (Carpenter 2011). During the 1920s, South Africa expanded the scope of existing import and export regulations on wildlife and placed protections on certain plant and animal species (ibid.). In 1933, South Africa participated in the International Conference for the Protection of Flora and Fauna of Africa (the 1933 London Convention) (Convention Relative to the Preservation of Fauna and Flora in Their Natural State 1933). Among other things, the parties agreed to establish national parks and “strict natural reserves” and to afford protection to species identified in two lists (Class A and B). Regarding the listed animals, the hunting of Class A animals was to be completely banned except for “special circumstances, solely in order to further important scientific purposes, or when essential for the administration of the territory.” Class B species were afforded a lower level of protection, but still could not be “hunted, killed, or captured, even by natives, except under special license granted by the competent authorities” (italics added). The parties also reserved the ability to extend the Class A and B protections to additional species within their respective territories.

The parties to the 1933 London Convention clarified that the prohibitions regarding the hunting of the identified species did not, as a matter of course, extinguish hunting rights already possessed by native Africans pursuant to a treaty, concession or administrative permission in “those areas in which such rights [had] already been definitively recognised by the authorities of the territory” (1933 London Convention). The plain language of the document, however, made no such exception for traditional hunting rights that had not previously been recognized by the
colonial powers (such as in previously unconquered territory), or in any way prevented parties from revoking existing native hunting rights.

Despite its participation in the 1933 London Convention, South Africa made few substantive changes to its existing conservation policy (Joubert 1974; Botha 2005). Joubert (1974:36) sums up the period from 1915 to the early 1950s as a “period of stagnation . . . during which time virtually no progress was made regarding conservation as a whole.” He describes Namibia as simply being too expansive to allow for effective governmental enforcement of hunting restrictions, particularly given that South Africa did not dedicate any full-time officials to the issue of nature conservation (ibid). Nevertheless, one seemingly minor policy change is worth discussing here. In 1919, the colonial government enacted taxes on the ownership of dogs, although an exception allowed rural white residents to keep one dog as a watchdog without paying the tax (no such exemption existed for black Namibians) (Authority 1919). Black Namibians traditionally used dogs in hunting, and so this seemingly innocuous tax may well have had a significant and disproportionate impact on the livelihoods of those groups (Botha 2005).

Starting in the 1950s, South Africa began much more aggressively pursuing nature conservation. The period from 1953 and 1972 saw dramatic increases in full-time staff associated with nature conservation and tourism (from 15 to 593), allocated budget (from R16,000 to R2,112,000), and formal conservation areas (from 3 to 12) (Joubert 1974:36). These efforts to protect wildlife coincided with increased restrictions on the utilization of wildlife by black Namibians. Despite the existence of Germany’s 1909 statute, the South African government had had generally permitted (whether voluntarily or as the result of a lack of enforcement capacity) black Namibians to use wildlife resources found on the native reserves (Botha 2005).
The 1950s saw a change to this *status quo* with the passage of an ordinance that strictly limited hunting across the country (Game Preservation Ordinance 1951) and the eviction of San subsistence hunters from Etosha National Park in the first half of the decade (Botha 2005). The ordinance prohibited the hunting of all game between September and April and prohibited entirely the hunting of protected\(^8\) or big game\(^9\) without a hunting license (Game Preservation Ordinance 1951). It contained some exceptions for private landowners and occupiers, providing them with the bounded ability to hunt some small game and kill game that threatened grazing, crops, or livestock. If they had a fenced property, the landowners had unlimited ability to kill any or all of a number of game species, but did not include any such exceptions for residents of native reserves or common property areas (ibid). The Game Preservation Ordinance also explicitly banned the use of dogs in hunting and generally forbade hunting with anything other than a rifle – a hunting tool that was beyond the financial means of most black Namibians (Game Preservation Ordinance 1951; Botha 2005). The strict enforcement of this Proclamation had the effect of ending subsistence hunting on the native reserves (Botha 2005).

In 1962, the South African government declared all wild game to be protected, state-owned assets (Libanda and Blignaut 2008:41). Six years later, in 1968, the government once again returned some rights to private landowners – granting them the ability to sustainably utilize the wildlife on their properties for tourism, meat, and trophy hunting (Summers 1999). Equivalent rights, however, were not granted to residents on communal lands, so wildlife remained the property of the state in the areas typically occupied by black Namibians (Alpert 1996; Boudreaux 2008).

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\(^8\) Protected game included elephant, eland, giraffe, hippopotamus, impala, rhinoceros, sable antelope, and zebra.

\(^9\) The following game were listed as big game: buffalo, oryx, hartebeest, kudu, ostrich, and tsessebe.
The early 1970s saw a brief decentralization of wildlife governance to regional
governments, with the Ovambo and Kavango regional assemblies passing their own legislation
allowing traditional authorities the ability to hunt certain game and to issue limited hunting
permits to their followers (Hinz 2003:24). In 1986, however, the Nature Conservation
Amendment Act reconsolidated decision-making and, in the process, repealed the Ovambo and
Kavango legislation. The next year, in 1987, the South African government passed an exemption
that allowed the Nyae Nyae San to engage in “traditional” hunting without a permit (ibid).
Ironically, the same white apartheid government that granted that exemption also took great
pains to direct what could legally be considered to be the San’s tradition practice – including
specifying the length of their bows and design of their arrows, the materials from which they
crafted their snares, and the method by which they could track down wounded animals (i.e., on
foot). Hinz (2003) astutely notes that this exemption both prohibited the natural evolution of
traditions and conflated environmental and cultural conservation.

In 1982, the Namibian Wildlife Trust (NWT) moved to address the rampant poaching of
black rhinos in the Kunene region – only 66 rhinos remained from a population estimated to
number from 250 to 350 in 1970 (Martin 1994:43). At this time, the entire governmental staff
tasked with patrolling the 9 million hectares of the Koakoveld (part of the current-day Kunene
region) consisted of two individuals: a government Nature conservator and his Herero assistant
(ibid). The NWT, through Garth Owen-Smith, a former governmental agricultural extension
officer and game ranger, began discussions with communities regarding issues of wildlife
poaching in the area and the communities’ loss of livestock due to a severe drought (Owen-
Smith and Jacobsohn 1989; Owen-Smith 2010). The result of these meetings was the creation of
a community game guard in which members of the local communities patrolled core rhino
territory, and whose salaries were paid by the NWT (Martin 1994; Jones 1999; Owen-Smith 2010). Incidents of rhino poaching dropped dramatically after the development of the community game guard, and the approach was subsequently expanded to communal lands in Namibia’s northeast Caprivi region (Martin 1994; Jones 1999; Owen-Smith 2010). After the expansion of the community game guard system to the Caprivi, Owen-Smith and Jacobsohn established an NGO, the Integrated Rural Development and Nature Conservation (IRDNC), dedicated to bringing wildlife benefits to residents in Namibia’s communal areas (Jones 1999).

By the late 1980s, tourism had dramatically increased in the former Kaokoveld which, until 1978, had been largely closed to white visitors (Owen-Smith and Jacobsohn 1989). In response to a rise in tension between the tourists and local residents, Owen-Smith and an anthropologist named Margaret Jacobsohn met with stakeholders near Puros, a spring found in the lower Hoarusib River that served as both a tourist attraction and an important source of water to local residents (ibid). As a consequence of those meetings, tourists were asked to pay a levy of R25 (approximately $5 at the time) to enter the area (Owen-Smith and Jacobsohn 1989; Jones 1999). The money was provided directly to the community at Puros, who were able to decide for themselves how it should be spent (Owen-Smith and Jacobsohn 1989). The success of this pilot “Puros project” in generating local support for conservation led to the creation of a second such project in the Caprivi in 1990 (Owen-Smith 2010).

d. Namibian Independence (1990–present): The Adoption of Community-Based Conservation as a Conservation Tool and Redress for Inequality

i. Land Policy: The continuation of communally owned property and the codification of land allocation rights of traditional authorities

Namibia gained its independence from South Africa in 1990. While it has engaged in some land reforms, such as reallocating white freehold estates to black Namibians, post-independence Namibia has retained the distinction between the freehold estates and the
communal-property lands. It also continues to observe and enforce the red line as an agricultural barrier.

In 1991, the newly independent Namibian government divided the country into administrative regions as part of an effort to undo South Africa’s racially oriented “Bantustan” policy (Tötemeyer 2010). Each of these administrative regions is divided into 6-12 local authorities, which function at the municipality, town, and village level. The local and regional authorities have elected councils, with the local authorities each electing one representative to the regional council (ibid). The Namibian Constitution, however, continues to recognize the existence of “customary” law, and expressly provides that customary and common law is valid to the extent that it does not conflict with statutory or constitutional law (Article 66 of the Namibian Constitution).

Customary law is determined and administered by “traditional authorities” which, as defined in the 2000 Traditional Authorities Act (the “TAA”), consist of a community chief and “senior traditional councilors” (Traditional Authorities Act, 2000). Under the TAA, community chiefs are either a member of the royal family of the traditional community or, if no royal family exists, an individual selected from within the traditional community. The senior traditional councilors are, at the discretion of the chief, either appointed by the chief or popularly elected by community members. Traditional authorities are empowered to preside over disputes regarding “any customary matter” between members of the community and exercise other customary powers. The Act limits the scope of the traditional authority’s jurisdiction to members

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10 The TAA defines a “traditional community” as being “an indigenous homogenous, endogamous social grouping of persons comprising of families deriving from exogamous clans which share a common ancestry, language, cultural heritage, customs and traditions, who recognises a common traditional authority and inhabits a common communal area, and may include the members of that traditional community residing outside the common communal area.”
of that community and to non-members that voluntarily submit (either expressly or by conduct) to the customary law of the community. While the Act differentiates traditional authorities from what it refers to as “government organs,” it nevertheless provides for traditional authorities to be remunerated for their services from the State Revenue Fund.

The TAA expressly bounds the powers of the traditional authority by statutory and constitutional law, governmental policies, and the authority vested in regional and local authority councils. In practice, however, these boundaries are not always clear. For example, the TAA (like its predecessor, the 1995 Traditional Authority Act\textsuperscript{11}) requires that traditional authorities ensure that their members sustainably use natural resources, but it does not include land allocation amongst the enumerated powers granted to the traditional authorities. Nevertheless, as they had done prior to independence, traditional authorities continued to allocate land for use by their members after the TAA’s adoption (Werner 2011).

In 2002, Namibia passed the Communal Land Reform Act (“CLRA”), recognizing the role of traditional authorities in allocating and cancelling customary land rights (Communal Land Reform Act, 2002). Yet, at the same time, the language of the CLRA also potentially constricted this power through the creation of regional boards to, \textit{inter alia}, “exercise control” over the practice through the ratification and registration of land right determinations (and absent such ratification, the allocation or cancellation has no legally recognized effect). In response to complaints by a number of traditional authorities over the language of the CLRA, the Minister of Lands of Resettlement affirmed that traditional authorities would retain their ability to allocate communal land (Haufiku 2013).

\textsuperscript{11} Traditional Authorities Act, 1995
ii. Environmental Policy: The statutory creation of conservancies and the involvement of traditional authorities in conservancy functions

After independence, Namibia’s Ministry of Wildlife, Conservation and Tourism (the predecessor to the current Ministry of Environment and Tourism (MET)) worked with Owen-Smith and Jacobsohn to formalize the game guard and tourist levy approach first adopted in Puros (Jones 1999). In 1995, the Ministry issued a policy entitled “Promotion of Community Based Tourism” (the “Conservancy Policy”). The Conservancy Policy’s stated goal is to provide “a framework for ensuring that local communities have access to opportunities in tourism development and are able to share in the benefits of tourism activities that take place on their land” (MET 1995). In particular, the Conservancy Policy notes the need to enhance the rights enjoyed by communities over tourism resources. It proposes that conservancies are the key to redressing past inequalities and views conservancies as a key tool by which communal residents could gain rights over environmental resources – particularly wildlife rights – and therefore attract tourism-related income. The Conservancy Policy states that the MET will support communities’ establishment of conservancies and tourism ventures. It also provides for the channeling of a “substantial share” of funds for investment in Namibian tourism to communal areas (ibid).

The next year, the Namibian government enacted the Nature Conservation Amendment Act of 1996 (“Conservancy Act”), granting conservancies the same rights enjoyed by the freehold commercial farmers. The Conservancy Act provides that any group of people residing on communal land can apply for conservancy status (MET 1996). To be recognized as a conservancy, applicants must have a registered membership, a legal constitution, a representative management committee, an outline of a benefit distribution plan, and defined geographic boundaries (Snyman 2012). Once a community is granted conservancy status, it possesses the
same *de jure* rights as commercial farmers to hunt, capture, cull, and sell huntable game (oryx, springbok, kudu, warthog, buffalo, and bush pig) (WRI 2005). Furthermore, the community has the right to apply to the MET for permits to use quotas of protected game for trophy hunting (ibid). The Conservancy Act does not confer any additional rights to the conservancy land itself or to mineral, fishery, or forest resources located within the conservancy’s territory (Harring and Odendaal 2012). And, it should be noted that the conservancies’ hunting rights are revocable by the MET Minister (MET 1996).

As written, the Conservancy Act would appear to allow the conservancies to set their own quotas regarding huntable game. However, based on interviews with traditional authorities and conservancy representatives in the Anabeb, Omatendeka, Puros, and Sesfontein conservancies (conducted in April 2017) and informal conversations with Namibian conservation practitioners (across several visits in 2016-2017) it appears that this is no longer the case. As relayed to me, there were concerns that the conservancies were overharvesting huntable game, often by hunting wildlife (or selling hunting rights to outsiders) to sell to meat processors in cities to the south. Consequently, Namibia’s Ministry of Environment and Tourism now generates quotas for huntable game for each of the conservancies.

The Conservancy Act does not expressly provide for the involvement of traditional authorities in the creation or management of the conservancies. Nevertheless, the role of the traditional authorities in land allocation and dispute resolution means that, in practice, they have a substantial degree of jurisdictional overlap with the conservancy administration. In recognition of this fact, guidelines promulgated by the Ministry of Environment and Tourism call for the involvement of traditional authorities in key areas of the conservancies’ formation and management (MET 2013). Of note, the guidelines observe that “[a]lthough the conservancy
legislation and regulations do not prescribe the role of Traditional Authorities in conservancies, in practice it is important for them to play a role because of their legal authority in land issues and their duty under the Traditional Authorities Act to play a role in conservation” (ibid.).

The guidelines specify the following roles for traditional authorities:

- Unless specified otherwise in the specific conservancies’ constitutions, individuals are only eligible for conservancy membership if, among other things, they have resided in the conservancy area for three years with the permission of the conservancy’s traditional authority.

- If a dispute arises that involves the conservancy committee as a party, the traditional authority is appointed as a mediator if the parties cannot agree on another intermediary.

- Traditional authorities should be represented on conservancy committees in an advisory role.

- Conservancies should consult with traditional authorities when determining zoning (such as grazing, tourism, or conservation areas) within the conservancies.

- Benefits to the traditional authorities from the conservancies should be stated in the conservancies’ benefit distribution plans.

Namibia recognized its first conservancies in 1998: the Nyae Nyae and Salambala in the east of the country and the Torra and ≠Khoadi-/>Hôas in the western Kunene region. As of this writing, 86 conservancies have been gazetted, covering approximately 166,045 km² and including an estimated 189,230 people (about 20% and 7.5% of the country’s total land area and population, respectively) (NACSO 2019).
e. **Structural Challenges to the Success of the Conservancy Program**

The conservancy model is a notable departure from the highly centralized and exclusionary policies of the German and South African governments, but its devolution of authority is incomplete. First, while it is subject to procedural limitations set out in the Conservancy Act, the MET retains the right to withdraw a conservancy’s recognition. As such, the conservancies lack permanent *de jure* rights to their wildlife resources. Second, the conservancies also lack the *de facto* capacity to fully govern those resources, as the MET currently issues permits to the conservancies for both huntable and protected game. Third, despite their formal recognition by the MET, the conservancies lack any easy means by which to regulate the entry of outsiders (Boudreaux 2008).

Regarding the lack of permanent *de jure* rights, in my interviews with multiple conservancy authorities and traditional leaders across four conservancies, no one ever mentioned the legal right of the MET to withdraw the recognition of the conservancies. Therefore, it may be that most conservancy residents are either unaware of the possibility or consider the revocation of their conservancy’s status to be highly improbable. Further, in their responses to my survey (administered across those same four conservancies), no participants made any allusions to the possibility of revocation. Consequently, the mere legal possibility of the withdrawal of conservancy recognition appears unlikely to undermine to have any real impact on the conservancies’ governance function.

The latter two limitations – the lack of a governance privileges and the inability to regulate entry – are potentially more significant threats to the success of the conservancy program. Starting with the issue of governance rights, it appears that residents are very cognizant of the inability of their conservancies to make their own decisions about wildlife harvesting. The topic frequently arose in my interviews of conservancy representatives and traditional
authorities. And, in response to a survey question asking them to identify who owned a range of
different species, less than one third of respondents answered that small herbivores and big
herbivores (31.23% and 31.5% of respondents, respectively) were owned, at least in part, by
their conservancy (these two sets of animals make up the “huntable game” identified in the
Conservancy Act). By comparison, around 42% of participants responded that the animals
were not owned by anyone, approximately 24% felt that they belonged to the government, and
less than 3% felt that the ownership was unknown or fell under a different category.

Whether control over some or all game species should be fully devolved to the
conservancies is a matter of debate (it is worth noting that, in my interviews with them, even
conservancy representatives and traditional authorities were split on this issue). And, CBNRM
participants can be motivated to conserve wildlife by a multitude of considerations, including
existence values and the belief that the presence of wildlife benefits the communities by
attracting tourism. Nevertheless, the fact that approximately two-thirds of survey respondents
lack a sense of legal ownership over wildlife in the conservancies raises a concern that the
conservancies fall short in fostering among their residents a sense of durable interests in wildlife
- a core pillar on which the success of CBNRM approach is predicated (Mbaiwa 2004; Lepetu, et

The inability of the conservancies to exclude outsiders also poses a challenge to their
long-term viability (Jones, et al. 2015). The economic incentives and self-policing at the heart of
the CBNRM approach is unlikely to hold modify the behavior of outsiders who are not
integrated into the local communities (EAL and GE 2015). Absent the ability to regulate entry,
the conservancies may struggle to prevent their wildlife from becoming an open access resource

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12 I conducted in-person interviews in April 2017 and administered a field survey in October and
November 2017. The design of this survey is further discussed in Chapter 3 of this dissertation.
The conservancies can certainly organize patrols to monitor outsiders once they have entered but, at least in the four conservancies included in my fieldwork, the patrols are unarmed and lack the legal authority to apprehend suspected poachers. As explained to me by one headman in the Anabeb conservancy, the patrols can only radio the police and hope they arrive in time to catch the suspects before they flee.

The extended drought in Namibia’s north has also caused a number of Himba pastoralists from the Epupa region to migrate southward into some of the Kunene conservancies. While I was unable to independently verify their claims, conservancy officials, traditional authorities, and survey respondents (excepting the migrants themselves) all regularly complained about how the “illegal” settlers had moved their cattle into restricted areas of the conservancies and were competing for grazing. These claims echo those made to other researchers working in northern conservancies (Bollig 2016). Lacking any other more expedient means of removing the settlers, the conservancies included in my fieldwork were forced to turn to the courts, filing an action in 2015 seeking the settlers’ eviction (ibid). At the time of this writing, over four years later, I have not been able to find any final determination of that lawsuit.

A fourth potential hurdle to the overall success of the conservancy model comes, not from the legal empowerment of the conservancies themselves, but from the symbolism attached to the conservancies as a post-apartheid form of empowerment. For many communities, a significant draw behind forming a conservancy was the legal status that accompanied it – a sense that recognition as a conservancy represented a reclaiming of land ownership that had long been

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13 These claims are admittedly quite murky. Bollig (2016) observes that the boundaries of the different usage zones in the conservancies can be ambiguous, disputed, and sometimes unknown even to registered conservancy members. Further, based on informal conversations with conservation practitioners, it appears that at least some of the settlers may have received initial permission from one of the multiple traditional authorities in their respective conservancies. Nevertheless, the situation remains that the conservancy authorities lack the means to easily address issues involving encroachment by outsiders.
denied them by the German and South African governments (Bollig 2016). As a result, despite its creation as means for affording communal areas the same wildlife management rights as freehold estates, a number of Namibia’s conservancies actually have marginal wildlife potential (NACSO 2019).

The administration of a conservancy requires expenditures on, among other things, employee salaries and transportation, and it is unlikely that many conservancies will ever realize sufficient wildlife revenue to cover their administrative costs. Of Namibia’s 86 conservancies, less than half (42) have entered into joint ventures with tourism lodges (NACSO 2019). In 2017, only 39 conservancies reported being able to cover their operational costs from their own income (out of 54 that provided this information) (NACSO 2018). The conservancy program has attracted a substantial amount of investment from NGOs and foreign agencies over its lifetime. However, this sort of funding is notoriously fickle, as donors are often tempted to switch their expenditures to support the next cure-all approach. If the conservancy program loses a significant amount of its non-governmental funding, the brunt of the costs for supporting the conservancies will fall squarely on the Namibian government. At that point, the government will have to choose whether to continue subsidizing those conservancies that do not have the potential to become self-sufficient.14

In short, the adoption of CBNRM in Namibia represents, in several important respects, a retreat from the paternalistic and increasingly rigid wildlife management policy utilized by Namibia’s colonial governments. Residents of communal lands now have a legally recognized interest in their wildlife resources, can elect a committee to make zoning and governance decisions, and are able to benefit from the revenues generated by tourism activities.

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14 Adams and Hulme (2001) identify this sort of scenario – where wildlife resources are insufficient to yield a sustainable revenue flow – as one in which community-based conservation is likely an inappropriate conservation policy.
decisions, and have the potential to profit from wildlife-based goods and services. However, this retreat is incomplete and provides only a partial return to the pre-colonial autonomy enjoyed by those living in the communal areas. This incomplete devolution of rights, and the marginal potential for many conservancies to realize sufficient wildlife-generated profits, represent an ongoing challenge to the long-term efficacy of Namibia’s conservancy model.

3. Common Criticisms of the CBNRM Approach and Formation of Specific Research Questions

The challenges of the incomplete devolution of rights and the limited regional economic potential of wildlife are not limited to Namibia’s CBNRM efforts (see, e.g., Levine and Wandesforde-Smith 2004; Balint and Mashinya 2006; Shyamsundar 2008; Rozemeijer 2009), and the inability of many CBNRM efforts to overcome these and other hurdles give rise to criticism that the approach simply does not work at either wildlife conservation or economic development (see Hutton, et al. 2005; Nunan 2006). For instance, Blaikie (2006:1947) asserts that “a generalized conclusion may be fairly confidently made that CBNRM programs in central and southern Africa have substantially failed to deliver the promises to both communities and the environment.” Sceptics of the approach also commonly contend that CBNRM relies on an unrealistic and romanticized concept of “traditional communities;” commercializes natural resources, which can motivate overharvesting; encourages elite capture, corruption, nepotism, and inequity at the local level; and is based on un-scientific postmodern logic (Child and Lyman 2005; Hutton, et al. 2005; Murphree 2009).15 These criticisms give rise to three specific inquiries that I address in the following chapters of this dissertation.

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15 Other common criticisms are that CBNRM tries to fix fictitious problems; leads to chaotic governance; disenfranchises national constituents that also have an interest in the governed resource; devolves governance to less competent local communities that lack the capacity to compete in a market economy, leading to their reliance on monopoly players; fosters reliance on tourism, which is an unstable and
First, calls for a return of wildlife conservation policy to the fortress model approach rely primarily on the perceived failure of the CBNRM approach, and of other approaches that seek to recruit local participation in wildlife conservation (such as allowing neighboring residents resource use rights within protected areas) (Terborgh 1999; Oates 1999; Kumar 2006). Yet, the calls for a return to centralized wildlife governance must also be weighed against the historical track record of that approach – indeed, the development of the CBNRM approach occurred, in part, because the perceived failures of centralized wildlife governance. The question, then, is not whether the CBNRM approach has a high rate of failure, but how its success rate compares to that of alternate policy approaches. Thus, in Chapter 2, I investigate whether, using a common measure of success, outcomes in CBNRM areas materially differ from those in governmental protected areas.

Second, critics challenge the reliance of the CBNRM model on the commodification of wildlife. This criticism generally focuses on the potential of commodification to encourage unsustainable harvesting (Murphree 2009). That concern calls into question the CBNRM model’s base economic assumption – that people will sustainably manage wildlife if they perceive that the benefits they receive from wildlife make their lives better. In Chapter 3, I address one part of that core assumption. Does the receipt of wildlife-derived economic benefits cause CBNRM residents to feel that their lives have improved? Relatedly, do certain benefits have more of a positive impact than others?

Finally, criticisms regarding the romanticism of rural “communities” as singular, homogenous entities are perhaps well founded, in that rural communities are (like those found elsewhere) inherently messy entities, with constituents that are heterogeneous in their interests

 unreliable source of income; and, due to the presence of middle-man actors, results in the local “producer” receiving only a small and inequitable portion of the wildlife-generate revenue (ibid).
and across a host of socioeconomic, racial, ethnic, cultural, and geographic variables. But, to what extent do each of these various forms of heterogeneity matter? The impacts of some forms of heterogeneity, such as wealth or income, have been the focus of robust bodies of literature, but that research largely takes place in a risk free environment, whereas participants in wildlife governance face the threat of losses from human-wildlife conflict (a topic that is further discussed in Chapter 4). The potential effects of other forms of heterogeneity, such as risk exposure, have only recently started being explored, leaving little guidance to be found in the academic literature. In Chapter 4, I investigate the following questions: Does the presence of either risk or economic heterogeneities impact collective action in CBNRM communities when those residents are faced with the threat of human-wildlife conflict? What is the impact of interactions between these two forms of heterogeneity?
CHAPTER 2

A CROSS-NATIONAL COMPARISON OF THE EFFICACY OF CBNRM AND NATIONAL GOVERNANCE APPROACHES ON THE PROTECTION OF THE AFRICAN ELEPHANT

As previewed in Chapter 1, an ongoing debate exists about whether wildlife conservation should focus primarily on the strict protection of wildlife resources (such as through the national governance of parks and game reserves) or on improving the livelihoods of rural communities through sustainable development (Oldekop, et al. 2016). Sometimes referred to as the “Back to Barriers” movement or the “New Conservation Debate,” this disagreement debate rests, in part, on differing underlying moral positions, but also focuses on perceptions about the relative efficacy of the two approaches (Miller, et al. 2011). In other words, is one approach inherently superior to the other?

CBNRM remains a popular sustainable development approach to wildlife conservation (Dyer, et al. 2014). But, given the debate over the efficacy of the approaches, surprisingly little research exists analyzing the relative efficacy of the CBNRM and national governance approaches. The small number of existing studies have produced conflicting findings. This paper contributes to that underdeveloped area of study by exploring the respective impacts of CBNRM and national governance on the protection of economically valuable wildlife. Specifically, it uses the proportion of elephant deaths resulting from illegal kills, recorded in 19 African countries between 2002-2015, to analyze the relationship between the two governance types and the likelihood that an observed elephant death will be the result of an illegal kill.

1. Background

   a. A Brief Review of the Protectionism versus Sustainable Use Debate

   A debate exists regarding whether species and biodiversity conservation efforts should focus on strict protectionism or sustainable use. Often alternately referred to as “fines and
fences” or the “fortress model,” the strict protectionism approach relies on a combination of the implementation of trade restrictions, the creation of protected areas, and the nationalization of wildlife, involving the adoption of laws restricting or prohibiting the harvesting of wildlife (Spiteri and Nepalz 2006).

The strict protectionism approach has been the dominant conservation approach for over a century (Adams and Hutton 2007). However, the approach is criticized as being costly, as the creation of protected land and the nationalization of wildlife require constant policing and enforcement of those resources, while trade restrictions necessitate increased border control and monitoring of domestic markets (Abensperg-Traun 2009; Murphree 2009). Critics also point out that the creation of protected areas displaces and/or restricts the traditional livelihoods of rural populations (Poirier and Ostergren 2002; Ormsby and Kaplin 2005; Adams and Hutton 2007; Agrawal and Redford 2009; Kabiri and Child 2014). Concerns over these issues led to the adoption in many places of CBNRM programs in the 1980s and 1990s (Roe 2008; DeGeorges and Reilly 2009; Abensperg-Traun 2009; Murphree 2009).

The CBNRM approach has itself been challenged, as critics contend that CBNRM regimes often fail to achieve many or all of their stated economic or environmental objectives (Blaikie 2006; Fletcher 2012). These perceived shortcomings have led to calls for a renewed focus on national governance approaches. Sustainable use critics stress the need for increased governmental intervention in conservation (McShane, et al. 2011; Miller, et al. 2011). This intervention can come in the form of traditional “strict” national parks in which no extractive human activity is allowed (Kramer, et al. 1997; Terborgh 1999; Foreman 2006) but can also consist of the consolidation of governmental control over geographic areas in which some extractive activity occurs (DeGeorges and Reilly 2009).
b. **Existing Comparisons of the Efficacy of the Two Approaches**

Relatively little empirical research exists on the overall effectiveness of formal protected areas in species conservation, and even less compares the performance of protected areas falling under different governance approaches (Western, et al. 2009; Craigie, et al. 2010; Ihwagi, et al. 2015; Dudley, et al. 2016). Overall, the collective impact of protected areas on wildlife conservation, regardless of management approach, is unclear. One recent international study found higher levels of wildlife conservation within protected areas than without (Gray, et al. 2016), while other multi-site analyses have been either inconclusive (Craigie, et al. 2010; Geldmann, et al. 2013; Dudley, et al. 2016;) or have found that protected areas have resulted in no significant improvement (Western, et al. 2009; Mtui, et al. 2016).

A small number of studies exist that compare the performance of CBNRM and national governance approaches at wildlife protection, and that literature is also inconclusive. Two studies have used worldwide meta-analyses to examine the performance of protected areas. The first study found no significant difference in performance when comparing protected areas falling under different Protected Area Management Categories designated by the International Union for Conservation of Nature (IUCN), such as would be found between restricted use national governance and sustainable use CBNRM areas (Gray, et al. 2016).

The second study analyzed the performance of 160 protected areas and determined that (a) areas reporting socioeconomic benefits more often report positive conservation outcomes, (b) areas focusing on sustainable use (such as those under CBNRM governance) provide greater economic benefits than do those focusing on stricter resource protection, and (c) sustainable use areas co-managed by communities and state entities generate greater economic benefits than do areas falling exclusively under either community or national governance (Oldekop, et al. 2016).
However, the relative effectiveness of sustainable use areas on wildlife conservation is unclear from that study, as the cases analyzed varied widely in how they measured their conservation objectives, involving “a range of ecological attributes from specific species to components of ecosystems (such as habitat cover or quality)” (ibid).

Three East African studies have also compared the performance of different governance approaches. Two studies of protected areas in Tanzania alternately found that CBNRM outperformed national governance\(^1\) (Lee and Bond 2018) and that strictly-protected national parks were more effective than multiuse areas lacking in onsite enforcement (Stoner, et al. 2007). The third study, which is of particular relevance to this analysis, used elephant poaching data to analyze the effectiveness of different management approaches within northern Kenya (Ihwagi, et al. 2015). The authors found that the average annual proportion of illegal kills was lower in national reserves (.26) than in community conservation areas (0.37), but that there was no significant difference in proportion between different management types if land use was not accounted for (e.g., if national governance included forest reserves, and community governance included both community conservation and community pastoral areas).

Overall, the paucity of existing research and the ambiguity of relevant literature precludes the generation of any predictions regarding the respective performance of CBNRM and national governance approaches at protecting wildlife of high economic value. As such, rather than generating initial hypotheses, this study instead engages in an exploratory analysis with no preconceived expectations regarding the direction of any associations between governance type and elephant protection.

\(^1\) Lee and Bond (2018) compare CBNRM-based Wildlife Management Areas with Game Controlled Areas (GCAs). The GCAs are managed by Tanzania’s Ministry of Natural Resources and Tourism and hunting within the GCAs is strictly forbidden without a license granted by the Director of Wildlife. As such, for the purpose of this analysis, this study categorizes GCAs as falling under national governance.
2. Design and Methods

a. Variable Identification and Selection

i. Proportion of illegal elephant kills as the outcome variable

The outcome variable used in this analysis is the proportion of illegal elephant kills (PIKE) provided by the Monitoring of Illegal Killing of Elephants (MIKE) program, which was established in 1997 by the Convention of International Trade in Endangered Species (CITES) (CITES 2016). Since 2002, the program has monitored 58 designated sites within Africa, encompassing approximately 30-40% of the total remaining elephant populations. For each observation site, participating countries annually report both the total number of observed elephant carcasses and the number of those carcasses that were deemed to result from illegal kills. As of 2015, over 14,600 carcasses had been recorded across the African sites (ibid).

The MIKE program has a standardized set of procedures and forms for patrols of observation sites.\(^2\) The standardized form allows for the recording of the specific cause of death (along with the gender, age, and other particularized information regarding the carcass) and provides broad criteria for identifying the circumstances under which the elephant died. However, publicly available MIKE data only differentiates between illegal and non-illegal deaths and does not reflect the manner of the deaths. Consequently, it is not possible to determine, for instance, whether an elephant died from natural causes or from a sanctioned trophy hunt or cull, as both types of death would be identified in the database as being non-illegal. Likewise, it is not possible to differentiate between illegal killings by an ivory poacher or a farmer killing an elephant to prevent crop depredation.

\(^2\) Available at https://cites.org/eng/prog/mike/tools_training_materials.
The PIKE measure reflects the proportion of the total observed carcasses that are identified as resulting from an illegal kill (CITES 2016). PIKE is a standard metric for comparing the relative levels of poaching in MIKE observation sites (Ihwagi, et al. 2015; Ihwagi, et al. 2018), and has been used to estimate regional and continental poaching rates of African elephants (Wittemyer, et al. 2014). One notable advantage of using PIKE is the intensity, frequency, and methods of patrols vary across MIKE sites, and PIKE remains a valid measure of poaching intensity even in the absence of information regarding the monitoring effort (Burn, et al. 2011; Ihwagi, et al. 2015). Additionally, it can be difficult to obtain precise local elephant population estimates, particularly in wooded areas (Morley and van Aarde 2007), and the use of PIKE allows for cross-site comparison when (as in this study) there are not accurate counts of total elephant populations or poached animals within those sites (Burn, et al. 2011). While cross-site comparisons based on PIKE do rely on a premise that baseline natural elephant mortality rates are roughly comparable across sites, the measure has previously been found to be robust even when accounting for outlier environmental events, such as severe drought, that might impact localized annual carcass counts (ibid).

It is important, however, to note a key limitation of using PIKE data in a multi-country analysis. No publicly available data exist regarding the location of individual elephant kills within the MIKE sites, as that information is exclusively held by the reporting countries (CITES 2016). Consequently, it is not possible to determine whether and where observed elephant deaths and/or illegal kills are geographically clustered within the MIKE observation sites. Unfortunately, it was not feasible to negotiate with each of the selected reporting countries for access to the locations of the individual observed elephant carcasses.

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3 Additional information regarding the generation of PIKE can be found in this Chapter’s Appendix.
The lack of data regarding specific carcass locations necessitates a coarser analysis than is found in within-country analyses where such information is made available. However, 28 of the 39 sites included in this study had at least 95% of their territory falling under a single form of governance. Thus, while the lack of individual carcass data prevents a conclusive comparison of the two approaches, the available data still provide insight into the two approaches’ respective overall performance and allow us to evaluate the general applicability of within-country studies. In short, the increase in insight gained by expanding the scope of analysis helps to offset the resulting loss of analytical precision.

It should also be noted that the assessment of any wildlife governance approach necessarily involves both efficacy and value concerns, and underlying “ethical commitments” guide the selection of criteria by which success is measured (Miller, et al. 2011). The protection of commercially valuable species represents only one possible measure of success (Adams and Hulme 2001). Governance approaches can also be evaluated on, among other outcomes, their economic or social impact on local communities or their preservation of overall biodiversity or habitat, and a vigorous debate exists regarding whether conservation efforts should primarily try to achieve anthropocentric or ecological goals. Without choosing a side in this debate, this study uses elephant protection as its outcome measure, as the safeguarding of charismatic and/or economically valuable wildlife is one criterion on which the success of wildlife policies is likely to be evaluated.

Finally, given that MIKE data is self-reported, it is possible that individual countries (or patrols) could have an incentive to either systematically under- or over-report the frequency of illegal kills. Such an incentive might theoretically arise, for instance, if reported successes or shortcomings in elephant protection could potentially be utilized to attract or retain substantial
donor funding. To be clear, no insinuation is being made that data manipulation is occurring here, but any analysis of MIKE data must nevertheless be conducted with this possibility in mind.

ii. Governance approach as the primary explanatory variable of interest

The primary explanatory variables of interest in this study are the percentages of the MIKE site areas and site borders falling under CBNRM and national governance. These percentages were generated as follows. Keyhole markup language (KML) files\(^4\) delineating the boundaries of the MIKE observation sites were obtained from the MIKE program’s website. These boundaries were then mapped using ArcGIS satellite mapping software. The mapping employed the Africa Albers Equal Area Conic projection, which ensures that areas on the map occupy an area proportional to their areas on the globe.

The boundaries of adjacent or overlapping national and community protected areas were mapped along with the MIKE sites. Map shapefiles for these two types of protected areas were primarily obtained from The World Database on Protected Areas (WDPA),\(^5\) which provides the boundaries of recognized protected areas, including many designated and maintained by communities operating within formal CBNRM programs. To ensure that all official national and community areas were mapped, a supplemental internet search was conducted of governmental agencies and any hunting outfitters, CBNRM support organizations, and wildlife-oriented non-governmental organizations operating within each of the 19 countries. Shapefiles for additional areas identified through the supplemental search were obtained either through (a) individual

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\(^4\) KML files contain geographic annotation for use by Google Earth and other internet-based geospatial programs.

\(^5\) This database is managed by the United Nations Environment Programme’s World Conservation Monitoring Centre, with support from IUCN, and its online interface can be found at https://www.protectedplanet.net/.
contacts with researchers publishing on or practitioners working within those countries, or (b) if existing files were otherwise unavailable, the creation of shapefiles from publicly available maps.

iii. Identification and selection of other explanatory variables

I considered explanatory variables for inclusion in the model if they have previously been identified in the academic literature as potentially impacting illegal elephant kills. At the local level, population density (Burn, et al. 2011; Maisels, et al. 2013), along with forest cover and human footprint, including the penetration of roads, (Blake, et al. 2007; Burn, et al. 2011) have all been found to predictors of illegal elephant kills. The presence of armed conflict and civil unrest has also been linked with the increased poaching of elephants and a decline in the range and number of wildlife species (Dudley, et al. 2002; Lemieux and Clarke 2009; Beyers, et al. 2011; Daskin and Pringle 2018).

At the national level, Burn, et al. (2011) found economic development (measured through the United Nations’ Human Development Index, a country-level composite score reflecting educational attainment, life expectancy, and income, frozen at the 2007 level) was significantly associated with an increase in illegal elephant kills. Increased corruption negatively impacts elephant conservation, as well as the protection of wildlife more generally (Burn, 2011; Maisels, 2013; Smith, et al. 2015). Finally, the global increase in the price of ivory has been identified as a factor contributing to elephant poaching (Douglas-Hamilton 2008-2009; Anderson and Jooste 2014; Challender and MacMillan 2014).

I selected variables for inclusion if they resulted in a drop in the Akaike information criterion (AIC) and Bayesian information criterion (BIC) scores for the model. This study ultimately used the following site, country, and common explanatory variables. At the site level,
the analyses used (1) percentage of the site covered by forest, (2) the average infant malnutrition level for each reporting year,⁶ and (3) an annual count of the number of episodes of conflict arising from either civil unrest or international conflict within a 250 km radius of the site boundaries. Country-level variables were (1) GDP per capita and (2) a rule of law score.⁷ The study also included a common variable (i.e., shared by all sites) in the form of the price of ivory for each year of the MIKE program. A full discussion of the selection process for the explanatory variables is contained in the Appendix to this chapter.

b. Site Selection

This study uses MIKE data from Central, Southern, and Eastern Africa, as those regions are defined by the program (Figure 1). MIKE data do not exist for North Africa, and data from the West Africa region are not used because (a) many of the participating countries in that region failed to report any data for at least half of the duration of the MIKE program, and (b) these range states have small and highly fragmented elephant populations (UNEP, et al. 2013). The small number and sporadic reporting of observed elephant carcasses makes it more difficult to draw accurate conclusions from West Africa data.

The exclusion of West African countries should not compromise the findings of the research, as approximately 99% of the remaining African elephant population is found within the southern (55%), eastern (28%), and central (16%) African regions (CITES, et al. 2013:2).

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⁶ Infant malnutrition levels are used here as a proxy for local income levels. They are episodically published for the selected MIKE sites, and so predicted values were generated for years in which published data were not available. This study used three alternate assumptions about the behavior of the missing data. The results of this analysis were not sensitive to the assumptions used, and so results for only one of the alternates is reported here: the assumption that infant malnutrition levels change linearly between reported values.

⁷ This study considered both the World Bank’s Corruption and Rule of Law measures. Both are standard governance measures and were found to be highly correlated. Rule of Law was determined to be a preferable measure for use in this analysis as it appears to be influenced less by outlier scores.
Additionally, all three regions have countries with established community-based conservation initiatives as well as national protected areas. As such, these three regions provide sufficient observations to explore the respective performances of the CBNRM and traditional conservation approaches.

In total, this study uses data from 39 observation sites located within 19 countries. Between 2002 and 2015, the participating countries submitted a combined 414 annual reports involving a total of 14,221 observed elephant carcasses.

![FIGURE 1: MIKE Observation Sites Included in Study](image-url)
c. **Method of Analysis**

This analysis employs a three-level binomial logit generalized linear model, with PIKE observations as the Level 1 units, observation sites as the Level 2 units, and countries as the Level 3 units.

The clustering of yearly observations within sites which are, in turn, clustered within countries, means that the data has intra-site and (to a lesser degree) intra-country correlated errors. As such, the data violate the assumption of independence required of ordinary regression models and, if unaccounted for, this violation can result in narrowed confidence intervals and the increased likelihood of Type 1 errors (Guo and Zhao 2000; Clarke 2008; McCoach and Adelson 2010). Mixed-effects models (also frequently known as multi-level models) are an accepted approach for addressing the effects of spatially clustered data (Omariba and Boyle 2007; McCoach and Adelson 2010).

A binomial regression model is used to address the fact that the number of observed elephant deaths varies widely among sites and years – from as few as one reported death to as many as 351 – resulting in the possibility of heteroskedastic measurement error. In other words, a single misidentified or unobserved elephant carcass will have a much greater effect on the PIKE of a site with a single observation than it will in a site with over 300 observations. The use of a binomial regression weights each site’s yearly PIKE according to its total number of observed elephant carcasses. In using a binomial mixed-effects regression, this study adopts the same approach used by Burn, et al. (2011) in analyzing PIKE data.

3. **Results**

As shown in Table 1, increases in either of the CBNRM and national governance variables are positively correlated with the odds of an illegal elephant kill, holding all other
explanatory variables at a constant value. For each additional 1% of a MIKE site that falls under CBNRM governance, the baseline odds that an observed elephant carcass results from an illegal kill increase by approximately 4%. Similarly, a 1% increase in the area under national governance is associated with an approximate 2% increase in the odds of an illegal elephant kill.  

\[
\begin{array}{l|c|c|c|c|c}
\text{VARIABLE EFFECTS} & \text{Estimate} & \text{Std. Error} & \text{P>|z|} & \text{95% Conf. Interval} \\
\hline
\text{Country} & 0.197 & 0.529 & 0.001 & 37.73 \\
\text{Site} & 1.85 & 0.692 & 0.888 & 3.85 \\
\hline
\end{array}
\]

* significant at p<0.05  **significant at p<0.01

\text{TABLE 1: Impact of Governance Approaches on the Odds of an Elephant Death Resulting from an Illegal Kill}

To make these results more tangible, it can be useful to examine the effect that these changes in governance might have in the real world. As shown in Figure 2, in a theoretical MIKE site with no formal governance and which is at the mean for all other control variables, we would expect to observe between 0.24 and 1.99 illegal elephant kills out of 34.58 total deaths (Figure 2). At 100% national governance, the expected range for illegal kills would increase to

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8 The percent increases refer to the percent by which the underlying odds change, not the magnitude of that change. For example, if a hypothetical site with no formal governance had a 25% chance of an elephant dying from an illegal kill, a 1% increase in the area under national governance would be predicted to result in a 1.95% positive change in those underlying odds. Therefore, with 1% national governance, these findings would predict that the site would have a 25.49% chance of an elephant dying from an illegal kill.
2.18-3.21. At 100% CBNRM governance, we would expect to find between 3.62 and 4.99 illegal kills.

FIGURE 2: Projected Number of Illegal Kills and 95% Confidence Intervals Under CBNRM and National Governance Approaches

Regarding the other control variables, ivory price and the presence of armed conflicts are positively correlated with the likelihood of illegal kills, while GDP per capita, the rule of law and infant malnutrition measures have a negative association.\(^9\) Forest cover has no statistical significance.

\(^9\) The finding that infant malnutrition is correlated with a decrease in the likelihood of an illegal kill is surprising and finds no support in the literature. An examination of a two-variable scatter provides a possible explanation for this anomaly. Approximately three-fourths of the observations in this study had an infant malnutrition value under 22%, regardless of the assumption used to estimate missing data. Only
4. Discussion

The first finding – that an increase in either type of governance is associated with an increase in the odds of an illegal elephant kill – is unexpected given that previous studies suggest that protected areas perform equivalently or better at protecting wildlife than do outside lands. It is possible that, in some instances, informal governance regimes could exist that outperform formal governance approaches because they could be better tailored to local conditions and have greater legitimacy among local populations. Yet, it seems unlikely that effective informal wildlife governance arrangements are found across the 19 countries included in this study.

One possible explanation for this finding is that the formal governance approaches were adopted in response to an existing threat of wildlife poaching and/or unsustainable use. In other words, areas under formal governance regimes may experience a higher likelihood of elephant poaching because those areas were predisposed to suffer from that problem. For instance, in the case study of Namibia’s conservancy program in Chapter 1, the initial efforts at CBNRM in the Kunene region were motivated largely by concerns over the unsanctioned hunting of wildlife. Unfortunately, available data does not permit a determination regarding the baseline risk of elephant poaching across the sites selected for this study, and so this potential explanation cannot be evaluated here.

21 observations (5.1% of the total observations) had infant malnutrition values above 35%, and those values were split between two sites (Zakouma, Chad and Gash Setit, Eritrea). Given that the vast majority of the data in this study has an infant malnutrition value of under 22%, and this data is positively correlated with the likelihood of illegal kills, it may be that the variable’s overall negative correlation is the result of a lack of observations for sites with high infant malnutrition levels. In other words, a lack of observations in the top half of the score range may allow a few sites to disproportionately influence the measured relationship between infant malnutrition and the likelihood of illegal kills. Fortunately, for purposes of this study, the exclusion of the infant malnutrition variable impacts only the magnitude of the CBNRM and national governance variables, not those variables’ direction or statistical significance. As such, while the impact of local economic conditions on poaching remains an important consideration that merits further study, the behavior of the infant malnutrition measure is not further addressed here.
A second potential explanation for the first finding is that it is the result of a relative lack of data for MIKE sites that include substantial areas without a formal wildlife governance approach. Only 91 of the 414 annual reports involve sites with less than 95% of their area falling under formal governance. Therefore, the sustained low PIKE of a few such sites likely skews the association between lower levels of formal governance and the odds of an illegal kill.

Setting aside the correlation between overall formal governance and PIKE, what can we learn from the finding that the increase in illegal kill odds associated with national governance is less than half of that associated with CBNRM?

One possible implication of this difference in magnitude is that the national governance approach is more successful than CBNRM at elephant protection. Such a conclusion finds some support in a 2015 investigation into East African elephant poaching. That investigation, conducted by the Elephant Action League and Global Eye, infiltrated four ivory tracking networks in East Africa (EAL and GE 2015). Among other things, the investigation determined that commercial poaching networks include rural and urban cells, both of which participate in elephant poaching. The report concluded that poaching by urban cells is “highly unlikely” to be impacted by the local income-generating activities relied upon by CBNRM programs (ibid). Thus, even if the CBNRM approach is able to eliminate the incentives for community members to engage in the illegal killing of elephants, participating communities may lack the capacity, legally and/or logistically, to prevent poaching by organized outside actors. By contrast, national governance approaches are predicated on exclusion and law enforcement and therefore have greater potential to deter both rural and urban participation in elephant poaching.

However, the residents in CBNRM areas are not necessarily impotent in preventing poaching by outsiders. For instance, each of the four conservancies included in my field research
(Chapter 3) had a team of full-time game rangers, who were tasked with monitoring the zoned wildlife areas to prevent grazing or hunting by locals and outsiders alike. The rangers were then able to alert police in the area about the presence of poachers. Granted, I heard multiple complaints by conservancy representatives and traditional authorities that the rangers were unarmed and lacked the capacity to make any arrests of violators. Nevertheless, if the conservancies in my field study are in any way representative of CBNRM areas elsewhere, communities may not all be as powerless to stop outside poaching as the Elephant Action League report suggests.

A second possible explanation could be that areas under CBNRM governance may face a higher baseline threat of poaching than do areas under national governance. Therefore, even if the CBNRM approach is comparable or better at elephant protection than the national approach, areas under CBNRM governance may nevertheless still be associated with a higher risk of illegal elephant kills. However, available data does not permit a determination regarding the baseline odds of illegal kills across the areas included in this study.

A third potential explanation for this finding is that the difference in the magnitude of the association with illegal kill odds is due to the fact that CBNRM is still a relatively new governance approach, and so there are fewer MIKE sites that include CBNRM than there are sites that contain land under national governance. Of the 39 sites included in this study, 10 contain at least some land under CBNRM governance, compared with 37 that contain any land under national governance. Consequently, the magnitude of the correlation between CBNRM governance and illegal kill odds may be unduly influenced by the performance of a few sites.

As a practical matter, the difference in efficacy between the two approaches appears to be relatively minor. The difference between the respective midpoints of the 95% confidence
intervals for the two approaches is approximately 1.61 deaths illegal kills (representing roughly 4.66% of the total predicted number of observed deaths). If we look at the extremes of the 95% confidence intervals, the difference in predicted illegal kills between the two approaches could be as much as 2.81 deaths (around 8.1% of observed deaths) or as low as .42 deaths (approximately 1.2% of observed deaths). A difference of less than 5% in the predicted outcome of a policy approach is certainly noteworthy, and suggests the need for further understanding about when CBNRM should be selected as a policy tool, but it is not to great as to suggest that traditional governmental approaches are unqualifiedly superior in the protection of high-value species such as elephants.

Given (a) the uncertainty regarding the reason for CBNRM’s higher association with the likelihood of an illegal kill, (b) the finding that both forms of governance are associated with an increase in illegal kill odds, and (c) the relatively small difference the predicted outcomes between the two approaches, perhaps the safest conclusion is that it is premature to declare one governance approach as clearly superior to the other at protecting high-value animals such as elephants. Rather, the results suggest that consistent success has proved elusive for both approaches.

A review of the raw data indicates that both governance approaches have experienced a wide range of conservation outcomes. Of the 27 MIKE sites included in this study that have at least 95% national governance, the median averaged PIKE (i.e., the PIKE for a site averaged across all of its yearly reports) is .586, and the averaged PIKE scores are nearly evenly distributed from 0 to 1. By comparison, only two sites consist entirely of CBNRM governance, but the averaged PIKE for those two sites (0.444 and 0.612) are roughly comparable to the median averaged PIKE for sites under national governance. And, when considering MIKE sites
with at least one-third CBNRM governance, the averaged PIKE ranges from a low of 0.275 to a high of 0.789, with a median averaged PIKE of 0.444 (albeit with a much smaller sample size). So, while some MIKE sites under full national governance have realized better elephant protection than have those incorporating CBNRM, others have performed worse, and the two approaches have a similar distribution of outcomes. In short, the data indicate that neither governance approach can yet be viewed as either a success or failure.

5. Conclusion

Despite the debate over whether wildlife policy should focus on protectionism or sustainable use, policymakers may be best served by avoiding a reliance on a single governance approach and instead adopt site-specific governance approaches that focus on relevant natural, institutional, and socio-economic factors.

A particularly apt example of the need for appropriately tailored policy can be found in the Mozambique’s western Tete province. In 1993, one year after the end of its civil war, Mozambique institute a CBNRM program, called “Tchuma Tchato” (“Our Wealth” in the Nyungwe language), in the Bawa community (Suich 2013a; Suich 2013b; Gerety 2018). The Tchuma Tchato program was viewed by policymakers as a well-suited for adoption elsewhere in the province, and it was adopted in the nearby Mágoë District during the time period of 1994-1995 (Suich 2013a:442).

Mozambique initiated the Tchuma Tchato program, in part, to ameliorate tensions between residents of Bawa and a nearby safari operator located on the southern shore of Lake Cahora Bassa (Gerety 2018). When the program was implemented, local communities were promised one third of the revenue from trophy hunting fees conducted by safari companies operating in the area, with another 35% going to the government to be used in support of
sustainable development in the area (ibid). To try to maximize the money flowing into the local communities, prices for trophy hunting were set three times higher in the Tchuma Tchato project areas than elsewhere in the country (Suich 2013a).

The initial results from the project were positive, with local hunters and poachers agreeing to work without pay to prevent the illegal harvesting of wildlife resources (Gerety 2018). However, the project quickly ran into problems. Rebounding wildlife populations resulted in greater human-wildlife conflict, particularly in the form of cop raiding by elephants (Filimão, et al. 1999; Gerety 2018). Additionally, the region did not have a well-developed tourism industry, and subsequent attempts at increasing tourism were unsuccessful (Suich 2013a). Consequently, the amount of money received by the communities fell well short of initial expectations (Gerety 2018). Finally, there appears to have been little accountability regarding the expenditure of the earmarked funds received by the government. In 1996, for example, the annual government allotment went missing; in 1998, an official spent the entirety of the funds on a combination of spare parts for his motorcycle and repairs to roads located outside of the Tchuma Tchato areas (ibid). Morale amongst Tchuma Tchato participants collapsed, and the program was further hamstrung when the program lost its foreign funding in the early 2000s (Suich 2013b; Gerety 2018).

In 2013, Mozambique announced the creation of the Magoé National Park in the Tchuma Tchato territory (AIM 2013). The new park was supposed to strengthen the Tchuma Tchato program by providing a breeding ground for wildlife and a boost to tourism in the region (AIM 2013; Jackson 2013; Gerety 2018). However, two years after its creation, the park had received virtually no funding and lacked the money to even begin operations (VOA 2016).
News reports suggest that both the Tchuma Tchato areas and the Magoé National Park are now suffering from rampant poaching. In the period of 2015-2017, poachers killed over six hundred animals, including at least 144 hippos, 111 buffaloes, 54 elephants, and two lions (COM 2018). An article in 2016 reported concern by Mozambique officials that continued elephant poaching in the Magoé National Park would lead to local extinction of the species (COM 2016).

In the case of the wildlife governance in Mozambique’s Tete region, both CBNRM and national conservation approaches were unsuccessful. The CBNRM effort floundered because the absence of a viable tourism industry, insufficient external funding, and lack of accountability among government officials meant that the communities ultimately realized little benefit from the wildlife they were tasked with conserving. The creation of a new national park failed because it never received enough funding to become anything more than a *de facto* open access resource. In short, failure of the two approaches in this case was not because either were inherently failed approaches, but because neither were well suited to local and/or national conditions at the time of their adoption.

To return to the question posed at the beginning of this chapter, what does this research tell us about whether CBNRM or governmental approaches are better? Based on my results, the answer appears to be very context specific. Scholars such as Andersson and Ostrom (2008) and McShane, et. al (2011) warn against anointing specific governance approaches as panaceas. The findings in this study lend credence to that warning, and caution against categorically dismissing particular governance approaches as failures. Brandon (1997) contends that effective conservation requires a nuanced approach that is tailored to a site’s unique mix of natural, institutional, and socio-economic characteristics. That advice appears to apply equally well here.
CHAPTER 3
GAINING FROM WILDLIFE? THE IMPACT OF DIFFERENT BENEFIT TYPES ON INDIVIDUALS’ VIEWS OF WHETHER WILDLIFE IMPROVES THEIR LIVES

In this chapter, I turn to the second line of inquiry introduced in Chapter 1. In practice, does the receipt of wildlife-derived benefits cause CBNRM residents to feel that their lives have improved? And, if so, are some types of benefits more likely than others to cause residents to feel that way?¹

As previously noted, CBNRM seeks to incentivize local communities to actively participate in wildlife conservation efforts by devolving to them control over, and the right to profit from, wildlife resources (Child and Barnes 2010; Pienaar, et al. 2013). The approach relies primarily on the distribution of wildlife-generated benefits to increase the tolerance of wildlife by rural populations (Scanlon and Kull 2009), hinging on the central tenet that people will participate in the sustainable management of wildlife if the perceived benefits associated with wildlife outweigh (or at least equal) the perceived costs (Thakadu 2005; Mogende and Kolawole 2016). In short, under CBNRM governance, “wildlife must pay its way” (Levine and Wandesforde-Smith 2004).

The need for wildlife to pay for itself arises because interactions between humans and wildlife are often antagonistic, and this conflict can result in significant economic and personal losses to people living alongside wildlife. This human-wildlife conflict (HWC) can impact communities in a multitude of ways, including livestock predation, depredation of managed wildlife, crop-raiding and destruction of food stores, attacks on humans, disease transmission (to

¹ This fieldwork on which this chapter relies was supported by the Wildize Foundation and Indiana University’s Ostrom Workshop.
both crops and humans), and foregone economic or lifestyle choices due to the presence of wildlife or restrictions related to conservation areas (Hill 2004; Woodroffe, et al. 2005; Dickman 2010; Barua, et al. 2013). The threat of HWC can cause rural residents to form negative opinions regarding “problem” wildlife, and to engage in “self-help” through the retaliatory or anticipatory killing of wildlife or the destruction of wildlife habitat (Pienaar, et al. 2013).

Despite the importance of economic benefits to the CBNRM model of conservation, little research exists regarding the actual impact of different benefit types on residents’ attitudes towards wildlife. This chapter contributes to that underdeveloped area of study by exploring the relationship between the receipt of specific wildlife-generated benefits and the likelihood that community residents will feel that wildlife improves their lives.

1. Theory

   a. Impact of Direct Benefits

      A limited body of research suggests that the receipt of any type of direct benefit should positively impact individuals’ feelings regarding whether wildlife makes their lives better, and that cash benefits (e.g., wages or cash payments) should have a greater effect than in-kind benefits such as meat distributions.

      More widely distributed and higher-value benefits increase residents’ support for conservation efforts (MacKenzie, et al. 2017; Angula, et al. 2018). Residents in CBNRM areas rarely directly receive benefits, compared with indirect community-level benefits such as schools or infrastructure improvements, but even a limited distribution of direct benefits can be significant, as these areas often lack economic opportunities (Gosling, et al. 2017).

      Households prioritize the receipt of food, cash, and jobs from CBNRM programs, with cash and employment being particularly desired (Pienaar, et al. 2014; Naidoo, et al. 2016). Cash
distributions can be especially important because they allow households to purchase provisions and pay for school fees (Scanlon and Kull 2009; Pienaar, et al. 2014). However, the potential impact of the receipt of meat is unclear. Suich (2013a) found that meat benefits have little impact on recipients’ perceptions of wildlife because of the benefit’s relatively low economic value. On the other hand, Störmer, et al. (2019) found that both meat benefits and the presence of high levels of hunting benefits were associated with positive attitudes towards wildlife, while Jones and Weaver (2009) observed that recipients of direct benefits such as meat and employment tend to have a more positive view of the impact of CBNRM on their lives.

The limited available research supports the following two hypotheses:

Hypothesis 1: The receipt of any type of direct household benefit will increase the likelihood that a community resident feels that wildlife improves his or her life.

Hypothesis 2: The receipt of direct monetary benefits, such as wages or cash distributions, will have a greater positive effect on recipients’ opinions regarding the impact of wildlife on their lives than will the receipt of direct non-monetary benefits such as meat distributions.

b. Other Relevant Variables

Gender, age, wealth, and education have all previously been identified in relevant literature as potentially impacting individuals’ views of wildlife. Females are often found to be less likely to view wildlife positively than are men (Kellert and Berry 1987; Kaltenborn, et al. 1999; Sundaresan, et al. 2012; Dressel, et al. 2015), although some studies have not found any meaningful gender-based differences in perceptions of wildlife (Parry and Campbell 1992; Dressel, et al. 2015; Störmer, et al. 2019) or have found that women perceive wildlife more
favorably (Barthwal and Mathur 2012; Dressel, et al. 2015). Age is often negatively associated with attitudes towards wildlife (Bjerke, et al. 1998; Kaltenborn, et al. 1999; Dressel, et al. 2015; Angula, et al. 2018). Increased education levels are frequently associated with positive views of wildlife (Bjerke, et al. 1998; Barthwal and Mathur 2012; Dressel, et al. 2015; but see Störmer, et al. 2019). Finally, wealth has sometimes been identified as impacting views of wildlife and wildlife conservation although no clear direction for that relationship exists, with prior studies finding both that wealth is positively correlated (Romañach, et al. 2007; Bandyopadhyay, et al. 2009) and negatively correlated (Kanapaux and Child 2011; Silva and Mosimane 2012) with attitudes toward wildlife.

2. Methods

a. Site Selection for Administration and Pre-Testing of Surveys

This study administered household-level surveys in four conservancies (formally recognized CBNRM areas) in Namibia’s northwestern Kunene region (Figure 1). The selected conservancies – Puros, Anabeb, Sesfontein, and Omatendeka (collectively referred to as the “study area”) – are coterminous and collectively cover 9,216 km² (Table 1), representing approximately 8% of the Kunene’s total area and around 5.7% of Namibia’s total conservancy land area.

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<tbody>
<tr>
<td>Anabeb</td>
<td>2003</td>
<td>1,570</td>
<td>1,498</td>
<td>N$108,890</td>
<td>N$1,443,220</td>
</tr>
<tr>
<td>Omatendeka</td>
<td>2003</td>
<td>1,619</td>
<td>2,541</td>
<td>N$326,633</td>
<td>N$604,597</td>
</tr>
<tr>
<td>Puros</td>
<td>2000</td>
<td>3,562</td>
<td>1,167</td>
<td>N$62,570</td>
<td>N$515,825</td>
</tr>
<tr>
<td>Sesfontein</td>
<td>2003</td>
<td>2,465</td>
<td>1,835</td>
<td>N$156,070</td>
<td>N$886,045</td>
</tr>
<tr>
<td>Pretest</td>
<td>2009</td>
<td>348</td>
<td>1,873</td>
<td>N$49,128</td>
<td>N$42,513</td>
</tr>
</tbody>
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The four conservancies were all established between 2000 and 2003, each have low population density, and each reported earning greater wildlife-generated income than losses from human-wildlife conflict during the time period of 2014-2017. The study area is remote, economically underdeveloped and, because of the arid climate, largely pastoral, with cattle, goats, and sheep being the predominant livestock. It is located above Namibia’s agricultural line, and the land is communally owned.

The ethnic composition of the study area is predominantly Himba and Herero (two closely related ethnicities that speak Otjiherero), with a sizeable Damara minority. Residents are found scattered in small settlements and follow multiple formal and informal traditional authorities.²

I pretested the survey instrument in the Otjambangu conservancy, which sits east of the Puros conservancy and north of the remaining three selected conservancies. The Otjambangu conservancy has a relatively higher population density and receives less conservancy income than the conservancies in the study area. Despite these differences, it was an appropriate location to test the survey language because it is similar to the selected conservancies in its ethnic composition, reliance on pastoralism, and the socio-economic status of the majority of its residents.

b. Survey Development and Pre-Testing³

I travelled to the study area in May 2017 in order to administer semi-structured interviews of, and obtain permission from, traditional authorities and conservancy representatives for each of the conservancies. My guide and translator, Sokoi, was an individual

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² As discussed in greater detail in Chapter 1, traditional authorities are entities tasked under Namibian law with enforcing the customary law of clan-based social groups based in common property areas.
³ This section summarizes the development and pretesting process. Supplemental information regarding that process is found in Appendix 2.
who had previously worked for more than a decade for a Namibian NGO that required him to travel extensively within the study area to meet with residents. Sokoi was equally fluent in both Otjiherero (spoken by the Herero and Himba residents) and Khoekhoe (spoken by Damara and Nama residents and referred to herein as Damara, as that is how it is colloquially referred to in the region). I was able to meet with representatives for all the conservancies and interviewed most of the traditional authorities or their formal representatives. I received permission from all the traditional authorities and conservancy leadership in the area to conduct my research.

Along with a review of relevant literature, I used information obtained from my interviews of traditional authorities and conservancy representatives to inform the content of the surveys. I then used language professors at the University of Namibia to translate the survey into both Otjiherero and Damara. Two alternate versions were created for each of the translations so that the order of two attitudinal questions could be reversed to ensure that the ordering of those questions did not influence the study’s findings.

I used four Namibian field assistants to administer the survey, all of whom were identified by Selma Lendelvo, the Head of the Life Sciences Division at the University of Namibia’s Multidisciplinary Research Centre. All the assistants were black and lived outside of the study area. Two of the assistants were female, and all were fully fluent in Otjiherero. One of the field assistants was fully fluent in Damara, and another also possessed functional fluency in Damara. Three of the four had prior experience in survey administration.

Initial feedback from my assistants suggested that the language used in the Herero and Damara translations did not reflect how people actually speak in the field. Therefore, prior to pretesting, the translated surveys were modified based on the assistants’ recommendations to match the spoken form of the two languages.
I pre-tested the survey over a period of two days in a valley in the Otjambangu conservancy that consisted of multiple clusters of residences. For the first test, all assistants were present for an administration of the survey to a single household. After the completion of the first administration, the participants were asked about their thoughts on the survey, including whether they were confused by any questions or if they thought any additional questions needed to be asked. After the conclusion of the first test, the field assistants and I discussed and implemented revisions to the survey language and design.

I conducted a second test after this meeting, this time breaking up the assistants into groups of two and having each group separately administer the survey to different households. As before, we met after the completion of the survey to discuss potential changes to the survey instrument. The following morning, we conducted a third administration of the survey, and this time each of the assistants individually administered the survey in separate households spread across the valley. As none of the assistants received any feedback indicating a need for any additional changes after this third administration, I concluded the pre-test.

The survey administration took place between September and December 2017. The assistants administered a single survey per household, regardless of its size. The population in the study area was widely dispersed with a low population density, preventing the use of probability sampling. To ensure that the respondents to the survey reflected the composition of the overall population as closely as possible, the research assistants used non-proportional quota sampling, actively seeking out (a) Damara and other minority ethnicities and (b) followers of the various traditional authorities located within the study area. I consulted with Sokoi to identify the most likely geographical location of the followers of the study area’s traditional authorities.4

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4 Because the legal authority of a traditional authority is not limited by geography, residents in a conservancy can follow traditional authorities that are based elsewhere.
For purposes of this study, I defined a household as “all people sharing the same cooking pot.” I adopted that definition on the recommendation of my field assistants, as this is a definition that is frequently used in household surveys in rural areas in Namibia. The papers were orally administered by the assistants, with answers being recorded on paper surveys. The surveys were administered in either Otjiherero or Damara, depending on the preference of the respondent.

To identify the appropriate survey respondent, the administrators requested to speak with the head of the household. If the head of the household was not available, the administrators surveyed the first available adult from the household. Measurement at the household level is appropriate given that direct benefits in these conservancies are often distributed at that level.

When not actively attempting to seeking participation from minority ethnic and traditional authority groups, the administrators attempted to randomize the pattern by which households were selected for participation. However, many settlements consisted of only a few households, so a randomized sampling approach was often not possible. The participants were assured of anonymity, their names were not recorded by the survey administrators, and the research received both Institutional Review Board approval and permission from Namibia’s Ministry of Environment and Tourism (Permit No. 2246/2017).

c. Selected Survey Questions

The survey consisted of six sections. The first two sections asked the participants to identify their conservancy, whether they and members of their households were a member, and which traditional authority they followed, if any. The third section involved questions about income sources and crop and livestock ownership. The fourth section of the survey asked about the respondents’ experiences with wildlife. The fifth section contained questions about the
respondents’ experiences with their respective conservancies. The last section contained some demographic questions as well as some attitudinal questions. The full survey is contained in the Appendix to this chapter.

This chapter analyzes the responses to several of the survey questions. Question 9 asked the respondents to identify if they or anyone in their household engaged in a number of livelihood activities, one of which was conservancy employment.

In coding these responses, I created a separate entry for each of the identified activities, including a default “other” category. Responses were coded as a binary 0/1 response. If the respondents affirmatively stated that they participated in an activity, their responses were coded as a “1;” otherwise they received a “0.” This question appeared near the beginning of the survey, immediately after questions involving conservancy residency, household size and traditional authority allegiance. I included “conservancy employment” in this category, rather than in the conservancy benefits question (Question 28, discussed below) because the employment represents a reliable and non-trivial source of household income similar to other forms of employment and regular economic activity.

Question 13 asked participants about the impact of wildlife on their lives.
Responses to this question were coded as -1 (worse), 0 (same), or 1 (better). This question appeared at the beginning of the section involving respondents’ experiences with wildlife. The wording was purposefully broad and appeared prior to any discussion of HWC or conservancy benefits (other than the option to select “conservancy employment” in Question 9, above). The question sought to avoid steering the respondents into a purely economic evaluation of the impact of wildlife, instead allowing them to weigh both monetary and non-monetary considerations (such as existence value or pride). In this way, the question provides a comprehensive measure of the respondents’ respective views of the impact of wildlife on their lives.

Question 28 asked the respondents to identify whether they had received certain benefits from the conservancy. These involved both “direct” benefits (i.e., benefits directly distributed at the household or individual levels) and “indirect” benefits (i.e., improvements such as boreholes or vehicles that may be generally available to residents of the conservancy but do not accrue directly to any of the individual households therein).
This was the second question in the section of the survey involving the respondents’ experiences in their conservancies and followed a question asking whether the respondents felt that the conservancy had impacted the number of problems they experienced from wild animals. As with Question 9, above, participants’ responses were coded on a binary 0/1 scale, with “1” indicating receipt of that benefit.

As discussed above, conservancy employment was included in the “livelihood activities” question, rather than the “conservancy benefits” question because of its potential impact on households’ overall income. Nevertheless, conservancy employment only exists because of the presence of the conservancy and, like other benefits, it is available only to eligible conservancy residents. As such, for purposes of this study, responses for conservancy employment (Question 9) were combined with responses regarding the receipt of conservancy benefits (Question 28).

d. **Survey Results**  

   i. **Demographic information**

   My field assistants administered surveys to a total of 284 households, of which 272 completed or substantially completed the entirety of the survey. Approximately 30% of
respondents resided in the Anabeb conservancy, while around 26% and 17% lived in the adjacent Sesfontein and Puros conservancies, respectively (Table 2). The lower representation of Puros residence is a result of that conservancy having a lower total population, lower population density, and significantly more remote clusters of population (for instance, one of the villages surveyed had no road leading to it and required my assistants to undertake a two-day donkey ride to visit it). It is important to note both that the number of respondents from the Omatendeka conservancy is very low, particularly in light of the fact that it is the most populated of the four conservancies, and that nearly 17% of respondents responded that they did not know the identity of the conservancy they resided in.

The high numbers in the latter category likely explains the low numbers in the former. My assistants indicated to me that they had interviewed numerous illegal settlers in Omatendeka who were initially suspicious that they (the assistants) were government spies. It is likely that these settlers were either (a) unaware of the identity of the conservancy into which they had relocated, or (b) were reluctant to disclose their location because of ongoing efforts by the communities to expel them. Additionally, some responses in the “don’t know” category involved individuals who had recently moved from Opuwo (the regional capital) or elsewhere to work in schools or businesses located in the Omatendeka conservancy, an occurrence that was less common in the other three conservancies that were located farther away from the larger towns. As such, it appears that most of the “don’t know” responses involved individuals from Omatendeka but, unfortunately, the surveys did not contain an entry for the assistants to indicate the specific conservancy in which they were administered.
Twelve of the 284 households dropped out of the survey within the first few questions. On those 12 aborted surveys, four respondents identified as residing in Omatendeka, another four stated they did not know which conservancy they lived in, two respondents lived in Puros and Anabeb, respectively, and two individuals dropped out of the survey before answering the residency question (the first question of the survey). If most of these “don’t know” responses are also from individuals residing in Omatendeka, it is possible that up to 8 of the 12 households that dropped out of the survey were located in the Omatendeka conservancy. If some or all of the four “don’t know” surveys were of illegal settler households, the lower completion rate among those households is likely the result of the initial suspicion that my assistants were government spies.

The four dropped “Omatendeka” surveys are harder to explain, as three of the four identified as either having been born in the conservancy or as having been in the conservancy since before its creation. While the completion rate for those identifying as living in Omatendeka was still quite high (approximately 87.1%), it is nonetheless substantially lower than for the other conservancies. Nevertheless, as this analysis aggregates the responses across conservancies and controls for conservancy-specific differences via a dummy variable, the difference in response rate amongst the conservancies is unlikely to have a significant impact on the reported findings.

Around one-fifth of the respondents identified as being ethnically Damara, with roughly a further 8% identifying as belonging to another minority ethnicity (Table 2). The remaining respondents identified as either Herero or Himba (these two groups are closely related and share a common language and traditional reliance on livestock as a measurement of wealth).

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5 Information was not obtained about the nonresponse rate of the households approached. However, because I used quota sampling, nonresponse rates are irrelevant, as (for purposes of sampling) households fulfilling quota criteria are considered interchangeable (Schwarz 2013; see also Couper 2000).
6 Forty-seven of the surveys were administered in the Damara language.
Approximately one-third of the respondents were women. Most of the households owned livestock and many also sometimes engaged in some type of subsistence-level farming.

TABLE 2: Gender, Ethnicity, Residency, and Crop and Livestock Ownership of Survey Participants

<table>
<thead>
<tr>
<th>Characteristic</th>
<th># of Respondents</th>
<th>% of Respondents</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Gender</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Female</td>
<td>92</td>
<td>34.33</td>
</tr>
<tr>
<td>Male</td>
<td>176</td>
<td>65.67</td>
</tr>
<tr>
<td><strong>Ethnicity</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Damara</td>
<td>53</td>
<td>19.63</td>
</tr>
<tr>
<td>Herero</td>
<td>136</td>
<td>50.37</td>
</tr>
<tr>
<td>Himba</td>
<td>60</td>
<td>22.22</td>
</tr>
<tr>
<td>Other</td>
<td>21</td>
<td>7.78</td>
</tr>
<tr>
<td><strong>Conservancy</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Anabeb</td>
<td>82</td>
<td>30.15</td>
</tr>
<tr>
<td>Omatendeka</td>
<td>27</td>
<td>9.93</td>
</tr>
<tr>
<td>Puros</td>
<td>45</td>
<td>16.54</td>
</tr>
<tr>
<td>Sesfontein</td>
<td>72</td>
<td>26.47</td>
</tr>
<tr>
<td>Don’t Know</td>
<td>46</td>
<td>16.91</td>
</tr>
<tr>
<td><strong>Crop/Livestock Ownership</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Crop</td>
<td>114</td>
<td>42.22</td>
</tr>
<tr>
<td>Livestock</td>
<td>234</td>
<td>86.67</td>
</tr>
</tbody>
</table>

The respondents’ ages ranged from 23 to 87, with a mean and median age of around 50 years (Table 3). The respondents ranged from having no formal education to having obtained a university degree, with the median respondent having received some primary-level schooling and the mean respondent having finished primary school. The average household size was 10.7 people and the median size was 9. The livestock wealth of the respondents’ households ranged from N$0 to N$876,000 in estimated value, using the market values provided to me in pre-survey interviews for cattle, sheep, and goats (the three forms of livestock most commonly found in the conservancies).
TABLE 3: Age, Household Size, Livestock Wealth, and Educational Attainment of Survey Participants

<table>
<thead>
<tr>
<th>Characteristic</th>
<th>Obs.</th>
<th>Mean</th>
<th>Median</th>
<th>Min.</th>
<th>Max.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Age</td>
<td>264</td>
<td>51.34</td>
<td>50</td>
<td>23</td>
<td>87</td>
</tr>
<tr>
<td>Household Size</td>
<td>270</td>
<td>10.71</td>
<td>9</td>
<td>1</td>
<td>80</td>
</tr>
<tr>
<td>Livestock Wealth (N$)</td>
<td>272</td>
<td>141,349.3</td>
<td>116,300</td>
<td>0</td>
<td>876,000</td>
</tr>
<tr>
<td>Education Level</td>
<td>268</td>
<td>2.04</td>
<td>1</td>
<td>0</td>
<td>8</td>
</tr>
</tbody>
</table>

ii. Responses to impact of wildlife and benefits received

In total, 259 of the 272 respondents answered the question of whether wildlife made their lives better. Of those, approximately 42% answered that wildlife made their lives better, 40% stated that wildlife made their lives worse, and the remainder replied that their lives were about the same (Table 4).

TABLE 4: Answers to Selected Survey Questions

<table>
<thead>
<tr>
<th>Question</th>
<th>Total Responses</th>
<th>Category</th>
<th>No. of “Yes” Responses</th>
<th>% of “Yes” Respondents</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wildlife make life better?</td>
<td>259</td>
<td>Better</td>
<td>109</td>
<td>42.08%</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Same</td>
<td>48</td>
<td>18.53%</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Worse</td>
<td>102</td>
<td>39.38%</td>
</tr>
<tr>
<td>Benefits Received</td>
<td></td>
<td>Meat</td>
<td>170</td>
<td>84.16%</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Funeral</td>
<td>136</td>
<td>68.34%</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Cash</td>
<td>33</td>
<td>16.42%</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Employment</td>
<td>38</td>
<td>14.07%</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Tuition</td>
<td>25</td>
<td>13.66%</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Loans</td>
<td>15</td>
<td>7.77%</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Infrastructure</td>
<td>56</td>
<td>28.43%</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Other</td>
<td>16</td>
<td>10.19%</td>
</tr>
</tbody>
</table>

Respondents to the survey most frequently reported receiving one or more of the following six direct benefit types (with the percentage of respondents indicating that they had

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7 Educational levels were coded from 0-8 using the following scale: 0 (none); 1 (some primary); 2 (finished primary); 3 (some secondary); 4 (finished secondary); 5 (vocation training); 6 (vocational degree); 7 (some tertiary); 8 (college degree)
ever received that benefit): meat (84.16%), funeral assistance (68.34%), cash benefits (16.42%), conservancy employment (14.07%), tuition assistance (13.66%), and business loans (7.77%) (the cash, tuition, business loans, and employment benefits are collectively referred to in this paper as “rare benefits”) (Table 1). Approximately 28% of respondents identified receiving indirect benefits in the form of infrastructure, such as community vehicles or boreholes.

The “other” category of benefits received noticeably fewer responses than did the other benefit categories but, of those individuals that answered this sub-question, around 10% received some additional benefit. A review of additional written comments on the surveys indicate that this category of benefits included direct services – such as the distribution of seeds for planting crop, construction materials for homes, and food to individuals in need – as well as indirect benefits, such as a school built by a safari hunting organization and money to fund conservancy-based soccer teams. Unfortunately, many of the “other” responses do not identify the additional benefit received, and so it is not possible to identify whether those specific benefits are direct (e.g., seeds or construction materials) or indirect (e.g., a school or soccer team). Thus, for purposes of this analysis, this category is included in the model as an indirect category of benefits.

None of the direct or indirect benefits are highly correlated. However, the rare benefits do evidence a raised level of correlation with each other compared to the relationship between the common benefits, between the rare and common benefits, and between the direct and indirect benefits (Table 5). For instance, cash benefits are correlated with each of the employment, loans, and tuition benefits at a phi correlation of greater than 0.4, and the tuition and loans benefits have a phi coefficient of nearly 0.5. By contrast, all the other relationships between the benefit types

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8 Funeral assistance consists of game meat along with an additional small cash payment to help defray funeral costs.
have a correlation coefficient of less than 0.4 (with most having a correlation coefficient of less than 0.3).

TABLE 5: Phi Correlation Coefficients of Variable Pairs

<table>
<thead>
<tr>
<th></th>
<th>Cash</th>
<th>Empl.</th>
<th>Funeral</th>
<th>Loans</th>
<th>Meat</th>
<th>Tuition</th>
<th>Infrastr.</th>
<th>Other</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cash</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Empl.</td>
<td>.472</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Funeral</td>
<td>.153</td>
<td>.048</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Loans</td>
<td>.41</td>
<td>.06</td>
<td>.153</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Meat</td>
<td>.193</td>
<td>.174</td>
<td>.363</td>
<td>.031</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Tuition</td>
<td>.445</td>
<td>.163</td>
<td>.166</td>
<td>.497</td>
<td>0.155</td>
<td>1</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Infrastr.</td>
<td>.31</td>
<td>.168</td>
<td>.211</td>
<td>.292</td>
<td>.186</td>
<td>.262</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Other</td>
<td>.021</td>
<td>.08</td>
<td>.194</td>
<td>.002</td>
<td>.136</td>
<td>.000</td>
<td>.113</td>
<td>1</td>
</tr>
</tbody>
</table>

e. Method of Analysis

This study used two separate sets of ordinal logit regression models to analyze responses. The first set of models investigated the individual impact of each of the different benefit types. The second set of models alternately used a total benefits variable as the explanatory variable of interest. Each of the benefit types and the total benefits variables were initially regressed separately without any control variables in order to establish a baseline relationship between each of those variables and the outcome measure. Since some respondents received more than one benefit, all benefit types were then regressed together, again without any control variables. By not incorporating control variables at this stage, this regression allowed for an evaluation of the relative impact of each of the benefit types without the risk that any significant associations were the result of a confounding control variable. Finally, both the total benefits and the benefit types models were regressed while progressively including all control variables.

In addition to the ordinal logit regression, I used t-tests to compare responses of those who received non-rare benefits with (a) those residents who reported receiving no benefits and
(b) residents who reported receiving additional rare benefits. These tests allowed the investigation of any possible cumulative impact of the receipt of particular benefit types.

The various benefit categories received differing response rates, with the “other” category receiving notably fewer responses than the others. To ensure that the findings of the study were not unduly influenced by this missing data, I generated predicted responses for missing data using the following assumptions:

1. Where individuals had answered any of the benefit category questions, non-responses indicated a lack of receipt of the corresponding benefit type.

2. Given that benefits are ordinarily distributed only to households with conservancy members,9 where individuals responded that they (a) did not know which conservancy they lived in or (b) were illegal residents, I also assumed that those individuals had not received any benefits.

I used this predicted dataset to re-run the full model regressions.

It is possible, of course, that the predicted data undercounted the number of individuals who received the various categories of benefits. However, the three benefit categories with the lowest response rate – other, tuition, and loans – each had a low percentage of recipients, so it is not unreasonable to assume that most of the assumed responses would correctly show a lack of receipt of those benefits. Further, the inclusion of the assumed “no” responses prevents the exclusion of surveys for which affirmative responses do exist. This resulting increase in statistical power allows for critical insight into the accuracy of the findings from the full un-supplemented model.

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9 Benefits are sometimes given by conservancies to long-standing residents that are not officially members.
3. Results

a. Ordinal Logit Regression

The total number of direct benefits received was positively and significantly correlated with the likelihood that respondents felt that wildlife made their lives better (Table 6). For each additional benefit received, the odds of a respondent feeling more positively about the impact of wildlife on his or her life increased.\(^\text{10}\) On the other hand, the individual benefits had notably different associations with the outcome measure. When each of the benefits were individually regressed with the outcome measure, the employment, funeral, and meat benefits all had a significant, positive association. By contrast, the business loans and cash benefits both evidenced a marked decrease in the likelihood of a positive response about the impact of wildlife, although the latter association was not statistically significant. Tuition benefits showed a positive but not significant association with the outcome measure.

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\(^{10}\) The odds ratio refers to the likelihood of a unit change in an outcome measure for every unit change in the variable of interest. Here, the odds ratio refers to the likelihood of a unit increase in whether respondents feel that wildlife makes their lives better given a unit change in a benefits variable. The same odds ratio value applies for both the change in the outcome measure at the 0-1 value (worse to same) or from 1-2 value (same to better). It should be noted that the percent change refer to the percent by which the underlying odds change, not the magnitude of that change. For example, using Table 6 as our reference, if a hypothetical individual that received no benefits had a 25% baseline likelihood of responding more positively about the impact of wildlife, the receipt of meat would increase those odds by 318%, resulting in a new likelihood of 79.5%. On the other hand, if that same individual originally had a 5% baseline likelihood, the receipt of meat would result in a new likelihood of 15.9%.
TABLE 6: (Model 1) Compilation of Separate Regressions of Benefit Types on Respondents’ Perceptions of Whether Wildlife Makes Their Life Better

| Benefit Type    | Odds Ratio | Standard Error | P>|z| |
|-----------------|------------|----------------|-----|
| Total Benefits (n=259) | **1.574*** | .148           | .000 |
| Cash (n=195)    | .897       | .349           | .779 |
| Employment (n=258) | **3.261*** | 1.251          | .002 |
| Funeral (n=193) | **2.479*** | .743           | .002 |
| Loans (n=187)   | **.375*    | .207           | .076 |
| Meat (n=196)    | **3.179*** | 1.225          | .003 |
| Tuition (n=177) | 1.39       | .622           | .461 |
| Infrastructure (n=191) | 1.349 | .421           | .338 |
| Other (n=151)   | .977       | .485           | .963 |

Table reflects ordinal logit results, with standard errors in parentheses, where *p < 0.1, **p < 0.05, ***p < 0.01 in two-tailed tests.

Regressing the different benefit types together yielded similar results to when they were regressed separately (Model 2, shown in Table 7). As before, the individual funeral, employment, and meat benefits remained positive and statistically significant at the 90% level or greater. The cash and loans benefits continued to evidence negative relationships and the tuition benefit a positive relationship with the outcome measure, but none of those relationships were statistically significant.

The direction and significance of the total benefits and individual benefit types remained largely unchanged after inclusion of control variables not including indirect benefits (Models 3 and 7). The total benefits, employment, and funeral variables were significantly and positively associated with the odds of an increase in the outcome measure. The meat benefit was no longer significant at the 90% level, although it still evidenced a positive relationship with residents’ views of the impact of wildlife on their lives. The association between the outcome measure and the tuition, cash, and loans variables remained largely unchanged from the earlier models.
Notable among the control variables, females were much less likely to perceive wildlife as having a positive impact on their lives than were males.

The results remained largely unchanged after the indirect infrastructure benefit was added to the models (Models 4 and 8). Individuals who received employment and funeral benefits continued to be more likely to have a positive view of the impact of wildlife on their lives, and the total benefits variable also retained its positive and significant relationship with the outcome measure. The individual meat and tuition benefits had a positive but non-significant association with the outcome measure, while the loans and cash had a negative, non-significant relationship. Additionally, females continued to be less likely to have a favorable view of the impact of wildlife on their lives.

The inclusion of the “other” category of benefits resulted in some notable changes to the models’ respective outputs (Models 5 and 9). The total benefits variable lost statistical significance, as did the individual employment benefit variable. On the other hand, the tuition benefit gained statistical significance and the magnitude of that relationship increased. However, these changes could be the result of a large decrease in both models in the total number of observations – from 157 to 122 for the direct benefits model and from 181 to 138 for the total benefits model. The remaining variables discussed above remained consistent in the direction and significance of their association with respondents’ views of the impact of wildlife on their lives.
## TABLE 7: Models of Relationship Between Benefits and Perceptions of Wildlife Impact

<table>
<thead>
<tr>
<th>Model</th>
<th>Individual Benefits</th>
<th>Total Benefits</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Observations</td>
<td>Total Benefits</td>
</tr>
<tr>
<td></td>
<td>168</td>
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### Table reflects ordinal logit results, with standard errors in parentheses, where *p < 0.1, **p < 0.05, ***p < 0.01 in two-tailed tests.**
When I reran the models included using the predicted data (Models 6 and 10), the significance and direction of association of the variables of interest largely reflected those found in the preceding individual and total benefit models. As found in most of the earlier model regressions, the total benefits, employment, and funeral benefits all showed positive and significant associations with the outcome measure. The business loans variable continued to evidence a negative association with the likelihood that respondents would feel that they benefited from wildlife, and this association was now significant at the 90th percentile level. Finally, females continued to be more likely to negatively perceive the impact of wildlife on their lives.

b. **Comparison of Means Test**

Thirty-eight respondents reported receiving only a single benefit type and, of those, 29 received meat, 7 received funeral assistance, and the remaining 2 received employment and a loan, respectively. When responses of the 29 meat recipients were compared to those of the 82 respondents who did not receive any type of benefits, the former group had a mean response regarding the impact of wildlife that was significantly higher than the latter (i.e., individuals who received only meat felt, on average, that wildlife had a more positive impact on their lives than did those did not receive any benefits) (Table 8).

<table>
<thead>
<tr>
<th>Benefits Received</th>
<th>Observations</th>
<th>Mean</th>
<th>95% Conf. Interval</th>
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<tbody>
<tr>
<td>No benefits</td>
<td>82</td>
<td>.573</td>
<td>.423 - .724</td>
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<tr>
<td>Meat benefit only</td>
<td>29</td>
<td>1.069</td>
<td>.718 - 1.42</td>
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Pr (|T| > |t|) = .003

Eighty-seven respondents reported that their households received exactly two benefit types and, of those, 78 received both meat and funeral assistance (three respondents reported as having received two benefits but did not answer regarding whether wildlife made their lives
better, and those three individuals are not counted here). When the responses of those 78
individuals were compared to the responses of individuals whose households had received one or
more additional rare benefit (in other words, three or more benefits, including both meat and
funeral assistance), the mean response for the two groups decreased slightly, although this
difference was not statistically significant (Table 9). When recipients of at least one additional
rare benefit were required to have received employment (in addition to any other rare benefits),
the mean response rose slightly, but this difference was also not statistically significant. When
conservancy employees were excluded from this analysis, however, the mean response for
recipients of tuition, cash, or loans benefits dropped markedly, and the difference between the
two mean responses was statistically significant.

<table>
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<tr>
<th>Table 9: T-Tests Between Meat and Funeral Recipients and Those that Received One or More Additional Rare Benefits</th>
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<tr>
<td>Group</td>
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<tr>
<td>Meat &amp; Funeral Benefits</td>
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<td>+1 or More Rare Benefits (regardless of type)</td>
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<td>+1 or More Rare Benefits (requiring employment)</td>
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<td>+1 or More Rare Benefits (excluding employment)</td>
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4. Discussion

The data provide qualified support for the first hypothesis – that the receipt of any type of
benefit will be positively associated with the likelihood that residents will feel that wildlife
improves their lives. Certainly, the number of benefits a household receives has a significant and
positive correlation with the likelihood that a resident feels that wildlife makes his or her life
better. However, nearly all single-benefit recipients (36 of 38 individuals) received either the
meat or funeral benefits. As such, it is not possible to determine whether the receipt of only a single rare benefit has any significant relationship with either of the alternate outcome measures.

Nevertheless, when analyzed more closely, the findings in this study suggest that different benefits may vary greatly in their influence on residents’ perceptions, and that the significance of the total benefits variable may be largely attributable to the impact of the meat and funeral benefits. Certainly, recipients of employment and funeral benefits appear to be much more likely to feel that wildlife improves their lives. And, even though not statistically significant in the final model, the meat benefit still evidenced a consistent positive association with the likelihood that a recipient will feel that wildlife makes her life better. The remaining three benefits (cash, loans, and tuition) all had consistent associations across the various model iterations, with cash and loans evidencing a negative relationship and tuition a positive relationship with the outcome measure. The cash benefit, however, retained a high $p$ value across all model iterations, while the tuition and loans variables each only reached statistical significance in some of the model iterations. As such, it is difficult to make any definitive statements about the potential impact of those three variables from the logit regression alone.

The comparison of the means t-test largely supports the logit findings that the meat, funeral, and employment variables have a meaningful positive relationship with respondents’ feelings about wildlife, while the remaining benefits do not (and the cash and loans benefits likely have a negative association). As seen in Tables 8 and 9, the receipt of meat appears to have a strong impact on recipients’ views of whether wildlife makes their lives better. And, in comparing residents that received both meat and funeral benefits, recipients of additional non-employment rare benefits had a significantly lower opinion of the impact of wildlife on their lives (a mean score that was half that of individuals who had received the additional employment
benefit). In short, once recipients received both meat and funeral benefits, the addition of any rare benefits other than conservancy employment appears to have had little (and sometimes a negative) effect on the recipients’ opinions regarding the impact of wildlife on their lives. Consequently, this data does not support the second hypothesis tested in this study – that monetary benefits should necessarily have a greater positive effect than in-kind benefits on peoples’ perceptions of the impact of wildlife on their lives.

Given these findings, what might explain the heterogeneity among the benefit types in their association with residents’ perceptions of the impact of wildlife on their lives? The lack of statistical significance for the meat benefit in the full model is consistent with previous findings by Suich (2013) and may be explained by the fact that meat is perceived by recipients as having lower economic value than the funeral or employment benefits, as they both provide cash payments.

The large, positive association of the funeral and employment benefits is also expected. Funeral benefits provide both in-kind and cash benefits and represents a social service that can be directly attributed to wildlife – both via the game meat provided and the payment of money by the conservancy, which exists to collect and manage wildlife-generated income. Absent these funeral benefits, the families of the deceased would have to rely exclusively on their own resources to feed funeral guests and cover any associated funeral costs. As such, funeral assistance represents significant economic value, the gift of which is strongly linked to wildlife at a time when the household may be most in need of financial and emotional support.

Regarding the employment benefit, jobs in many CBNRM areas are scarce and any form of employment is highly coveted (Naidoo, et al. 2016; Gosling, et al. 2017). Thus, individuals who have secured conservancy employment are highly likely to feel like their lives have
significantly improved. Since the employment is through the conservancy, which was created for
the purpose of governing wildlife, those individuals are also likely to attribute the employment to
the existence of wildlife.

The remaining variables’ lack of significance, however, is puzzling. Each of the business
loans, tuition, and cash benefits have tangible monetary value and should therefore be expected
to have a significant, positive correlation with the outcome measure. Even more surprising is the
fact that, notwithstanding their lack of statistical significance, the cash and loans benefits
evidence a negative correlation with residents’ views of wildlife impact. Two potential
explanations for these findings are that they are the result of either (a) error or (b) some sort of
psychological phenomenon.

a. **Validity of Results**

One possible explanation for the surprising findings for the cash and business loans
variables is that they result from an error in the design of this study, either in the form of
sampling errors or a problem in the design of the survey questions. However, the survey
instrument was pre-tested in the field prior to administration without incident. Further, the
behavior of the meat, employment, and funeral benefits are each consistent with findings from
other similar studies (as discussed in subsection 1(a) of this chapter), and the percentage of
respondents that received cash, tuition assistance, and business loans are in line with
expectations for benefit types that are not widely distributed within the four conservancies.
Störmer, et al. (2019), for instance, conducted interviews in 18 Namibian conservancies and
found a similar frequency of meat distributions (64% of respondents received meat compared to
62.5% in this study) and a low frequency of conservancy employment (21% of respondents
received employment compared to 13.97% in this study).
Finally, while surprising, these findings are not entirely unprecedented. Muyengwa (2015) surveyed residents of five conservancies in Namibia’s Zambezi region and asked them to rate their support for their respective conservancy (1-5, with “1” indicating that they strongly supported the conservancy and “5” indicating that they strongly disliked the conservancy). That study found that participants who received meat were significantly more likely to report supporting their conservancy, whereas the receipt of either cash or employment benefits was not associated with any sizeable or significant change in participants’ ratings. Cash payments in Muyengwa’s study were associated with participants having a less favorable view of their conservancies, and that negative association persisted (albeit without ever attaining statistical significance) across all the author’s regression models.

Muyengwa’s outcome variable differs from the one used in this study in that it asked about support for conservancies, rather than perceptions of wildlife. Consequently, the participants’ views in Muyengwa’s study may be shaped by non-wildlife-related factors such as corruption, responsiveness (or lack thereof), or perceived favoritism in conservancy governance. On the other hand, Namibia’s conservancies are designed to allow residents to utilize and benefit from their wildlife resources, so it is not unreasonable to think that participants’ views of their conservancies, and whether they personally benefit from wildlife, may be linked in my study. For instance, both support for conservancies (in Muyengwa’s study) and perceptions of wildlife impacts (in my study) are similarly correlated with the receipt of different types of benefits, and the availability and distribution of benefits in both study areas is likely to be influenced by the same categories of institutional variables.

In light of the feedback received during pretesting, the similarity of many of my findings with those found in other studies, and the strong parallels between my results and those reported
by Muyengwa (2015), the difference among benefits in this study appears unlikely to be attributable to methodological error.

b. Psychological Explanation

A second possible explanation for the heterogeneity in the behavior of the individual benefits types is that their differences result from some sort of psychological phenomenon. Both the psychological and behavioral economic literatures recognize that individuals do not formulate opinions about either the quality of their lives or the desirability of events they experience through the use of any objective measure, but they instead do so by comparing their current situation or experience with a subjectively derived standard (Medvec, et al. 1995; McBride 2010).

The psychology literature notes that people engage in “counterfactual thinking,” in which they compare their current reality to a hypothetical “counterfactual” reality and become more or less satisfied with their current situation as a result (Epstude and Roese 2008; Roese and Morrison 2009). For instance, two employees may perceive a 5% pay raise very differently depending if that amount exceeded or fell short of their respective expectations. While both would receive the same objective increase, the employees would each weigh the increase against different counterfactual standards. The employee who expected a higher pay raise would likely feel disappointed in the size of her raise, whereas the similarly situated employee who expected a smaller raise would feel elated.

Research suggests that people more frequently spontaneously generate “upward” counterfactuals – in which they imagine how their current reality might have turned out better – and these sorts of counterfactuals are typically associated with negative emotions such as disappointment or regret (Hafner et al. 2016). The use of counterfactuals appears to be a
universal tendency and has been observed across different countries and cultures (Epstude and Roese 2008; Gilovich, et al. 2013; Hafner, et al. 2016).

Similarly, behavioral economists have applied Kahneman and Tversky’s (1979) prospect theory, originally developed to explain decision making under uncertainty (Arkes, et al. 2010), to explain how individuals evaluate their current economic status by comparing it with various reference points, including their prior expectations about what their current status would be (McBride 2010; Castilla 2012). For example, Castilla (2012) found that individuals’ satisfaction with their current income level was positively correlated with the extent to which they had achieved their aspirational income goals for that stage of their life. Additionally, past outcomes shape current aspirations, so that past achievements can lead people to amend their reference points (Mentzakis and Moro 2009; McBride 2010; Bertoni and Corazzini 2015). Research suggests that people adapt their reference points upwards in response to gains more than they do downwards in response to losses (Arkes, et al. 2008; Chao, et al. 2017). I identified only one study on the cross-cultural applicability of reference point adaptation, which found that individuals across different cultures utilize reference points and modify them in response to changing outcomes, although the extent and responsiveness of that modification may be influenced by cultural factors (Arkes, et al. 2010).

Applying counterfactual theory to the facts of this study, if recipients of rare benefits expected to receive even more benefits than they did, they may feel disappointed at receiving a distribution of benefits of less than the anticipated amount. This expectation could arise, for instance, among individuals who are well-connected to those in charge of distributing conservancy benefits and might therefore have anticipated enjoying a dramatic economic windfall.
Applying the concept of reference points to these same facts, past recipients of rare benefits such as cash or loan payouts may have subsequently adjusted their economic aspirations upwards, so that they are disappointed when they do not later receive the same (or any) rare benefits. The unwillingness of the recipients to lower their reference points after they did not receive a second cash or loan payments would mean that they would also continue to feel disappointed by the future lack of cash or loan distributions.

The survey data analyzed in this chapter does not allow for any evaluation of the capacity of the counterfactual and reference points theories to explain the observed difference in behavior between the different individual benefit types. Nonetheless, evidence from my pre-survey interviews and the existing literature suggests that these sorts of psychological drivers may be occurring among residents in the Study Area.

\[i. \quad \text{Evidence of increased expectations}\]

A limited body of evidence from both my field research and the academic literature suggests that participants may elevate their expectations for CBNRM over time in response to the receipt of benefits. During my pre-survey interview of Head Lady Allina Karutjaiva, she stated that people in the Sesfontein Conservancy had expected that the creation of the conservancy would generate employment, improve the livelihoods of its members, increase wildlife populations to attract tourists, and balance the grazing of livestock in the area (Interview of Head Lady Allina Karutjaiva, April 17, 2017). When I asked her about whether those goals have changed since the Sesfontein Conservancy’s formation, Head Lady Karutjaiva responded that the needs and expectations of community members had increased over time, and that these increases resulted in community members placing growing demands on the conservancy (ibid).\[11\]

\[11\] An investigation into “expectation creep” within CBNRM areas would appear to be a fruitful area for future research. I attempted to conduct follow-up interviews with traditional authorities in the study area.
Welch (2018) observed a similar evolution in residents’ demands over time in Namibia’s N#a-Jaqna Conservancy. Welch writes that the traditionally marginalized San people were motivated to form the conservancy “not simply to benefit economically from natural resources but also to establish and/or strengthen an economic, social, and political base from which to pursue a path towards developmental goals on terms acceptable to local residents” (ibid at 12). Welch witnessed the conservancy continually struggle with increasing expectations by its residents regarding the conservancy’s role in generating economic development and poverty alleviation (ibid).

Welch (2018) tells one story that is particularly relevant to a discussion of elevated reference points. During Welch’s years working with the conservancy, it annually distributed N$300 payments at the end of the year to its members – an amount unofficially referred to as the Ju|’hoan Christmas Bonus. In December 2007, a rumor was widely circulated that the conservancy was planning on distributing an additional N$300 payment. When the additional payment failed to materialize (with the members receiving only their regular end-of-year cash bonus), some members were angry, having “become attached to the idea of having extra cash around the holiday time” (ibid at 144).

Snyman’s (2014) study of CBNRM areas in southern Africa (Botswana, Malawi, Namibia, South Africa, Zambia, and Zimbabwe) is also relevant to this analysis. Snyman studied the attitudes of individuals living in or adjacent to protected areas with active ecotourism lodges (the protected areas ranged from national parks to community-designated areas). In Namibia, these areas involved seven conservancies, including three of the four (Puros, Sesfontein, and via satellite phone, but I was unable to do so. It appears that Sokoi (the guide that I used in my earlier trips to the area) either suffered some calamity or decided to forego the money promised him at the end of the follow-up research and ran away with the advance payment I sent him. Either way, he ceased communicating with me and I was unable to conduct the interviews as planned.
Anabeb) selected for my study. Among other things, Snyman analyzed the relationship between the number of years that the lodges had been operating and respondents’ views of whether (1) there had been a positive change in their village; (2) tourism had reduced poverty in the area; and (3) conservation was important (ibid).

Snyman found seemingly contradictory relationships between those variables. The variable reflecting the number of years a lodge had been operational was positively and significantly associated with perceptions of whether tourism had caused positive changes in the respondents’ villages and had reduced poverty in the area (Snyman 2014). Yet, that same variable had a negative, although not statistically significant, relationship with respondents’ perceptions of whether conservation was important. Snyman hypothesized that the negative association between the time-of-operation and importance-of-conservation variables resulted from a combination of increased human-wildlife conflict and the failure of the lodges to match both their promises and the residents’ expectations (ibid).

Snyman’s conflicting findings, however, would still be surprising if the respondents’ evaluations were based solely on static, initial expectations regarding the impact of the lodges. If, over time, respondents were increasingly likely to feel that the lodge tourism had resulted in a positive change in their villages, and were also more likely feel that tourism had reduced poverty, it would stand to reason that residents would perceive the impacts of the ecotourism ventures as progressively fulfilling their expectations. Consequently, we should expect this progress to affirm to respondents, rather than undermine, the importance of conservation. If, on the other hand, we allow for the possibility that the respondents’ raised their expectations based on past outcomes, rather than possessing static expectations throughout the time of the lodges’ operation, the conflicting perceptions make more sense. Under this scenario, the respondents
could feel that the lodges had made a positive impact on their villages and reduced poverty while still being frustrated that the perceived benefits of those lodges failed to match their current, heightened expectations. This disappointment could fuel disillusionment with conservation efforts.

\textit{ii. Evidence of elite capture}

The data suggest that elite capture is occurring within the study area. Among the respondents, individuals who received one of the rare benefits were more likely to receive an additional rare benefit than were individuals who only received meat or funeral benefits. Aggregated across all four conservancies, and using recipients of trophy meat as the group most likely to receive any benefits at all, 35.3\% of that group also received at least one rare benefit. However, half (49.98\%) of the recipients of one rare benefit received at least one additional rare benefit, as opposed to the 12.46\% of recipients that should be expected if the distribution of rare benefits was truly random. The distribution of rare benefits suggests that these benefits may be disproportionately captured by well-connected conservancy members. In their survey responses, most respondents also indicated that they believed that favoritism was occurring within the conservancies, as did some participants in pre-survey interviews. And, elite capture of conservancy benefits would be consistent with that found in other conservancies in Namibia and in community-based wildlife areas elsewhere in Africa (Groom and Harris 2008; Naidoo, et al. 2016; Morton, et al. 2016; MacKenzie, et al. 2017; Lubilo and Hebinck 2019).

This evidence of elite capture, taken together with the preliminary evidence that residents may raise their expectations for the conservancies over time, suggests that recipients of one-time/sporadic cash and loans benefits (and, to a lesser degree, tuition payments) may have raised their reference points, aspiring for those sorts of benefits to be regularly distributed. When they
were not, the residents were more likely to be disappointed than were those who had never received those benefits. Local elites may have pre-existing elevated reference points, imagining a world in which they were the recipients of largesse exceeding the value of benefits they received. In both of these scenarios, the past receipt of rare benefits may have conditioned residents to be disappointed with their current status and, consequently, less likely to perceive wildlife as positively benefitting their lives.

This sort of subjective evaluation of benefits also helps to explain the consistent positive association of employment and funeral benefits with the outcome measure. Given the scarcity of employment in the region, recipients of conservancy employment would presumably be more likely to compare their current reality to a counterfactual in which those opportunities did not exist. Given the scarcity of regular wage employment in the region, wages from the conservancy employment probably allow the respondents to match their aspirational economic goals. Likewise, recipients of funeral benefits are would likely compare their current reality to a counterfactual one in which they were solely responsible for all funeral costs and meat, and so a reality in which they receive assistance would seem clearly preferable to one in which they do not.

On the other hand, the counterfactual realities for meat distribution are perhaps more ambiguous. Meat benefits can be an important source of protein for households lacking the means to regularly purchase meat. But, the economic value of meat is relatively low, and an increased availability of meat is also associated with larger wildlife populations and an increase in crop and livestock depredation. Additionally, the receipt of meat is unlikely to allow recipients to match any aspirational economic goals. As such, it is unsurprising that meat should have a less clear association with respondents’ perceptions of the impact of wildlife on their lives.
5. Conclusion

Assuming that the difference in the benefits’ respective performance is due to some sort of psychological phenomena such as counterfactual thinking or the heuristic use of reference points, what practical lessons can be learned? First, these findings suggest that it is important that residents of community-based conservation programs develop reasonable expectations regarding the types and magnitude of the benefits and problems they are likely to experience because of their participation. It is natural for residents to develop some degree of expectations regarding anticipated benefits (Fabricius 2004). However, if too large a discrepancy exists between expectations and reality, residents in CBNRM programs can become frustrated and disillusioned (Musumali, et al. 2007). If expectations for CBNRM programs are tempered, both from the start and after the receipt of occasional benefits, the residents may be less susceptible to building up unrealistically high counterfactuals/reference points upon which to judge the benefits they receive.

Second, these results highlight the need for the open and transparent distribution of benefits to community members. In the absence of such transparency, residents are much more likely to suspect (rightly or wrongly) that others are benefitting more than they are, or that funds are being captured or mismanaged by conservancy leaders. This suspicion, in turn, may lead community members to be less satisfied with the benefits that they have received as they compare the received value to the amount they believe they might have been entitled to.

Third, the data suggests that predictable, regularly distributed benefits may be more effective than larger, episodic or single-purpose distributions such as the tuition, loans, or cash benefits. In a study of community-based conservation areas in Botswana, Pienaar et. al (2013) proposed using wildlife revenues for the creation of a community “conservation corps” offering
residents the opportunity to earn wages while engaging in community-oriented work (either full-time or day laborers), such as anti-poaching enforcement, patrolling at night to ward off predators, or constructing fences and kraals. In survey responses, residents of those communities were supportive of the idea of a conservancy corps and indicated a willingness to work for relatively low wages, leading the authors to conclude that the availability of these types of wildlife-funded job opportunities would be feasible and contribute to the success of those community-based conservation initiatives. Finances permitting, a similar approach might be effective in the conservancies included in this study, with short-term employment opportunities supplementing pre-existing full-time conservancy jobs. In this way, a greater proportion of conservancy residents would have the opportunity to receive some degree of employment benefits.
CHAPTER 4

CBNRM IN A HETEROGENEOUS ENVIRONMENT: THE POTENTIAL ROLE OF RISK AND ECONOMIC HETEROGENEITY ON COLLECTIVE ACTION

Stefan Carpenter
Ursula Kreitmair*1

Communities involved in CBNRM efforts are not homogenous entities, but rather vary both across and within communities on a range of measures (Agrawal and Gibson 1999; Vedeld 2000; Varughese and Ostrom 2001; Poteete and Ostrom 2004; Perez-Verdin, et al. 2009; King and Peralvo 2010). This heterogeneity can present a challenge to a community’s ability to collectively manage its natural resources (Ruttan 2006; Ruttan 2008; Andersson and Agrawal 2011). To date, socioeconomic and cultural forms of heterogeneity have received the most attention, but other forms of heterogeneity may have similarly significant impacts on cooperation. Risk heterogeneity is present in a range of natural resource problems, such as air pollution (Stewart, et al. 2014), climate change (Beck 2013), or forest fires (Syphard, et al. 2007). Yet, despite its prevalence, the impact of heterogeneous risk exposure on collective action remains understudied.

In this chapter, we used a common-pool-resource (CPR) game in a lab setting to investigate the effects of economic and risk heterogeneity, and interactions between the two

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1 The division of labor for the research presented in this chapter was as follows. I approached Dr. Kreitmair with the idea of modifying a prisoner’s dilemma game to account for the presence of human-wildlife conflict and the risk of increased losses that CBNRM participants face from the conservation of certain wildlife species. I was primarily responsible for review of relevant experimental and field-based literature. Based on a review of that literature, Dr. Kreitmair and I jointly conceived of the focus of the experiment on risk and economic heterogeneity in the presence of human-wildlife conflict. Dr. Kreitmair was primarily responsible for designing the experiment, although we jointly developed the experiment’s instructions and descriptive language. Dr. Kreitmair was solely responsible for the administration of the experiment. I had the initial responsibility of analyzing and interpreting the resulting data, and in authoring this chapter, with final edits being conducted in concert with Dr. Kreitmair.
forms of heterogeneity, on the collective governance of wildlife. We selected these two forms of heterogeneity because both are commonly found in CBNRM areas (see, e.g., Thakadu 2005; Lemessa, et al. 2013). While the existing academic literature discusses the need to recognize and account for heterogeneity in rural communities (Kumar 2005; Thakadu 2005; Spiteri and Nepalz 2006), much remains to be learned about how different types of heterogeneity (and the simultaneous presence of multiple types of heterogeneity) impact support for and involvement in community-based wildlife conservation.

The results of our experiment indicate that, in settings where cooperation results in increased risk of resource loss (such as when cooperation might prohibit the preventative killing of wildlife to limit human-wildlife conflict), unidimensional heterogeneous groups (i.e., varying in wealth or risk) are less likely to cooperate. At the individual level, wealth heterogeneity depresses cooperation across the board, while risk heterogeneity reduces the likelihood of cooperation by high risk individuals but increases the likelihood of cooperation by low risk individuals. When wealth and risk heterogeneities interact, we find that levels of group cooperation only drop slightly or may even increase for groups with unbalanced heterogeneity, where comparatively poor individuals faced a high risk of depredation and vice versa. We hypothesize that the difference in performance between the two mixed-heterogeneity groups may be the result of participants engaging in a psychological phenomenon known as ‘social comparison.’
I. LITERATURE REVIEW

a. Risk and Economic Heterogeneity in the Governance of Wildlife

Risk heterogeneity can occur in CBNRM areas because human-wildlife conflict (HWC)\(^2\) is often strongly correlated with proximity to core wildlife areas such as forests or national parks and, as such, households located closer to wildlife habitat tend to experience more frequent depredation (De Boer and Baquete 1998; Naughton-Treves 1998; Dickman 2010; Nijman and Nekaris 2010; Sogbohossou, et al. 2011; Granados and Weladji 2012; Regmi, et al. 2013). For instance, in the area surrounding Kruger National Park in South Africa, nearly 1 in 5 households located within three kilometers of the park border had experienced losses from HWC over a two-year period (Anthony, et al. 2010:232). That number dropped by roughly half (1 in 10) for households located between 3.1-7 kilometers of the park border, and half again (approximately 1 in 20) for households located between 7.1-15 kilometers.

Likewise, among communities surrounding Kahna National Park in India, the modeled probability of crop loss from HWC dropped from as high as 0.83 for households at the park border to as low as 0.3 for households 20 kilometers away (Karanth, et al. 2012:8). Households located within the park’s administrative buffer (a band of 1,005 km\(^2\) surrounding the 940 km\(^2\) core park) experienced a 0.74 modeled probability of livestock predation, compared to a 0.55 probability for households located outside of the buffer. Wildlife attacks on humans in the area surrounding Nepal’s Chitwan National Park followed a similar spatial pattern: 74% of all such attacks occurred within 1 kilometer of the park boundary (Silwal, et al. 2016:4).

Even within closely clustered households, the risk of HWC can vary significantly depending on the physical distribution of properties (Linkie, et al. 2007; Hartter, et al. 2011;)

\(^{2}\) Not including depredation by birds, rodents, or insects
Granados and Weladji 2012; Lemessa, et al. 2013). In a study of 25 villages bordering Uganda’s Kibale National Park, 90% of crop damage from wildlife occurred within 530 meters of the park boundary (Mackenzie and Ahabyona 2012:75), a similar statistic to that reported elsewhere (Lemessa, et al. 2013). Properties closer to the boundaries of wildlife areas serve as a buffer to those farther away (Mackenzie and Ahabyona 2012), such that the “best defense for a farmer is to have their field buffered by two or three others” (Hartter, et al. 2011:84).

Differences in susceptibility to HWC are not limited solely to proximity, however, as population density and natural and manmade barriers can serve to shield some locations from crop and livestock predation (Hartter, et al. 2011, Granados and Weladji 2012). In one example, a village on the border of the Maputo Elephant Reserve (Mozambique) reported far lower frequency of crop predation than did communities located farther from the reserve’s borders, a difference that the study’s authors attributed to its comparatively higher population density and the fact that the village was buffered from the reserve by the Maputo River (De Boer and Baquete 1998). Similarly, in a study of conflicts between farmers and elk in Canada, researchers found that elk preferred to visit crop fields that were proximate to forest cover and located far away from roads (Hegel, et al. 2009).

In addition to risk, communities can be economically heterogeneous, both internally and with respect to each other (Mehta and Heinen 2001; Thakadu 2005; King and Peralvo 2010; Sarker and Røskaft 2011; Snyman 2012). One study determined that conservancies in Namibia had per capita incomes ranging from N$66.8 to N$4,477.07 (Silva and Mosimane 2012:34). Within Namibia’s Mashi conservancy, inland households tend to be wealthier and have greater food security than households located along the edge of the Kwando River (Kanapaux and Child
In China, households located within the Przewalski’s gazelle’s home range had a mean income that varied geographically from 9,724 – 131,247 yuan (Hu, et al. 2009:555).

Risk and economic heterogeneity can overlap in CBNRM communities, with poorer households at the greatest risk of HWC because their lack of resources can force them to the edges of core wildlife areas (Hartter, et al. 2011; Kanapaux and Child 2011). The resulting increase in HWC can, in turn, exacerbate the poverty of those households by reducing food supplies and limiting economic opportunities (ibid). On the other hand, particularly in areas that engage in raising open-range livestock such as in Africa’s communal lands and Brazil’s Pantanal region, risk and economic heterogeneity are not necessarily related. In those areas, wildlife lives amongst and relies on the same natural resources as the livestock across an economically heterogeneous population (see, e.g., Zimmermann, et al. 2005; Romañach, et al. 2007).

b. The Potential Impact of Heterogeneity on Wildlife Governance

Why might heterogeneity matter for the collective governance of wildlife? Competing schools of thought have traditionally viewed heterogeneity as either potentially supporting or undermining collective action (Adhikari and Lovett 2006). On the one hand, heterogeneity can result in user groups possessing markedly different interests that impede agreement on core governance rules (Thakadu 2005; Adhikari and Lovett 2006). Awareness by users of these differences undermines trust and can generate conflict that weakens local management institutions (Poteete and Ostrom 2004; Adhikari and Lovett 2006; Chand, et al. 2015). On the other hand, as proposed most famously by Mancur Olson (1965), heterogeneity may facilitate collective action by enabling certain stakeholders to take the lead on collective efforts. Thus, in situations where the interests of different groups are aligned, heterogeneity may provide an opening for leaders of various groups to work together to guide governance of the resource
(Vedeld 2000). We turn to field and experimental literature to generate predictions about the specific impacts of risk and economic heterogeneities, and their interaction, on community-based wildlife governance.

i. The effect of individual wealth levels and economic heterogeneity on CBNRM

Thakadu (2005) observes that, from his experience working in Botswana’s CBNRM program, socioeconomic heterogeneity tends to undermine wildlife governance efforts, as it becomes more difficult to reconcile diverse interests within and between socioeconomic subgroups. Based on interviews conducted in communities in Botswana, Kerapeletwe (2005) offers a more ambivalent view of the impact of wealth heterogeneity, determining that wealthier individuals in the villages surveyed were less likely to follow resource governance rules, but that the negative impacts of heterogeneity were ameliorated as the economic stakes of cooperation increased. Finally, King and Peralvo (2010:275) examined the impact of economic heterogeneity in a wildlife-based CBNRM community in South Africa and found it challenging to isolate a single wealth effect, instead concluding that “wealth is a complicated category that demonstrates the intersections between environmental, economic, and cultural processes within particular settings.”

The impact of wealth heterogeneity is also unclear across the broader field-based common-pool resource literature. For example, a sample of literature involving studies of forest user groups suggests that economic heterogeneity could potentially undermine collective action (Chand, et al. 2015), have either a positive or negative association with collective action, depending on the degree of inequity and the distribution of resource benefits (Alix-Garcia 2008; Naidu 2009; Pradhan and Patra 2012), or no have consistent relationship with the efficacy of the groups’ resource governance (Adhikari and Lovett 2006). In their review of forest user groups in
Nepal, Varughese and Ostrom (2001) found that the presence of economic heterogeneity presents challenges, but that it is not determinative, and much of its impact hinges on other factors such as institutional arrangements.

A few studies based on meta-analysis have investigated economic heterogeneity and found that it generally undermines collective governance, but even those findings are subject to some important qualifications. In a review of 94 case studies involving common-pool fishery and irrigation system management, Ruttan (2008) determined that economic heterogeneity will generally have a negative impact on collective resource governance except where both (a) economically advantaged stakeholders gain by providing the collective good, and (b) the actions of those select stakeholders are sufficient to benefit the entirety of the group. Bardhan and Dayton-Johnson (2002) reviewed empirical irrigation studies and determined that heterogeneity of all forms (including wealth and income heterogeneity) is likely to undermine cooperation. After studying 228 forest user group cases, Andersson and Agrawal (2011) generally agreed with Varughese and Ostrom (2001) (above) when they concluded that economic heterogeneity has consistently negative impacts, but that those impacts can be ameliorated by effective local institutions. Two other recent meta-analyses discussed the potential importance of economic heterogeneity but were unable to include it as an explanatory variable because of insufficient data about that measure in the studies surveyed (Brooks, et al. 2012; Brooks 2017).

The behavioral experiment literature is also mixed regarding the potential impact of economic heterogeneity. Chan et al. (1996; 1999) found that, under the right conditions, economic heterogeneity increased group contributions in a public goods game. Similarly, Cardenas et al (2002) and Hayo and Vollan (2012) conducted field experiments in Columbia and
Southern Africa, respectively, and found that economically heterogeneous groups realized better outcomes in a public goods game than did homogenous groups.

Other experiment-based studies suggest that economic heterogeneity is likely to have a deleterious impact on collective action (Isaac and Walker 1988; Rapoport and Suleiman 1993; Cherry, et al. 2005; Anderson, et al. 2008; Seçilmiş and Güran 2012; Fischbacher, et al. 2014), including an experiment conducted in Columbia (Cardenas 2003) and a meta-analysis of experiments conducted in Nigeria and Ghana (Rosenbaum, et al. 2016). Of particular interest, Anderson et al (2008) and Fung (2014) both found economic heterogeneity to only have a significant impact under certain conditions. Anderson et al (2008) observed an impact on collective action only when the participants were aware of the heterogeneity; absent knowledge of inequality, the heterogeneity had no impact. Fung (2014) examined both income heterogeneity and the skew of those incomes (such as 15-30-45 or 20-20-70) and determined that symmetric heterogeneity decreased public goods contributions while asymmetric heterogeneity did not.

While there is a conspicuous lack of agreement among the available literature regarding the anticipated impact of economic heterogeneity, a majority of the studies we reviewed suggest that, when participants are aware of it, economic heterogeneity is likely to undermine collective action. As such, our first hypothesis is as follows:

Hypothesis 1: Under conditions of full information, the presence of economic heterogeneity will decrease groups’ overall rate of contribution to collective action.

Setting aside, for the moment, the issue of heterogeneity, the impact of individual changes in wealth is also unclear. Field studies of wildlife oriented CBNRM are conflicting, alternately finding that wealth is positively correlated (Romañach, et al. 2007; Bandyopadhyay,
et al. 2009) or negatively correlated (Kanapaux and Child 2011; Silva and Mosimane 2012) with participation in wildlife conservation.

Public goods experiments have examined the proportion of endowments contributed by participants in varying endowments. Multiple studies found that high-endowment players contributed a lower proportion of their endowments than did low-endowment players (Rapoport and Suleiman 1993; Chan, et al. 1996; Cardenas, et al. 2002; Cherry, et al. 2005; Buckley and Croson 2006; Visser and Burns 2015; Hargreaves Heap, et al. 2016; Kingsley 2016). However, at least two studies found no significant difference between players of different endowment levels regarding the amount of endowments contributed to a collective pool (Hofmyer, et al. 2007; Seçilmiş and Güran 2012). These experiments differ in design from ours, in that (as discussed below) our design required a binary decision regarding contribution to collective action. Nevertheless, the results of these experiments suggest that individuals with lower endowments may remain more invested in collective action than do their wealthier counterparts.

Taken together, the bulk of the field survey and experimental literature supports the following hypothesis:

Hypothesis 2: Participants’ endowment will be negatively correlated with their likelihood of contributing to collective action.

ii. Evidence of the effect of risk and risk heterogeneity on CBNRM

We identified no field studies that explicitly examined the potential impact of risk heterogeneity on CBNRM. However, several studies have found a relationship between exposure to HWC and negative attitudes toward wildlife and/or specific wildlife species (Songorwa, et al. 2000; Spiteri and Nepalz 2006; Carter, et al. 2014). Additionally, Kahler and Gore (2015) found that the species which conservancy residents in Namibia perceived as posing the most risk for
HWC were moderately correlated with species identified by those same individuals as being at the highest risk of being poached. Read together, this literature suggests that, in a heterogeneous environment, individuals with a higher perceived risk of experiencing HWC may have a less favorable opinion of wildlife and be more likely to feel the need to engage in retaliatory poaching. As such, their interests may diverge from stakeholders with a lower perceived risk of HWC, and this divergence of interests could undermine collective wildlife governance in an environment of heterogeneous risk.

Whereas field research on risk heterogeneity in wildlife conservation appears to be largely barren, a few experiments have focused more generally on the impact of risk heterogeneity on collective action. Fischbacher et al (2014) introduced heterogeneity of payouts into a public goods game, where half of the group would receive a low payout and the other a high payout, but it was unclear at the beginning of each round of play which player would receive which payout. They observed that this heterogeneity significantly reduced unconditional contributions (i.e., contributions that were not pre-conditioned on how many other group members contributed).

Théroude and Zylbersztejn (2019) recently conducted an experiment that is particularly relevant to this study. The authors introduced both group and individual-level environmental risk into a voluntary contribution game. In both risk scenarios, participants could choose to contribute to a group pool, from which they would receive an individual payout based on the magnitude of the total investment by the group that round. To introduce risk, the authors made the payout from the group pool probabilistic, rather than automatic. In the homogenous risk treatment, a single random draw took place each round that determined whether the participants received the payout (ibid:9-10). In the heterogeneous risk treatment, each participant received their own random
number draw each round. The authors found no variance among the treatments in the overall patterns of group cooperation (in other words, all of the treatments followed a pattern of contributions typically found in these sorts of games, with heightened earlier cooperation followed by a marked decay of cooperation over time) (ibid). Contribution in the heterogeneous risk treatment was not significantly different from that in the riskless control at any stage. Groups with homogenous risk experienced improved cooperation early in the game but that effect disappeared thereafter (ibid). An experiment by Rohde and Rohde (2011) provides a possible explanation for the lack of observed difference between groups with heterogeneous risk and homogenous risks, in that they found that individuals’ attitudes toward risk were only weakly influenced by the risk that others faced.

A growing body of experimental literature has also examined the role of increased individual risk in collective action. Among experiments similar in design to the one employed here, both Gangadharan & Nemes (2009) and Cardenas, et al. (2017) found that risk associated with payouts from collective investments decreased participants’ contributions. On the other hand, Levati and Morone (2013) found that, when the worst outcome of a game still allowed for efficiency gains, risk had no appreciable effects on contributions. We were not able to identify any experiments that found that increased risk was positively correlated with collective action.

Recently, a number of researchers have used “threshold games” to examine the impact of risk and risk heterogeneity on collective attempts to avoid catastrophic “tipping points” from environmental hazards such as climate change. In these threshold games, participants must collectively contribute a certain amount into a group pool by a given time (often the last round of the game), and the failure to do so results in a risk of losing some or all of the post-game payouts from the pool. Of note, Robbins (2017) found that groups with homogenous risk coordinated
more than those in which group members faced heterogeneous risk of harm from failure to meet the threshold. However, Abou Chakra et al (2018), used a simulated experiment to predict that risk heterogeneity alone should have no appreciable impact and, when paired with wealth heterogeneity, it should actually increase cooperation, with the magnitude of that increase being positively correlated with the degree of wealth inequality. In examining only the issue of risk (and not introducing risk heterogeneity), threshold game experiments suggest that cooperation is positively correlated with the increased risk of harm from missing a target threshold (Milinski, et al. 2008; Santos and Pacheco 2011).

The nature of the risk in those threshold games differs from that examined in this experiment in that, in our game (as further discussed below) there is a risk associated with contributing, whereas in the threshold game the risk is associated with not contributing enough to avoid catastrophic loss. Hence, while these studies provide guidance to behavioral expectations in the presence of risk, they do not provide expectations for how behavior might differ under risk heterogeneity – where individuals face different odds of incurring a loss. However, there is compelling evidence that when individuals face greater risk, i.e., lower odds of a successful gamble, they are less likely to take the gamble. First, if gains and losses remain the same (as in our experiment) and just the risks change, higher risks also equate to lower expected returns. The marginal payoff from investing in the group project is lower, suggesting that fewer individuals will take that action. Second, many individuals are risk averse (Holt and Laury, 2002), suggesting that individuals are less likely to cooperate in our game as risks increase.3

These individual tendencies also have implications for behavioral expectations of risk heterogeneity. In this context, as cooperation by high risk individuals drops, cooperation by low

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3 This, however, depends on the participants’ reference point – i.e., whether they see the potential outcome as gains or losses (Kahneman and Tversky 1979).
risk individuals should also drop because studies suggest that about 50% of individuals will only cooperate if others do so as well (Fischbacher et al., 2001; Frey & Meier, 2004):

Hypothesis 3: The presence of risk heterogeneity will reduce collective action at the group level compared to homogenous risk settings.

Hypothesis 4: An increased risk of loss will negatively impact the likelihood that participants will contribute to collective action.

Given that the available literature suggests that both economic and risk heterogeneity will have a negative impact on collective action, we would expect that environments with both risk and economic heterogeneity would also experience decreases in collective action. Similarly, as the literature suggests that both elevated risk and endowment levels will be correlated with lower levels of contribution, we would anticipate that individuals with higher endowments and higher levels of risk would be least likely to contribute to collective action. Therefore, our fifth and sixth hypotheses are as follows:

Hypothesis 5: The presence of both risk and economic heterogeneity will have a negative impact on the likelihood that participants will contribute to collective action.

Hypothesis 6: Individuals that have both high risk and high endowments will have a lower likelihood of contributing than will individuals with other combinations of risk and endowment magnitudes.

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4 As noted above, Abou Chakra et al (2018) predicted that risk heterogeneity could increase cooperation in an economically heterogeneous environment. However, that research involved a simulated experiment in a threshold game (i.e., the risk was associated with noncooperation, rather than cooperation as in our game), and so the participants in our experiment faced a differing set of motivational drivers than assumed by the authors of that study.
II. A Model of Risk Heterogeneity

To formalize the impacts of risk heterogeneity in the context of human-wildlife conflict, consider the following hypothetical scenario. An individual resides in a CBNRM community that collectively earns income from the presence of wildlife through tourism. The money earned by the community is shared equally amongst the community’s residents. This individual has an identified economic interest (such as crops, livestock, or infrastructure) that is at risk of harm from wildlife (in the form of predation or infrastructure damage). The individual can prevent wildlife from harming her economic interest by engaging in the preventative killing of problem wildlife. However, the resulting diminishment of local wildlife stocks decreases the community’s income from tourism which, in turn, results in each of the community members receiving a smaller distribution of community funds. If, on the other hand, the individual chooses to engage in less effective but non-lethal deterrent effects, wildlife stocks do not suffer, but there is a chance that the individual suffers significant personal losses from HWC. The others in this individual’s community have similar livelihoods and are faced with the same HWC deterrence choices.

This scenario is, of course, very much a simplification of a complex problem. Among many things, it ignores the psychological costs of living with wildlife and the opportunity costs associated with guarding crops, livestock, and other resources against HWC (Hill 2004). It also sets aside the problem of elite capture and/or inequitable distribution of benefits within CBNRM communities. Finally, it does not encompass climatic, ecological, or economic factors associated with precipitation levels, species recovery rates, or tourism patterns, respectively. Nevertheless, this simplified scenario represents the foundational premise of the CBNRM approach discussed
throughout this dissertation: people must earn money from wildlife in an amount that at least equals the costs imposed by HWC.

The scenario for our hypothetical individual can be formalized as the following game: Individual, $i$, is part of a group $N$ (i.e., $i \in N$ where $N = \{1, \ldots, n\}$). Every period she chooses whether to contribute to a group effort, $a_i$ (e.g. collectively conserving wildlife stocks by not engaging in the preventative or retaliatory killing of wildlife). If she contributes, $a_i = 1$, otherwise, $a_i = 0$ (i.e. she undermines the collective conservation effort to protect herself from HWC). If, and only if, she engages in the collective effort ($a_i = 1$), she runs the risk, $p_i$, of incurring a cost, $c$ (e.g., the loss of crops, livestock, or infrastructure due to HWC) which is subtracted from her private earnings she receives, $e$. There is a benefit associated with participating in the collective effort – every group member receives a benefit, $b$, for each group member that also decides to contribute. Thus, the expected payoff function is:

$$E(\pi_i) = e - a_i p_i c + b \sum_{i}^{N} a_i$$

To qualify as a commons dilemma individual and group incentives must be at odds. In other words, in a linear game such as this, the following must hold:\footnote{Standard assumptions of rational, risk-neutral, self-regarding, payoff-motivated actors apply.}

Defection is the dominant strategy for individuals if the payoff for defecting is greater than that for participating:

$$\pi_i^{a_i=0} > E(\pi_i^{a_i=1})$$

$$e + b(A - 1) > e - p i c + bA$$

$$p_i c > b$$

Where $A = \sum_{i}^{N} a_i$. 
The expected loss to the individual from human-wildlife conflict must be greater than the benefit derived from the conserved wildlife stocks. Therefore, the individual best response is to defect, \( a_i = 0 \), and the Nash equilibrium is defection by all members, \( A = 0 \).

At the group-level, contribution is socially optimal if the payoff for contribution is greater than that for defection:

\[
E(\pi_{N}^{A=N}) > E(\pi_{N}^{A=N-1})
\]

\[
N(e - \bar{p}c + Nb) > N(e + (N - 1)b) - \bar{p}_{i \in a_i=1}(N - 1)c
\]

Assuming homogeneous risks:

\[
Nb > pc
\]

The group benefits from an individual’s contribution if the total group benefit from the conserved wildlife stocks outweighs the individual’s expected loss.\(^6\) Therefore, the social optimum for the group is full contribution \( A = N \).

### III. Experimental Parameters and Implementation

Behavioral laboratory experiments are a widely accepted approach to developing our understanding of why and when collective resource governance succeeds or fails (Deadman, et al. 2000). In the real world, the collective governance of resources takes place in a mélange of institutions and variables. The shear complexity of these situations can make it difficult to (a) determine the impact of any particular variable, and (b) draw generalizations across communities or resource types. By strictly controlling the circumstances under which individuals interact, experiments allow the researcher to isolate and investigate the impact of specific variables of interest. This allows the formulation of generalizable theory which can then be tested across a range of conditions in the field.

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\(^6\) Note, this assumes independent probabilities of experiencing a loss due to human-wildlife conflict.
The experiment used was programmed and conducted using Z-Tree (Fischbacher 2007). While the participants were all in the same room throughout the duration of the experiment, their interaction with one another was solely through the computer interface and they did not communicate with one another or otherwise interact face-to-face. Each participant saw the aggregated decisions of their fellow group members, but it was not possible for the participants to identify which of their group members made any particular decision.7

The experiment was conducted in late 2018 and early 2019 at a large Midwestern University. The participants were recruited from that University’s undergraduate student body and were from diverse majors. Participants were randomly assigned to groups of four and received a participant number (1-4) within the group. Participants remained in the same group and retained the participant number during the experiment. We used five different group treatments (further discussed below) and repeated each treatment seven times. In total, 28 participants were assigned to each of the treatments, with 140 individuals ultimately participating in the experiment.

Before participating in any stage, each participant received on-screen and printed instructions (worded in a neutral manner), which they could read at their own pace.8 All instructions were also read publicly. The experiment lasted approximately 45 minutes. Participants received payoffs in terms of Experimental Currency Units (ECU) and at the end of

---

7 In the real world, CBNRM participants often interact face-to-face, and they may or may not have anonymity in their decision-making. For this experiment, we chose anonymity and the lack of communication so that we could focus as narrowly as possible on the impacts of risk and economic heterogeneities. It is quite possible that communication and/or transparent decision-making could have a material impact on these two forms of heterogeneity (and on their interaction), and this may be a fertile ground for future research.

8 Full instructions are available in the appendix.
the experiment these payoffs were exchanged into US dollars at a rate of 65 ECU = $1. This amount was added to the $5 show-up payment and was paid out in private.

The experiment was comprised of two stages. Before proceeding to the decision-making phase of each stage, participants completed in quizzes to ensure their comprehension of the stage design. Stage 1 was a risk elicitation task based on Holt and Laury (2002). In this stage, participants were asked to make ten choices between a certain payout or a 50% chance of an alternate 250 ECU payout (Table 1). The certain payouts began at 25 ECU and increased in 25 ECU increments until they matched the alternate payout. Assuming the participants have a linear risk tolerance, their choices allow for the quantification of their relative risk tolerances based on the stage at which they opt for the certain over the alternate payout. Participants were paid for one of their ten choices, selected at random, with the payoff determination withheld until the end of the experiment.

**TABLE 1: Stage 1 Risk Tolerance Decisions**

<table>
<thead>
<tr>
<th>25 ECU for certain</th>
<th>or</th>
<th>50% 0, and 50% 250</th>
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<tbody>
<tr>
<td>50 ECU for certain</td>
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<tr>
<td>75 ECU for certain</td>
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<td>100 ECU for certain</td>
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<td>125 ECU for certain</td>
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<td>150 ECU for certain</td>
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<td>175 ECU for certain</td>
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<td>200 ECU for certain</td>
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<td>225 ECU for certain</td>
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<tr>
<td>250 ECU for certain</td>
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Stage 2 was a binary choice common pool resource game that was repeated for 15 rounds. Participants received a token endowment each round, \( e_i \), the value of which was determined by the treatment they were in. In each round, participants decided whether to contribute in the collective group effort for that round \( (a_i) \). If the participants chose to invest in the group effort, they incurred a risk of losing 30 tokens, \( c_i \), from their endowment. These risks were independent of one another across treatments.

Regardless of the action the participants took each round, they received a benefit from any group member’s contribution in the group effort (including their own) in the amount of 8 tokens for each participant that contributed \( (b) \). In heterogeneous treatment conditions, there were two types of participants with varying risk, \( p_i \), and endowments, depending on the treatment. After each round, individuals received information on i) how many group members invested in the group project, ii) whether a loss occurred (if they contributed), and iii) their earnings for the round. Information from previous rounds was available to participants when they made their decisions.

We used five treatments (one control and four variations) to explore heterogeneity effects (Table 2). All treatments were identical in group payoffs to ensure that group incentives remained the same, so that differences in contribution across groups could be attributable to changes in individual incentives. The Homogeneous (HOM) treatment had symmetric players, each with medium levels of endowment (42 tokens per round) and risk (50% chance of incurring a loss). The Heterogeneous Wealth (HET WEALTH) treatment endowed all participants with the same risk (50%) but provided two participants with high endowments (48 tokens) and two participants with low endowments (36 tokens). In contrast, the Heterogeneous Risk (HET RISK) treatment provided all participants with the same starting endowment (42 tokens) but varied the
risk: two participants faced a 70% chance of incurring a loss and the remaining two faced a 30% chance. The final two treatments, *Balanced Heterogeneity (HET BAL)* and *Unbalanced Heterogeneity (HET UNBAL)*, combined different endowments and different risks. In the Balanced Heterogeneity treatment, two participants received high endowments (48 tokens) and high risk (70%), while the remaining two received low endowments (36 tokens) and low risk of losses (30%). In the Unbalanced Heterogeneity treatment, the two individuals with high endowments (48 tokens) faced a small chance of loss (30%), while those with low endowments (36 tokens) faced a high chance of loss (70%).

<table>
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<th>Table 2: Overview of Treatments</th>
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<tr>
<td><strong>Treatment</strong></td>
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<tr>
<td>Homogeneous Group (HOM)</td>
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<td>Heterogeneous Wealth (HET WEALTH)</td>
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</tbody>
</table>

Note: All participants face the same cost, c, of 30 tokens, and receive the same benefit, b, (8 tokens) from the group for each individual that decided to contribute.

The participants received full information about their respective risk and endowments and about the risk and endowment levels of the others in their group. So, for example, in the heterogeneous treatment, a participant with low risk and high wealth would be informed both of
her own risk and endowment (0.3 and 48 ECU, respectively) and of the fact that one other group member would share have the same wealth/risk combination, while the other two members would have a high risk/low wealth combination (0.7 and 36 ECU, respectively).

We used a panel data model to determine the impact of the treatments on (a) total group cooperation, and (b) the likelihood that a participant chose to contribute in the collective effort. The use of panel data models is appropriate here, because models of this type allow the researcher to better capture the complexity of human behavior (Hsiao 2007).

IV. Results

The 140 undergraduate student participants earned, on average, $19.10, inclusive of payouts from both stages and the $5 show up fee. In Stage 1, approximately 90% of the participants evidenced the expected linear risk tolerance, consistently selecting the probabilistic payout at lower guaranteed payouts (or always/never selecting it) and, at a certain guaranteed payment value, switching to consistently selecting guaranteed payments thereafter. However, 16 participants (10.67%) did not evidence linear risk preferences. This group of participants intermittently switched their responses; some used a pattern (alternating selecting the certain or probabilistic payouts) whereas other choices appear to be random.

Additionally, four participants (including two of those without linear risk preferences) appear to have erroneously flipped the direction of their responses, choosing to accept very low guaranteed payments, while turning down the maximum guaranteed payment. For instance, one such participant opted to choose the certain payment for all of the risk tolerance decisions (from 25 ECU to 225 ECU), except for the final one, in which the participant opted for a 50% chance of a 250 ECU payment, rather than a guaranteed 250 ECU payment. Similarly, another
participant selected the guaranteed payouts for the first five decisions (choosing 25 ECU through 125 ECU) but selecting the riskier option in all subsequent decisions.

To avoid issues of non-linear risk preferences and the potentially flipped responses, the responses for Stage 1 were coded to reflect the total number of decisions in which the participants selected the probabilistic outcome. Subjects had a mean score of 4.01 on this measure of risk tolerance (with a maximum risk tolerance of 10 and a minimum risk tolerance of 0), suggesting slight risk aversion (consistent with risk aversion observed by Holt and Laury 2002).9

a. Results for Total Group Contribution

In Stage 2, across all treatments, the groups averaged 2.08 individuals (out of 4) investing in the group effort each round. However, as shown in Figure 1, the level of cooperation varied across the treatments. On average, the heterogeneous wealth and risk treatments each experienced approximately 0.5 fewer participants contributing than in the homogeneous treatment, while the balanced and unbalanced heterogeneous treatments realized roughly 0.25 less and more contributing participants, respectively. Overall, group contributions were slightly higher from rounds 1-10 than they were in rounds 11-15, although the control and two of the treatments experienced a noticeable increase in average contributions in the final round (Figure 2).

---

9 To ensure that the findings from this project are not the result of this coding decision regarding the “flipped” responses, we reran the final model to using the maximum accepted risk, both with and without reversing the “flipped” decisions. These alternate regressions did not return materially different results and so they are not addressed further here.
FIGURE 1: Average Number of Participants Cooperating (by Contributing to the Group Pool)

FIGURE 2: Average Number of Participants Cooperating by Round
To test whether these differences in contribution rates were significant, we compared the group contribution rates across the different treatments using Mann-Whitney two-sample tests (Table 3). The results of that comparison suggest the following impacts of heterogeneity. The presence of heterogeneity, either in wealth or risk, reduced a group’s ability to cooperate compared to a homogenous environment, except where low risk was paired with high wealth and vice versa (HET UNBAL). Single forms of heterogeneity (HET WEALTH and HET RISK) appeared to have the greatest impact in discouraging collective action. There was no significant difference in collective action between wealth or risk heterogeneity and, to the extent that any difference exists, it appeared to be minor. Multidimensional heterogeneity, surprisingly, fared better than single forms of heterogeneity when compared to a homogenous environment. When risk was positively correlated with wealth (i.e., high wealth individuals also faced high risk, as in the HET BAL treatment) groups were more cooperative but still not as cooperative as groups that faced unbalanced risk and wealth heterogeneity, and groups that were homogeneous.

**TABLE 2: Mann-Whitney Two-Sample Test of Group Contributions Across Treatments (15 periods)**

<table>
<thead>
<tr>
<th></th>
<th>HET WEALTH</th>
<th>HET RISK</th>
<th>HET BAL</th>
<th>HET UNBAL</th>
</tr>
</thead>
<tbody>
<tr>
<td>HOM</td>
<td>3.231***</td>
<td>3.080***</td>
<td>1.659*</td>
<td>-1.171</td>
</tr>
<tr>
<td>HET WEALTH</td>
<td>--</td>
<td>-0.241</td>
<td>-1.947*</td>
<td>4.886***</td>
</tr>
<tr>
<td>HET RISK</td>
<td>--</td>
<td>--</td>
<td>-1.766*</td>
<td>4.833***</td>
</tr>
<tr>
<td>HET BAL</td>
<td>--</td>
<td>--</td>
<td>--</td>
<td>3.103***</td>
</tr>
</tbody>
</table>

* z-score
210 combined observations for each test
* p-value ≤ 0.1, ** p-value ≤ 0.05, *** p-value ≤ 0.01
Although randomization allows for simple two-sample comparisons, to gain a more nuanced understanding of treatment effects we also analyzed group data using a panel data structure and relevant control variables in random effects ordered logit regression models. As shown in Table 3, Models 1 and 2 examined the relationship of the treatments to the likelihood that the group would experience an additional contribution from a participant. Model 1 included treatment dummies only. In Model 2, we included a range of group-level control variables: i) the previous round’s group-level investment total (to account for path dependency), ii) average number of losses in the previous period (to account for changes in perceived risk after a loss occurs), iii) average risk preferences\(^{10}\) (to account for variation in risk tolerance across groups), iv) a “single-group session” dummy reflecting when sessions included only enough participants to conduct the experiment with one group, meaning that individuals were aware of who else was in their group,\(^{11}\) and v) period (to account for time trends in the likelihood of cooperation).

Contribution rates were not significantly different compared to the HOM treatment, although the lower likelihoods of contribution in the unidimensional heterogeneity were large and mirror the findings from Figure 1. Additionally, contribution rates in the HET WEALTH treatment were significantly lower than in the HET UNBAL treatment in both models (Model 1 p=0.0510, Model 2 p=0.0613).

\(^{10}\) Measured by averaging group members’ total risk tolerance elicited in Stage 1, described above.
\(^{11}\) In most sessions, we were able to run multiple simultaneous games, so that participants were unsure as to whom was in their group. However, because this experiment involved student volunteers, participant turnout did not always allow sufficient numbers to conduct the experiment with more than one group in a session. Awareness of group membership may have created peer-pressure that could have changed the likelihood of cooperative behavior by members of those groups.
### TABLE 3: Group Investment Decisions

<table>
<thead>
<tr>
<th>Group Investments</th>
<th>Model 1</th>
<th>Model 2</th>
</tr>
</thead>
<tbody>
<tr>
<td>HET WEALTH</td>
<td>0.304</td>
<td>0.329</td>
</tr>
<tr>
<td></td>
<td>[0.264]</td>
<td>[0.241]</td>
</tr>
<tr>
<td>HET RISK</td>
<td>0.334</td>
<td>0.709</td>
</tr>
<tr>
<td></td>
<td>[0.299]</td>
<td>[0.720]</td>
</tr>
<tr>
<td>HET BAL</td>
<td>0.573</td>
<td>0.824</td>
</tr>
<tr>
<td></td>
<td>[0.599]</td>
<td>[0.815]</td>
</tr>
<tr>
<td>HET UNBAL</td>
<td>1.739</td>
<td>1.992</td>
</tr>
<tr>
<td></td>
<td>[0.613]</td>
<td>[0.499]</td>
</tr>
<tr>
<td>Lagged Group Investment</td>
<td>1.203</td>
<td></td>
</tr>
<tr>
<td></td>
<td>[0.282]</td>
<td></td>
</tr>
<tr>
<td>Lagged Average Loss</td>
<td>0.638</td>
<td></td>
</tr>
<tr>
<td></td>
<td>[0.456]</td>
<td></td>
</tr>
<tr>
<td>Average Risk Preferences</td>
<td>2.626***</td>
<td></td>
</tr>
<tr>
<td>(Average Total Risk)</td>
<td>[0.000]</td>
<td></td>
</tr>
<tr>
<td>Single-Group Session</td>
<td>2.292</td>
<td></td>
</tr>
<tr>
<td></td>
<td>[0.106]</td>
<td></td>
</tr>
<tr>
<td>Period</td>
<td>0.929***</td>
<td></td>
</tr>
<tr>
<td></td>
<td>[0.000]</td>
<td></td>
</tr>
<tr>
<td>Observations</td>
<td>525</td>
<td>490</td>
</tr>
<tr>
<td>$\chi^2$</td>
<td>5.117</td>
<td>75.35***</td>
</tr>
</tbody>
</table>

Note: Regressions are random effects panel ordered logit models with errors clustered at the group level.

- Standard errors in parentheses
- All results displayed as odds ratios
- * $p$-value ≤ 0.1, ** $p$-value ≤ 0.05, *** $p$-value ≤ 0.01

b. **Results for Individual Contribution**

Individual contribution levels varied across different treatments and sub-treatment groupings. In the HET WEALTH treatment, across 15 rounds, participants contributed toward the group goal the same number of times regardless of their endowment (Figure 3). This result
suggests that, although wealth heterogeneity may decrease collective action overall (compared to the HOM treatment), there is no difference in cooperation across the relatively poor and the relatively wealthy. Rather, the mere presence of wealth heterogeneity appears to negatively impact cooperation.

**FIGURE 3: Mean Rate of Cooperation of Individuals by Treatment and Type**

By contrast, risk levels seem to change the individual likelihood of contributing toward the group goal based on the individual level of risk the participant faces. In the HET RISK treatment, there is a clear difference in how often high-risk individuals cooperated compared to low-risk individuals, with low-risk individuals cooperating at nearly twice the rate as those facing high risk. This difference was amplified in the multidimensional heterogeneity treatments. These findings suggest that risk is a strong driver of cooperative behavior but may be moderated by wealth.

To account for individual and group effects over time, we also used random effects logit models to analyze the impacts of heterogeneity on individual contribution levels (Table 4). We
included dummy variables for low risk, high risk, low endowment, and high endowment treatments,\textsuperscript{12} comparing these groups to individuals in the HOM treatment (where both risk and endowment are at medium levels). We included these variables rather than treatment dummies because, given heterogeneity, individuals faced different incentive structures within each treatment, which resulted in different contribution behavior (as seen in Figure 3).

Whereas changes in endowment had no significant impact on the likelihood of contribution, risk had a large, statistically significant association with contribution likelihood. In settings of with low risk, individuals were more than 4 times as likely to cooperate compared to those facing moderate risk in the HOM treatment, while high risk individuals were approximately 63\% less likely to cooperate (Model 3).

To account for interactions between the two types of heterogeneities, we included interaction effects between the risk and endowment dummy variables (Model 4). Endowment remained insignificant in this model, while risk remained a notable influence on participation. As in Model 3, participants in this analysis continued to display a significantly lower likelihood of contributing to the group pool. The low risk dummy variable lost significance in this model, but this loss does not necessarily mean that it no longer drove an increased likelihood of contribution. Rather, the extent to which low risk impacted cooperation appears to have depended on the endowment setting. When faced with both low risk and low endowment, participants were more than 6 times as likely to contribute toward the group project compared to those in the HOM treatment (who were in a medium risk/medium endowment setting). When individuals with low risk received high endowments, their likelihood of contributing increased.

\textsuperscript{12} These dummies looked at risk and endowment levels across multiple treatments. For instance, low-endowment participants in the HET WEALTH treatment were lumped together with low-endowment participants in the HET BAL and HET UNBAL treatments.
even further, and they were over 18 times as likely as those in the HOM treatment. Interestingly, low-endowment individuals facing high risks were more likely to cooperate than the low risk individuals in the HET RISK treatment.

**TABLE 4: Individual Investment Decisions**

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Low Risk</td>
<td>4.208***</td>
<td>1.037</td>
<td>1.289</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>[0.006]</td>
<td>[0.961]</td>
<td>[0.708]</td>
<td></td>
<td></td>
</tr>
<tr>
<td>High Risk</td>
<td>0.377**</td>
<td>0.213**</td>
<td>0.287**</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>[0.032]</td>
<td>[0.029]</td>
<td>[0.033]</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Low Endowment</td>
<td>1.070</td>
<td>0.384</td>
<td>0.450</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>[0.886]</td>
<td>[0.176]</td>
<td>[0.230]</td>
<td></td>
<td></td>
</tr>
<tr>
<td>High Endowment</td>
<td>1.026</td>
<td>0.396</td>
<td>0.414</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>[0.955]</td>
<td>[0.213]</td>
<td>[0.179]</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Low Risk X Low Endowment</td>
<td>6.658*</td>
<td>4.530</td>
<td>20.21***</td>
<td>11.75***</td>
<td></td>
</tr>
<tr>
<td></td>
<td>[0.082]</td>
<td>[0.117]</td>
<td>[0.000]</td>
<td>[0.000]</td>
<td></td>
</tr>
<tr>
<td>Low Risk X High Endowment</td>
<td>18.85***</td>
<td>14.65***</td>
<td>63.23***</td>
<td>42.26***</td>
<td></td>
</tr>
<tr>
<td></td>
<td>[0.006]</td>
<td>[0.004]</td>
<td>[0.000]</td>
<td>[0.000]</td>
<td></td>
</tr>
<tr>
<td>High Risk X Low Endowment</td>
<td>5.254*</td>
<td>4.111</td>
<td>3.073*</td>
<td>3.738**</td>
<td></td>
</tr>
<tr>
<td></td>
<td>[0.095]</td>
<td>[0.105]</td>
<td>[0.089]</td>
<td>[0.029]</td>
<td></td>
</tr>
<tr>
<td>High Risk X High Endowment</td>
<td>1.759</td>
<td>1.453</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>[0.539]</td>
<td>[0.634]</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lagged Investment Decision</td>
<td>1.341</td>
<td></td>
<td></td>
<td>3.141***</td>
<td></td>
</tr>
<tr>
<td></td>
<td>[0.223]</td>
<td></td>
<td></td>
<td>[0.001]</td>
<td></td>
</tr>
<tr>
<td>Lagged Loss</td>
<td>0.963</td>
<td></td>
<td></td>
<td>0.607</td>
<td></td>
</tr>
<tr>
<td></td>
<td>[0.856]</td>
<td></td>
<td></td>
<td>[0.157]</td>
<td></td>
</tr>
<tr>
<td>Lagged # of Others Investing</td>
<td>1.030</td>
<td></td>
<td></td>
<td>0.935</td>
<td></td>
</tr>
<tr>
<td></td>
<td>[0.718]</td>
<td></td>
<td></td>
<td>[0.653]</td>
<td></td>
</tr>
<tr>
<td>Risk Preferences (Total Risk)</td>
<td>1.362***</td>
<td></td>
<td></td>
<td>1.199</td>
<td></td>
</tr>
<tr>
<td></td>
<td>[0.001]</td>
<td></td>
<td></td>
<td>[0.350]</td>
<td></td>
</tr>
<tr>
<td>Single-Group Session</td>
<td>1.760</td>
<td></td>
<td></td>
<td>3.000**</td>
<td></td>
</tr>
<tr>
<td></td>
<td>[0.163]</td>
<td></td>
<td></td>
<td>[0.015]</td>
<td></td>
</tr>
<tr>
<td>Period</td>
<td>0.954***</td>
<td></td>
<td></td>
<td>0.948*</td>
<td></td>
</tr>
<tr>
<td></td>
<td>[0.001]</td>
<td></td>
<td></td>
<td>[0.066]</td>
<td></td>
</tr>
</tbody>
</table>

$\chi^2$  40.46***  87.82***  154.6***  77.96***  219.1***
In Model 5 we tested for the robustness of our findings by including a range of individual level control variables: i) the participant’s investment decision in the prior round (to account for path dependency), ii) whether the participant experienced a loss in the previous period (to account for changes in perceived risk after a loss occurs), iii) how many group members contributed in the previous round (to account for conditional cooperation), iv) the participant’s risk preference (elicited in Stage 1 of our experiment), and v) period (to account for time trends in the likelihood of cooperation).

In this model, high risk remained a significant factor in dampening cooperative behavior. Similarly, low risk continued to increase the likelihood of cooperation, although this effect was only significant in the high endowment settings. Among the control variables, only risk preference and period had significant effects on individual contributions: the higher the participant’s tolerance for risk, the more likely she was to contribute, and with every passing period individuals became about 5% less likely to contribute.

Finally, to test for individual-level interaction effects between wealth and risk (i.e., to explain why low risk is significant in the low endowment setting only), we ran Models 6 and 7 using only data from the multidimensional heterogeneity treatments (HET BAL and HET UNBAL). Model 6 included only dummy variables for the different heterogeneity categories in these treatments – high risk/low wealth, low risk/low wealth, and low risk/high wealth – with our reference group for these analyses being the high risk/high wealth individuals in the HET BAL treatment. Model 7 included the same control variables we used in Model 5.13

---

13 Model 7 is the only model to show significance of the single-group session dummy – suggesting that participant awareness of who was in their group may have been particularly important in the multidimensional heterogeneity treatments.
Both models indicated an interaction effect among those participants facing high risk. Those individuals tended to be more cooperative when they also received low endowments (more than 3 times as likely to cooperate in our experiment). Model 7 suggests that the opposite is true for participants facing low risk; they were more likely to cooperate when they received high endowments.\textsuperscript{14}

V. Discussion

a. Evaluation of Hypotheses

Returning to our hypotheses regarding group effects, the analysis is somewhat supportive of Hypotheses 1 and 3 (economic and risk heterogeneity will total group contributions, respectively), in that the Mann-Whitney tests suggest that unidimensional heterogeneity in wealth and risk reduce a group’s ability to act collectively. Our regressions analysis showed the same effects, but the results lacked statistical significance. The degree of support generated for Hypothesis 5 (the presence of both risk and economic heterogeneity will reduce collective action) is more nuanced. One of the combined heterogeneity treatments (HET BAL) resulted in lower cooperation than the HOM control treatment. On the other hand, both forms of combined heterogeneities result in greater contribution rates than in the single heterogeneity treatments, and the HET UNBAL treatment realized greater group contributions than the control treatment. As such, we partially reject Hypothesis 5.

Regarding our hypotheses about the effects of heterogeneity on individual contributions, these results suggests we must reject Hypothesis 2 (endowment levels will be negatively correlated with participants’ likelihood of contributing), as there is no significant evidence that

\textsuperscript{14} When using Wald tests to compare coefficients of Low Risk X Low Endowment and Low Risk X High Endowment, there is a significant difference between cooperative behavior in Model 7 (p=0.0848) but not Model 6 (p=0.2040).
participants receiving high endowment showed less inclination to cooperate, regardless of their risk levels. Our findings supported Hypothesis 4 (increased risk will negatively impact the likelihood of contributing), in that we found that, when the risks of cooperation increased, participants were significantly less likely to cooperate. Finally, we found overwhelming support for Hypothesis 6 (individuals with high risk/high endowment will be less likely to contribute than those with other combinations of endowment and risk). Our analysis found that high risk, high endowment participants were significantly less likely to contribute compared to individuals facing the other endowment-risk combinations in the multidimensional treatments.

b. A Closer Examination of the Results

What might explain the difference in performance between the two mixed heterogeneity treatments? And why did wealth appear to ameliorate the negative impact of risk exposure, even though it had no independent significant impact on participant contributions? One potential explanation might be found in social comparison theory. That theory maintains that people evaluate their own situation by comparing it to that of their peers, and then use that reference point as a benchmark to establish relative gains or losses (Wang, et al. 2016). A body of experimental literature has found that individuals’ tolerance for risky behavior is impacted by whether they perceive the outcome of that behavior as constituting a potential social gain or loss, with gains being associated with risk tolerance and losses with risk aversion (Kahneman and Tversky 1979).

Individuals who perceive their economic status as being lower than their social reference point may be likely to undertake risk in order to “make up ground;” those with higher economic status than their reference points may be more risk adverse if the losses they suffer might cause them to drop below their social reference point (Lim 2018). However, there is some evidence
that if the higher economic status individuals will remain above their social reference point, even after suffering losses, they will exhibit greater risk tolerance than will those who, even if the risk pays off, are unable to surpass their reference point (Linde and Sonnemans 2011).

To provide a theoretical real-world application of the social comparison theory, imagine an investment fund manager whose pay includes both a salary and performance bonuses tied to the fund’s performance. Rather than comparing her salary to some abstract measure, such as the mean national income, the manager is instead likely to use the mean salary of similarly situated individuals (i.e., other investment fund managers) as her social reference point. If our hypothetical manager perceives that she is earning less than the average fund manager, she might undertake riskier investment decisions in order to increase the size of her bonus. The prospect of achieving a social gain (here, getting closer to her social reference point) would foster a greater tolerance for risk. On the other hand, if our manager perceived that she was earning more than her reference point, we would expect that her concern over a social loss (losing her bonus and dropping below the reference income) would outweigh her desire for even more social gain, and so she would become less tolerant of risk.

The shifting tolerance of risk, predicated on the use of social reference points, appears to explain the findings of our experiment. Applying the social comparison theory first to the HET UNBAL treatment, we note that the low risk/high endowment subgroup received an endowment of 48 ECU, with a risk of losing 30 ECU from contributing to the collective pool. When members of that subgroup contributed to the group pool, their worst possible outcome in any round was 26 ECU (48 endowment – 30 loss + 8 from the collective pool). If only one additional member of the group also contributed, they would receive nearly the same value (34 ECU) as the endowment given to the high risk/low endowment group (36 ECU). Faced with a
low risk of suffering a loss (30%), members of the low risk/high endowment subgroup could expect to earn much more than that amount over the course of the game, and so they would be relatively tolerant of the risk of loss.

The high risk/low endowment subgroup, on the other hand, occupied an economic status below that of their social comparison reference point (the low risk/high endowment subgroup). To the extent that they were motivated to “catch up” to the endowment of their comparators (and the social comparison theory suggests that they would be), the members of this subgroup had to rely on payments from the group pool to make up the difference. Because the payments from the group pool provided the possibility of a social gain – i.e., bringing those participants closer to the endowments received by other participants in the group – the low endowment recipients were relatively tolerant of the increased risk associated with contributing. As such, the social comparison theory would predict that all the participants in the HET UNBAL treatment would be relatively tolerant of the higher risk associated with contributing to the collective pool. And, in fact, the participants did just that, contributing at mean rates that were 22.86 and 10.95 percentage points higher than their low risk and high risk counterparts in the HET WEALTH treatment, respectively.

Turning now to the HET BAL treatment, we find a misalignment between subgroups regarding tolerance of risk. As with the low endowment group from the unbalanced treatment, the low risk/low endowment group here would have been incentivized to contribute in order to overcome their lower economic status and achieve a gain relative to their social comparison reference point. On the other hand, the high-risk group was already ahead of their reference point while facing a high risk of loss. Absent the prospect of social gain, members of this subgroup would have had less intrinsic motivation to contribute than would the other subgroups in the
mixed heterogeneity treatments. Thus, unlike in the HET UNBAL treatment, where both subgroups would have been motivated to contribute to the group fund, the motivations of the subgroups in the HET BAL were mismatched. The low risk group was highly motivated to contribute and did so at a much greater rate (71.9% mean rate of contribution compared to 58.57%) than their counterparts in the HET RISK group, who had received the same endowment level as their groupmates. The high income/high risk group, by contrast, had much less motivation to risk a loss, and consequently contributed at a slightly lower rate than their HET RISK comparators (29.52% mean rate compared to 32.38%).

c. Generalizability of Our Findings

There is a growing awareness that the reliance of behavioral experiments on western university students may limit its generalizability to other sections of society, particularly when it comes to differing cultures and socioeconomic groups (see, e.g., Henrich, et al. 2005; Levitt and List 2007a; Levitt and List 2007b; Levitt and List 2008; Benz and Meier 2008; Henrich, et al. 2010; Slonim, et al. 2013; Goeschl, et al. 2020). Certainly, the world experienced by student participants at a midwestern university differs in myriad ways from that experienced by livestock holders in remote areas of northwest Namibia. As such, we interpret our results as indicative, rather than dispositive: our results provide implications about the impact of risk and economic heterogeneities on collective action, with the expectation that these implications should be examined in the field to assess their applicability across a range of conditions.

That said, even though our experiment occurred within the artificial confines of a laboratory, its results do find some support in field studies. Some researchers and practitioners have observed the potential negative impact of economic heterogeneity on CBNRM efforts in the field (Kerapeletswe 2005; Thakadu 2005). Additionally, our finding regarding the significance of
individual risk levels are bolstered by Kahler et al.’s (2013) field study of two Namibian conservancies, in which it was determined that poaching was most likely to occur in areas where residents perceived the greatest risk of human-wildlife conflict.

Finally, our findings regarding the relationship of risk and endowment combinations on the likelihood of individual and group contributions also find some preliminary support from field research. In particular, two studies of Namibian conservancies found that poorer residents were more likely to seek formal conservancy membership (Silva and Mosimane 2012; Kanapaux and Child 2011), even when those residents faced higher risks of predation (Kanapaux & Child 2011). As previously noted in this dissertation, all residents in Namibia’s conservancies are required to follow conservancy regulations regarding the killing of wildlife, and the formal recognition as a member simply entitles that individual to share in the distribution of direct benefits. Nevertheless, residents can (and do) refuse to become formal members because of philosophical differences with their conservancy’s ideology (Silva and Mosimane 2012). Therefore, by seeking to become members, these individuals are at least affirming their support for, and intent to comply with, their conservancies’ efforts at wildlife conservation.

VI. Conclusion

We conducted a common-pool resource experiment that examined the impact of risk and economic heterogeneity on group performance and individual decision-making. We found that both the presence of economic heterogeneity and risk heterogeneity was associated with a significant decrease in the likelihood of contribution to a collective fund, although the presence of heterogeneity impacted individual decision making differently. Economic heterogeneity decreased cooperation by all group members equally, suggesting that the mere presence of that form of heterogeneity negatively impacted collective action. The presence of risk heterogeneity,
on the other hand, depressed the cooperation of participants facing a high risk of loss but encouraged participation by those facing low risk.

When wealth and risk heterogeneities interacted, we observed that levels of group cooperation remained elevated compared to that found in the unidimensional environments, although they differed in their relation to homogenous groups. Cooperation decreased for groups with balanced heterogeneity (where participants had both high risk and endowments, and vice versa) relative to homogenous groups but increased (albeit not significantly) for groups with unbalanced heterogeneity (where individuals with low endowments faced high risk of loss and vice versa). We theorize that the difference between these two groups may be the result of a psychological phenomenon known as social comparison, in which individuals tend to measure their status using a social reference point and then adjust their tolerance for risk depending on whether the risky outcome is likely to cause them to gain or lose relative to that reference point.

Taken together, our findings highlight that individual risk levels, economic heterogeneity, and the interaction between risk and economic heterogeneities have the potential to significantly impact the real-world success of CBNRM programs. More broadly, communities are often heterogeneous across a range of measures, such as race, religion, caste, and primary forms of livelihood, to name only a few. And, the presence, distribution, and interaction of those various heterogeneities is likely to vary considerably between and within those communities. Rather than merely copying policy approaches that were successful elsewhere, policymakers would be best served to first consider the makeup of the individual communities of interest, and to adopt policy that has the flexibility to accommodate the impacts (both good and bad) of the communities’ various forms of heterogeneities.
CONCLUSION

This dissertation addresses the broad topic of the viability of the wildlife-based CBNRM approach. I build on two existing bodies of literature that debate (1) whether wildlife conservation should focus on sustainable development approaches such as the CBNRM approach, and (2) the appropriateness of approach’s core design features, namely the monetization of wildlife and the inclusion of oftentimes diverse populations under the single moniker of “community.” I use these debates to generate three related inquiries. Regarding the debate over protectionism vs. sustainable development, I ask whether, using a common measure of success, wildlife governance outcomes in CBNRM areas materially differ from those in governmental areas? To address criticisms of the reliance of CBNRM on economic benefits, I ask whether (a) the receipt of wildlife-generated benefits cause recipients to feel that their lives have improved, and (b) whether different types of these benefits vary in the direction and magnitude of their impact on such perceptions? Finally, in response to criticisms that the CBNRM often incorrectly assumes homogeneity, I explore the potential impact of risk and economic heterogeneity, and the interaction between the two types of heterogeneity, on CBNRM governance efforts.

I approach these three inquiries using a multi-part, mixed-method approach. First, I utilize a political history of the adoption of CBNRM in Namibia to explore the approach’s theoretical foundations and resulting criticisms, which I use to introduce the specific inquiries informing this dissertation. Second, I use a multinational, longitudinal database of elephant deaths to assess the relative efficacy of CBNRM and governmental approaches at protecting African elephant populations. Third, I analyze data from a field-based survey of Namibian households to evaluate the relationship between the receipt of benefits and stakeholders’ perceptions of the impact of wildlife on their lives. Finally, I use a behavioral laboratory
experiment to generate hypotheses about the possible impacts of risk and economic heterogeneity on collective action in CBNRM programs. This research advances our understanding of CBNRM by addressing key gaps in the existing academic literature. It also gives rise to additional questions, which can serve as the foundation for future research into both the efficacy of the CBNRM approach and the factors impacting the success of CBNRM projects.

The remainder of this conclusion proceeds as follows. First, I will summarize the key findings and theoretical contributions of my research. Second, I will explore the policy implications of my research findings. Finally, I will set out the limitations of my studies and identify outstanding questions and possible areas of future research.

1. **Key Findings and Theoretical Contributions**

In Chapter 2, I analyze an elephant carcass database and find that an increase in both CBNRM and governmental governance approaches is associated with an increased likelihood of a carcass being the result of an illegal kill. While increased CBNRM governance is associated with a comparatively larger increase in the likelihood of an illegal kill, the available data does not allow for a determination of the reasons for this difference. I conclude that it is premature to anoint either approach as clearly superior.

From a theoretical standpoint, the findings in Chapter 2 represent one of the few efforts in the academic literature to systematically compare the performance of CBNRM and governmental approaches both across a range of sites and using a common measure of success. By highlighting the wide range of outcomes for both policies and the inability of either approach to achieve consistent success, my findings help inform the ongoing debate about whether sustainable use or protectionist policies are more effective at wildlife conservation. While the outcome measure used here was specific to the African elephant, the import of my research has broader potential
applicability. My approach of using GIS mapping to overlay governance approaches with other relevant factors like incidents of armed conflict or corruption or poverty indicators can be replicated either at the local level (for example, to take advantage of information regarding the specific location of elephant carcasses) or using a different outcome measure (such as harvesting data for fisheries or forests).

In Chapter 3, I use data from a field survey conducted in northwest Namibia and find that the likelihood of CBNRM residents feeling that wildlife makes their lives better is positively correlated with the total number of different wildlife-generated benefits they receives. However, a more pointed analysis finds that individual benefit types differ drastically in the magnitude and direction of their association with residents’ perceptions. I theorize that this difference may be explained by the fact that, rather than relying on an objective comparison of costs and benefits, CBNRM residents gauge the impact of the benefits of their lives by using subjective, fluid reference points.

My data is derived from a survey of households in four conservancies located in Namibia’s Kunene region, but my findings are potentially relevant to wildlife-based CBNRM efforts everywhere. As I observe in Chapter 3, CBNRM programs across the globe are largely based on the fundamental economic premise that wildlife must “pay its way” by generating benefits that CBNRM participants perceive as equaling or outweighing any harm they experience from human-wildlife conflict (HWC). My research is one of the first efforts to examine the impacts of different benefit types on participants’ perception of whether wildlife is actually “paying its way.” Though the exact mix of benefit types examined in my research may not be found in other CBNRM areas, my core finding – that different benefit types can have notably different associations with residents perceptions of the impact of wildlife on their lives – is
potentially applicable to the majority of CBNRM programs worldwide. Further, my hypothesis that CBNRM participants may adjust their reference points can apply to other community-governed resources like fisheries or forests. Even in the absence of HWC, participants in those communities must still decide whether to participate in community’s governance efforts, and their perceptions of whether they are receiving the expected level of economic benefits will inform their decision-making.

In Chapter 4, I and a coauthor find that, when participants in a behavioral laboratory experiment are faced with a risk of loss, different types and combinations of heterogeneity have markedly different associations with collective action. Specifically, in our experiment, the presence of economic heterogeneity was negatively associated with group cooperation, but the participants’ individual decisions whether to cooperate were unrelated to their respective endowment levels. On the other hand, we find that no statistically significant relationship existed between the presence of risk heterogeneity and group cooperation (although the data is suggestive of a negative association), but individual risk levels were significantly associated with a decreased likelihood of cooperation.

When the two forms of heterogeneity were combined, we find that unbalanced groups (where participants with high levels of risk receive low endowments, and vice versa) were more likely to cooperate than were balanced groups (where high risk participants received high endowments, and vice versa). Similar to the community members’ evaluation of their benefits in Chapter 3, we hypothesize that the experiment participants’ risk tolerance was affected by their use of reference points, with recipients of lower endowments becoming more risk tolerant in order to “catch up” to their peers, and recipients of higher endowments becoming more risk adverse in order to avoid “falling back.”
This behavioral experiment is one of only a few to introduce risk into a collective action game, and it appears that it is first to explore the interaction between economic and risk heterogeneities. Given that the two forms of heterogeneity often overlap in the field, our findings present testable implications that are highly relevant to CBNRM wildlife governance everywhere. Further, our findings, and associated hypothesis about the role of peer comparison on the risk tolerance of participants, has broader applicability to the collective governance of other resources, as risk and economic heterogeneities are each found across a wide range of environmental collective action situations.

2. Future Research

The research in Chapter 2, in which I analyzed the relative efficacy of CBNRM and national conservation approaches, suggests the need for future within-country research. Data for the specific locations of elephant carcasses is held by the respective countries in which the sites are located. While it was not feasible for me to negotiate with the 19 different countries included in my study, it may be possible for researchers to obtain this information from one or two countries possessing multiple observation sites. That data would allow a more fine-grained analysis than I was able to conduct. Additionally, it is possible that some of the reporting countries have pre-existing data that would allow for the establishment of a baseline proportion of illegal elephant kills. If that information exists, it could permit a more nuanced understanding of the potential efficacy of the two governance approaches.

In Chapter 3, my core finding is that different benefit types can have substantially different associations with the perception by conservancy residents of whether wildlife makes their lives better, and I suggest that these differences may be the result of residents comparing their current status to a subjective “reference point” or “counterfactual reality.” My suggestion is
based on (a) my pre-survey interview with one of the traditional authorities in the area, and (b) a couple of other studies of communities within southern Africa. Future, in-person interviews within the study area would provide additional insight into why the different benefit types appear to have such a disparate impact on residents’ perceptions in my study. Similar in-depth interviews in CBNRM locations elsewhere in southern Africa would speak the applicability of my hypothesis across a range of circumstances.

Finally, in Chapter 4, I present findings about the possible impact of economic and risk heterogeneities, and their interaction, on the collective governance of wildlife. Because these findings are derived from a behavioral laboratory experiment conducted at a midwestern university in the US, future research could answer whether (a) these findings would emerge if we conducted this same experiment in other settings, and (b) these forms of heterogeneity would have the same relationship with cooperation in actual CBNRM communities. To answer these questions, this experiment should be replicated in other environments, and the implications of these findings should be explored in the field.

3. Policy Implications

a. Policy Cannot be Divorced from Historical Legacies

Current-day policies are enacted in the shadow of the past, and so policymakers must understand the history of the areas in which policy is to be implemented, and how that history may influence the performance of the selected policy approach. While this admonishment is applicable across the globe,¹ it is particularly germane in Africa and elsewhere in the global south, where many countries continue to grapple with social and economic issues from their

¹ For instance, residents of an eastern state in the United States, such as Connecticut, could be anticipated to view the establishment of a protected wildlife refuge in their state very differently than might those in a western state, such as Nevada, where the federal ownership of land is more controversial.
colonial pasts (see, e.g., the account of colonial and postcolonial history on CBNRM efforts in Zimbabwe by Alexander and McGregor (2000)).

In Namibia, colonial and apartheid policies of relocation, dispossession, and favoritism have exacerbated (if not created) rivalries between different ethnicities and traditional authorities. Additionally, Namibia’s colonial and apartheid rulers spent approximately a century trying to stamp out the autonomy of its black citizens and the functioning of many of their informal institutions. Combined, these two legacies mean that conservancies can continue to struggle with issues of favoritism, accountability, and institutional capacity. For instance, participants in my survey frequently stated their belief that conservancy authorities favored themselves, their families, and/or their ethnicities, and (as I note in Chapter 3) my survey data does suggest a degree of elite capture of higher value benefits. CBNRM requires strong institutions to consistently succeed (Armitage 2005), and the lack of empowerment previously afforded to many rural communities means that participating communities need sufficient time to develop and/or evolve their local institutions to meet that demand (Fabricius and Collins 2007).

The lasting inequalities of colonial governance can also lead to a critical difference in understanding between stakeholders about the desired goals of CBNRM. The goal of formal protected areas, for example, is relatively straightforward: the conservation of the natural resources that lie within. Of course, some particularly successful national parks may generate positive externalities for their surrounding communities. But, these sorts of benefits are secondary; the primary goal of national parks, wildlife refuges, and other governmental protected areas is protectionism.

The goals of CBNRM and other sustainable use policies are often much murkier, and perceptions of those goals can vary widely amongst stakeholders. For example, as discussed in
Chapter 1, Namibia’s conservancy program found support among both wildlife conservationists and disenfranchised communal residents. However, the former viewed the approach as a means of improving wildlife conservation in a remote area in which formal policing was largely absent. The latter primarily supported the creation of conservancies to foster the legal empowerment and economic development that had long been denied rural communities by the pre-independence governments.

These disparate viewpoints about the primary focus of CBNRM, shaped in large part by the lingering effects of the parties’ historical experiences, can lead to dissatisfaction on all sides. Wildlife conservationists lament the perceived failure of CBNRM areas at protecting wildlife, while communities chafe at what they see as the reluctance of governments and third-party stakeholders to allow them full control over their own wildlife resources (the same right granted often granted to more privileged private landowners). Therefore, when selecting the appropriate wildlife governance approach, policymakers must weigh the approaches’ historical baggage and clearly identify the approach’s desired outcome to all parties.

b. There are no “Failed” Policy Approaches in Wildlife Governance

Wildlife policy, like all environmental policy, is implemented in a complex environment, and its outcome can be influenced by any number of environmental, cultural, sociopolitical, and geopolitical factors. Further, as noted throughout this dissertation, stakeholders can have a range of disparate, and sometimes seemingly irreconcilable, definitions of what constitutes a “successful” policy outcome, with some using ecological criteria and others using anthropocentric measures. Compounding these two issues – the multitude of potentially confounding variables and the potential for disagreement over the primary focus of wildlife governance – is the fact that wildlife faces a host of threats. This dissertation has largely focused
on the preventative or retaliatory killing of wildlife, where rural populations are incentivized to harm wildlife in order to ward off future conflict with that wildlife. However, wildlife can also be threatened by other human activities including, but not limited to, commercial poaching, habitat loss, industrial/mining/agricultural runoff, accidental poisoning (often of scavenging birds such as vultures or condors), and climate change.

Some wildlife policy approaches may generally be more appropriate than others for certain combinations of these three sets of variables. Nevertheless, despite efforts to identify the circumstances under which different policy approaches are most likely to succeed (e.g., Ostrom 1990; Brandon and Wells 1992; Mukamuri 2009), the success or failure of a policy approach in any given situation remains far from preordained. Compare, for example, the number of poaching incidents over the last 15 years in the neighboring conservancies of Anabeb and Sesfontein (two of the four conservancies surveyed in Chapter 3). The two conservancies are so similar in their ecological and socioeconomic composition that they were combined into a single conservancy until 2003. Yet, between 2003 and 2017, Anabeb reported 5.6 incidents of wildlife poaching per year\(^2\) compared to Sesfontein’s annual average of 0.47.\(^3\) Of course, the rate of poaching could have been significantly higher without the introduction of conservancies, and the reasons for the difference in poaching rates is beyond the scope of this dissertation. However, the difference is illustrative of the potential variability in outcomes experienced by the same policy approach implemented in substantially similar situations.

Additionally, the initial circumstances informing the selection of a particular policy approach are not static, and changes to those conditions could undermine a formerly successful

\(^2\) This number reflects both commercial and subsistence poaching incidents.
approach and vice versa. For instance, the multi-year drought in Namibia’s northwest (referenced in Chapter 1), led to increased HWC and the influx of outsiders into the Kunene conservancies, challenging the capacity of the conservancies’ institutions. Closer to home, the decline in the number of hunters in the United States (and the corresponding decrease in revenue from the issuance of hunting permits) poses a threat to the ability of many states to effectively manage their wildlife resources (Rott 2018).

Policymakers share the universal human distaste for ambiguity (Christensen 1985) and, as such, they seek to attain at least the “illusion of certainty” (Gunder 2008). Policymakers rush to adopt target outcomes and policy prescriptions for even the murkiest of problems (Christensen 1985) and pronounce “unambiguous rules” about future courses of action (Gunder 2008). Rather than recognizing that policy approaches are inherently experimental in nature, with an unavoidable risk of failure and unexpected outcomes, policymakers may be all too ready to listen to policy advocates and declare certain approaches to be panaceas and others to be failed or outdated. However, the wholesale adoption of a single type of policy approach limits the flexibility of policymakers to both (a) match policy approaches with the sorts of site-specific variables discussed above, and (b) respond to future changes in those variables.

c. Policy Makers Should Not Presume Community Residents are Static Entities

As noted in Chapter 4, it is often observed in the academic literature that rural communities are not the conveniently homogenous groups that policymakers assume (or wish) them to be. It is also important to note that these messy, heterogeneous communities are not static (Leach et al. 1999), but rather are composed of individuals who can and do shift their expectations and perceptions about CBNRM outcomes. The axiom that wildlife must pay its way
may hold true throughout the lifespan of a CBNRM program, but it is possible that members of the community will at some point raise that price.

The fact that CBNRM participants may raise their expectations regarding wildlife-generated benefits should be viewed as neither surprising nor objectionable, as urban dwellers and residents of developed countries are encouraged to do just that. Workers are expected to strive for promotions and pay increases, consumers to purchase ever nicer homes or larger televisions. Our own satisfaction with these new promotions, raises, or material gains is fleeting. We appreciate them, but we soon chase after new goals, and our frustration at failing to attain them is not ameliorated by our prior achievements.

Given this near universal tendency to raise our expectations over time, there is no reason that we should expect individuals within CBNRM communities to act differently. Distributions of trophy meat and the occasional cash distribution may have a large initial impact on CBNRM residents because these benefit distributions likely mark the first time that they have been able to (legally) benefit from and enjoy a degree of ownership over the wildlife resources. Over time, however, CBNRM residents are likely to wish for more substantial changes. In my pre-survey interviews, for example, some individuals voiced the desire for their conservancy to spend money to drill or improve nearby boreholes (Interview of Head Lady Allina Karutjaiva, April 17, 2017; Interview of Headman Govan Tjipombo, April 21, 2017) or to pipe in spring water (Interview of Secretary to the Kandjii Traditional Authority Benson Jatemuna and Gangea Traditional Authority Organizer Vetatumiza Kozohura, April 12, 2017). One traditional authority, gesturing to the dusty ground around her tent with frustration, lamented the fact that, despite her status, she still lacked a permanent home to live in or a shady place in which to sit (Interview of Head Lady Allina Karutjaiva, April 17, 2017).
Any discontentment that residents may have with the status quo of wildlife benefit distribution may increase with the intensity of tourism activity in the area. As one Himba traditional leader explained to me, local residents watch wealthy tourists zip by in their new Land Cruisers, with all of their expensive gear, and it naturally makes them want to acquire those sorts of possessions themselves (Interview with Puros Chairman Albertus Uararavi, April 19, 2017). Their inability to achieve these new goals can lead residents to feel that they do not benefit from the CBNRM model (however objectively true or false that may be), potentially leading to a loss of community participation in CBNRM efforts. The literature indicates that community satisfaction with CBNRM can ebb over time (Snyman 2014), suggesting that residents need to perceive greater opportunities for advancement with the passage of time.

To address this natural “expectation creep,” policymakers must facilitate CBNRM communities in the generation of progressive and meaningful long-term economic changes. While meat and cash distributions may be important for encouraging initial community buy-in, CBNRM policies should be designed to afford residents consistent and available opportunities for employment and continued economic advancement. Policymakers may be able to ameliorate expectation creep by (a) incentivizing transparency and community participation in the collection and use of conservancy income, and (b) ensuring the continued empowerment of CBNRM areas in their own economic affairs.

Regarding the issue of transparency and community participation, in both my survey responses and my pre-survey interviews, I encountered repeated confusion and suspicion regarding the use of conservancy income and the allocation of benefits. Many, if not most, of the residents in the four conservancies included in my field study appear to have little or no knowledge of how much income is collected by their conservancies, the decision-making process
by which benefit distributions and other conservancy expenditures are determined, or who receives what amount and type of benefits. In the absence of this information, many residents suspect that they are receiving less than others in their communities – a fact that is likely to decrease how they perceive the value of the benefits that they have received.

Regarding the need for greater empowerment, most tourism profits in CBNRM areas accrue to outsiders (Walpole and Thouless, 2005). Some CBNRM areas have entered into arrangements without outside operators, pursuant to which ownership of lodges accrue to the communities after a certain number of years. However, residents in those areas generally lack the training and financial resources to operate their own tourism companies, and so the communities often choose to instead lease the lodges back to their outside partners. A few CBNRM communities do fully own and operate their own companies, including the Masaaai-owned Il Ngwesi Lodge in Kenya, the Grootberg Lodge in Namibia’s ≠Khoadi //Hoas Conservancy, and the travel company Conservancy Safaris Namibia, which is jointly owned by Namibia’s Puros, Orupembe, Sanitatas, Okonjombo, and Marienfluss Conservancies. These sorts of community-owned enterprises, however, are uncommon.

When it comes to trophy hunting, CBNRM areas tend to be even less empowered. In Namibia, for instance, the conservancies receive a quota allowing the hunting of a certain number of desirable trophy animals (such as lions or elephants), with the ability to apply for additional hunting permits as needed for individual nuisance animals. The permits are auctioned off to hunting outfitters. In talking to traditional authorities and conservancy officials, it became clear to me that the conservancy leadership had little involvement in the auction process and lacked even basic information about how much the operators were ultimately charging for those hunts.
Rather than relying on employment by outside for-profit tourism and trophy hunting companies, CBNRM areas could theoretically run their own hunting or photo-tourism ventures and capture the entirety of the profits for their communities. Operating their own tourism businesses could increase the total income of the communities, which could then be used to increase the value of the benefits received by their constituents and provide residents with a sense of ownership over these enterprises.

While the relationship between employment and happiness is complex, the academic literature indicates that a sense of ownership of a company significantly increases employees’ job satisfaction (Pendleton et al. 1998; Dyne and Pierce 2004; Mayhew et al. 2007; Peng and Pierce 2015). It appears that most of these studies were conducted in industrialized settings, and so it remains to be investigated whether the psychological benefits associated with company ownership would also apply to the rural CBNRM-based tourism industry. Nevertheless, providing residents with a clear ownership interest in wildlife-based tourism enterprises, and allowing their communities to reap the full economic benefits of those efforts, may facilitate CBNRM programs to better cater to the evolving needs of their residents.
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182


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I. Supplemental Information Regarding PIKE

The MIKE program has a standardized set of procedures and forms for patrols of observation sites. The standardized form allows for the recording of the specific cause of death (along with the gender, age, and other particularized information regarding the carcass) and provides broad criteria for identifying the circumstances under which the elephant died. However, publicly available MIKE data only differentiates between illegal and non-illegal deaths and does not reflect the manner of the deaths. Consequently, it is not possible to determine, for instance, whether an elephant died from natural causes or from a sanctioned trophy hunt or cull, as both types of death would be identified in the database as being non-illegal. Likewise, it is not possible to differentiate between illegal killings by an ivory poacher or a farmer killing an elephant to prevent crop depredation.

II. Selection of Explanatory Variables

The following criteria was used to select the explanatory variables included in the model used in this study.

a. Site-Level Variables

i. CBNRM and national governance percentage

Satellite mapping was used to overlap maps of national and community protected areas with the MIKE sites and, for each year data was reported under the MIKE program, to generate a percentage of each site falling under both types of governance. Many MIKE sites contain land areas falling outside of either CBNRM or national governance. Consequently, while these two percentages are negatively correlated (-.6633), a finding regarding the association of one
governance approach with illegal kill odds does not allow a determination of the association of
the other. Therefore, the national and CBNRM governance variables were both included in the
model.

In some circumstances, such as in Zambia and Tanzania, CBNRM governance in
bordering lands is expected to “buffer” formal protected areas from outside pressure by allowing
local communities to benefit from wildlife that move outside the protected areas’ boundaries
(Roe, et al. 2009; Lindsey, et al. 2014). It also stands to reason that bordering lands under
national governance may perform the same function. CBNRM and national buffer variables were
generated that reflect the percentage of the MIKE sites that were bordered by land under either
CBNRM or national governance, respectively. An initial analysis of the buffer variables found
that they had no statistical significance and their inclusion raised the AIC and BIC of the model.
Therefore, the buffer variables were not included in the final model.

Of course, specific sites undoubtedly differ to the extent that they match the paradigmatic
strict protectionism or sustainable use approaches. For instance, some national governance areas
may allow trophy hunting by safari operators or other limited resource usage, just as some
CBNRM programs can, in practice, afford little opportunity for communities to actually manage
or profit from wildlife resources. Nevertheless, this project is interested in the overall
effectiveness of the two governance approaches, accepting that their exact characteristics may
derfer between sites. Therefore, to preserve analytical power, the categories of national and
CBNRM governance include all approaches falling within their respective spectrums.
Forest cover, human footprint, and population density

For forest cover, this analysis considered the Global PALSAR-2/PALSAR/JERS-1 Mosaic and Forest/Non-Forest dataset, generated by the Japan Aerospace Exploration Agency. This dataset provides a single average binary determination of forest cover at a 25m resolution for the period of 2007-2010. GIS mapping was used to generate a percentage of each site under forest cover.

For the human footprint variable, this study used the Global Human Footprint Dataset (Geographic) from the Last of the Wild Data Version 2, 2005 (LTW-2), generated by the Wildlife Conservation Society (WCS) and Center for International Earth Science Information Network (CIESIN). This dataset uses 1km raster grids reflecting a single aggregate value for population density, land use, infrastructure, and human access (via roads, railroads, coastlines, and navigable rivers) over the time period 1995-2004. GIS mapping of these datasets was used to generate an average total score of human footprint for each of the MIKE observation sites.

The population density variable was determined using data from WorldPop. WorldPop provides population estimates at five-year intervals starting in the year 2000. Population estimates are given in a people-per-pixel value, readjusted to reflect UN population estimates at a 1km² resolution. Average site values were obtained for the years 2000, 2005, 2010, and 2015, and estimates were generated for intervening years by assuming a linear change between the given values.

These variables were not found to be highly correlated (between -.37 and .55), and therefore each variable was independently considered for inclusion into the model. The human

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4 Available at http://www.worldpop.org.uk/data/.
footprint and population density variables were not found to be statistically significant in preliminary analysis, and their inclusion increased the model’s AIC and BIC. Therefore, only forest cover was included as an explanatory variable.

iii. Infant malnutrition

Burn, et al. (2011) found economic development (measured at the country level using the United Nations’ Human Development Index, a country-level composite score reflecting educational attainment, life expectancy, and income, frozen at the 2007 level)) was significantly associated with an increase in PIKE values. Development levels, however, can vary widely within countries across both space and time. Therefore, this study sought to control for economic development at the site level, while allowing these values to change over time.

No measures that directly reflect local economic development across all the selected MIKE sites, and therefore World Bank infant malnutrition data was used as a proxy variable for income. Infant malnutrition values have been shown to be negatively associated with income levels (Petrou and Kupek 2010). The World Bank episodically provides infant malnutrition values for designated subnational regions for all but two of the countries included in this study. A single weighted infant malnutrition score was calculated for each MIKE site based on the percentage of the site’s border that is either adjacent to or falls within a World Bank region, regardless of national boundaries.

Because subnational infant malnutrition values were not available for each year of the study, the following assumptions were used to generate approximate values for the missing years. Changes between available data points were assumed to occur linearly, and yearly values between those points were generated accordingly. In Botswana, only one subnational data point was available, while multiple data points were available for national infant malnutrition levels.
Therefore, to allow for the generation of approximate subnational values, subnational infant malnutrition was assumed to increase or decrease by the same percentage as national values. No subnational values were available in South Africa or Eritrea, and so national-level infant malnutrition data was used instead.

For many of the selected sites, the span of yearly measurements provided by the World Bank was insufficient to generate estimated infant malnutrition for all the years for which MIKE data is available. For example, a site could have reported MIKE data for the years 2002-2015, but infant malnutrition data only exists for the years 2005-2012, leaving six years (2002-2004 and 2013-2015) for which no infant malnutrition estimates exist. Three different treatments were used to generate possible infant malnutrition values for those missing years: (1) over the course of the years for which data is missing, infant malnutrition values were assumed to regress to the site’s mean; (2) infant malnutrition values were assumed to continue to increase or decrease at the same linear rate, capped at 50% and 150% of the sites’ mean; and (3) infant malnutrition values were frozen at the last level for which data is available.

All three sets of infant malnutrition values separately decreased the model’s AIC and BIC, and so the variants were separately included in three alternate models. As the findings of model were not sensitive to the treatment used to generate the missing data, only the findings based on the assumed linear change are reported here.

**iv. Armed conflict**

This study used data from the Armed Conflict Location & Event Data Project (ACLED), which documents different types of incidents of political violence, demonstrations, and armed conflicts, and provides geometric coordinates (i.e., latitude and longitude) for each incident.

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5 Available at https://www.acleddata.com/.
Among other things, ACLED codes its data to differentiate incidents based on participants and whether the incident involved violence. A brief description is also provided about each incident.

In order to be included in this study, the incidents needed to involve some form of violent conflict evidencing an ongoing struggle involving governmental and/or rebel forces. In total, this study selected for inclusion 22,185 incidents of conflict for the period of 2002-2015. Satellite mapping was used to generate a 250km buffer around each MIKE site.\textsuperscript{6} For each MIKE site, a yearly count was generated reflecting the number of incidents incurring within the site and its 250km buffer.

This study includes those with the following coding (with corresponding ACLED codes in parentheses):

1. Sole military action (10)
2. Military versus military (11)
3. Military versus rebels (12)
4. Military versus political militia (13)
5. Military versus communal militia (14)
6. Military versus other (18)
7. Sole rebel action (20)
8. Rebels versus rebels (22)
9. Rebels versus political militia (23)
10. Rebels versus communal militia (24)
11. Rebels versus protesters (26)
12. Rebels versus civilians (27)
13. Rebels versus others (28)
14. Other actor versus civilians (78)

\textsuperscript{6} Little guidance exists regarding the size of the radius that should be used in this analysis. A search identified only two articles that discuss the impact of distance from borders on poaching incidents, and both make clear that distance can have a significant impact on poaching levels. Poulsen, et al. (2017) found an increase in elephant population density as the distance from the national border increased from 0 to 20km, while Eloff and Lemieux (2014) determined that 50\% of poaching in South Africa’s Kruger National Park occurred within 5km of the park borders. However, neither of these articles addressed the spatial effects of armed conflict. Thus, to minimize the risk of underestimating the effect of armed conflict, a 250km radius is used here. While admittedly arbitrary, this distance represents a not inconsequential travel by vehicle, and a multi-day travel by foot, and therefore it allows for spillover effects from regional conflict while still allowing for substantial subnational variation amongst MIKE sites.
Non-violent events (such as two-sided peaceful protests) were excluded, as were violent incidents involving only civilians (such as civilian riots in response to civilian protests), or non-rebel or non-governmental agents (such as the killing of civilians by private security agents). While this study does not include violence inflicted on civilians, protesters, or rioters by government forces, it does include violence by rebel forces against civilians and protesters. Incidents coded as “other actor versus civilians” are selected for inclusion only if they evidence either (a) activity by state or rebel forces (for example, violence by governmental forces from a neighboring country), or (b) violence by coalition forces in response to armed conflict involving rebel or governmental forces (for instance, the killing of civilians by African Union forces in response to an attack by rebel forces).

Rebel violence against civilians and protesters was included because the involvement of rebel forces necessarily reflects the ongoing presence of armed rebellion, whereas violence enacted by governmental forces does not. In other words, governmental attacks on civilians, protesters, or rioters could be singular events that are not part of a larger, long-lasting pattern of violent instability. On the other hand, attacks by rebel forces require the presence of an armed rebel force, meaning that, at minimum, a strong potential for armed conflict and instability exists.

b. Country-Level Variables

i. Rule of law, corruption, political stability, and state fragility

In addition to corruption, this study considered the other commonly used governance measures of rule of law and political stability. Data for each of the rule of law, political stability, and corruption measures were obtained from the World Bank Governance Indicators. This study

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found a high degree of correlation (0.7 or greater) between the three measures, and so only one was selected for inclusion in the final models.

The rule of law measure was selected over the others because its inclusion resulted in the greatest reduction of both AIC and BIC and, based on a two-variable scatter, it appeared to be less influenced by outlier countries than the corruption measure. The rule of law measure is a continuous measure (from -2.5 to 2.5), with higher scores indicating a greater rule of law.

ii. **GDP per capita**

Burn, et al. (2011) previously did not find GDP per capita to have any statistically significant impact on the proportion of illegally killed elephants. Nevertheless, GDP per capita was considered for inclusion here because countries with higher wealth may have more resources available to devote to elephant protection. The variable was included in the final analyses because its inclusion reduced the core model’s AIC and BIC.

c. **Common Variable**

i. **Price of Ivory**

Poached ivory is a black market good, and so it is difficult to determine a singular global price for ivory for the duration of the MIKE program. However, studies have provided estimates for ivory prices in China for each of the years 2002, 2006, and 2008-2016 (Martin and Stiles 2003; Stiles 2004; Stiles 2009; Gabriel, et al. 2011; Stiles, et al. 2011; Gao and Clark 2014; Vigne and Martin 2017).

Because China is often recognized as being the most influential of the markets for poached ivory (Underwood, et al. 2013; Milliken, et al. 2016), and it is the country for which the

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8 Since the implementation of the ban on international trade of ivory, some non-African countries have purchased stockpiles of ivory through sanctioned “one-off” sales, and subsequently legally sold some of that ivory to their domestic ivory carvers and artisans. The prices of these legal sales are not used in this study.
most complete recent record of ivory prices is available, this study uses Chinese ivory prices as a proxy for global ivory prices. Many elephant range states have their own domestic ivory markets, and it is likely that ivory prices in those markets are significantly lower than in China. However, the available literature provides little longitudinal data regarding ivory prices in African markets. Therefore, for purposes of this study, it is assumed that local market prices will generally track the relative increases and decreases of Chinese prices.

While the price of illegal ivory is available for each year after 2007, it does not appear that any such prices have been determined for the years 2003-2005 and 2007. Prices for those years are generated by assuming a linear change between known ivory prices. In this way, barring any wild short-term swings in prices, a linear projection allows for the generation of “ball-park” estimate of ivory prices for those years for which data is missing.
APPENDIX 2 (CHAPTER 3)

I. Supplemental Field Research Information

Prior to first traveling to the study area, I sent each of the traditional authorities in the area a letter stating that I would be traveling to the area to obtain permission to conduct my study, the purpose of the study, and the date range in which I would be in the study area. Thus, while I was not able to meet with all of the traditional authorities in the study area (some were out of the area, including several that had traveled south to attend a ceremony held by South Africa thanking them for siding with South Africa during Namibia’s independence movement), those with whom I did not meet alerted Sokoi prior to my arrival that they consented to my administering the surveys to their followers.

My field assistants were Melanie, Ellen, Erich, and Ben. I met with Melanie, Ellen, and Erich in Windhoek in August 2017 to train them on the administration of the survey and to explain the reasoning behind the survey’s questions and organization. After the initial training session in Windhoek, I traveled with the three assistants to Opuwo, a town of approximately 7,500 that is the capital of the Kunene region. One individual originally selected to serve as a fourth field assistant did not show up to the training session in Windhoek, so I retained Ben upon my arrival in Opuwo. Ben was also recommended by Selma Lendelvo, and he had prior experience in survey administration and (as an additional logistical benefit) owned a 4x4 vehicle.

I asked Ben to conduct the first administration of the survey during the pretest phase, as he had missed the orientation in Windhoek, and I viewed it as a chance for him to familiarize himself with the survey and to receive feedback from the other assistants regarding his performance (the administration was conducted in Otjiherero).
II. Full Survey

The following is the full English language of the version of the paper survey used in the field.

**Selection Criteria**

For each household visited, ask for the head of the household. If the head of the household is not available, ask to speak to an adult member of the household.

**Opening Instructions**

This survey is being conducted as part of a research project that seeks to learn about the experiences and opinions of people living in this area. We have received permission from the conservancy and the traditional authorities to conduct this survey, but your participation is completely voluntary. Although we appreciate your full participation, you may stop the survey at any time, for any reason, and you may choose not to answer any of the questions if you wish. It is anticipated that this survey will take around 45 minutes.

All of your answers will be completely anonymous (which means that no one will be able to know what you answered). Nevertheless, it might be possible for others in the area to identify you based on some of your answers and so, to prevent this from happening, your answers will only be shared with the researchers leading this project. Otherwise, the primary risk associated with this survey is the inconvenience you might experience from participation. The possible benefit of your participation would be that the results of this research may be used to improve laws and government practices in this region.

If you have any questions about this research, you can contact Selma Lendelvo at the University of Namibia.

If you do not fully understand what a question is asking please let me know and I will be happy to clarify. Likewise, if you need me to repeat anything, please let me know.

Do you have any questions for me, or do you need more time to decide whether you would like to participate?

**Questions**

This first set of questions asks some general background information about you.

**QUESTION 1: Which conservancy do you live (stay) in?** __________________________________

□ DON'T KNOW

**QUESTION 2: How long have you lived (stayed) in this [conservancy/ if person answered “don’t know” to question 1, use the term “area” instead]?** __________________________________
QUESTION 3: Where did you live (stay) before you came to this [conservancy/ if person answered “don’t know” to question 1, use the term “area” instead]?  
________________________________________________________________________________

IF THE PARTICIPANT ANSWERED “DON’T KNOW” TO QUESTION 1, SKIP TO QUESTION 5

QUESTION 4: Are you a member of the [USE NAME OF CONSERVANCY MENTIONED IN Q1] conservancy?

☐ YES (proceed to Question 4(A))
☐ NO (proceed to Question 4(B))
☐ DON’T KNOW (proceed to Question 4(B))

QUESTION 4(A)

i. For how many years have you been a member? ____________________________

ii. How did you become a member? ______________________________________

QUESTION 4(B)

i. Have you ever been a member of this conservancy?

☐ YES (ask why the person is no longer a member)

____________________________________________________________________________

☐ NO

☐ DON’T KNOW

ii. Are you eligible to be a member of this conservancy?

☐ YES

☐ NO

☐ DON’T KNOW

QUESTION 5: Have you ever been a member of any [other] conservancies?

☐ YES (proceed to Question 5(A))

☐ NO (proceed to Question 6)

☐ DON’T KNOW

QUESTION 5(A)

i. Which conservancies? ____________________________________________
ii. Are you still a member of that/those conservancies?
   Conservancy ________________________:
   □ YES  □ NO  □ DON’T KNOW
   Conservancy ________________________:
   □ YES  □ NO  □ DON’T KNOW
   Conservancy ________________________:
   □ YES  □ NO  □ DON’T KNOW

QUESTION 6: Do you follow any traditional authority or authorities?
   □ YES (proceed to Question 6(A))
   □ NO (skip to household questions below)
   □ DON’T KNOW (skip to household questions below)

QUESTION 6A:
   i. Which traditional authority(s)?
   ___________________________________________________________

   (IF THEY IDENTIFIED THE CONSERVANCY THEY LIVE IN QUESTION 1):
   ii. Is your traditional leader based in the (USE NAME OF CONSERVANCY MENTIONED IN Q1) conservancy?
      □ YES
      □ NO
      If NO: Is there a traditional leader who represents you in that conservancy?
      □ YES Name:________________________________________________
      □ NO
      □ DON’T KNOW

This next set of questions asks some background questions about other people in your household. A person is in your household if they share the same cooking utensils as you. When you are answering these questions, you are not answering about yourself, but rather for people who stay in your household. If any of those people must live (stay) elsewhere for work, they would be considered part of this household if they primarily live (stay) here and consider this to be their household when they are not working.

QUESTION 7: How many people other than you live in your household?
   ADULTS (18+) ____________
   CHILDREN (under 18) ____________
IF 0 OTHER PEOPLE LIVE THERE, SKIP TO THE NEXT SECTION (IN GREY)
IF 1 OR MORE OTHER PEOPLE LIVE THERE, PROCEED WITH QUESTIONS IN THIS SECTION

QUESTION 8: Are any of them currently a member of any conservancies?
□ YES (proceed to QUESTION 8(A))
□ NO (proceed to QUESTION 8(B))
□ DON’T KNOW (proceed to QUESTION 8(B))

QUESTION 8(A)
i. Which conservancies? (ask for total number of members of each conservancy)
_____________________________________________________________________
_____________________________________________________________________

QUESTION 8(B)
i. Are they eligible to become a member of any conservancy? (IF YES, record which conservancies)
_____________________________________________________________________
□ YES □ NO □ DON’T KNOW

ii. Have they ever been a member of a conservancy? (If they answer “NO” or “DON’T KNOW” SKIP TO NEXT SECTION IN GREY)
□ YES □ NO □ DON’T KNOW

iii. IF YES: Which Conservancies?
_____________________________________________________________________
_____________________________________________________________________

This next set of questions asks about both you and your household. Therefore, when answering questions about your household, please include yourself in your answers.

QUESTION 9: Please identify whether your household is engaged in the following activities, whether for money or for your own use (SELECT ALL THAT APPLY).
□ Livestock Farming (ex. goats, cattle, sheep)
□ Crop Farming (ex. millet, sorghum, maize, peanuts)
□ Conservancy employment (select this if someone receives a paycheck as a full or part-time conservancy employee)
□ Tourism (ex. sales to tourists, employment at tourism lodges or campsites)
□ Employment outside of conservancy boundaries (ex. work in Opuwo or Swakopmund)
□ Pension/social grant
□ Other (such as firewood, mining, etc. Identify this below)
QUESTION 10: How many of each of the following would you estimate your household has:  
Cattle: ____  
Goats: ____  
Sheep: ____  

QUESTION 11: Which of the following best describes the amount of crops that your household farms?  
☐ After household use, there is normally enough left over to sell or give to others  
☐ After household use, there is sometimes enough left over to sell or give to others  
☐ After household use, there normally is not enough left over to sell or give to others  

QUESTION 12: Approximately how much money does your household earn each year from each of the areas of the activities you identified above. Include the cash value of goods obtained through barter.  
___________________________________________________________________________________  
___________________________________________________________________________________  

This next set of questions asks about your opinions and experiences with wildlife and about the experiences of others in your household. For these questions, when I use the term “wildlife,” I am asking only about animals such as giraffe, springbok, lions, cheetahs, and elephants, and not about domesticated animals such as goats, sheep, cattle, or dogs.  

NOTE – THIS SECTION ONLY DISCUSSES WILDLIFE, NOT CONSERVANCIES  

QUESTION 13: Overall, has wildlife made your own life better, worse, or about the same?  
☐ Better  ☐ Worse  ☐ About the Same  

QUESTION 14: Overall, has wildlife made the lives of those in your household better, worse, or about the same?  
☐ Better  ☐ Worse  ☐ About the Same  

QUESTION 15: For each of the following types of animals, please select whether having the animal near your household would be good for you and your household, bad for you and your household, or would make no difference to you and your household  
1. Small herbivores (such as springbok, steenbok, or duiker)  
   ☐ Good  ☐ Bad  ☐ No Difference
2. Larger herbivores (such as zebra, gemsbok, ostrich, or giraffe BUT NOT ELEPHANT OR RHINO)
   ☐ Good ☐ Bad ☐ No Difference

3. Small carnivores (such as jackal, genet, or bat-eared fox)
   ☐ Good ☐ Bad ☐ No Difference

4. Lion
   ☐ Good ☐ Bad ☐ No Difference

5. Hyena
   ☐ Good ☐ Bad ☐ No Difference

6. Elephant
   ☐ Good ☐ Bad ☐ No Difference

7. Rhino
   ☐ Good ☐ Bad ☐ No Difference

8. Leopard
   ☐ Good ☐ Bad ☐ No Difference

9. Cheetah
   ☐ Good ☐ Bad ☐ No Difference

QUESTION 16: Please identify who owns each of the following types of wildlife. (Examples might be local, state, national govt.)

1. Small herbivores (such as springbok, steenbok, or duiker).

2. Larger herbivores (such as zebra, gemsbok, ostrich, or giraffe BUT NOT ELEPHANT OR RHINO).

3. Small carnivores (such as jackal, genet, or bat-eared fox).

4. Lion.

5. Hyena.


7. Rhino.

8. Leopard.

QUESTION 17: Over the last 5 years, about how many times per year have you or your household experienced any physical harm, loss of crops or livestock, or damage to your property from wildlife?

1. Physical harm to people  Estimated number of times/year _______
2. Loss of crops or livestock Estimated number of times/year ______
3. Damage to property Estimated number of times/year ______

QUESTION 18: Within the last 12 months, have you or your household experienced any physical harm, loss of crops or livestock, or damage to your property from wildlife?

1. Physical harm to people  □ YES □ NO Estimated # of times____
   Type/Cost of harm ______________________
2. Loss of crops or livestock □ YES □ NO Estimated # of times____
   Type/Cost of harm ______________________
3. Damage to property □ YES □ NO Estimated # of times____
   Type/Cost of harm ______________________

QUESTION 19: In your opinion has there been an increase in these types of problems in the last 5 years?

1. Physical harm to people □ YES □ NO
2. Loss of crops or livestock □ YES □ NO
3. Damage to property □ YES □ NO

QUESTION 20: Have you ever applied for compensation for loss of livestock resulting from wildlife? □ YES □ NO

IF YES: Were you satisfied with how the process worked? If not, please explain:
____________________________________________________________________________
____________________________________________________________________________

IF NO: Are you eligible to receive compensation? □ YES □ NO (If yes, explain why no application has been made)
____________________________________________________________________________
____________________________________________________________________________

IF APPLIED OR PERSON IS ELIGIBLE: where does the compensation money come from? (check all that apply, BUT DO NOT PROVIDE OPTIONS TO RESPONDENT)
Local Government □
QUESTION 21: When people lose crops or livestock to a wild animal, they can ask for the animal to be killed, but that process can take a long time and the request is not always granted. As a result, people can sometimes feel that their only option to stop future losses is to kill the animal without permission. Do you think that people around you currently kill problem animals without permission?

- YES
- NO
- DON’T KNOW

QUESTION 22: Thinking of scenario in the last question, when someone has lost crops or livestock to a wild animal, but permission has not been given to kill that animal, do you think that five years ago people around you killed that problem animal more often, less often, or about the same as they do today?

- MORE OFTEN
- LESS OFTEN
- ABOUT THE SAME
- DON’T KNOW

FOR QUESTIONS 23 & 24, USE THE CONSERVANCY OF RESIDENCE INDICATED IN QUESTION 1 AS INDICATED. IF THEY ANSWERED “DON’T KNOW” USE THE PHRASE “AROUND HERE.”

NOTE – REPEAT THAT THESE ANSWERS WILL BE CONFIDENTIAL AND ANONYMOUS

QUESTION 23: In the last seven years, there have been some instances of rhino poaching in the Kunene region. Have you heard of any rhinos being killed [in the CONSERVANCY/AROUND HERE] in the last seven years?

- YES
- NO

IF “YES” CONTINUE, OTHERWISE SKIP TO QUESTION 26.

QUESTION 24: Do you think that anybody living [in the CONSERVANCY/AROUND HERE] have participated in the killing of rhinos?

- YES
- NO
- I DON’T KNOW

IF “YES” AND PERSON IDENTIFIED CONSERVANCY OF RESIDENCE CONTINUE, OTHERWISE SKIP TO QUESTION 26.
QUESTION 25: Do you think that any members of the [CONSERVANCY] have participated in the killing of rhinos?

☐ YES
☐ NO
☐ I DON’T KNOW

QUESTION 26: The recent killing of rhinos has reoccurred after many years where no rhinos were killed. Why do you think that rhinos have started being killed recently?
___________________________________________________________________________________
___________________________________________________________________________________
___________________________________________________________________________________

IF THE PARTICIPANT ANSWERED “I DON'T KNOW” TO QUESTION 1 SKIP TO QUESTION 47, OTHERWISE CONTINUE

This next set of questions asks about your experience in the conservancy in which you currently stay.

QUESTION 27: Has your conservancy caused your household to experience a different number of problems with wild animals than you would have experienced without the conservancy being formed?

☐ A lot more problems than if the conservancy hadn’t been formed
☐ A little more than if the conservancy hadn’t been formed
☐ About the same than if the conservancy hadn't been formed
☐ A little less than if the conservancy hadn't been formed
☐ A lot less than if the conservancy hadn't been formed

QUESTION 28: Have you or anyone in your household received any of the following from your conservancy (select all that apply)?

1. Cash payments ☐ YES ☐ NO
2. Meat from animal killed by a trophy hunter ☐ YES ☐ NO
3. Use of infrastructure improvement such as a borehole or conservancy vehicle ☐ YES ☐ NO
4. School scholarship or other educational expenses paid ☐ YES ☐ NO
5. Money or meat for funeral ☐ YES ☐ NO
6. Money to start a business ☐ YES ☐ NO
7. Other. ☐ YES ☐ NO
QUESTION 29: Overall, have the benefits you have experienced from your conservancy outweighed the cost of living with wildlife?

☐ YES
☐ NO
☐ They are about the same

QUESTION 30: Do some people in your conservancy experience more benefits than others?

☐ YES ☐ NO
IF YES explain: __________________________________________________________

QUESTION 31: Have you or anyone in your household received benefits from a conservancy other than the one in which you live?

☐ YES ☐ NO
Explain _____________________________________________________________________
____________________________________________________________________________

QUESTION 32: Does your conservancy have rules saying what people can and cannot do?

☐ YES ☐ NO

IF NO, SKIP TO QUESTION 36, OTHERWISE CONTINUE TO QUESTION 32(A).

QUESTION 32(A)
Do the rules apply to everyone living in the conservancy, or only to some people?

☐ Everyone
☐ Only to some people
Explain______________________________________________________________________
____________________________________________________________________________

QUESTION 33: Do you think that members of your conservancy tend to follow the conservancy rules?

☐ YES ☐ NO
Explain why you think this: __________________________________________________
____________________________________________________________________________

QUESTION 34: Do you think that non-members living in your conservancy tend to follow the conservancy rules?
□ YES  □ NO  
Explain why you think this: _____________________________________________________
___________________________________________________________________________________
___________________________________________________________________________________
___________________________________________________________________________________

QUESTION 35: Do you feel that the conservancy rules are fair?

___________________________________________________________________________________

QUESTION 36: Please state the degree to which you agree with the following statement: My individual decisions have no impact on whether my conservancy succeeds. (vary order of this and Q#39)
□ Strongly Agree  □ Somewhat Agree  □ Somewhat Disagree  □ Strongly Disagree

QUESTION 37: Are you able to participate in choosing the people on the conservancy committee in your conservancy?

___________________________________________________________________________________

QUESTION 38: Do the people making decisions in your conservancy respond to the needs of you and people like you?
□ YES  □ NO  
Explain why you think this: _____________________________________________________
___________________________________________________________________________________
___________________________________________________________________________________

QUESTION 39: Please state the degree to which you agree with the following statement: The success of my conservancy depends on the individual decisions that I make.
□ Strongly Agree  □ Somewhat Agree  □ Somewhat Disagree  □ Strongly Disagree

QUESTION 40: Do you feel that decision-makers in your conservancy tend to favor people in any of the following groups?

1. Own family □ YES  □ NO
2. Own village □ YES  □ NO
3. Own traditional authority □ YES  □ NO
4.Own ethnicity □ YES  □ NO
5. Other □ YES  □ NO
Explain Any Yes Answers, at least in pretest
QUESTION 41: Do you think that the committee in your conservancy takes conservancy money without permission to spend on themselves or their families?

☐ Yes
☐ No

QUESTION 42: Overall, do you feel that your conservancy has made your life better, worse, or about the same as if the conservancy did not exist?

☐ Better
☐ Worse
☐ About the same

Explain

__________________________________________________________________________________

__________________________________________________________________________________

__________________________________________________________________________________

QUESTION 43: Overall, do you personally feel that the creation of your conservancy was a good or bad thing?

☐ Good
☐ Bad
☐ Neither good nor bad

Explain

__________________________________________________________________________________

__________________________________________________________________________________

__________________________________________________________________________________

QUESTION 44: In the past two years, have you attended any conservancy meetings?

☐ YES  How many? ________________________________

☐ NO

IF NO, WHY NOT?

__________________________________________________________________________________

__________________________________________________________________________________

QUESTION 45: Who has the responsibility for looking after wildlife in your conservancy? (choose all that apply)

☐ Government
☐ Non-governmental agencies (Save the Rhino Trust, WWF)
☐ Conservancy employees (game guards, rhino guards, lion guards)
☐ Conservancy members
☐ People living in the conservancy that are non-members
QUESTION 46: The following questions ask your thoughts regarding particular situations occurring within a conservancy. For each situation, please choose the thought that most closely matches your own thoughts.

In this situation, a person kills wildlife without permission from the conservancy committee for food for his/her own household.

i. If a conservancy member did this:
   □ This behavior is understandable and the person should not be punished
   □ This behavior is understandable but the person should be punished
   □ This behavior is not understandable but the person should not be punished
   □ This behavior is not understandable and the person should be punished

ii. If a non-member conservancy resident did this:
   □ This behavior is understandable and the person should not be punished
   □ This behavior is understandable but the person should be punished
   □ This behavior is not understandable but the person should not be punished
   □ This behavior is not understandable and the person should be punished

iii. If a non-member living outside the conservancy did this:
   □ This behavior is understandable and the person should not be punished
   □ This behavior is understandable but the person should be punished
   □ This behavior is not understandable but the person should not be punished
   □ This behavior is not understandable and the person should be punished

In the second situation, an individual kills wildlife without permission from the conservancy committee to sell for food to people outside the conservancy.

iv. If a conservancy member did this:
   □ This behavior is understandable and the person should not be punished
   □ This behavior is understandable but the person should be punished
   □ This behavior is not understandable but the person should not be punished
□ This behavior is not understandable and the person should be punished

v. If a non-member conservancy resident did this:
□ This behavior is understandable and the person should not be punished
□ This behavior is understandable but the person should be punished
□ This behavior is not understandable but the person should not be punished
□ This behavior is not understandable and the person should be punished

vi. If a non-member living outside the conservancy did this:
□ This behavior is understandable and the person should not be punished
□ This behavior is understandable but the person should be punished
□ This behavior is not understandable but the person should not be punished
□ This behavior is not understandable and the person should be punished

In the third situation, an individual kills wildlife without permission from the conservancy committee to stop the wildlife from harming his/her crops or livestock.

vii. If a conservancy member did this:
□ This behavior is understandable and the person should not be punished
□ This behavior is understandable but the person should be punished
□ This behavior is not understandable but the person should not be punished
□ This behavior is not understandable and the person should be punished

viii. If a non-member conservancy resident did this:
□ This behavior is understandable and the person should not be punished
□ This behavior is understandable but the person should be punished
□ This behavior is not understandable but the person should not be punished
□ This behavior is not understandable and the person should be punished

ix. If a non-member living outside the conservancy did this:
□ This behavior is understandable and the person should not be punished
□ This behavior is understandable but the person should be punished
□ This behavior is not understandable but the person should not be punished
□ This behavior is not understandable and the person should be punished
In the last situation, an individual kills wildlife without permission from the conservancy committee to sell the horns or tusks.

x. If a conservancy member did this:

☐ This behavior is understandable and the person should not be punished
☐ This behavior is understandable but the person should be punished
☐ This behavior is not understandable but the person should not be punished
☐ This behavior is not understandable and the person should be punished

xi. If a non-member conservancy resident did this:

☐ This behavior is understandable and the person should not be punished
☐ This behavior is understandable but the person should be punished
☐ This behavior is not understandable but the person should not be punished
☐ This behavior is not understandable and the person should be punished

xii. If a non-member living outside the conservancy did this:

☐ This behavior is understandable and the person should not be punished
☐ This behavior is understandable but the person should be punished
☐ This behavior is not understandable but the person should not be punished
☐ This behavior is not understandable and the person should be punished

This last section asks some final follow-up questions.

QUESTION 47: How much do you agree with the following statement? “Taking care of myself and my family is the most important thing, even if it means doing something that would not be good for my community?” (alternate order with Q#52)

☐ Strongly Agree ☐ Somewhat Agree ☐ Somewhat Disagree ☐ Strongly Disagree

QUESTION 48: How old are you? ____________________________

QUESTION 49: Do you identify as any of the following ethnicities? (Select all that apply)

☐ Damara
☐ Himba
☐ Herero
☐ Other ___________________________________________________________________________
QUESTION 50: Please select the highest level of schooling that you have completed.

- None
- Some primary education (grades 1-7)
- Finished primary education
- Some secondary education (grades 8-12)
- Finished secondary education
- Received vocational training (also mark highest level of other schooling completed)
- Received a vocational degree (also mark highest level of other schooling completed)
- Some tertiary education (college or university)
- Received a college or university degree

QUESTION 51: Please select the highest level of schooling that anyone other than you in your household has completed.

- None
- Some primary education (grades 1-7)
- Finished primary education
- Some secondary education (grades 8-12)
- Finished secondary education
- Received vocational training (also mark highest level of other schooling completed)
- Received a vocational degree (also mark highest level of other schooling completed)
- Some tertiary education (college or university)
- Received a college or university degree

QUESTION 52: Do you agree with the following statement? “Taking care of my community is the most important thing, even if it means doing something that would not be good for myself or my family?”

- Strongly Agree
- Somewhat Agree
- Somewhat Disagree
- Strongly Disagree

QUESTION 53: Do you feel that you can trust the people who live in the area around you?

- YES
- NO

QUESTION 54: Compared to FIVE YEARS AGO, is your life today better, worse, or about the same?

- BETTER
- WORSE
- ABOUT THE SAME
QUESTION 55: Compared to LAST YEAR, is your life today better, worse, or about the same?

☐ BETTER  ☐ WORSE  ☐ ABOUT THE SAME

QUESTION 56: In ONE YEAR FROM NOW, will your life be better, worse, or about the same as it is today?

☐ BETTER  ☐ WORSE  ☐ ABOUT THE SAME

QUESTION 57: In FIVE YEARS FROM NOW, will your life be better, worse, or about the same as it is today?

☐ BETTER  ☐ WORSE  ☐ ABOUT THE SAME

QUESTION 58: Is there anything that I have not asked about that you would like to tell me?

___________________________________________________________________________________
___________________________________________________________________________________

QUESTION 59: How satisfied were you with the survey experience?

___________________________________________________________________________________
___________________________________________________________________________________

INTERVIEWER OBSERVATIONS/NOTES:

Respondent Gender: ☐ MALE  ☐ FEMALE

___________________________________________________________________________________
___________________________________________________________________________________
___________________________________________________________________________________
APPENDIX 3 (CHAPTER 4)

I. Experiment Instructions

The following instructions were provided to participants in the experiment.

Instructions

Now that the experiment has begun, please make sure that your cell phones are switched off. **We ask that you do not talk with one another and do not turn around or look at other participants' screens.**

If you have a question after reading the instructions, please raise your hand and the experimenter will answer your question in private.

Welcome

You will receive $7 for showing up for this experiment. You can also earn additional money by participating fully in this experiment. The total amount of money you can earn depends on your decisions during this experiment and, in part, on the decisions of paired participants. The experiment will not take more than 1 hour. You are free to leave at any time. However, if you choose to do so before the end, you will only receive the $7 show-up fee. Please read the following instructions carefully.

*Private Decisions*

Please note that your decisions and earnings are private. Your decisions are recorded using your experimental subject ID given to you by the experimenter, not your name or your student ID. At the end of the experiment, you will be asked to enter your name into the computer. This information is to process your payment only - it will not be used in any other way.
Today's Experiment

Payments
Your decisions will earn you Experimental Currency Units (ECUs). At the end of the experiment your ECUs will be exchanged into US dollars at a rate of 65 ECUs = $1. You will be paid in US dollars.

Stages
Today's experiment will consist of two stages. You will receive instructions at the beginning of each stage.

Stage 1 Instructions

In Stage 1, you will make decisions in 10 different scenarios.

Decision Task
You have been given 10 different scenarios where you must choose between the alternatives A and B. In each of the 10 decisions, A gives a guaranteed payment. If you choose B, your payment depends on chance.

Example: In the following setting you must decide whether you prefer A, in which you receive 175 ECU guaranteed, or B, in which there is a 50% chance that you receive 250 ECU and a 50% chance that you receive 0 ECU.

<table>
<thead>
<tr>
<th>A</th>
<th>Please indicate your choice</th>
<th>B</th>
</tr>
</thead>
<tbody>
<tr>
<td>175 ECU guaranteed</td>
<td>A☐☐☐☐B</td>
<td>50% chance of 250 ECU and 50% chance of 0 ECU.</td>
</tr>
</tbody>
</table>
The guaranteed payment when you choose A will change from scenario to scenario. The chances and payments for B remain the same: In all 10 scenarios, you will have a 50% chance of receiving 250 ECU and a 50% of receiving 0 ECU.

---

_Earnings in Stage 1_

*At the end of the experiment, the computer will calculate your earning for Stage 1. Only one of the 10 decisions you make will be used to compute these earnings.* One of the 10 decision scenarios is selected at random and your corresponding decision will be used to calculate your earnings.

To select the decision scenario the experimenter will draw a card from a shuffled deck of cards number 1 through 10 (corresponding to the decision tasks 1-10). Each decision task has the same probability of being picked. The draw will take place in public at the front of the room and the scenario that is picked is the same for **everyone** in the room.

Once the decision scenario has been picked, another card will be randomly picked from a deck of two cards that contains one face card (a Jack) and one non-face card (#2). **The card drawn will determine the earnings of everyone that picked B in this decision scenario:**

- If the Jack is picked, those that picked B will receive the high payoff 250 ECU.
- If the #2 card is picked, those that picked B will receive 0 ECU.

**Everyone** that picked A for this decision scenario will receive the guaranteed payoff.

Please raise your hand if you have any questions. Otherwise, click Continue to proceed to the quiz.
Stage 2 Instructions

You have completed Stage 1. Before making decisions that affect your earnings for Stage 2, you will answer another short quiz designed to check your understanding of the decision task.

Please note that your earnings in this stage do not depend on your decisions from the previous stage.

Groups and Member Number

You have been randomly assigned to a group of 4 persons. You will remain in this group until the end of the experiment. Each group member has been assigned a member number between 1 and 4. Your member number (and your group members' numbers) will remain the same throughout the experiment.

Please note: Group assignments and member number assignments have been assigned randomly across all of the participants in today's experiment.

Decision Periods

Stage 2 consists of 15 decision rounds.

Decision Task

Every round, you decide on your own whether to invest in a group project. Your own individual payoffs will change depending on the decisions made by you and the other members of your group.

Every round, each group member receives an initial allocation of ECUs – this is his/her Starting Balance. (You will be informed on the next page what your Starting Balance is.) In each round, each group member must individually choose whether to invest in the group project. The group project also returns ECU: for each member that invests in the group project, all four people in the group (including that member) receives 8 ECU each. A group member does not need to invest to receive this payout. So, for example, if two members invested in the project, every member of that group would receive a return of 16 ECU from the project (in addition to his/her Starting
Balance), regardless of whether they themselves made an investment. If all four members invest, every person in the group would receive an additional 32 ECU each.

Group members do not need to pay any ECUs to invest, but investment in the group project is risky. Every individual that decides to invest in the group project has a chance of losing 30 ECUs from their Starting Balance. NOTE: The group project investor will still receive the group project payout even if they lose the 30 ECUs.

Your per round earnings are therefore calculated as follows:

<table>
<thead>
<tr>
<th>You did not invest in the group project</th>
<th>You invested in the group project</th>
</tr>
</thead>
<tbody>
<tr>
<td>Starting Balance ECU + 8ECU \times \text{Number of individuals who invested in the group project}</td>
<td>(Starting Balance – 30) ECU + 8ECU \times \text{Number of individuals who invested in the group project}</td>
</tr>
<tr>
<td>(Starting Balance ECU + 8ECU \times \text{Number of individuals who invested in the group project})</td>
<td>(Starting Balance ECU + 8ECU \times \text{Number of individuals who invested in the group project})</td>
</tr>
</tbody>
</table>

Information you will receive after each round:

- How many group members invested in the group project
- Whether you invested in the group project
- If you did, whether a loss occurred
- Your earnings for the round

[1] Starting Balances and Probability of Losses [HOMOGENEOUS_WM_Rm Treatment]

In your group, everyone receives the same Starting Balance and faces the same probability of loss:

- Everyone receives a Starting Balance of 42 ECU each.
- Everyone each faces a 50% risk of losing 30 ECU if he or she decided to invest in the group project.

To determine whether an investor loses 30 ECU, the computer will randomly select a number for that individual investor between (and including) 1 and 100. If the number is 50 or lower, the
investor loses 30 ECU from his or her Starting Balance. If the number is greater than 50, the investor keeps the 30 ECU.

**Important**: Risks are independent of one another across individuals and rounds. This means, whether a group member loses 30 ECU *does not* impact the likelihood of another group member losing 30 ECU if he or she invested in the group project. And, (not) losing 30 ECU one round, *does not* affect the likelihood of loss the following round.

### [2] Starting Balances and Probability of Losses [HETEROGENEOUS_WEALTH_Rm Treatment]

In your group, members have different Starting Balances but everyone faces the same probability of loss.

- Two members receive a Starting Balance of **48 ECU** each and the other two members receive a Starting Balance of **36 ECU** each.
- Everyone each faces a **50% risk** of losing 30 ECU if he or she decided to invest in the group project.

To determine whether an investor loses 30 ECU, the computer will randomly select a number for that individual investor between (and including) 1 and 100. If the number is 50 or lower, the investor loses 30 ECU from his or her Starting Balance. If the number is greater than 50, the investor keeps the 30 ECU.

**Important**: Risks are independent of one another across individuals and rounds. This means, whether a group member loses 30 ECU *does not* impact the likelihood of another group member losing 30 ECU if he or she invested in the group project. And, (not) losing 30 ECU one round, *does not* affect the likelihood of loss the following round.

### [3] Starting Balances and Probability of Losses [HETEROGENEOUS_RISK_Wm Treatment]

In your group, members have the same Starting Balances but face different probabilities of loss.

- In your group, everyone receives a Starting Balance of **42 ECU** each
Two members each face a 30% risk of losing 30 ECU if he or she decided to invest in the group project. The other two face a 70% risk.

To determine whether an investor loses 30 ECU, the computer will randomly select a number for that individual between (and including) 1 and 100.

- For individuals with a 30% risk of loss, if the number is 30 or lower, the investor loses 30 ECU from his or her Starting Balance. If the number is greater than 30, the investor keeps the 30 ECU.
- For individuals with a 70% risk of loss, if the number is 70 or lower, the investor loses 30 ECU from his or her Starting Balance. If the number is greater than 70, the investor keeps the 30 ECU.

**Important**: Risks are independent of one another across individuals and rounds. This means, whether a group member loses 30 ECU does not impact the likelihood of another group member losing 30 ECU if he or she invested in the group project. And, (not) losing 30 ECU one round, does not affect the likelihood of loss the following round.

**You have been randomly assigned to face a [loss probability here] chance of losing 30 ECU if you invest in the group project.**

[4] *Starting Balances and Probability of Losses [HETEROGENEOUS_BALANCED Treatment]*

In your group, members have different Starting Balances and face different probabilities of loss.

- Two members receive a Starting Balance of 48 ECU each and face a 70% risk of losing 30 ECU if he or she decided to invest in the group project.
- The other two members receive a Starting Balance of 36 ECU each and face a 30% risk of losing 30 ECU if he or she decided to invest in the group project.

To determine whether an investor loses 30 ECU, the computer will randomly select a number for that individual between (and including) 1 and 100.

- For individuals with a 30% risk of loss, if the number is 30 or lower, the investor loses 30 ECU from his or her Starting Balance. If the number is greater than 30, the investor keeps the 30 ECU.
• For individuals with a 70% risk of loss, if the number is 70 or lower, the investor loses 30 ECU from his or her Starting Balance. If the number is greater than 70, the investor keeps the 30 ECU.

Important: Risks are independent of one another across individuals and rounds. This means, whether a group member loses 30 ECU does not impact the likelihood of another group member losing 30 ECU if he or she invested in the group project. And, (not) losing 30 ECU one round, does not affect the likelihood of loss the following round.

You have been randomly assigned to receive a Starting Balance of [Starting Balance here] ECU and face a [loss probability here] chance of losing 30 ECU if you invest in the group project.

[5] Starting Balances and Probability of Losses [HETEROGENEOUS_UNBALANCED Treatment]

In your group, members have different Starting Balances and face different probabilities of loss.

• Two members receive a Starting Balance of 48 ECU each and face a 30% risk of losing 30 ECU if he or she decided to invest in the group project.
• The other two members receive a Starting Balance of 36 ECU each and face a 70% risk of losing 30 ECU if he or she decided to invest in the group project.

To determine whether an investor loses 30 ECU, the computer will randomly select a number for that individual between (and including) 1 and 100.

• For individuals with a 30% risk of loss, if the number is 30 or lower, the investor loses 30 ECU from his or her Starting Balance. If the number is greater than 30, the investor keeps the 30 ECU.
• For individuals with a 70% risk of loss, if the number is 70 or lower, the investor loses 30 ECU from his or her Starting Balance. If the number is greater than 70, the investor keeps the 30 ECU.

Important: Risks are independent of one another across individuals and rounds. This means, whether a group member loses 30 ECU does not impact the likelihood of another group member
losing 30 ECU if he or she invested in the group project. And, (not) losing 30 ECU one round, 
*does not* affect the likelihood of loss the following round.

You have been randomly assigned to receive a Starting 
Balance of [Starting Balance here] ECU and face a [loss 
probability here] chance of losing 30 ECU if you invest in 
the group project.
Stefan Carpenter
Curriculum Vitae

EDUCATION

Ph.D. Joint Public Policy, School of Public and Environmental Affairs and Department of Political Science, Indiana University, Bloomington, expected June 2020
Fields: Environmental Policy, Public Policy, Comparative Politics
Committee: Lauren MacLean (chair), Bill Blomquist, Jennifer Brass, David Konisky, Vicky Meretsky
Dissertation: Unpacking the Efficacy of Community-Based Wildlife Governance: The Influence of Economic Benefit Types, Risk, and Heterogeneity on Collective Action
Dissertation Abstract: The dissertation uses three different methodologies (quantitative, experimental, and survey-based) to investigate community-based natural resource management (CBNRM) of wildlife. The survey-based portion analyzes household level questionnaires administered in Namibian pastoral communities to identify factors impacting community attitudes regarding wildlife in the presence of human-wildlife conflict (such as wildlife depredation of crops or livestock). The quantitative portion compares the efficacy of national and CBNRM governance at protecting African elephant by analyzing elephant kill data collected by the Convention on International Trade (CITES). The experimental portion uses a behavioral “prisoners’ dilemma” game to examine the impact of risk heterogeneity, economic heterogeneity, and the interaction between the two, on collective action and individual decision-making in community-based wildlife areas.

J.D. University of Pennsylvania, Philadelphia, 2007
Certificate of Environmental Studies

M.Ed. Boston University, Boston, 2004

B.A. Department of History, Amherst College, Amherst, 1999

ACADEMIC APPOINTMENTS

2019-2020 Visiting Assistant Professor
Department of Environmental Studies
Eckerd College, St. Petersburg, FL

PUBLICATIONS

Peer Reviewed and Refereed


In Progress

Carpenter, Stefan. “Gaining from Wildlife? The Impact of Different Benefit Types on Individuals’ Views of Whether Wildlife Improves Their Lives.”


Carpenter, Stefan. “The Impact of Climate Change on Community-Based Wildlife Management: A Case Study from Northwest Namibia.”

FELLOWSHIPS

Federal

Fall 2015- Spring 2016  Federal Foreign Language and Area Studies (FLAS) Fellowship for the study of Swahili, IU Bloomington

Research

Spring 2019  Bentley Dissertation Writing Fellowship, Department of Political Science

Fall 2018  Research Fellowship, Department of Political Science (funded through a Carnegie Fellows award)

Fall 2017  Bentley Dissertation Research Fellowship, Department of Political Science

Summer 2017  PhD Merit-Based Fellowship, School of Public and Environmental Affairs

Fall 2016- Spring 2017  Research Fellowship, School of Public and Environmental Affairs

Summer 2016  Research Stipend, School of Public and Environmental Affairs

Summer 2015  Research Stipend, School of Public and Environmental Affairs

Fall 2014- Spring 2015  Research Fellowship, School of Public and Environmental Affairs
Summer 2014 Research Stipend, Ostrom Workshop, Indiana University

Teaching
Fall 2013- Spring 2014 Teaching Fellowship, Department of Political Science

AWARDS
2016 Best Paper in Environmental and Public Policy, Association of SPEA PhD Students (ASPS) Conference, IU Bloomington

PRESENTATIONS
Invited Panelist/Lecturer

Papers
2019 “The Impact of Climate Change on Community-Based Wildlife Management: A Case Study from Northwest Namibia.” Sustainability and Development Conference, University of Michigan, Ann Arbor, MI, October 11-14.
2018 “Community-Based Governance in the Presence of Human-Wildlife Conflict: Examining the Impact of Different Benefit Types in Four Namibian Conservancies.” Sustainability and Development Conference, University of Michigan, Ann Arbor, MI, November 9-11.

GRANTS

2017         Stefan Carpenter. “Investigation into Factors Impacting the Success of Community-Based Conservation in Northern Namibia.” Wildize Foundation ($10,000).
2016         Stefan Carpenter. “Investigation into Factors Impacting the Success of Community-Based Conservation in Northern Namibia.” Ostrom Workshop Research Award, IU Bloomington ($7,500).

TEACHING EXPERIENCE

Eckerd College, St. Petersburg, Visiting Assistant Professor
Fall 2019     Introduction to Environmental Studies (2 sections)
              Marine Protected Species
Winter 2019   Human-Wildlife Interactions
Spring 2020   Fisheries Governance
              Wildlife Politics and Policy
              Marine Protected Species

Indiana University, Bloomington, Assistant Instructor
Spring 2014   American Political Controversies
Fall 2013     Constitutional Law

University of South Florida, Tampa, Instructor of Record
Spring 2011   Topics in Construction Law

University of Pennsylvania, Philadelphia, Teaching Assistant
Fall 2006-Spring 2007       Sports Business Management

Other Teaching
2004         Instructor, SAT and Reading Skills, Learning Skills-Correct Read, Inc., Amherst MA
MEDIA APPEARANCES

PROFESSIONAL SERVICE
Reviewer, Journal of Environmental Management (regular); Conservation Letters (occasional)
Pro Bono Representation, Sea Turtle Oversight Protection Inc., Miami, FL; Everglades Law Center, Inc., Winter Haven, FL
Judge, International Environmental Moot Court Competition, Stetson University College of Law, Tampa

PROFESSIONAL EXPERIENCE
2011-2013 Associate, Shankman Leone, P.A., Tampa, FL
2009-2011 Associate, Carlton Fields, St. Petersburg, FL
2007-2009 Associate, Hunton & Williams, Washington, DC
2005 Legal Intern, Land, Environment and Development (LEAD) Project, Legal Assistance Centre, Windhoek, Namibia
2003-2004 Assistant Coach, Men’s Track & Field, Massachusetts Institute of Technology
2002-2003 Assistant Coach, Women’s Track & Field, Massachusetts Institute of Technology
2000-2002 Head Indoor/Assistant Outdoor Coach, Track & Field, Mount Holyoke College, South Hadley, MA
1999-2000 Assistant Coach, Track & Field, Mount Holyoke College, South Hadley, MA

PROFESSIONAL ASSOCIATIONS
Association for Environmental Studies and Sciences
Society for Conservation Biology
The Wildlife Society
Association for Public Policy Analysis and Management
American Political Science Association
Midwest Political Science Association
Southern Political Science Association
District of Columbia Bar Association
Pi Alpha Alpha (Public Affairs and Administration Honor Society)