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Conservation, Economics, and Management of Hunting on Private Land: An International, National and California Analysis

By

Luke Thomas Macaulay

A dissertation submitted in partial satisfaction of the

requirements for the degree of

Doctor of Philosophy

in
Environmental Science, Policy & Management

in the

Graduate Division

of the

University of California, Berkeley

Committee in charge:

Professor Lynn Huntsinger, Chair
Professor Reginald Barrett
Professor David Zilberman

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Abstract

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Privately owned land accounts for significant areas of land internationally, nationally, and in California. In the U.S. and elsewhere, private land tends to support high levels of biodiversity because land with more productive natural resources was settled and privatized first. These lands, which are integral to conservation goals, are often the most vulnerable to habitat degradation and loss through changes in land-use and fragmentation. In 1930, Aldo Leopold encouraged the development of an incentive scheme to better conserve private lands in the U.S. where hunters would pay landowners for access to conserved wildlife habitat and game populations that could be sustainably harvested. Although a wide body of literature has discussed this approach, much of the research is either theoretical or limited to particular regions and these studies have rarely tested for an explicit connection to whether conservation is ultimately improved as a result of paid hunting.

The goal of this dissertation is to evaluate the economic, conservation and management aspects of hunting on private land internationally, nationally, and in California. The first chapter uses a case study approach to explore the environmental and economic issues surrounding hunting in the context of Spain and California. The study found that increased game management rights in Spain appears to yield improved economic return, but at an environmental cost.

The second chapter evaluates the scale, distribution and conservation aspects of spending to access private land for fishing, hunting, and wildlife-watching in the United States. This study found that that approximately 440 million acres of private land, an estimated 22% of the contiguous land area of the U.S. and 33% of all private land in the U.S., are either leased or owned for wildlife-associated recreation. Much of these lands are private rangelands and forestlands. Land utilized for hunting accounted for 81% of that total, while land utilized for fishing and wildlife-watching, although comparatively small, likely includes riparian zones and areas with high environmental or amenity values. Hunters own or lease properties of larger size classes than anglers or wildlife-watchers, providing a possible economic incentive for maintaining large unfragmented properties that provide a variety of conservation benefits. Results show that Americans annually spend an estimated $814 million in day-use fees, $1.48 billion for long-term leases, and $14.8 billion to own private land primarily for wildlife-

associated recreation. Hunting, in particular big game hunting, comprises some of the largest contributions to payments for wildlife-associated recreational use on private land. This finding suggests that hunting may be an important market-based mechanism to maintain large unfragmented parcels of wildlife habitat.

Chapter 3 utilizes interviews with a random sample of landowners in California to evaluate conservation practices associated with hunting enterprises. This study found relatively low adoption of hunting enterprises among landowners, and that there were mixed conservation outcomes associated with hunting. Landowners who enrolled in the California Department of Fish & Wildlife's Private Land Management program, which provides enhanced game management rights to landowners in exchange for habitat improvement practices, performed the most comprehensive habitat improvement practices, including riparian zone restoration and adjustment of grazing practices to enhance cover and forage resources for wildlife. Many other landowners, however, earned some income from hunting, but either did not implement additional conservation practices to enhance wildlife habitat or performed practices that could cause some ecological problems, such as planting of feed crops that can create openings for invasive noxious weeds to be established on a property. This study found significant opportunities in California to not only increase adoption of hunting enterprises, but to engage in educational efforts to encourage ecologically-friendly wildlife management practices as a way to enhance both revenue from hunting enterprises and conservation outcomes.

The final chapter focuses on the development of methods to better understand the population characteristics of the Columbian black-tailed deer (*Odocoileus hemionus columbianus*) in order to improve harvest recommendations. Using camera-traps on a 2,500 acre private ranch in San Benito County, California, the study estimates the density and sex ratios of deer by using a Bayesian spatial mark-resight model. It also evaluates the effect of using bait in developing population estimates. The study found that deer densities on the property are estimated to be 9.9 (SE 0.91) individuals/km^2, and that antlered bucks make up only 11% (SE 1%) of the population. Bait increased encounter rates of deer by a factor of 3.7, showing that the use of bait can help reduce the length of time that cameras must be operational and may create more precise population estimates due to increased detectability of deer.

In conclusion, this dissertation found the game management rights for hunting were important for economic return from hunting on private land, but without regulation may result in negative environmental impacts. Across the United States, hunting contributes significantly to landowner income, especially to properties of larger size classes in rangeland and forestry habitats, which suggests that hunting provides an economic incentive to maintain large unfragmented properties. In the context of California, programs that give landowners greater game management rights in exchange for habitat improvement practices resulted in benefits for landowners and the environment. Finally, this dissertation has developed a statistical model that can be utilized to evaluate population parameters for one of the most economically important game species in California. In sum, recreational hunting can provide income to the private landowner and with the appropriate regulations, education and management, can incentivize the enhancement and maintenance of wildlife habitat on private land.

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Chapter 1

Hunting in Managed Oak Woodlands: Contrasts Among Similarities



The California tule elk (*Cervus canadensis ssp. nannodes*) is a sub-species of elk once almost lost to commercial overhunting. Successful reintroduction efforts have allowed renewed hunting opportunities of great value to landowners. (Photograph by B. Voelker)

Abstract: Distinct cultural and legal histories governing the property rights that regulate wildlife and land tenure in California and Spain have created dissimilar hunting systems. The differences that are manifest in the methods of hunting, the economic return to landowners, the actions taken to manage game species, and the accompanying environmental effects. Private landowners in Spain retain greater control of game species, while in California, the state and federal government exerts greater authority. After providing background on the game species and systems of hunting in California and Spain, a review of the legal and cultural history illustrates how distinct systems evolved in places that are similar in many other ways. In terms of economics, hunting revenue in Spain is often greater than in California, due to higher hunter participation rates, fewer governmental restrictions that limit the commercialization of hunting, and greater liberties in hunting methods and game management practices. As such, income from hunting provides a greater incentive for Spanish landowners to maintain areas of habitat for game species. Some of the greatest contrasts between these places are illustrated in wildlife management practices, where Spanish landowners can implement far more intensive practices to manipulate populations of game species. Numerous environmental effects can result from these management practices, which include changes to vegetation, erosion, genetic impacts, invasive species introductions, and impacts to non-game species.

Keywords: Hunting California Spain Game Wildlife management Property rights Predator control

1.1 The Basics

The oak woodlands of California and Spain share commonalities of climate, vegetation, livestock production, and biodiversity of global importance. Yet distinct cultural and legal histories governing property rights over wildlife and land tenure have created dissimilar hunting traditions. The geography, game species, hunting participation rates, land tenure, and other factors have all shaped these unique systems, and the legal and cultural history of hunting illustrates how these hunting systems evolved. This historical discussion elucidates the economic, legal, and social roots of present-day hunting, while the final sections dwell on wildlife management practices, and the associated environmental impacts of hunting in Spain and California oak woodlands.

1.1.1 Geography and Game Species

Three regional autonomías, or autonomous regions—Castilla-La Mancha, Extremadura, and Andalucía—contain a sizable part of the Spanish dehesa (Huntsinger et al. 2013) and include

what historically and today are Spain's most productive hunting lands. Forty-two percent of all Spanish hunting areas are contained in these autonomías, where two-thirds of all big game harvests take place (Metra and Seish 1985; López Ontiveros and Verdugo 1991). The dehesa, in association with surrounding and understory shrublands, provides quality food and cover for popular game species including Iberian red deer (Cervus elaphus hispanicus), wild pig (Sus scrofa), and occasionally mouflon (Ovis orientalis musimon) or fallow deer (Dama dama) (Blas Aritio 1974). High quality hunting in the early 1900s made the Spanish dehesa a destination for big game hunters from Europe and lands farther afield (Delgado and Muñoz 1982). Iberian red deer and wild pigs make up the largest share of Spain's annual harvest, reaching averages of 100,000 Iberian red deer and 180,000 wild boar (Carranza 2010; Garrido Martín 2011) (Tables 1.1 and 1.2).

The hunting of small game in Spain is a secondary activity, although preferred by some hunting classicists because it permits solitary and contemplative travel in a natural realm (Delibes 1982). Small game species predominate in more agri- culturally developed areas. Popular small game species include thrushes (Turdus spp.), the European rabbit (Oryctolagus cuniculus) and the red-legged partridge (Alectoris rufa). While over 3 million partridges are harvested every year, few are from natural populations, with most birds farm-raised and released onto dehesa sites before the start of each year's hunting season (Garrido Martín 2011) (Figs. 1.1 and 1.2).

The oak woodlands of California are located in foothill and coastal regions from sea level to an elevation of 1300 m in the southern Sierra Nevada. The main oak habitat is in California's Central to North Coast region and in a vast ring around the Central Valley. These offer food and cover for big game animals, especially in terms of acorns that yield significant nutrition for wildlife in the fall when herbaceous forage is desiccated and of minimal value. Migratory deer in many parts of California descend in the fall from montane environments to lower elevation oak woodlands where they over-winter. Big game species commonly hunted include the native mule deer (Odocoileus hemionus, classified into as many as six subspecies) and non-native wild pigs. California quail, mourning dove, wild turkey, pheasant, rabbit, jackrabbit, and squirrels are the small game and upland game commonly hunted in oak woodlands and on adjoining cropland (Figs. 1.3 and 1.4) (Table 1.3).

While waterfowl is hunted on water impoundments or rivers in the oak woodlands of California and Spain, most duck and goose hunting occurs in wetland habitats or flooded agricultural fields, areas outside the focus of this chapter.

Table 1.1 Predominant game species in the oak woodlands of California and Spain

California	Spain
Predominant big game species	*Predominant big game species*
Black-tailed/Mule deer *(Odocoileus hemionus* ssp.)	Iberian red deer *(Cervus elaphus hispanicus)*
Wild pig *(Sus scrofa)*	Wild pig *(Sus scrofa)*
Predominant small game/upland game	*Predominant small game/upland game*
Turkey *(Meleagris gallopavo)*	European rabbit *(Oryctolagus cuniculus)*
Quail *(Callipepla californica)*	Red-legged partridge *(Alectoris rufa)*

Table 1.2 2005–2009 average annual harvest of game species in Spain (Garrido Martín 2011)

Species	Average harvest
Big game	
Wild boar (*Sus scrofa*)	183,739
Red deer (*Cervus elaphus*)	102,220
Roe deer (*Capreolus capreolus*)	22,591
Fallow deer (*Dama dama*)	8,805
Mouflon (*Ovis musimon*)	6,150
Spanish ibex (*Capra pyrenaica*)	2,722
Barbary sheep (*Ammotragus lervia*)	601
Chamois (*Rupicapra rupicapra*)	485
Iberian wolf (*Canis lupus signatus*)	80
Small game	
Thrush (*Turdus* spp.)	6,120,587
European rabbit (*Oryctolagus cuniculus*)	5,628,208
Red-legged partridge (*Alectoris rufa*)	3,803,460
Common quail (*Coturnix coturnix*)	1,287,014
Granada hare (*Lepus granatensis*)	1,202,869
European turtle dove (*Streptopelia turtur*)	840,888
Eurasian woodcock (*Scolopax rusticola*)	75,611

Fig. 1.1 Game birds: Spain's red-legged partridge (*Alectoris rufa*).



Fig. 1.2 Changes in legislation, including the hunting law of 1970, means that an overwhelming proportion of Spanish woodland properties are posted as some form of reserved-access land (cotos). (Photograph by P. F. Starrs)



Fig. 1.3 Game birds: the wild turkey (*Meleagris gallopavo*) in California. (Photograph by L. T. Macaulay)

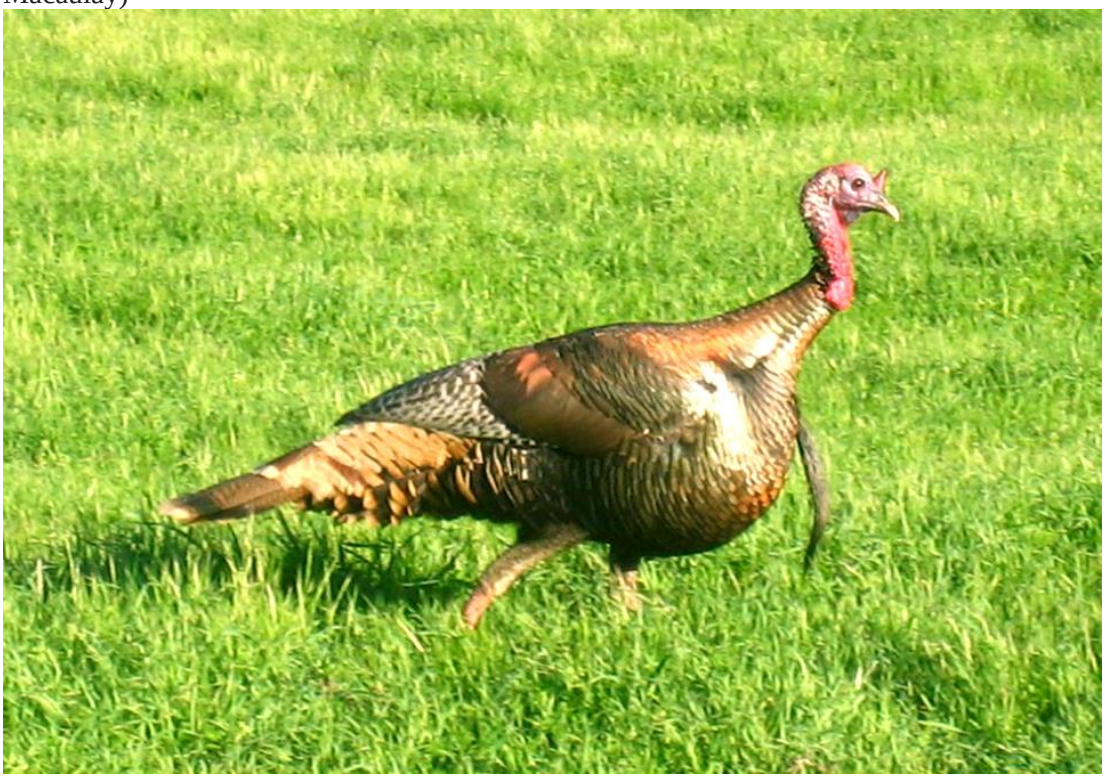


Fig. 1.4 Game birds: the California quail (Callipepla californica). (Photograph by L. T. Macaulay)



Table 1.3 2004–2008 averaged annual harvest of game species in California (CDFG)

Species	Average harvest
Big game	
Mule deer (*Odocoileus hemionus*)	40,470
Wild boar (*Sus scrofa*)	19,864
Small game	
Dove (*Columbidae* spp.)	1,759,337
Quail (*Callipepla* and *Oreortyx* spp.)	626,970
Farmed ring-necked pheasant (*Phasianus colchicus*)	342,710
Wild ring-necked pheasant (*Phasianus colchicus*)	107,082
Black-tailed jackrabbit (*Lepus californicus*)	63,137
Cottontail rabbit (*Sylvilagus* spp.)	55,977
Western gray squirrel (*Sciurus griseus*)	50,310
Wild turkey (*Meleagris gallopavo*)	26,202

1.1.2 Hunting Seasons

California deer hunting seasons close prior to the breeding season or rut. The seasons open at varying times across the state, with archery-only hunting commencing in August–September, and general gun hunting from September to early November (CDFG 2012b). Wild pigs, a non-native and invasive species in California, are hunted year-round. Bears are

occasionally found in California's oak woodlands, and the bear season often coincides with the deer season (Fig. 1.5).

In Spain, the general season for hunting runs from early October to late February, closing just before the onset of spring breeding for most game species. Hunting after February may adversely impact pregnant ungulate females, since they are easily captured by hunting dogs even when not a target of the hunt. Red deer are hunted in the Spanish montería—an organized hunt with dogs driving big game toward waiting hunters—within the general hunting period from early October to mid-February, and by stalking during the rutting season (from September to opening of the general season in October) (Aguayo 1986). One notable exception is the roe deer season, which generally runs from mid-April to the end of July with additional hunting periods in September and October.

Fig. 1.5 Wild pigs harvested in California. (Photograph from the collection of R. Barrett)



1.1.3 Structural Organization of Hunting Operations

In Spain, two main hunting venues exist, the coto social (or coto deportivo) and the coto privado (Barbosa et al. 2004). The coto social is organized by a community and maintained for recreational use. Hunts historically occur on community hunting grounds and are generally oriented towards small game without a profit motive (Alagona et al. 2013). In modern times, these clubs may secure the use of a private property, or in some cases, a community actually

owns a dehesa where a productive use of the land is hunting. The coto privado involves hunts on private lands oftentimes for profit. Viability and profitability of these properties has increased since 1970 due to four significant changes: (1) landowners can enclose wildlife behind high game fences, which makes wildlife management much easier; (2) there is more interest in big game hunting in Spain; (3) small game is increasingly scarce on community lands due to disease outbreaks that have significantly reduced European rabbit populations; and (4) the Hunting Law of 1970 has made it easier for associations to own an estate, lowered taxation on hunting lands, and allowed landowners to prevent trespassing on marked hunting reserves (López Ontiveros and Verdugo 1991; Barbosa et al. 2004; Martínez Garrido 2009).

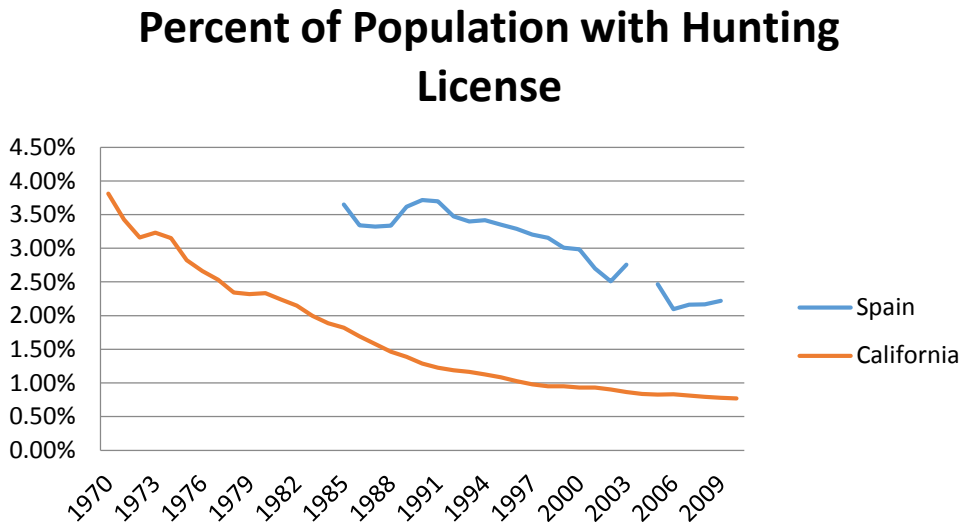
Significant areas of public land (most of which is owned by the U.S. Forest Service) in California are open to free hunting without any reservation or membership. Some public lands require reservations or special tags drawn by lottery to hunt. Hunting operations on private lands in California generally fall into two main types: (1) year-long leases, where hunters pay for access to the property year-round with restrictions on game taken, and (2) day hunts, where hunters pay for access to a property for a given number of days with a specified amount of game to be taken. Although there is a wide array of hunting styles in California, there is no significant cultural and historical distinction between the social hunt and private hunting as in Spain; however, there are social and cultural differences between what one researcher deems ‘‘sport hunters,’’ ‘‘meat hunters,’’ and ‘‘ecologistic hunters,’’ with varying priorities and motivations among the types (Kellert 1980). Small game hunting is often practiced by youth or as a secondary quarry for many hunters in California (Shaw 2008).

1.1.4 Trends in Hunting Participation

A feature shared by hunters in California and Spain is their enthusiasm for the rhythms and associations of hunting. Nonetheless, a demographic shift in hunters appears to be underway, with hunting license sales decreasing in California and Spain. As one California journalist reported in 2003, after discussions with Department of Fish and Wildlife legislative analysts, ‘‘the growing urban populations in California are ‘less supportive of hunting’’’ (Stienstra 2003). Yet the overall populations of California and of Spain are increasing, if only gradually since 2008, so while the total count of licensed hunters decreases, the percentage of hunters with licenses in the population drops still more (Fig. 1.6). Of late the pace of decline in hunter interest has slowed slightly, with increasing sales of tags to hunt wild pigs and bear. In California, a decreasing overall percentage of hunting license holders tracks with a general decline in the state’s most popular game species, the mule deer and black-tailed deer, which are particularly effected by increasing numbers of predators and a decrease in readily available habitat.

Curiously, while urbanized populations may be less familiar with hunting’s traditions, schedules, and attractions, there is rising interest in hunting from a new generation seeking to reconnect with food sources (Marris 2012). In the U.S., the most recent national survey on hunting participation has revealed an increase in the number of hunters of 9 % from 2006 to 2011, which could be indicative of a renewed desire to learn how to hunt, clean, and cook one’s food from sources in the wild (Pollan 2006; Goldstein 2010; Shaw 2011; USDOI and USDOC 2011; Heyser 2012).

Fig. 1.6 In both Spain and California the percentage of population that hunts is decreasing. (CDFG 2012c; MARM 2012)



1.1.5 Land Ownership

The majority of dehesa habitat in Spain is under private ownership and that is not government-owned. The properties where hunting occurs reflect three distinct histories of land ownership: one in aristocratic proprietorship; another with Church, or ecclesiastical, control of property; and finally, an ownership by village communities of land that is hunted. This history has opened up a variety of ways in which hunters and landowners access and manage game species (Gómez Mendoza 1992; López Ontiveros 1993).

Approximately 50 % of California is public land, and much of that is open to free hunting access. However, more than 80 % of oak woodlands in California are privately owned (CDF-FRAP 2003). Hunter success rates on private holdings are generally much higher than on public lands because private lands generally contain higher quality habitat, food, and water resources and because private landowners can regulate hunting pressure (Scott et al. 2001; CDFG 2012a).

1.2 Brief History of Hunting

1.2.1 Spain

As Upper Paleolithic and Magdalenian cave art bears witness, interest in the taking of game in Spain was widespread even in prehistoric and ancient times. Hunters in more recent years derive from two main types, the first of aristocratic background who traditionally sought game on the properties of friends or relations, and the other residents of towns and of communities owning land suitable for hunting (Alagona et al. 2013). Wealthy landowners would hunt big game or small game, maintaining the right to exclude others, and sustaining a privileged

land use with the force of law by employing hired gamekeepers or guards. Hunting by rural residents often focused on small game, taken by snare, shotgun, rifle, or net. Historically, these village hunters—almost exclusively male—hunted for recreation and for food.

During the nineteenth and first half of the twentieth century, hunting in Spain was a popular activity, though one restricted in practice. The divide between the wealthy landowning gentry, who hunted socially with friends, and the rural village residents and poachers who hunted for food or recreation was distinctive and large. This is much-studied for parts of Andalucía, with enduring accounts issuing especially from researchers and historians working in Córdoba (Alagona et al. 2013). Hunting and the montes of Spain are remarkably linked, in topics laced with political and social significance (Gómez Mendoza and López Ontiveros 2001) (Fig. 1.7).

Fig. 1.7 From 1940s issues of Caza y Pesca, a monthly Spanish sports magazine, came two visions of ‘‘hunters’’ in the Spanish countryside: at right, a group of loden-garbed gentry; at left,a poacher, whose work has been discovered by horse-mounted Guardia Civil, the Spanish national police force that was especially vigilant in rural areas. (Photographs composited by P.F. Starrs)



Beyond historical accounts, there is a vast literature on twentieth-century hunting in Spain that extends even into cinema, and unease about invidious distinctions of class and access are a feature of many narratives (Almazán 1934). While residents of villages might hunt community properties, some hunted game illegally, trespassing onto the private land of others. These hunters were furtivos, whose actions were, as the name suggests, furtive in their pursuits. Poaching was not only an act of villager disregard and disdain for upper class norms, it was an act of defiance toward any ruling authority, including the long-lived Franco government, as depicted in literature and films such as Carlos Saura’s La caza (1965), Borau’s Furtivos (1975), Berlanga’s La escopeta nacional (1977), and Brasó’s El mundo de Juan Lobón (1989; TV miniseries). In latter-day Spain, however, fenced properties enclose valuable game animals and exclude rural hunters who once regarded taking game by poaching as an adventurous sport and a

birthright. Little wonder that many properties now managed principally for game hire gamekeepers who reside on the property to protect against illicit access.

Well into the 1970s, two distinct groups participated in a montería. The shooters were one group and the rehaleros or podenqueros, dog fanciers and handlers, made up the other. In today's commercially-organized hunts, however, the rehaleros are itinerant professionals hired for wages by hunt organizers—a marked departure from older montería traditions, and a change spoken of with regret by long-time ''monteros.'' Hunting dog owners until the last 30 years or so were prized and fully-fledged members of the hunting party. Traditionally, demonstrating the prowess of a rehala (the dog pack) earned the rehalero prestige at least equal to any vested in the successful shooter (Gibert Buch 1975).

1.2.2 California

California's history of hunting exploitation began at the arrival of Native Americans approximately 14,000 years ago, with early arrivals practicing subsistence hunting, which continued among some Native American groups into the early 1900s. While Spanish missions marked the first European settlements in California, their impact on wildlife resources was relatively minor compared to the influx of immigrants during the Gold Rush of 1849, when the population ballooned from 15,000 non-Native Americans to hundreds of thousands (Starr and Orsi 2000).

The immense wildlife resources present at the time of European arrival made it seem impossible that hunting could be a threat to their sustainability, but with settlers increasing in numbers, market hunting, and improved firearm technology many species began to decline and reach extinction levels. Wildlife were harvested as a source of food without any limit or regulation (CDFG 2002). Grizzly bears and wolves were extirpated from California in 1922 and 1924, respectively, and once-immense numbers of tule elk, a California subspecies of Cervus canadensis, were reduced to a herd of 28 animals protected by a single rancher near Bakersfield (McCullough et al. 1996). Populations of many wildlife species declined precipitously in the late 1800s, some due to overhunting, others because of habitat loss and human encroachment. Federal and state regulations were enacted in the early 1900s to create hunting seasons and bag limits, with the support of sportsmen hunters, and against the protests of some market hunters. These regulations have led to the recovery of many game species and are the foundation of the current system of state regulation of wildlife in North America—known as the North American Model of Wildlife Conservation (Tober 1981; Geist 1988) (Fig. 1.8).

Fig. 1.8 Animals taken, whether in Spain and California, are always an object of hunter pride, as here at a property in California. (Photograph from the collection of L.T. Macaulay)



Table 1.4 Wildlife practices in Spain and California

Wildlife practice	California	Spain
Shrub management	X	X
Predator control	X	X
Food planting	X	X
Wildlife fencing (2 m+)		X
Sale of meat from hunting		X
Supplemental feeding		X
Restocking and transporting native big game		X
Both sex harvest on native deer		X
Landowners set bag limit for big game		X
Hunting over bait		X
Hunting at night		X
Driving wildlife with dogs		X

X denotes commonly and legally practiced

1.3 Economics of Hunting

1.3.1 Governance, Property Rights, and Wildlife Management

Although a number of Spanish and Mexican legal principles were borrowed in the early state history of California, divergent legal histories governing wildlife since colonial times have resulted in distinct property rights over wildlife that have led to readily apparent differences in the wildlife management regimes of each place (Table 1.4). In Spain, landowners have much greater ability to manipulate and control the movement and management of wildlife, while in California regulations prohibit landowners from fencing, relocating, and feeding wildlife. If wildlife is nominally considered res nullius[1] in both locations, Spanish landowners have much greater rights to control and prevent the escape of game animals, capturing de facto ownership, even if not de jure ownership. Although game species are managed in the public interest by the state and federal government in California, private landowners do control access to property where the game resides, and can charge a fee for hunters to hunt on their property.

In the United States there have been three major historical influences on the development of wildlife governance (Huffman and Wallace 2011). The first was rebellion against oppressive English policies, including the restriction of hunting to those with sufficient wealth or status. The second was the need of colonists to hunt for food and clothing. The third was the belief that given the huge expanse of America and abundant wildlife, restrictions on hunting would hinder economic growth. Eventually, laws and policies developed granting states and the federal government some aspects of managing wildlife in the public interest. There remains a strong sentiment among hunters in the United States that hunting is a time to experience nature, share adventure with friends and family, and to acquire food, and that it should be open to all regardless of wealth or status (IAFWA

2002). There is a tension between this widespread view of the role of hunting and the practice of trophy hunting for high fees, which is often seen as the domain of wealthier individuals. While there is historic opposition to paying for access to hunting opportunities, in California the deer herd has been in decline for the past several decades and the success rate of public land hunting has dropped significantly. As a result, individuals may stop hunting, or become willing to pay for access to higher quality hunting opportunities that occur on private lands.

In many rural areas, it was once and to some extent still is a common courtesy to allow friends and community members, or even polite visitors, to hunt on one's land for free or in exchange for other favors. One study found that even in 2004 nearly two-thirds of ranches on oak woodlands had hunting on their property, but fewer than ten percent charged fees for hunting access (Huntsinger et al. 1997).

The most visible manifestation of the differences between the two countries in property rights over wildlife is the enclosure of game species behind *2 m high game-fencing (Spanish: malla or valla cinegética), a common practice in Spain but illegal in California.

An extensive literature discusses the intricacies of property rights of fugitive resources, which Ostrom (1990) terms ‘‘common-pool resources’’ (Ostrom 1990; Lueck 1995). An essential quandary for these resources is that they are often jointly produced by more than one person's property, yet these resources, when harvested, are generally not jointly used. For

[1] Res nullius is a Latin term borrowed from Roman law. It holds that when an object (which conspicuously includes wildlife) is unowned by any particular person or entity it is ownerless property and free to be owned by anyone. When such an object is found, however, it may in some circumstances belong to the first person who takes it (as in beachcombing).

example, deer may live on, reproduce on, and utilize the habitat on several properties that are owned by different individuals, but when a deer is hunted and harvested, it is usually harvested and consumed by one person and is no longer available for someone else to harvest.

Fig. 1.9 The preparation, cooking and consumption of meat after a hunt in California is an important part of the California hunting experience: here a wild boar is barbecued. (Photograph from the collection of R. Barrett)



The lack of game-fencing in California and in most areas of Spain means game animals are a fugitive resource because they can easily move off a landowner's property and are a type of common-pool resource. Organizing the harvest of common-pool resources is ''usually an uncertain and complex undertaking'' and if the right conditions are not present, many of these resources may be utilized in a way that reduces the productivity and quality of the resource (Ostrom 1990). The limitations of institutional management and challenges to cooperative efforts such as the presence of free riders can result in difficulties in managing the fugitive resource. For example, male deer in many areas of California are often harvested at two years of age—the first opportunity that they may be legally harvested—due to the possibility that one's neighbor may harvest it first. As such, deer in California and in unfenced areas of Spain often exhibit an age structure that is skewed to younger males and very low proportion of male animals, which can lead to diminished overall harvest and biological problems (Ostrom 1990; Clark 2010; McCullough 2001; CDFG 2002; Milner et al. 2007; Pérez-Gonzalez and Carranza 2009). Game-fencing in Spain, while a costly investment, allows those landowners in Spain to control the fugitive wildlife resource and thereby avoid having to cooperate with others in management and

harvest the resource—though not without creating distinct management problems and biological quandaries (Díaz et al. 2009).

Other important property rights differences between California and Spain is the ability of Spanish landowners to provide supplemental feed to wildlife, set bag limits on big game, and sell live game animals to other landowners. Feeding allows landowners to maintain larger game populations, and owners of Spanish hunting dehesa properties can retain meat from the animals taken in a hunt, which yields an additional income source. The situation is significantly different in California, where wildlife may not be fed, sold (dead or alive), or transported, and bag limits are largely state-controlled. While hunters may self-consume or donate meat from the hunt, they may not sell it on the open market. Of note, California law requires possession of the meat, prohibiting hunters from allowing an animal's meat to go ''needlessly to waste'' (CFGC 2012) (Fig. 1.9).

1.3.2 Commercialization

Hunting in California is far less commercialized than in Spain due to four main factors: (1) cultural opposition to charging fees for hunting, (2) fewer hunters in California as a percentage of the population (Fig. 1.6); (3) the extensive public lands that are open to free hunting, and (4) the vastly different regulatory structures and property rights regimes that govern wildlife. The more state-oriented property rights regime in California and the rest of the U.S. developed in large part as a reaction to overexploitation of wildlife by profit-motivated market hunting (Stine

1980; Tober 1981). As such, there is deep-seated suspicion and long-standing opposition to programs that would allow landowners to profit from wildlife (Fitzhugh 1989).

Greater freedoms to manipulate and control wildlife in Spain makes for a more intensively managed and significantly commercialized hunting industry. A significant shift has occurred in the role of hunting in Spain from 1970 to today. Before 1970, hunting was in the main a leisure activity for socializing, and lacked an overt focus on generating cash income, whether in the realm of large landholders or village hunting associations of small means. The Hunting Law of 1970 modernized the previous Hunting Law from 1902, and made possible the expansion of commercialized hunting and intensified management as discussed above (Structural Organization of Hunting Operations). Since the incorporation of the law in 1970, hunting increasingly is managed as a for-profit commercialized business, because hunting resource rent can surpass returns from grazing or field crops (López Ontiveros and Verdugo 1991; Vargas et al. 1995; Martínez Garrido

2009). While this increases motivations to maintain better habitat for game species, increased fencing and feeding in Spain has yielded high ungulate densities on hunting properties and can lead to deleterious environmental effects as discussed below (Olea and San Miguel-Ayanz 2006; Díaz et al. 2009).

California has developed a pair of programs to help landowners earn income from recreational hunting of wildlife on their land in exchange for habitat improvement or expanded public access. The Private Lands Management program in California has a contentious history but mimics a Texas program that provides landowners greater harvest flexibility over game

species in exchange for improving habitat. Landowners must submit a lengthy wildlife management plan to the California Department of Fish and Wildlife and must obey requirements such as informing neighbors of their intent to enroll in the program. The SHARE program provides private landowners with a liability waiver and a cash payment in exchange for allowing public access to a property for hunting or other forms of outdoor recreation (CDFG 2012d).

1.3.3 Costs, Revenue, and Net Profit

Costs and revenues are known to be highly variable between woodland properties and from year to year (Loomis and Fitzhugh 1989; Peiró and Seva 1996). This chapter draws on data from a 1987 survey of 55 ranches with hunting in California and from three case studies in Spain on properties of 658, 4000, and 7000 ha (Loomis and Fitzhugh 1989; Campos et al. 1995; Lenzano and Zamora 1999). Cost and revenue data for hunting properties is relatively limited, but new economic research on hunting is underway in each location to better understand profitability in this industry (Macaulay in prep, RECAMAN in Oviedo et al. 2013). In general the economic activity, in terms of average costs, revenues, and profit per hectare, is several times greater in Spain than in California. All cost and revenue figures from this section have been converted to 2011 US dollars.

1.3.3.1 Costs

Tables 1.5 and 1.6 illustrate a sample of the differences in costs between a case study of a fenced Spanish hunting property and an average of a subset of 41 ranches in California with [10 % oak cover. These tables should only be compared generally and with caution as different accounting methods were used.

Personnel wages and vegetation management expenses account for a significant portion of total costs on fenced Spanish properties, while administrative and infrastructure costs are significantly higher in California. Economic data from unfenced properties in Spain is unavailable, but would presumably be considerably lower, especially in personnel costs, as there is less need for hired hands to census and cull the red deer population and less need of a guard to protect game animals, as there are fewer valuable trophies present (Campos et al. 1995) (Tables 1.5 and 1.6).

1.3.3.2 Revenue

Lenzano and Zamora (1999) estimate income from the sale of 76 positions at about $700 per person on an unfenced montería hunt, yielding $53,200 in revenue for a red deer montería.

Added income of $5,000 can come from on-site wild boar, with $5,000 from the sale of venison and boar meat after the hunt (Lenzano and Zamora 1999). This leads to a total revenue stream of $63,200 resulting in approximately $96/ha in revenue (income before costs). The value of a montería position can vary from $500 to $6,000, with most montería positions costing $800–1,200 (Campos et al. 1995). The sale of live deer for restocking dehesa properties can offer an additional source of income. If the value of live animals exceeds the meat value, landowners may find it more profitable to capture and sell live animals instead of culling their deer herd (Vargas et al. 1995).

Table 1.5 Costs of managing a hunting property in Spain. The Lenzano and Zamora case study of a 658 ha dehesa is one of the few in Spain that includes specific costs of operating a hunting estate, summarized in the table below; dollars normalized to 2011 (Lenzano and Zamora 1999; BLS 2012)

Costs	2011 dollars/ha	Percent of total (%)
Guard	26.5	50.3
Dog packs—Rehalas	6.5	12.4
Bidders	1.4	2.8
Food	1.4	2.6
Horses/Mules/Oxen	1.3	2.5
Vehicles	1.0	1.9
Veterinarian	0.9	1.7
Guides	0.6	1.1
Personnel cost subtotal	39.6	75.3
Brush clearing	8.3	15.7
Pruning trees	0.4	0.7
Enhancing oak stands	0.2	0.3
Vegetation management cost subtotal	8.8	16.7
Housing (each year)	1.4	2.8
Roads	0.8	1.4
Ponds	0.5	1.0
Water wells (each year)	0.3	0.5
Fences (each year)	0.2	0.3
Infrastructure costs subtotal	3.2	6.1
Taxes on Luxuries	0.6	1.1
Renewal of Registration	0.4	0.8
Montería Hunting Permit	0.0	0.1
Administrative costs subtotal	1.0	1.9
Total Costs/ha	52.6	100.0

In California, research has shown that hunter success rate, percentage of trophies, and the wealth of hunting participants added to revenues (Loomis and Fitzhugh 1989). Additional

analysis of the data shows that proportion of a ranch with oak habitat significantly improved revenue in California hunting operations. Ranches in the study on average earned $19.54/ha, while those with 10 % or more oak woodland cover received $21.09, and those with 15 % cover or more earned $22.47/ha on average (calculated from data in Standiford 1989).

Table 1.6 Average costs of 41 Ranches in California with hunting and C 10 % oak woodland cover [adapted from data from (Loomis and Fitzhugh 1989; Standiford 1989)]

Cost	2011 dollars/ha (BLS)	Percent of total (%)
Non-professional wages: cooks, operators, etc.	1.5	22.7
Guides	0.7	9.7
Accountants	0.1	1.9
Attorney and consultants	0.1	1.5
Personnel wages subtotal	2.4	35.8
Vegetation management	0.1	2.0
Vegetation management cost subtotal	0.1	2.0
Vehicles: maintenance	0.3	4.5
Vehicles: depreciation	0.3	4.5
Irrigation water	0.2	2.9
Improvements (roads fences and water): maintenance	0.2	2.6
Phone	0.2	2.4
Road construction	0.1	2.0
Ranch equipment: depreciation	0.1	1.8
Other annual costs	0.1	1.7
Ranch equipment: maintenance	0.1	1.2
Supplemental feeding	0.1	1.0
Improvements (roads fences and water): depreciation	0.1	1.0
Predator control	0.0	0.7
Other costs (gas, electric, structure maintenance)	0.4	5.3
Infrastructure/Maintenance costs subtotal	2.3	32.3
Liability insurance	1.0	15.0
Lease of hunting rights on other lands	0.5	7.0
Advertising	0.5	6.9
License, permit, and legal fees	0.1	1.7
Administrative costs subtotal	2.1	30.5
Total costs	6.8	100

1.3.3.3 Net Profit

A significant issue to consider is how costs factor into final profitability calculations. Some costs, as with water provisioning, are undertaken for livestock, yet benefit wildlife. Comparisons of net profit among hunting operations is difficult. While accounting differences make a detailed comparison impossible, the profitability of Spanish dehesa hunting properties appears significantly higher than an equivalent California site. Spanish studies show net profit (owner hunting net operating margin) of $23.80–72.47/ha (Campos Palacín et al. 2001), while other studies tally net losses when costs such as capital depreciation are incorporated. Lenzano and Zamora (1999) suggested that by using a third party intermediary a landowner could receive a return of $6/ha, but under self-management, profitability increased to $37/ha in an unfenced property, with increases up to $42–63/ha for fenced properties while amortizing fencing investments. Net profit in California is estimated at $5.68–7.31/ha. There is significant variation in profitability levels, with 30 out of 41 ranches operating profitably, and 11 ranches taking losses (calculated from data in Standiford 1989).

While data suggests Spanish dehesas may earn 10 times as much from hunting than California ranches, these figures should be compared with caution given differences in accounting for costs and revenues (Standiford and Howitt 1993).

1.4 Wildlife Management

Wildlife management can generally be divided into two major types:

1. Top-down population regulation: actions taken by humans that directly impact the movement and population composition of game species and predators, such as harvest, culling, fencing, predator control and restocking; and

2. Bottom-up management practices: actions taken by humans that influence or improve resources needed by wildlife. These are generally improvements to food, cover, and water resources for the benefit of wildlife species.

1.4.1 Top-Down Population Regulation: Harvest, Fencing, Restocking, and Predator Control

1.4.1.1 Harvest Methods and Regulations

A major component of any wildlife management program is managing the harvest of animals by hunting. Harvest rates are impacted by the method of hunting, bag limits, and the age and sex of animals harvested. Distinct regulations in Spain and California govern each of these aspects of hunting. California's hunting methods are far more restrictive (Table 1.7).

Table 1.7 Hunting methods in Spain and California

Hunting method	Spain	California
Big game		
Spot and stalk, i.e. rececho	X	X
Stand hunting, i.e. aguardo, espera	X	X
Montería	X	
Gancho	X	
Hunting over bait	X	
Hunting at night	X	
Small game		
Opportunistic, i.e. la caza al salto	X	X
Walking lines, i.e. ojeo	X	X
Driven shooting	X	
Live decoy hunting	X	
Use of sighthounds (galgos)	X	

Big Game

Common to Spain and California are the spot and stalk (*rececho*) and stand-hunting methods. In spot and stalk hunting, hunters use binoculars to spot game animals from a vantage point where a large area of terrain can be surveyed. They then stalk the animal to get close enough to take a shot. In Spain, *rececho* is the most expensive hunting method for a hunter to undertake, and is usually used to harvest a single trophy animal. In California it is one of the primary ways to hunt, and generally costs no more than hunting from a blind or stand. Stand hunting involves placing hunters in a concealed or elevated position (the stand) where they wait for an animal to approach. Stand hunting (*aguardo* or *espera*) is often done over bait in Spain, may be performed at night, and is generally used for selective wild boar hunting. In California, stand hunting is used by hunters who wait in stations above locations frequented by game, or they may wait over grain fields, since hunting over bait is illegal.

Unique to Spain is the *montería*, where 50–75 hunters take up shooting positions in fixed locations on a property for a single day of the year. The *rehaleros* or beaters move into the lower reaches of the property with one or more *rehalas* or *recovas*, a group of 20–25 hounds, and chase animals from hiding in the often dense brush and into the open where hunters shoot the game, often on the move. The *gancho* is a small-scale *montería*, usually focused on wild boar. In California dogs can be used to hunt non-native wild pigs, but the method usually involves following dogs to the game rather than dogs driving animals to hunters waiting in designated locations.

Small Game

Small game methods of hunting are generally similar in Spain and California, although Spanish hunters may use packs of dogs, as in big game hunting, to drive small game toward hunters (Vargas et al. 2006). With rabbits, hunters can utilize methods of "*la caza al salto*," which roughly translates as "hunting on the jump," where a solitary hunter with or without dogs

walks an area and flushes rabbits. Additionally, rabbit hunting can be performed with sighthounds that rely on vision and speed to hunt. Spanish hunters usually use the galgo, a breed similar in appearance to the greyhound (Figs. 1.10 (at left) and 1.11).

Fig. 1.10 Dogs are an essential element in the drive hunt, or montería, of Spain. (Photograph by P. F. Starrs)



Fig. 1.11 Dogs await transport and release with eager anticipation before a hunt in the Sierra Morena. (Photograph by L. T. Macaulay)



California considers rabbits, jackrabbits, and squirrels as small game, and generally classifies game birds including quail, dove, and wild turkey as upland game. In California, small game mammal hunting is practiced with and without dogs, but lacks the widespread popularity and cultural significance of hunting rabbits and hares in Spain. Blogger Hank Shaw describes a prevalent attitude toward jackrabbits: ‘‘Most ‘‘normal’’ hunters wouldn’t waste a shell on hares…. [T]hese folks view jackrabbits as beneath them’’ (Shaw 2008). Small game mammal hunting is generally practiced by youth or opportunistically as a secondary quarry, when hunting for other game.

Upland game bird hunting of quail and turkey generates more hunter interest than small mammal hunting (Table 1.3). Live decoys are illegal in upland game bird hunting in California, although artificial decoys are utilized, particularly for turkey hunting. Quail in California are generally hunted by individuals or small groups of hunters. The practice of walking lines is less productive in California quail hunting, given the prey’s preference for thicker habitat cover and the pattern in which they flush. Hunters will use dogs to locate and retrieve birds, but rarely to drive the animals. Smaller bag limits for wild turkey result in lower harvest numbers, which belie the significant interest in wild turkey hunting in California, particularly during the spring mating season, when male turkeys are lured within shotgun range by mimicking the call of a female hen.

Bag Limits and Antlerless Harvest

California’s Department of Fish and Wildlife generally imposes a bag limit of two deer, and various limits on small game. Because wild pigs are nonnative in California and have high fecundity, there is neither a closed season nor bag limit. In Spain, each regional government, or autonomía, sets its hunting regulations. Generally, there is no limit on the big game species, but small game species have various limits.

While Spanish landowners generally control the age and sex of harvest on big game on their properties, the California Department of Fish and Wildlife retains control over bag limits and the age and sex of harvests. Due to political controversy dating back to the state’s first antlerless harvest in 1956, regulatory authorities prohibit female and juvenile male harvest of deer in most parts of the state (Fitzhugh 1989). This has led to high hunting pressure on male deer and sex ratios that, while variable, are oftentimes far less than 0.5 bucks-to-doe ratio, with some estimates as low as 0.05 bucks per doe (CDFG 2012e). Despite research that shows density-dependent responses in California deer and increased buck harvests as a result of removing female deer, political opposition persists, and today the California Department of Fish and Wildlife (prior to 2013, the California Department of Fish and Game, or CDFG) describes California as having the “dubious distinction” of being the only state in the United States where doe hunts cannot be carried out even when such hunts are biologically justified (McCullough 2001; CDFG 2002).

A similar tradition of not harvesting female red deer historically existed in Spain. This was rooted in a belief that hunting females would drive populations to local extinction, which was occurring just after the Civil War in the late 1930s and 1940s. However, after the hunting law in 1970 with populations increasing for most Spanish big game species, management policies embraced the need to regulate population size and the ratio of males to females to

improve game animal condition, increase the value of trophies, and reduce the impact on vegetation and the risk of traffic collisions (Carranza 2010).

1.4.1.2 Fencing

The most significant wildlife management consideration on Spanish hunting lands is whether a property is fenced. On unfenced (open) properties, traditional agricultural practices and livestock are usually the main productive activities and big game hunting is of secondary importance (Vargas et al. 1995). Given a propensity of wildlife to stray onto adjacent properties, unfenced lands generally have low intensity management, since only very large properties or those in cooperative agreements with neighbors can reap the benefits of targeted harvest and habitat improvement strategies. These smaller unfenced properties tend to manage for a greater number of wildlife than for high quality trophies. Unfenced hunting lands have been documented to have a 0.25 male-to-female sex ratio, and an age structure where over 45 % of the males are juveniles (Pérez-González and Carranza 2009; Torres-Porras et al. 2009).

Fig. 1.12 Game fences represent a significant act of landowner control over a dehesa property. Fences allow game species containment and increased profits by intensified management. The problems associated with fenced properties made this particularly controversial when game fences first arrived, but the fences are now very common. (Photograph by P. F. Starrs)



Fenced properties require significantly different management practices than open properties because the wildlife cannot escape the property, allowing managers to more accurately census the population of game species and control the harvest. This includes knowing the sex ratio and age structure of the population, which allows a landowner to practice much more effective population management through selective hunting, allowing for more stags to mature to trophy quality and increasing the overall number of valuable males in the population.

Although fenced properties may entail significant benefits for a landowner in terms of quality and quantity of trophies, and can result in a wildlife population with more natural characteristics and age structure, their management requires resources to maintain healthy wildlife populations and habitats. This is particularly true in the Spanish Mediterranean ecosystem known for high variability in rainfall and in resource production. A landowner must maintain the fence itself, ensure a water supply within the fence, and provide adequate forage for

the animals within the property to prevent excessive grazing and habitat degradation in times of drought. Many fenced properties cull to reduce population size, particularly of females and of smaller, non-trophy males to counteract selective pressures of trophy hunting for large males (Martínez et al. 2005; Mysterud and Bischof 2010).

Dehesa properties may also need to provide supplemental summer feed, especially in times of drought, to maintain a wildlife population that cannot migrate off the property to find food. Furthermore, gamekeepers and guards are required to police a property and prevent poaching of valuable trophy animals.

As with other aspects of Spanish hunting management, fencing regulations are controlled by regional governments. Most set a minimum size and enforce permeability restrictions on fencing. In southwestern Spain a big game hunting operation has a minimum area requirement of 700 ha—also the minimum size for a fenced estate. The tendency, however, is to enclose larger areas. Experts in Andalucía recommend that properties smaller than 2,000 ha obtain a certificate of quality as a prerequisite for fencing (Carranza and Vargas 2007; AAMAA 2008). Fences must have regular openings to allow the movement of non-game species such as the protected Iberian lynx (Lynx pardina). Fences retain deer species better than wild boar, which easily burrow under a fence. Enclosures for wild boar (cercones) are in general prohibited for their association with unnatural, high-density management, which is linked to an increase of diseases. However, these do occur and their numbers are increasing (Fig. 1.12).

1.4.1.3 Restocking and Farmed Game Species

Restocking and farming of game species plays a major role in both small and big game hunting in Spain (Barbosa et al. 2004), but in California the practice is utilized almost exclusively with exotic game birds, notably the ring-necked pheasant. In Spain, red deer, red-legged partridges, rabbits, roe deer and wild boar may be restocked, with red-legged partridges the most common among farmed game species. Vargas et al. (2006) estimated that one-third of red-legged partridges harvested in Andalucía were farmed birds, usually released into the field prior to a hunt. For other species, restocking is less common, but can involve animals from game farms, or from live captures in other areas, which are generally used to re-establish or reinforce natural populations.

For stocking game species in Spain, a permit is needed to move game animals and the approval depends on regional policy. Generally, only native species or subspecies can be transported, although enforcement of this regulation varies. There is no current regulation against the release of animals from game farms, provided that they belong to native varieties of the region where they are going to be released, although effective enforcement is not common. There is a growing tendency toward limiting import, export, transfer, or release of live game.

In California, three-quarters of exotic ring-necked pheasant harvested are from stocked game bird farms (CDFG). For native big game species, restocking is illegal unless undertaken by California Department of Fish and Wildlife to reestablish game species (CFGC). The state does license a number of game bird farms that exclusively utilize farmed birds.

1.4.1.4 Predator Control

California permits predator control on a number of species. Steel-jaw leg traps and poison have been banned, but box traps, padded leg traps, and snares are legal subject to their being checked every 24 h. Mountain lions (Puma concolor) are legally protected by a statewide voter referendum in 1979, and predator control of these populations is illegal, except when lions are killed for public safety concerns or with depredation permits granted by the state. Many landowners and hunters blame this law for a decline in deer populations in California, although research suggests that a combination of factors, including habitat change, is affecting deer populations (Longhurst et al. 1976; Kucera and Mayer 1999). Bobcats (Lynx rufus) can be harvested with the purchase of a special tag and a hunting license. There is no legal season on red foxes (Vulpes vulpes). Gray foxes (Urocyon cinereoargenteus) are considered a furbearer species with a separate season (Fig. 1.13).

In Spain, landowners practice predator control most intensively for predators of small game such as the red fox and the common magpie (Pica pica). While there is intensive effort to control red fox populations, studies have found that red fox populations are rarely controlled by these measures, and that oftentimes other non-target predator species may be most greatly impacted (Virgós and Travaini 2005).

Fig. 1.13 Not all hunters are older, or male. This wild boar eventually gained the young huntress an award for taking a silver medal specimen, in a hunt north of Córdoba. It was not her first hunt. (Photograph by P. F. Starrs)



The Iberian wolf (Canis lupus signatus) is the only predator that can significantly impact red deer populations. Wolf populations are increasing in central and northern Spain, while in southern Spain they appear to be declining. In Extremadura, wolves disappeared in the early 1990s (Rico et al. 2000), and in Andalucía there is only a small breeding population in the Sierra Morena. Authors describe an antagonistic relationship of rural landowners with wolves and some illegal killing may occur, but disease and traffic fatalities likely account for more deaths than illegal hunting or predator control (Rico et al. 2000). Ultimately, the wolf's range is so limited today that wolves are rarely encountered on dehesas (Blanco and Cortés 2002).

1.4.2 Bottom-Up Practices: Shrub Management, Food Plots, Supplemental Feed, and Water

In California and Spain, landowners practice a variety of habitat management actions to improve the food, cover and water needs for game species, which we will refer to as bottom-up management practices. In both California and Spanish oak woodlands this generally involves clearing understory brush, planting food plots, and providing water sources. In Spain, supplemental feeding is common (Fig. 1.14).

Fig. 1.14 Much of the dehesa is given to occupation by brush, if not regularly cleared of its aromatic understory vegetation. Sometimes, however, to give access to hunters, strips (or manchas) will be cut into the understory, as in this aerial photograph near Hornachuelos, Córdoba. Oaks were carefully left behind, in keeping with legal strictures. (Photograph by P. F. Starrs)



1.4.2.1 Shrub Management

While the level of shrub, or brush, presence and growth in a given area is dependent on a multitude of factors such as grazing, soil characteristics, topography, fire history, and climate, many landowners will clear brush to improve herbaceous production or to stimulate fresh and more palatable brush growth. While some brush species provide important cover and forage for ungulates and other wildlife, others provide little nutritional value for game species and can reduce the overall forage production for game and livestock. California deer prefer foraging on brush during the spring when new growth appears (Evans et al. 1976). Palatable species include manzanita (Arctostaphylos spp.), chamise (Adenostoma fasciculatum), ceanothus (Ceanothus spp.), and oaks (Quercus spp.) (Sampson 1963). A less palatable species rarely eaten by deer in California is coyote brush (Baccharis spp.). Palatable brush species in Spain include species of phyllirea (Phyllirea spp.), madrone (Arbutus spp.) and mytle (Myrtus spp.), while less palatable species include rockrose (Cistus spp.), heather (Erica spp.), and mastic (Pistacia lentiscus) (Rodríguez Berrocal 1993; Bugalho and Milne 2003). Many of these brush species are highly resilient and persistent in oak woodlands, and often return at various time intervals after clearing. Rockrose, Jara pringosa (Cistus ladanifer), will return aggressively 5–8 years after it is cleared. Even brush species that provide important forage for wildlife grow into tall thickets beyond the reach of deer, so some managers clear brush to encourage herbaceous production and new brush growth (Sampson 1963). After clearing, California landowners may retain brush piles as habitat, in particular for quail and a variety of other game and nongame wildlife (Gorenzel et al. 1995). In Spain, brush piles may shelter partridges and rabbits, but creating brush piles is not a widespread practice.

In California, ranchers will sometimes use tracked vehicles to drag a large chain with 60-pound links across brush areas to remove brush and enhance habitat for deer. Brush-clearing costs can be substantial. One study in Andalucía found that 17 % of expenses on a hunting property were devoted to clearing brush (Lenzano and Zamora 1999), while marginal expenses were devoted to pruning and enhancing oak cover. Discing every 4–6 years is the most commonly used practice to limit brush encroachment with hand-grubbing of brush and grazing utilized to reduce the frequency of follow-up discing treatments (Huntsinger et al. 1991).

Traditional management of the dehesa involves clearing all brush species, leaving older oaks interspersed with herbaceous vegetation. While this improves livestock feed and some agricultural returns, recent research suggests that maintaining a mosaic of brush species in dehesa habitat is ideal for game species (Carranza 2010). Leaving a mosaic of brush in the dehesa contributes to natural regeneration of oak trees (Chaps. 5, 8), adds to biodiversity and provides an increasingly important component of the diet of big game species during the summer drought period (Bugalho and Milne 2003; Plieninger et al. 2003; Díaz 2009; Carranza 2010).

While controlled burning can produce similar benefits, it is rarely used in Spain because escaped fires could harm valuable cork oak bark, and because arson historically was a means of expressing dissent with political authority by vandalizing the property of landowners who fence land (Huntsinger et al. 1991). In California prescribed burns are made difficult by air pollution regulations, which limit the days that a rancher can conduct a burn, and it is logistically difficult to coordinate fire-fighting personnel on short notice to reduce the liability risk of the fire escaping a property.

1.4.2.2 Food Plots

Ranchers in both Spain and California plant improved pastures for wildlife. In some places in California, this includes planting fields of barley or other grains to attract wild pigs. In Spain, land managers will sometimes sow fields to supplement the dietary needs of wildlife on the property, especially in closed or fenced properties.

1.4.2.3 Supplmental Feeding

Direct feeding of wildlife is generally illegal in California, although difficult to enforce. Several decades ago in Spain, supplemental feeding was a practice seen as beneficial to game and wildlife. Today, ecological knowledge and management policy generally discourages interfering with natural processes, although feeding threatened species is often justified and practiced. Recent decreases in population sizes of rabbits, for example, has resulted in supplementary feeding, although feeding to promote concentration of animals for hunting purposes is discouraged. For big game, supplementary food and making salt available is only allowed during extreme drought months in late summer. Many private land managers nonetheless provide food without control by authorities. For example, a wildlife study in Spain describes 23 t of feed being placed out for pig hunting on a 920 ha property (Braga et al. 2010).

1.4.2.4 Water Provisioning

Spanish and California landowners provide wildlife with supplemental water sources. While this historically was to serve livestock, wildlife species benefit. Water sources can range from stock ponds that collect and store surface water runoff to troughs that are filled with groundwater pumped from aquifers. Ranchers in California equip ‘‘wildlife friendly’’ water troughs with ramps to allow the escape of birds, bats, and other animals that may fall into the trough. Water troughs may be inset at near ground level for ease of access of game species. Ranchers in drier areas of California’s oak woodlands may construct ‘‘guzzlers’’ designed to provide water for small or big game species (Bleich et al. 2005). In Spain, the most common and traditional practice is to build stock ponds that fill with water pumped from a shallow aquifer or from surface runoff. These are designed to maintain water storage until the end of the summer dry period.

1.5 Environmental Effects

There are many positive environmental effects from management for hunting. Most notably, wildlife habitat is conserved in a relatively undisturbed state, especially when compared to conversion of oak woodlands to intensive agriculture or residential development. Because habitat loss is the greatest threat to biodiversity worldwide, this conservation of open areas provides critical habitat for many species in biodiversity hotspots (Brooks et al. 2002). Standiford and Howitt (1992, 1993) have demonstrated that incorporation of hunting revenue provides an incentive to retain oaks on woodland properties. The economic return from hunting ventures reduces the economic incentive to use the oak woodlands in less sustainable ways, resulting in greater habitat conservation and increased provisioning of ecosystem services including the following:

- Habitat conservation
- Reduce opportunity cost for habitat conservation
- Nutrient cycling
- Provisioning of food (game, livestock, crops)
- Water filtration
- Air filtration
- Carbon sequestration
- Aesthetic value
- Recreational experiences
- Cultural significance
- Spiritual inspiration

Table 1.8 A generalized overview of the complex mix of environmental effects of wildlife management actions

Environmental Effect	Wildlife Management Action
Animal Density	Fencing
Overgrazing and Vegetation Impacts	Feeding
Disease Impacts	Game Farming/Translocations/Harvest
Fragmentation	Fencing
Genetic Impacts	Fencing
Domestication	Feeding
Changed mating behaviors	Harvest
Hybridization	Game Farming/Translocations
Inbreeding	
Genetic drift	
Artificial selection and breeding	
Loss of local genetic adaptations by introgression	
Population Structure	Harvest
Age structure	
Sex ratios	
Altered dispersal patterns	
Erosion and water quality	Fencing
	Feeding
	Brush management
Invasive species	Brush management
	Discing
	Food plots
	Game farming/Translocations
Impacts on protected or non-target species	Predator control

Management actions for hunting have particular impacts on the environment, and our focus in this section will evaluate those effects. The impacts can be described by two mechanisms: (1) the effects of people on an environment through wildlife management actions, and (2) the effects of game animals on their habitat, which is largely determined by the species present and animal density. These are inextricably linked and will be discussed in an integrated way.

A brief and generalized overview of environmental impacts of management actions is classed by management action and environmental effect in Table 1.8. Much of the discussion will focus on environmental effects in Spain due to the greater range of wildlife management practices legally available.

1.5.1 Animal Density: Overgrazing and Disease/Parasite Concerns

Management actions of fencing, feeding, game farming/restocking, under-harvest and excessive predator control can lead to high animal densities and increased risks of vegetation degradation and disease outbreak. Overgrazing can result in vegetation community simplification with a loss of palatable species, detrimental effects on biodiversity and ecosystem function, and erosion (Murden and Risenhoover 1993; Mysterud 2006; Acevedo et al. 2008). Risks of disease and parasitic outbreaks also increase with high animal density (Andrews 2002; Vicente et al. 2007; Castillo et al. 2011)

Disease emergence or increased disease risk is a frequent consequence of ungulate overabundance (Acevedo et al. 2008). Several studies show a prevalence of tuberculosis-like lesions in southern Spain, which is noted for its intensively managed and fenced hunting properties (Vicente et al. 2007). Recent research increasingly posits that although overabundance plays a significant role in disease and parasite transmission, management practices that promote the clumping of animals, even of different species, at water sites or supplementary feeding sites can exacerbate disease even at low animal densities (Vicente et al. 2007; Castillo et al. 2011).

1.5.2 Erosion and Water Quality

Erosion is a concern on oak woodlands because the loss of fertile topsoil can reduce the vegetative capacity of a property. High animal density resulting from fencing or feeding can lead to over-grazing, and loss of many plants that anchor soils into the ground increasing erosion risks and reducing water quality (Kauffman and Krueger 1984). Brush clearing practices can leave large areas of land without vegetative cover, increasing erosion risks and impacting water quality in riparian zones (Sampson 1963).

1.5.3 Fragmentation

Fences increase fragmentation of a landscape. While fences generally are designed only to prevent the movement of certain game species, they can make migration of other species more difficult and produce unwanted genetic impacts by separating animal populations. Furthermore, fences can increase predation at crossing points, increase mortality of birds and other animals that get caught in the fence, and can lead to animals exceeding carrying capacity of a landscape (Andrews 2002; Hayward and Kerley 2009).

1.5.4 Genetic Impacts of Fencing, Feeding, Harvest, and Restocking

Genetic impacts that result from management activities can include the domestication of wild species, changed mating behaviors, inbreeding, genetic drift, hybridization, artificial selection, and loss of local genetic adaptations.

Fencing isolates populations, encouraging both genetic drift and inbreeding. For red deer populations in southwestern Spain, fences encourage genetic differentiation between neighboring properties, accounting for some level of inbreeding (Martinez et al. 2002). But the full role of fences on inbreeding is complex. In the dehesa, fenced lands are some of the only areas that sustain a natural age structure of red deer populations. In unfenced areas, overhunting of males leads to female-biased populations with mostly young males (Pérez-González and Carranza 2009). Research has shown that transmission of genetic variability is compromised in the paternal lineage in open lands compared to fenced ones (Pérez-González and Carranza 2009). The result is that inbreeding does not differ on average between open and fenced properties (Martinez et al. 2002) and cases of extreme inbreeding are found even more easily in open lands (Pérez-González et al. 2010a, b). Inbreeding can lead to decreased survival, increased vulnerability to disease, reduced fitness, and decreased lifetime breeding success. Biologists recommend prioritizing strategies to minimize the effects of inbreeding (Mysterud and Bischof 2010).

Feeding tends to increase the gathering of animals and have potential effects on natural mating behaviors. In Spanish hunting properties with supplemental feeding, the placement of feed can be more important in female red deer aggregation during the breeding season than the distribution of natural resources (Pérez-González et al. 2010a, b). Males then shift their strategy from harem defense to territoriality around the presence of supplemental feed (Carranza et al. 1995). Supplemental feeding also tends to increase harem size, but brings a reduction in sexual harassment of females by males (Sánchez-Prieto et al. 2004). The evolutionary consequence benefits males who control the larger female harem groups (Carranza et al. 1995). In red deer populations with mature males (mostly fenced populations), supplementary feeding may further increase the degree of polygyny and affect male mating success (Pérez-González and Carranza 2011).

Harvest has a major impact on population density, age structure, and sex ratio, leading to selective pressures that influence the genetic makeup of game animals. Several studies discuss how harvesting trophy-quality game selects for smaller and potentially less fit animals (Martínez et al. 2005; Mysterud 2010). Compensatory culling targeting low-quality yearlings may successfully counter the selective effects of trophy hunting, although combinations of human mediated management practices may lead to semi-domestication (Torres-Porras et al. 2009; Mysterud 2010; Mysterud and Bischof 2010).

Game farming and restocking practices mix genetic lines and leads to hybridizing among game animals that used to be separated by significant geographic distance. These practices increase the risk of losing local adaptations due to genetic introgression from non-local populations. Indiscriminate translocation practices throughout Europe introduced genetic matrilines of Scottish red deer and Eastern European red deer in some Spanish red deer populations (Martinez et al. 2002; Fernández-García et al. 2006). Recent regulations have attempted to reduce this practice, requiring genetic tests of trophies prior to their entry into Spanish records to ensure they are not hybrids. Some regions of Spain have begun to require these genetic tests prior to authorizing translocations, and some landowners utilize these genetic tests to purge hybridization from their herds (Carranza et al. 2003; Carranza 2010).

Genetic impacts are not limited to big game species. The red-legged partridge faces threats from farm-raised animals, including parasite and disease transmission, and genetic dilution through introgression of farmed bird genes (Vargas et al. 2006; Blanco-Aguiar et al. 2008).

1.5.5 Population Structure

In California, buck-only harvest regulations result in female-biased sex ratios and the selection for smaller-antlered deer because larger-antlered males are often preferentially removed from the population. California has not implemented any compensatory culling scheme to reduce this effect. Bucks that lack forked antlers (with at least two points on each side) are illegal to harvest. While this is designed to prevent the harvest of one-year-old males, there are examples of 2–3 year old bucks that do not grow branched antlers being selectively protected and presumably allowed to breed. This artificially selects for deer whose antlers never mature beyond the size of a one-year-old buck, which runs counter to expected breeding fitness characteristics.

Harvest practices can impact the age structure and sex ratios of game species, leading to altered dispersal patterns (Pérez-González and Carranza 2009). The phenomenon of artificially skewed age and sex ratios is captured effectively by Pérez-González and Carranza (2009) who found that red deer sex ratios in fenced properties was approximately 0.79 males to females, while unfenced populations exhibited a 0.25 male to female ratio. They also found lower percentages of adult males in unfenced properties than fenced ones, at 54 versus 74 %.

The population characteristics of the unfenced properties described above are often the result of overharvest of adult males, which leads to reduced male mating competition. This altered population structure can affect genetic exchange between populations in unfenced areas. Pérez-González and Carranza (2009) found that low male mating competition changed natural dispersal patterns to female-biased dispersal instead of the expected male-biased dispersal for polygynous species such as the red deer.

A study comparing two harvest methods of wild boars, espera (stand hunting) vs. montería (driven hunts by dogs), showed that the montería was much less selective, yielding harvest of animals across all age classes and genders, while espera hunting was much more selective for mature males (Braga et al. 2010).

1.5.6 Invasive Species

Management actions that clear vegetation can create openings for invasive species. In California, perennial native grasslands are usually extirpated from an area after plowing, discing, or planting of food plots. Invasive weeds, such as yellow star thistle (Centaurea solstitialis), and non-native grasses are usually the dominant species that re-vegetate plowed areas.

Translocations of non-native game animals can lead to invasions by the game animals that outcompete native species or disrupt ecosystem function. This can spread exotic diseases or

parasites to native species. Wild boars have spread throughout many oak woodlands in California, causing changes in the ecosystem (Wilcox and Van Vuren 2009). In Spain, the Barbary Sheep (Ammotragus laervia) has expanded in certain areas and can compete with native ungulates, in particular the Spanish ibex (Capra pyrenaica) (Carranza 2010). Translocation of rabbits has been implicated to some extent in the spread of both myxamatosis and rabbit hemorrhagic disease (RHD), which have decimated rabbit populations throughout Spain (Delibes-Mateos et al. 2008). Since the rabbit is the primary prey species for at least 29 top predators in Spain, its decline has led to declines in predator species, including the Spanish imperial eagle (Aquila adalberti) and the Iberian lynx, which are highly dependent on rabbits (Delibes-Mateos et al. 2008) (Fig. 1.15).

Fig. 1.15 So significant is hunter success in wild boar hunting in Spain that proud landowners will often create a trophy plaque showing the ‘‘defenses’’ of the boar, especially for a mature animal, as seen here at a dehesa in northern Córdoba. (Photograph by A. Caparrós)



1.5.7 Protected and Non-Target Species

Predator control in California’s oak woodlands does not currently threaten any endangered species, although historical hunting and culling of predators such as the gray wolf (Canis lupus) and the grizzly bear (Ursus arctos horribilis) extirpated those species from the state. While intensive predator control was historically a part of California’s management for hunting and livestock, recent regulations banning poisons and limiting methods of trapping have decreased the impact of predator control. Mountain lions are protected, and while ranchers can obtain a permit to kill a mountain lion that has preyed on livestock, the administrative process may take several days, during which time the mountain lion often moves far from a kill site.

Spain’s predator control practices appear somewhat detrimental to non-target species. While the target of predator control tends to be the red fox or the common magpie, many other species are affected by illegal or non-selective predator control. Studies have found declines in populations of the common kestrel (Falco tinnunculus) and the common genet (Genetta genetta) on hunting properties where intensive predator control is practiced (Beja et al. 2009). Additional research shows that non-selective control measures of box traps, snares, and illegal poison that is intended for red foxes may not significantly impact fox populations, but in fact adversely affect species such as the badger (Meles meles), wildcat (Felis silvestris), and stone marten (Martes foina), which are unable to cope with intensive predator control (Virgós and Travaini 2005).

1.6 Conclusions

The oak woodlands of Spain and California are ecologically similar, but significant disparities in history and governance yield dramatically different hunting systems on the two sites. These distinctions are manifest in the methods of hunting, the economic return to landowners, game management practices, and the environmental impacts of such management.

Income from hunting can provide an economic incentive to maintain areas of relatively undisturbed wildlife habitat. Properties that earn revenue from hunting are more common in Spain and often earn more than those in California. Current law and economic conditions in Spain make it favorable for landowners to continue to utilize areas of dehesa for recreational hunting, and reinforces a trend away from traditional livestock operations, brush clearing, and agricultural use that were the norm even a decade or two ago. However, while properties managed for income as hunting reserves may be profitable (Oviedo et al. 2013), they are not lucrative enough to entirely replace the traditional multiple uses of dehesa habitat. In California, historical precedent, widespread public lands, lower hunting participation rates, limitations on commercialization of wildlife, and increasing restrictions on hunting and wildlife management practices all serve to reduce the amount ranch owners earn from hunting. Nonetheless, the local food movement and the current interest in wilderness experiences are shaping a new narrative about hunting as an ecologically sound way to connect to one's food source while also obtaining organic meat. Even though hunting can provide sustainable economic return on dehesas and wooded oak ranchlands, uninformed wildlife management practices that are too narrowly focused on game animals can cause environmental degradation (Díaz et al. 2009). However, thoughtful management practices that seek to improve habitat for a variety of species can not only maintain but improve environmental values on these properties.

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Chapter 2

The role of wildlife-associated recreation in private land use: Providing the missing baseline

Abstract: As the pace of habitat loss accelerates worldwide, private land use is increasingly important in conservation policy. A wide body of literature has discussed the potential conservation incentives that wildlife-associated recreation (hunting, fishing, and wildlife-watching) could provide on private lands, yet we lack a good baseline of the spending and land-use characteristics of wildlife-associated recreation from which to evaluate its significance. This analysis pools three nationwide surveys conducted over an 11-year period to analyze spending to access private land for hunting, fishing, and wildlife-watching in the U.S. and to evaluate that spending in the context of private land conservation and land-use policy. Results show that approximately 440 million acres of private land, an estimated 22% of the contiguous land area of the U.S. and 33% of all private land in the U.S., are either leased or owned for wildlife-associated recreation. Much of these lands are private rangelands and forestlands. Land utilized for hunting accounted for 81% of that total, while land utilized for fishing and wildlife-watching, although comparatively small, likely includes riparian zones and areas with high environmental or amenity values. Hunters own or lease properties of larger size classes than anglers or wildlife-watchers, providing a possible economic incentive for maintaining large unfragmented properties that provide a variety of conservation benefits. Results show that Americans annually spend an estimated $814 million in day-use fees, $1.48 billion for long-term leases, and $14.8 billion to own private land primarily for wildlife-associated recreation. These findings underscore the influence of recreational users on the land use of vast areas of private land in the U.S. Current policy measures such as use-tax policies and incentive programs for hunting can be enhanced and expanded in order to reap additional conservation benefits on these lands.

Keywords: economics, conservation, hunting, fishing, wildlife-watching, policy incentives

2. 1 Introduction

Habitat loss due to anthropogenic causes is the primary source of biodiversity decline and extinctions worldwide and is continuing at a rapid pace (Wilcove et al. 1998). In the U.S., private land accounts for approximately 60% of land area and harbors high levels of biodiversity due to historical homesteading patterns where lands with more productive natural resources were settled first (Scott et al. 2001). These lands, which are integral to the conservation of biodiversity, are often the most vulnerable to habitat degradation and loss through land-use conversion and fragmentation (Knight 1999; Maestas et al. 2003).

In 1930, Aldo Leopold recognized the potential to better conserve U.S. private lands by an incentive scheme where recreational users would pay landowners for access to conserved wildlife habitat (Leopold 1930). Over 80 years later, despite a wide body of literature discussing this approach, we still do not have an accurate understanding of the scale and distribution of such

spending across the U.S. Publications on paid recreational use on private land have generally been limited to regional or agricultural areas or have relied on theoretical economic models (Geist 1988; Rasker et al. 1992; Standiford & Howitt 1992; Anderson & Hill 1995; Zivin et al. 2000; Benson 2001a; Bulte & Horan 2003; Zhang et al. 2006; Brown & Reeder 2007; Mozumder et al. 2007; Poudyal et al. 2012).

Some of the best national estimates can be found in survey reports by the U.S. Fish & Wildlife Service (USFWS) every 5 years, but relatively small samples result in large standard error estimates, leaving large gaps in our understanding of this land use (USFWS 2001, 2006, 2011). This study improves upon the USFWS reports by pooling three survey years to considerably increase sample size and precision of estimates and allowing the calculation of numerous estimates that are not possible using a single year of data. An analysis of the data revealed no significant differences in variables of interest across time, so pooling the data resulted in improved geographical precision without the loss of information about significant changes over time.

This study seeks to i) better understand the land area and land cover utilized for wildlife-associated recreation ii) assess the size classes of properties that are utilized for different types of recreation, iii) describe the scope and type of spending to own or access private land for wildlife-associated recreation in the U.S., iv) evaluate the regional variation in this land use, and v) discuss some of the policy implications of these findings.

Although the conservation impacts of recreational use can vary greatly (Reed & Merenlender 2008; Macaulay et al. 2013), studies have shown that under the correct governance structures, payments for recreation can motivate habitat conservation and yield a positive conservation effect (Oldfield et al. 2003; Lindsey et al. 2007; Dickson et al. 2009). Furthermore, research has shown how financial incentives can benefit conservation by influencing land-use decisions (Lubowski et al. 2008; Rashford et al. 2010), but recreational use has not been explicitly considered as an influence in models for land-use decision-making.

2.2 Methods

For this research, I analyzed data from the USFWS National Survey on Fishing, Hunting and Wildlife-Associated Recreation (National Survey), which was conducted by the U.S. Census Bureau. This survey gathers information every five years about participation in and spending for fishing, hunting, and wildlife-watching. To corroborate lease estimates on farm and ranch land in the U.S. and to better understand what types of properties are being used for recreation, I analyzed data from the U.S. Department of Agriculture's (USDA) Agricultural Resource Management Survey (ARMS).

The National Survey is a multistage probability sample with coverage in all 50 states. The National Survey was conducted in two stages: 1) an initial screening of households to identify likely sportspersons and wildlife-watching participants and 2) the selection of two independent samples of sportspersons (hunters and anglers) and wildlife-watchers from which detailed interviews were conducted to collect detailed data about their wildlife-related recreation during the survey year. This results in two data sets in each survey year, one for sportspersons and one for wildlife-watchers. This study uses National Survey data from the three most recent

survey years (2001, 2006, and 2011), which were independently sampled and asked the same questions each survey year about what recreationists pay to access or own private land. I pooled the three years to improve the precision of estimates, increasing the sample size to a total of 93,725 observations with 4,957 observations of individuals who leased, owned or paid fees to access private land for wildlife-associated recreation. Pooling across time sacrifices temporal detail in order to improve geographical understanding (Verma et al. 2009). Because analysis found no significant differences in variables of interest across time, pooling allowed for improved geographical analysis, including calculation of estimates by region and property size class, without losing information about how these estimates change over time. As a result, the results reported are estimates for an average year over the course of 2001-2011. See Appendix A for a more detailed explanation on the improvements in estimates achieved by pooling the data.

I evaluated three ways individuals pay to access private land for recreation: i) day-use fees, ii) leases, and iii) owning land. Day-use fees are payments to access or use private land during single or multi-day recreational trips. Leases are generally understood to be agreements for seasonal or year-round access to private land that are renewed on an annual or multi-year basis. Estimates of land ownership for wildlife-associated recreation include only those landowners who self-identify as owning the land primarily for the purpose of a wildlife-associated recreational activity. The survey questionnaire requested a single value for the amount spent to own land, which included mortgage payments, down payments, taxes and maintenance expenses. The inclusion of these various payments into a single amount combines different accounting concepts into a single value. As a result the values for ownership are difficult to compare to other standard accounting measures and exhibit relatively large standard errors (see Appendix B). Expenditures from 2001 and 2006 were converted to 2011 dollars using the Consumer Price Index.

The ARMS is an annual survey of farm businesses and operators (independently sampled every year) that is designed to be representative of the continental U.S. The farm population includes all establishments which produced and sold, or would normally have sold, at least $1,000 of agricultural products during the previous year. This survey asks the same questions each year about income from recreational sources such as hunting, fishing, and horseback riding. I use thirteen years (1999-2012) of ARMS data from a similar time frame as the National Survey to corroborate leasing estimates obtained from the National Survey. Similar to the National Survey, I pooled the 13 years of data to improve precision, obtaining an overall sample of 227,000 farms and ranches, with 5,800 of those farms and ranches earning income from recreation.

This ARMS survey has important differences from the National Survey in that it likely excludes much of the approximately 422 million acres of private forestlands in the U.S. (Nickerson et al. 2011) and uses a broader definition of recreation by including horseback riding and other non-wildlife-associated recreation activities. To address the lack of forestland coverage in the ARMS, I use preliminary results from the U.S. Forest Service (USFS) National Woodland Owners Survey (NWOS). These preliminary results, released via direct communication with the survey administrators, were used to corroborate use of private forestland for hunting or other recreation (Butler et al. 2014). Although important differences in these surveys makes direct comparisons challenging, these independently conducted surveys provide the ability to substantiate the surprisingly large area estimates from the National Survey as well as assess the land cover type utilized for recreation.

The National Survey and ARMS administrators calculate sampling weights that are used to scale up survey responses to estimate population totals. The concept behind sampling weights is that an individual surveyed with a sampling probability of π_1 represents $1/\pi_1$ individuals in the population, where $1/\pi_1$ is the sampling weight. If X_i is a measurement of variable X on person i, the following equation is used to estimate the total, $\hat{T}_x$:

$$\hat{T}_X = \sum_{i=1}^{n} \frac{1}{\pi_i} X_i .$$

The variance estimate is

$$\widehat{\text{var}}\left[\hat{T}_X\right] = \sum_{i,j} \left(\frac{X_i X_j}{\pi_{ij}} - \frac{X_i}{\pi_i} \frac{X_j}{\pi_j} \right) .$$

This estimator of the population total is called the Horvitz-Thompson estimator (Horvitz & Thompson 1952), and mean estimates are derived by dividing the estimated total by the population size, $\widehat{N}$, which is the sum of the sampling weights or

$$\hat{N} = \sum_{i=1}^{n} \frac{1}{\pi_i}.$$

I calculated 95% confidence intervals for estimates by multiplying standard error estimates by 1.96 (Lumley 2011). In the National Survey and ARMS, the sampling weight variable is used to calculate national estimates of variables for the single survey year. In order to obtain an average annual estimate from multiple survey years in this pooled data analysis, the sampling weights were divided by the number of survey years pooled (Lumley 2011).

In calculating totals across the separate sportsperson and wildlife-watching surveys within the National Survey, I summed the subtotals and the standard error estimates. To calculate means across the two surveys, I used the Graybill-Deal estimator (assuming independence of the mean estimates) to create an unbiased composite mean between the two estimates (Graybill & Deal 1959). Using estimated means of the wildlife-watcher survey, μ_1, and sportsperson survey, μ_2, I calculated a weight, b, based on the estimated variance, S^2, of each mean $b=S_1^2/(S_1^2 + S_2^2)$. I used this weight to estimate the composite mean, μ^*, using the equation $\mu^*=(1-b)\mu_1 + b\mu_2$.

The estimates in this manuscript could contain non-sampling error such as measurement error or non-response error. The U.S. Census Bureau, in administering the survey, employs quality control procedures throughout the planning, collecting, and processing of data to minimize these errors (U.S. Census Bureau 2012). Response rates ranged between 66% and 90% for the National Survey, which is relatively low compared to many survey studies, which helps to minimize the risk of non-response bias (Groves 2006). See Appendix C for further information on the methods, analysis, and the survey questions utilized in analysis.

2.3 Results

2.3.1 Land area owned or leased

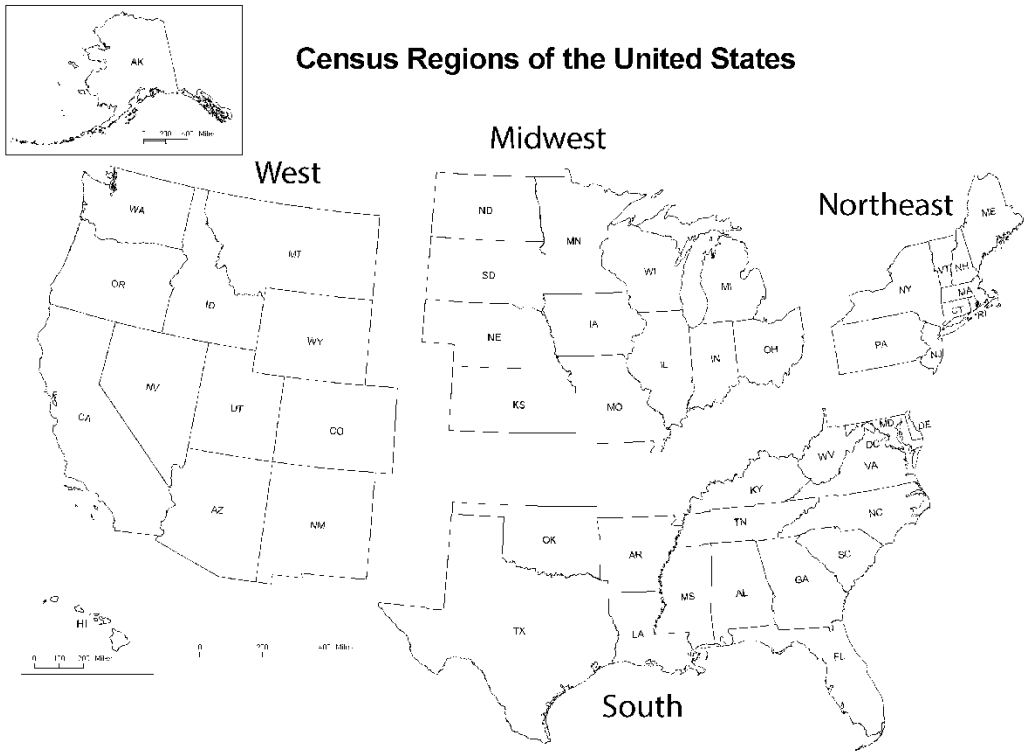
Approximately 1.3 billion acres, or 60% of the total U.S. land area, is privately owned (Nickerson et al. 2011). An estimated 440 million acres (SE 72 million acres) of private land are either leased or owned for wildlife-associated recreation. Leased lands account for 235.6 million acres and owned lands account for 204.5 million acres. Land used for hunting makes up the majority of this area, accounting for approximately 93% of all lands leased for wildlife-associated recreation, and 66% of all lands owned for wildlife-associated recreation (Table 1). Hunting lands contrast from fishing and wildlife-watching lands in that 38% of the estimated hunting land area is composed of owned properties, while over 80% of the 50 million acres used for wildlife-watching are owned, and 83% of the 34 million acres used for fishing are owned.

Table 1: Descriptive statistics on the land area used for wildlife-associated recreation in U.S., including estimates of payments on a per-person and per-acre basis.

	Total Acres (in millions)	SE	% of total within activity type	% of overall total	% of land area	Mean size of property (acres)	SE	% in clubs	SE	Mean number in club	SE	Mean acres per person (adjusted for clubs)	SE	Mean amount paid per person per acre	SE
Fishing															
Lease Total	**6.8**	**2.0**	**20%**	**2%**	**0.3%**	**423**	**168**	**59%**	**5%**	**71**	**24**	**33**	**10**	**685**	**113**
Northeast	0.2	0.1	0%	0.0%	0.1%	94	58	66%	13%	141	83	5	3	515	127
Midwest	0.6	0.5	2%	0.1%	0.1%	35	20	65%	8%	29	15	10	8	698	150
South	3.5	1.4	10%	1%	0.6%	681	329	48%	7%	50	25	38	14	766	214
West[a]	2.5	1.4	7%	1%	0.2%	1,088	920	78%	10%	148	97	142	81	472	157
Own Total	**27.5**	**4.0**	**80%**	**6%**	**1.1%**	**61**	**9**	**29%**	**2%**	**46**	**12**	**32**	**4**	**3,085**	**407**
Northeast	0.5	0.2	1%	0%	0.4%	85	53	29%	6%	179	74	5	2	3,714	1,096
Midwest	9.3	2.2	27%	2%	1.8%	52	13	34%	4%	13	5	29	7	2,340	600
South	17.2	3.3	50%	4%	2.9%	71	13	24%	3%	49	22	45	8	3,513	708
West	0.6	0.1	2%	0.1%	0.0%	17	7	35%	6%	33	20	8	2	3,183	939
Lease and Own Total	**34.3**	**4.5**	**100%**	**8%**	**1.4%**	**134**	**35**	**35%**	**2%**	**55**	**12**	**33**	**4**	**2,675**	**340**
Hunting															
Lease Total	**220.1**	**39.2**	**62%**	**50%**	**9.1%**	**2,399**	**387**	**91%**	**2%**	**19**	**2**	**225**	**38**	**15**	**3**
Northeast	18.2	14.7	5%	4%	15.7%	859	308	79%	8%	21	5	299	243	41	15
Midwest	23.2	7.3	7%	5%	4.4%	577	155	78%	8%	12	3	172	57	42	18
South	149.5	24.1	42%	34%	25.4%	2,566	415	94%	1%	18	2	194	26	8	2
West[a]	29.2	26.2	8%	7%	2.4%	14,873	11,813	71%	11%	185	147	2,208	1,942	49	29
Own Total	**135.8**	**17.4**	**38%**	**31%**	**5.6%**	**327**	**106**	**39%**	**2%**	**12**	**3**	**109**	**13**	**264**	**43**
Northeast	13.1	3.7	4%	3%	11.3%	117	25	43%	6%	8	2	63	16	441	165
Midwest	55.8	13.5	16%	13%	10.6%	147	27	35%	3%	5	1	106	25	240	66
South	60.1	10.1	17%	14%	10.2%	615	282	42%	4%	20	8	129	20	149	38
West	6.6	2.3	2%	2%	0.6%	462	148	44%	6%	20	11	147	47	847	274
Lease and Own Total	**355.9**	**43.2**	**100%**	**81%**	**14.7%**	**1,325**	**217**	**64%**	**2%**	**17**	**2**	**169**	**19**	**158**	**25**
Wildlife-Watching															
Lease Total	**8.7**	**3.2**	**17%**	**2%**	**0.4%**	**298**	**116**	**75%**	**7%**	**96**	**47**	**63**	**21**	**903**	**374**
Northeast[a]	0.0	0.0	0%	0.0%	0.0%	1	0	46%	20%	4	1	1	0	912	404
Midwest[a]	1.3	0.8	3%	0.3%	0.2%	106	71	76%	12%	80	72	50	30	264	102
South[a]	7.3	3.1	15%	2%	1.2%	464	188	80%	9%	57	26	91	34	876	543
West[a]	0.0	0.0	0%	0%	0.0%	116	96	83%	13%	478	320	1	0	2,162	1,929
Own Total	**41.2**	**6.4**	**83%**	**9%**	**1.7%**	**55**	**8**	**25%**	**3%**	**32**	**12**	**41**	**6**	**2,799**	**981**
Northeast	3.7	0.9	7%	1%	3.2%	23	5	21%	5%	12	5	17	3	1,506	314
Midwest	14.2	3.3	28%	3%	2.7%	55	12	20%	5%	28	15	49	11	771	265
South	16.3	4.2	33%	4%	2.8%	70	15	28%	5%	31	18	47	11	1,667	579
West	7.0	3.5	14%	2%	0.6%	65	27	31%	7%	58	53	52	26	11,845	6,248
Lease and Own Total	**49.9**	**7.2**	**100%**	**11%**	**2.1%**	**87**	**18**	**31%**	**3%**	**52**	**17**	**45**	**6**	**2,617**	**881**
Total Leased	**235.6**	**42**	*NA*	**54%**	**9.7%**	**494**	**155**	**85%**	**2%**	**26**	**6**	**104**	**25**	**133**	**32**
Total Owned	**204.5**	**25**	*NA*	**46%**	**8.4%**	**57**	**11**	**33%**	**2%**	**26**	**7**	**55**	**7**	**1,607**	**272**
Total Leased & Owned	**440**	**51**	*NA*	**100%**	**18.1%**	**59**	**15**	**56%**	**2%**	**26**	**6**	**59**	**10**	**153**	**45**

[a] Indicates estimates based on fewer than 30 observations

Fig.1: U.S. Census Regions



An analysis of these land areas by U.S. Census Region (see Fig. 1) reveals that land in the South and Midwest contributes substantially to the total land area across the three recreation types. Of the 34 million acres owned and leased for fishing, 50% is owned fishing land in the South, and 27% is owned fishing land in the Midwest. Of the 356 million acres owned and leased for hunting, 42% is leased hunting land in the South, 17% is owned hunting land in the South, and 16% is owned land in the Midwest. Of the 50 million acres owned and leased for wildlife-watching, 33% is owned land in the South, 28% is owned land in the Midwest, and 15% is leased in the South. Because the Northeast Census Region is less than a quarter the size of the other census regions, comparisons by percentage of land area reveal that the Northeast, while contributing relatively small amounts nationally, has a statistically similar proportion of its land area devoted to ownership for hunting (11.3%) as the Midwest (10.6%) and South (10.2%), as well as percent of land area owned for wildlife-watching (3.2%), compared to 2.7% for the Midwest and 2.8% for the South. The South has an estimated 25% of its land area being utilized for hunting leases. Estimates also show the Northeast with 15.7% of its land being used for hunting leases, however large standard error estimates in that case make the estimate somewhat unreliable. In sum, the South and Midwest contribute the most significant areas of land to wildlife-associated recreation, while the Northeast contributes proportionally similar amounts in hunting leases, hunting ownership, and wildlife-watching ownership, but not fishing. Private land in the West is notably a minor contributor to totals, especially given its greater size than any other census region.

Leased properties on average are substantially larger than owned properties across all three recreation types, however a much greater proportion of leased properties are leased by

clubs or groups rather than individuals. After accounting for these groups on a per-person basis, the differences between mean number of acres leased or owned per person is not significantly different for fishing properties or wildlife-watching properties, although leased hunting properties are significantly larger on a per person basis than owned hunting properties.

2.3.2 Land cover utilized for wildlife-associate recreation

Of the 1.3 billion private acres in the U.S., there are approximately 422 million acres of forestland, 402 million acres of cropland, 380 million acres of rangeland, and 142 million acres of urban, special use, and miscellaneous land (Nickerson et al. 2011). I utilized data from two independent surveys, the ARMS and the NWOS, to better understand the land cover types used for wildlife-associated recreation. The ARMS data was used to calculate the estimated area of cropland and rangeland in the U.S. that earns income from farm-based recreation, while the NWOS data was used to estimate private forestland used for recreation in the U.S. (Butler et al. 2014). The ARMS data shows that 99 million acres (SE 6.1 million acres) of properties classified as livestock ranches (which is used as proxy for rangeland land cover) earn income from recreation and 15.4 million acres (SE 1.0 million acres) of cropland earn income from recreation. Preliminary results of the NWOS reveal that 155 million acres (SE 2.4 million acres) of private forestland are leased for hunting and 51 million acres leased for recreation besides hunting (it is possible that some of these lands are leased for both hunting and other recreation, resulting in overlap in these acreage estimates). Combining estimates from these two survey efforts over the three main land cover types in the U.S. (rangeland, forestland, and cropland) yields an estimated 270-321 million acres (depending on amount of overlap between hunting leases and non-hunting recreation leases on forestlands) of land leased or earning income from recreation broadly defined. These estimates support the findings of the NSFWHAR, which estimates 236 million acres leased for the more narrowly defined subset of wildlife-associated recreation, which for example excludes horseback riding. Furthermore, they suggest that forestlands and rangelands are the most significant contributors to the area of land leased for wildlife-associated recreation, with croplands contributing a comparatively small amount.

I could only assess land cover types for owned lands on private forestlands through the NWOS because the ARMS does not gather information on whether recreation is a motivating factor for owning the land. Preliminary NWOS estimates show that owners of 212 million acres (SE 2.54 million acres) of forestland say that hunting is “very important” or “important” for their reasons for ownership and owners of 178 million acres (SE 2.46 million acres) say that other recreation besides hunting is “very important” or “important” for their reason for ownership. These estimates provide support for the more narrowly defined estimate of owning land primarily for wildlife-associated recreation from the National Survey, which estimates 205 million acres are owned “primarily” for wildlife-associated recreation with 135 million acres of that estimate owned primarily for hunting. Similarly to the leased lands, these estimates from the NWOS suggest that forestlands are a significant contributor to the total amount of private land owned primarily for wildlife-associated recreation.

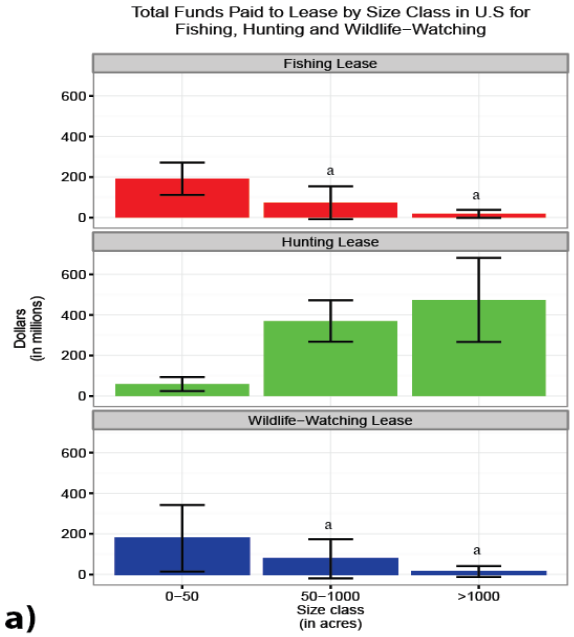
2.3.3 How hunters, anglers, and wildlife-watchers utilize properties of different size classes

Analysis of the National Survey shows that on a nationwide basis, hunters spend significantly more money than anglers or wildlife-watchers to lease lands of larger size classes, paying an estimated $370 million to lease property between 50-1,000 acres, and $473 million to lease properties greater than 1,000 acres (Fig. 2a). Analysis of spending to own land revealed that hunters as a whole also spent significantly more to own properties greater than 1000 acres, and spent significantly less than anglers and wildlife-watchers for properties less than 50 acres (Fig. 2b)

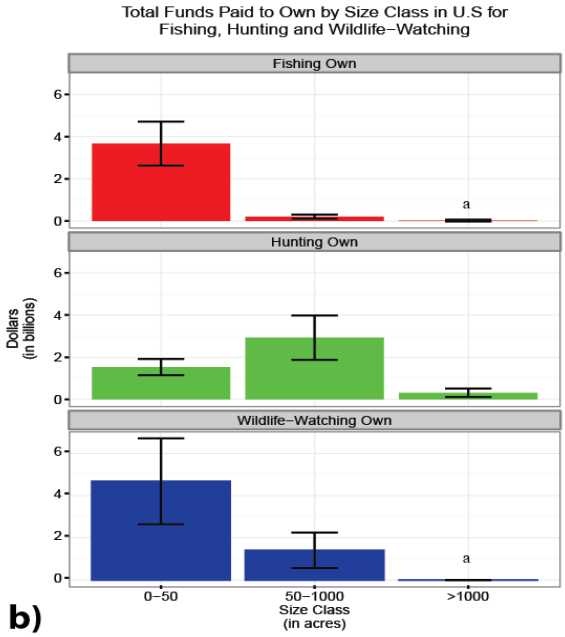
A similar pattern emerges when evaluating land area utilized in different size classes (as opposed to spending on different size classes). An estimated 176 million acres (SE 39.0 million acres) of land utilized for hunting leases is composed of properties greater than 1000 acres, which accounts for 80% of leased hunting lands and 75% of land that is leased for all types of wildlife-associated recreation (Fig. 2c). The land owned primarily for hunting follows a similar pattern, with more of the land area used for hunting falling into the largest two size classes than for fishing or wildlife-watching (Fig. 2d). The 50-1000 acre size class contributes most to the land area owned for fishing and wildlife-watching, but these totals remain smaller than the amount of land used for hunting in this size class.

Figure 2: How recreation types utilize different sized properties. Hunters as a whole lease and own properties in larger size classes than anglers or wildlife-watchers. a) Total annual expenditures for leases (95% CI). b) Total annual expenditures to own land primarily for recreation (95% CI). c) Total land area under lease for wildlife-associated recreation (95% CI). d) Total land area owned primarily for wildlife-associated recreation (95% CI). [a] *estimates that are based on less than 30 observations.* [b] *indicates subsets of data that have greater than 10% of their observations top-coded to protect the anonymity of respondents, which likely results in greater uncertainty in these estimates than standard error bars indicate.*

Dollars Spent

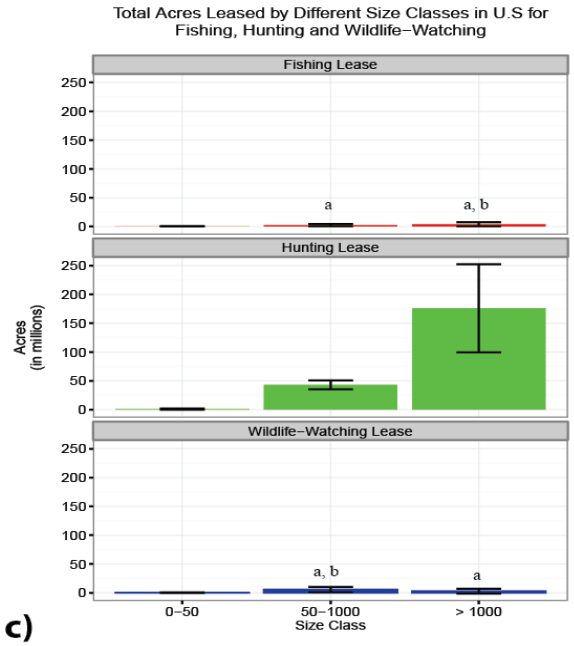


a)

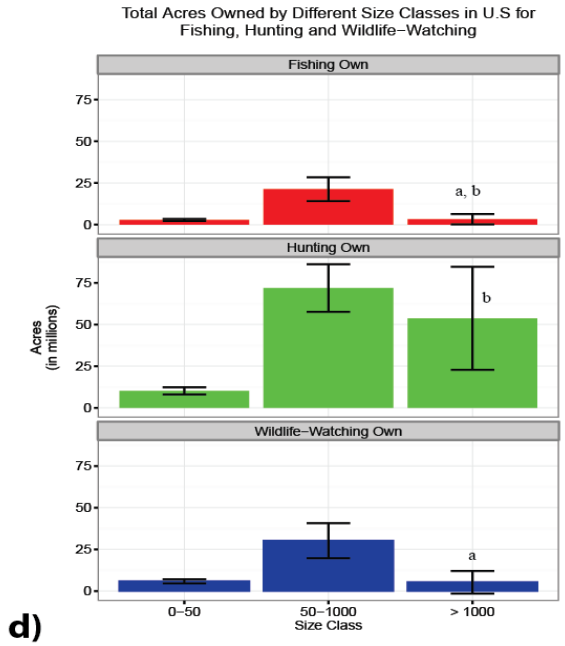


b)

Acres



c)



d)

Table 2: Price paid per acre for different property size classes (only categories with significant differences between size classes).

Size Class (acres)	Fishing Own	SE	Hunting Lease	SE	Hunting Own	SE	Wildlife-Watching Own	SE
<50	20,862	8,384	699	206	1,197	364	13,034	6,207
50-1000	23.6	6.0	19.8	2.5	72.8	13.4	88.6	25.4
>1000	35.2[a]	20.6	7.3	0.5	13.2	3.9	4.0[a]	2.6

[a] Indicates estimates based on fewer than 30 observations

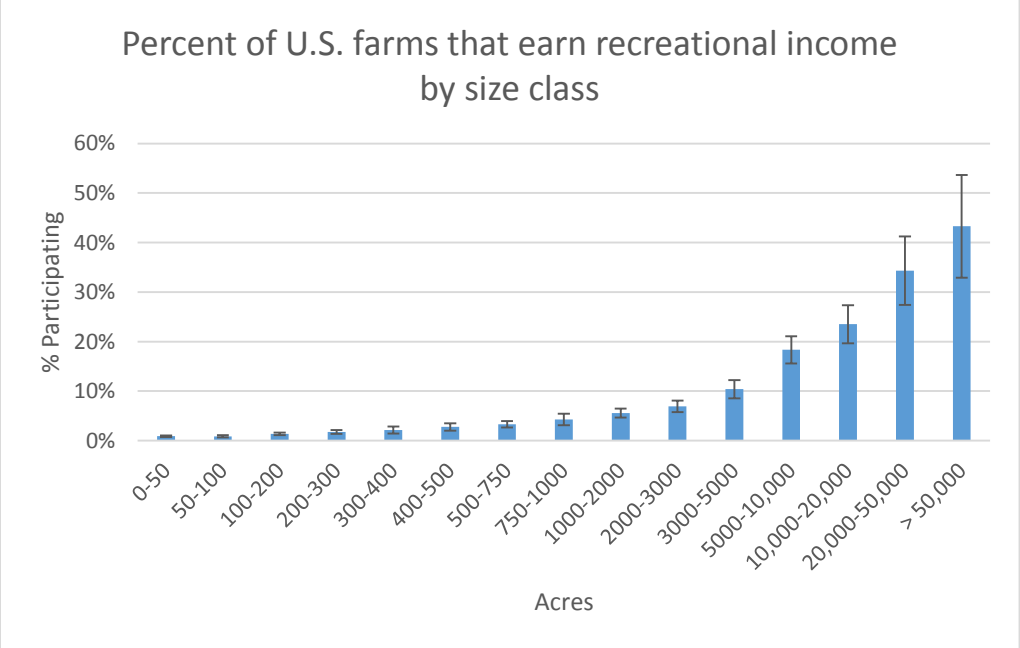
When evaluating spending on a per-acre basis, a pattern emerges of people paying significantly less per acre for larger properties in all cases except for fishing leases and wildlife-watching leases, which exhibited large standard errors (Table 2). This shows that while hunters utilize larger properties and spend more on those larger properties on a whole compared to fishing and wildlife-watching, on an individual basis, they have similar patterns of paying significantly less on a per acre basis for larger properties.

The ARMS data confirms this pattern for cropland and rangeland. Although only 2.5% of all farms and ranches earn recreational income, larger farms and ranches are far more likely to participate in earning recreational income (Fig. 3). This pattern helps to explain how large areas of land are utilized for recreation despite relatively small participation rates by landowners.

2. 3.4 Annual expenditures

According to the National Survey, hunters, anglers, and wildlife-watchers in the U.S. spend $17.068 billion (SE $1.952 billion) annually to lease, access, and own lands for wildlife-associated recreation. Spending for owning accounts for approximately 87% of these funds totaling $14.777 billion (SE $1.920 billion), and spending for leases and day-

Figure 3: Percent of farms and ranches participating in earning recreational income based on size class (95% CI).



use fees is $1.477 billion (SE $234 million) and $814 million (SE $88 million) respectively (Table 3). Annual spending to own land primarily for wildlife-associated recreation is not significantly different between fishing, hunting, and wildlife-watching; however, hunters spend significantly more than anglers or wildlife-watchers on both leasing and day use fees, mostly due to the higher participation rate and greater number of hunters who lease land and the far greater average amount of money paid for day-use fee access for hunting. Participation rates of paying for access through day-use fees, leases or ownership vary widely depending on the activity: 20.8% of hunters, 7.2% of anglers, and only 3% of wildlife-watchers pay for some sort of access to private land.

Table 3: Descriptive statistics on the annual expenditures and participation rates of owning land, leasing land, or accessing through day-use fees. Total participants do not sum due to overlap of activities.

	Total Annual Spending (in millions)	SE	% of overall total	% of total within activity type	Mean Payment	SE	Partici-pants (in 000s)	SE	% Parti-cipating	SE
Fishing										
Fee Total	**174**	**30**	**1%**	**4%**	**128**	**21**	**1,356**	**84**	**4%**	**0%**
Northeast	49	19	0.3%	1%	1,503	1,211	174	36	4%	1%
Midwest	27	5	0.2%	1%	103	37	354	39	4%	0.4%
South	69	21	0.4%	2%	56	9	537	60	4%	0.5%
West	28	6	0.2%	1%	120	43	291	25	5%	0.4%
Lease Total	**285**	**59**	**2%**	**7%**	**1,563**	**264**	**212**	**22**	**1%**	**0.1%**
Northeast	23	7	0.1%	1%	920	255	33	9	1%	0.2%
Midwest	49	12	0.3%	1%	879	149	66	13	1%	0.1%
South	202	57	1%	5%	2,378	485	93	13	1%	0.1%
West[a]	11	5	0.1%	0.2%	641	123	20	7	0%	0.1%
Own Total	**3,906**	**535**	**23%**	**89%**	**4,968**	**627**	**874**	**46**	**3%**	**0.1%**
Northeast	455	161	3%	10%	5,217	1,731	91	13	2%	0.3%
Midwest	1,275	272	7%	29%	4,527	810	325	29	4%	0.3%
South	1,692	411	10%	39%	4,804	1,109	381	33	3%	0.3%
West	484	132	3%	11%	7,435	1,984	77	9	1%	0.1%
Fee, Lease and Own Total	**4,365**	**540**	**26%**	**100%**	**1,964**	**240**	**2,323**	**97**	**7%**	**0.3%**
Hunting										
Fee Total	**556**	**58**	**3%**	**9%**	**591**	**51**	**941**	**52**	**7%**	**0.4%**
Northeast	29	8	0.2%	0.5%	324	84	85	12	4%	1%
Midwest	99	34	1%	2%	454	137	213	32	5%	1%
South	352	42	2%	6%	674	65	533	38	11%	1%
West	76	17	0.4%	1%	619	110	110	12	6%	1%
Lease Total	**913**	**119**	**5%**	**15%**	**947**	**84**	**1,037**	**72**	**8%**	**1%**
Northeast	42	20	0.2%	1%	700	299	68	11	4%	1%
Midwest	101	39	1%	2%	779	300	129	26	3%	1%
South	756	111	4%	12%	992	87	819	66	16%	1%
West[a]	14	5	0.1%	0.2%	1,100	206	21	5	1%	0.3%
Own Total	**4,780**	**581**	**28%**	**76%**	**4,249**	**496**	**1,268**	**60**	**10%**	**0.5%**
Northeast	743	173	4%	12%	3,794	889	206	26	11%	1%
Midwest	2,367	440	14%	38%	5,164	902	531	41	13%	1%
South	1,318	314	8%	21%	3,081	714	487	36	10%	1%
West	352	122	2%	6%	8,173	2,649	45	6	2%	0.3%
Fee, Lease and Own Total	**6,249**	**600**	**37%**	**100%**	**2,444**	**224**	**2,722**	**99**	**21%**	**1%**

Table 3 continued

Wildlife-Watching										
Fee Total	**84**	**23**	**0.5%**	**1%**	**73**	**20**	**1,045**	**83**	**2%**	**0.1%**
Northeast	34	21	0.2%	1%	159	89	205	44	2%	0.3%
Midwest	5	1	0.0%	0.1%	37	8	200	35	1%	0.2%
South	30	7	0.2%	0.5%	86	20	352	47	1%	0.2%
West	15	3	0.1%	0.2%	39	5	288	39	2%	0.3%
Lease Total	**280**	**99**	**2%**	**4%**	**2,156**	**720**	**144**	**26**	**0.2%**	**0.0%**
Northeast[a]	15	7	0.1%	0.2%	912	404	21	8	0.2%	0.1%
Midwest[a]	23	11	0.1%	0.4%	904	453	34	11	0.2%	0.1%
South[a]	162	66	1%	3%	2,202	733	79	22	0.4%	0.1%
West[a]	79	72	0%	1%	5,801	5,779	10	6	0.1%	0.0%
Own Total	**6,091**	**1,122**	**36%**	**94%**	**6,800**	**1,182**	**1,007**	**64**	**2%**	**0.1%**
Northeast	850	198	5%	13%	4,107	871	214	30	2%	0.2%
Midwest	1,224	360	7%	19%	5,017	1,430	293	35	2%	0.2%
South	1,945	506	11%	30%	6,106	1,494	349	40	2%	0.2%
West	2,073	914	12%	32%	16,397	6,142	152	21	1%	0.1%
Fee, Lease and Own Total	**6,454**	**1,128**	**38%**	**100%**	**3,210**	**552**	**2,125**	**107**	**3%**	**0.2%**
All Fees	**814**	**88**	**5%**	*NA*	**163**	**23**	*NA*[b]	*NA*[b]	**2%**	**0.2%**
All Leases	**1,477**	**234**	**9%**	*NA*	**1,087**	**119**	*NA*[b]	*NA*[b]	**0.3%**	**0.0%**
All Ownership	**14,777**	**1,920**	**87%**	*NA*	**5,201**	**573**	*NA*[b]	*NA*[b]	**2%**	**0.1%**
All Wildlife-Associated Recreation	**17,068**	**1,952**	**100%**	*NA*	**2,480**	**244**	*NA*[b]	*NA*[b]	**5%**	**0.2%**

[a] Indicates estimates based on fewer than 30 observations

[b] Unable to estimate due to inability to subtract out overlapping participation across activities.

Individuals pay significantly more on a per acre basis to own hunting and fishing properties than they do for leased properties. Estimates for mean amounts paid per person per acre for wildlife-watching have large standard errors and are not significantly different. Despite the similar number of acres utilized on a per person basis for most types of recreation, as described in section 2.1 above, individuals owning property primarily for recreation pay substantially more on a per person, per acre basis. This is not surprising given the much greater range of costs (mortgage payments, etc.) and benefits (exclusive control, investment benefits, etc.) that are part of owning land versus leasing land.

Regionally, there are no substantial differences in mean payments across all recreation types, nor for fishing or wildlife-watching participation rates; however, participation rates vary significantly for hunting. The South has much higher participation rates of hunters paying day-use fees (10.6%) and leasing (16.4%), while other regions have less than 6% of hunters paying day-use fees, and less than 3.6% leasing land. Interestingly, participation rates of individuals who *own* land primarily for hunting are statistically similar (9.8%-12.7%) across the Northeast, Midwest, and the South. The West has the lowest participation rates in leasing and owning private land for all categories of wildlife-associated recreation.

2.3.5 Spending on day-use fees for specific types of wildlife-associated recreation

The National Survey gathered information on private land day-use fees for specific subsets of hunting (big game, small game, migratory game, and other animals) and fishing (i.e. saltwater, freshwater, great lakes). Hunting accounted for an estimated 68% of the $814 million spent annually in day-use fees, and big game hunting accounted for most of that, contributing $407 million (SE $51 million) to the total (Table 4). Additional calculations to estimate types of spending on leases using a ratio based on days spent hunting various types of game yielded similar patterns for spending for hunting leases and properties owned for hunting (see Appendix D). Wildlife-watching and fishing day-use fee spending were comparatively low, accounting for $174 million and $90 million respectively.

2.4 Discussion

These results provide much needed baseline information for land use planners and policymakers in assessing the significance of recreational use of private lands. Results showed that a vast area, estimated at 22% (SE 2.6%) of the land area of the contiguous U.S., and 33% (SE 3.8%) of all the private land in the U.S., is leased or owned for wildlife-associated recreation. Participants spend $14.8 billion, $1.5 billion, and $814 million annually on owning, leasing, and day-use fees, respectively. In comparison, the value of U.S. cattle and calf production was $49.5 billion on the estimated 614 million acres of public *and* private rangeland and pasture in 2013 (USDA - ERS 2014). Although the annual spending for wildlife-associated recreation is smaller than the entire U.S. cattle industry, it represents a source of income from lands that, by virtue of their use, presumably support wildlife and wildlife habitat. As such, it is worthwhile to compare these estimates to some of the largest U.S. land and wildlife conservation programs which also help support wildlife and wildlife habitat. Land conservation easements have conserved 47 million acres of land in conservation easements at an annual cost of $1.7 billion (Land Trust Alliance 2011). The federal government's Conservation Reserve Program (a farmland retirement program) takes approximately 30 million acres out of farm production at an annual cost of approximately $2.6 billion (USDA 2012). Finally, the Pittman-Robertson Act fund, which comes from taxes on firearms and ammunition, provides approximately $400 million a year to various conservation efforts and has acquired or leased 4.8 million acres of private lands,

Table 4: Total annual day-use fees for different types of recreation on private land.

Day Use Fee Type	Spending (in millions, 2011 U.S. dollars)	SE	% of Total
Fishing Fee Total	**174**	**30**	**21%**
Freshwater	90	10	11%
Saltwater	74	28	9%
Great Lakes	11	4	1%
Hunting Fee Total	**556**	**58**	**68%**
Big Game	407	51	50%
Migratory Bird	73	17	9%
Small Game	69	13	9%
Other Animals[a]	8	3	1%
Wildlife-Watching Fee Total	**84**	**23**	**10%**
Fee Total	**814**	**88**	**100%**

[a] Indicates estimates based on fewer than 30 observations

and helps fund operation of 33 million acres of public land (U.S. Fish and Wildlife Service 2013). The funds spent and the area affected by recreational use on private land are an order of magnitude greater than these conservation initiatives, showing the influence that this recreational use has on land that supports wildlife in the U.S. It is critical to note that the land that is owned or leased for wildlife-associated recreation may not be permanently conserved and may not always be managed to improve biodiversity or benefit threatened species. As such, these findings point out the importance of providing educational materials and promoting policy tools to encourage these landowners and lessees to utilize sustainable practices and to seek more permanent conservation of these lands. Furthermore, it illustrates the potential impact of policies that incentivize conservation practices on private land used for wildlife-associated recreation.

Results show that hunters, anglers, and wildlife watchers have distinct patterns of land use. Eighty-one percent of the land area leased or owned for wildlife-associated recreation is used for hunting, which represents 17.8% (SE 2.2%) of the land area of the contiguous U.S. Furthermore, 50% of all day-use fees are spent to hunt big game (Table 4). These figures suggest that hunters, particularly big game hunters, are influential in affecting management practices on significant areas of land. For example, managers of the 825,000 acre King Ranch in Texas are careful to protect riparian zones to enhance habitat for hunted game species (DeLaney 2011). Freshwater resources and wetlands associated with the 34 million acres (SE 4.5 million) utilized for fishing are also critical for conservation as freshwater resources host increased biodiversity, many rare and endangered species, and often entirely different species pools than surrounding habitats (Dudgeon et al. 2006). The 50 million acres (SE 7.2 million) of land owned and leased for the primary purpose of wildlife-watching may also have important conservation value, due to high amenity values that often correlate with other important natural resource features, such as designated wilderness areas and increased environmental quality (Smith & Krannich 2000). The fact that much of the land utilized for wildlife-associated recreation is on rangeland and forestland property supports previous research suggesting that recreational activities are in general complementary and compatible with grazing and forestry operations (Marcouiller & Mace 1999; Marshall et al. 2014).

Previous research has found a saturation effect of property size on landowner's willingness to pay for amenity values such as recreation, meaning that as the size of property increases beyond a certain point, landowners obtain few additional personal benefits from amenity values (Oviedo et al. 2012). This effect appears to hold true in the case of wildlife-associated recreation, but the saturation point for hunters appears to be at much larger property sizes than for anglers or wildlife-watchers (Fig. 2). This underscores an important distinction between types of recreational users and the impacts they can be expected to have on land-use and land fragmentation, i.e. hunters are likely to seek properties of larger size classes, as exhibited in Figure 2, while wildlife watchers or anglers seem to prefer comparatively smaller properties for their activities. The smaller land-size preferences of wildlife-watchers or anglers could affect landowner decisions, which may result in higher densities of development and increased land fragmentation for fishing and wildlife-watching compared to hunting.

The ARMS data and other research (Teasley et al. 1999) shows that only 2-3% of private landowners earn income or lease their land for recreation, but this low participation rate belies a larger land area leased for wildlife-associated recreation. This is due to a highly skewed land ownership pattern in the U.S., with many small properties and fewer large properties (USDA - NASS 2012). Because larger farms have a significantly greater likelihood of participation in

earning recreational income (Fig. 3), a much larger area of land is leased for recreation than might be expected given the small percentage of landowners that lease their land for recreation.

Major regional differences in leasing and ownership for wildlife-associated recreation are likely affected by the vast areas of public land open for free recreational use in the West, and, in contrast, the relatively small areas of public land areas in the rest of the country. Increased participation in leasing for recreational access in the South may come from cultural mores of paying for access to private land and friendlier policies for leasing (Gentle et al. 1999). Low participation in leasing in the Northeast and Midwest may be a result of more lenient trespass laws and state programs like Wisconsin's Forest Tax law and Michigan's Commercial Forest Act, which together give tax breaks on 5.2 million acres of forestland under the condition of providing free access for recreation (Michigan DNR 2011; Rickenbach 2011).

2.4.1 Policy Implications

Much conservation policy relies upon direct regulation approaches such as the Endangered Species Act, which can result in unintended consequences of incentivizing the elimination of species or habitat from a property prior to listing in order to avoid regulation. However, these results illustrate that there is an enormous private sector market that is directly tied to wildlife that can be harnessed to encourage positive conservation outcomes and reduce negative impacts. Creating incentives within this private sector mechanism has the potential to influence landowners that may be wary of regulatory approaches.

Research on the role of habitat improvement practices associated with fishing and wildlife-watching on private land is lacking, and research on conservation practices associated with hunting has been equivocal, with some authors finding conservation benefits (Butler & Workman 1993; Oldfield et al. 2003), and others finding few differences in practices (Benson 2001b). As such, policy mechanisms and enhanced educational efforts may be helpful in spurring implementation of habitat improvement practices on private lands utilized for wildlife-associated recreation.

Numerous states including Texas and California have developed voluntary programs to improve wildlife habitat by providing incentives of enhanced hunting opportunities on private land, often in the form of increased flexibility of timing and amount of harvest. These programs appear to have had some success in incentivizing habitat improvement practices, with enrollment of 23 million acres in Texas, and 1 million acres in California. Evaluating the successes and challenges of these state programs would be a first step in identifying sound policies that can then be expanded to other states. For example, policies could be created that discourage practices that can result in adverse impacts to ecosystems and wildlife, such as containment fencing for ungulates, feeding wildlife, planting food plots, and game breeding (Macaulay et al. 2013). Policymakers could develop similar incentive programs for fishing and wildlife-watching to yield additional conservation gains on significant areas of private land. If these programs prove successful in enhancing conservation objectives, additional policy incentives to expand the area of land that is leased or owned for wildlife-associated recreation could be pursued. These could take the form of tax write-offs for landowners or subsidies for recreational users to access private land.

Use-value property tax assessment is a widespread policy tool that is used to maintain open space by appraising property at its agricultural or timberland use rather than its value on the real estate market (Anderson 1993). In some states, like Texas and Oklahoma, land devoted to

wildlife management purposes is eligible for use-value assessment based on productive value of agriculture (Oklahoma 2014; TPWD 2014). Texas explicitly requires landowners to perform at least three out of seven specified wildlife management practices on their property to qualify for the assessment. However, in other states, like New York or Kansas, land utilized for recreation or wildlife management instead of agriculture are not eligible for this agricultural use-value assessment (Kansas 2014; New York State 2014). Making wildlife-associated recreation eligible for use-value property tax benefits has the potential to discourage conversion of these lands to other uses, to reduce the intensity of agricultural production, and yield improved conservation practices if the use-value assessment is tied to conservation practices.

Economic models suggest that financial benefits that accrue to wildlife habitat due to spending for wildlife-associated recreation should slow the conversion of that habitat to other uses (Lubowski et al. 2008; Rashford et al. 2010; Mann et al. 2014). These results show the magnitude of wildlife-associated recreation and highlight its potential to influence conversion of wildlife habitat. Indeed, the value of recreation may help to explain why owners of arable land have not converted their land to crop production to the extent expected in the face of increasing profitability of agriculture in recent years (Swinton et al. 2011). Given the contributions of wildlife-associated recreation, policymakers should consider the potential environmental consequences of land-use change that could result from proposals that reduce landowners' ability to earn income from wildlife-associated recreation.

Diverse revenue streams have also been identified as a mechanism to enhance conservation of working lands by encouraging a more heterogeneous landscape and maintaining year-to-year cash flows for activities that may have high up-front costs or be subject to large fluctuations in income (Goldstein et al. 2006; Bowman & Zilberman 2013). Recreational income has been shown to provide a stable and complementary income source to ranchers (DeLaney 2011), making them less susceptible to financial stress due to droughts or fluctuations in agricultural or forestry markets, which could drive negative environmental impacts or subdivision and fragmentation of properties. Additional research is needed to evaluate whether additional income from recreational use is associated with improved conservation outcomes due to greater financial stability of the landowner.

2.4.2 Conclusion

The findings of this study show that recreational users of private land spend significant money to access vast areas of private land, and represents a private sector opportunity that can be harnessed to achieve public conservation goals. Fortunately, many amenity landowners are particularly interested in environmental protection (Smith & Krannich 2000), and may be particularly receptive to outreach and education efforts and voluntary incentive programs. State Cooperative Extension agencies can play a crucial role in disseminating information about conservation practices and enrolling landowners in programs that encourage sustainable recreational use. Future research can help to clarify the extent to which income from wildlife-associated recreation drives habitat improvement and influences decisions regarding land-use change in the absence of policy mechanisms. To identify the best policy approach, research should evaluate the efficacy and conservation outcomes of current use-value tax policies and state voluntary incentive programs. Finally, determining the most effective way to influence landowner behavior, whether through education or policy tools, will be crucial in developing programs that have the most conservation impact.

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Chapter 3

Evaluating Recreational Hunting as an Incentive for Habitat Conservation on Private Land in California

Abstract: Private land plays a crucial role in the conservation of biodiversity in California, yet these lands are the least protected and most prone to environmental degradation. In 1930, Aldo Leopold recognized the potential to better conserve private land by an incentive scheme where recreational users would pay landowners for access to conserved wildlife habitat. While research has shown that significant funds are spent to utilize large areas of private land for wildlife-associated recreation, this study seeks to specifically understand whether this recreational use actually results in improved conservation practices. I use interviews with private landowners and private land managers in California who owned > 404 ha (1000 ac) to evaluate whether those landowners with recreational utilization are more likely to perform habitat conservation practices. Interviews revealed that while not widely adopted, hunting was the primary recreational use on many private lands surveyed, and that some conservation practices were performed to enhance the hunting enterprise. There appears to be an important role for extension and outreach efforts to better inform landowners of the earning potential from recreational hunting as well as the conservation practices that could enhance both wildlife habitat and revenue for a hunting enterprise.

Keywords: biodiversity, recreation, management, ranching, rangelands, wildlife

3.1 Introduction

Wildlife-associated recreational use on private land has been shown to be a significant source of income for many landowners in the United States, but its conservation impact is poorly understood (Bagi and Reeder 2012, Brown and Reeder 2007, Carpio and others 2008, George and others 2011; Macaulay, n.d.; Nickerson and others 2001). Many studies have hypothesized that recreational use creates an incentive to perform conservation practices and maintain habitat, but many of these studies rely on theory or anecdotal accounts (Benson 2001a, Rasker and others 1992). Empirical research on conservation practices associated with recreational use has been equivocal, with some authors finding conservation benefits (Butler and Workman 1993, Oldfield and others 2003, Standiford and Howitt 1993), and others finding few differences in practices (Benson 2001b). Furthermore, some practices designed to maximize returns from recreational use may resort to activities that result in short-term gains for particular species of interest, but may degrade habitat for other species (for example, planting food plots, introductions of non-native species) (Macaulay and others 2013). Models evaluating economic drivers of land use change have not explicitly considered the role of income or benefits from recreation as motivating land use decisions (Lubowski and others 2008, Rashford and others 2011). This research seeks to provide 1) a better understanding of the economics of recreational use and the type of recreational activities pursued and 2) whether landowners with recreational benefits on their land implement more conservation practices that those without.

To better inform this issue, this study is gathering detailed information from private landowners and managers to specifically determine how recreational use may influence decisions regarding land-use and conservation practices. This information will help to inform how policy, government programs, and Cooperative Extension efforts may be used to encourage practices that not only help to conserve natural resources, but also provide increased recreational benefits on private land.

3.2 Methods

A semi-structured interview questionnaire was developed and tested in a classroom and in a focus group of experts and students in rangeland management. Thirty-seven semi-structured interviews were conducted with 27 private landowners or land managers, five commercial hunting operators, three real estate appraisers, and two regulatory personnel in the spring and summer of 2012. The majority of private landowners interviewed were randomly sampled from all individuals owning more than 404 ha (1000 ac) in the Central Coast mountain range of California between Santa Clara County in the north and San Luis Obispo County in the south. I interviewed several landowners who owned or managed land in the southern San Joaquin Valley and in Mendocino County. I acquired regional ownership information from county assessors from the area from which to randomly sample landowners.

Due to the wide-ranging nature of semi-structured interviews, not all questions in the interview questionnaire were fully answered due to either time constraints, lack of immediate knowledge of the answer by the interviewee, or unwillingness of interviewees to answer certain questions. As a result, much of the results presented here are of a qualitative nature and do not constitute a generalizable knowledge of landowners in California. The results have served to inform future survey and interview efforts, which are currently ongoing.

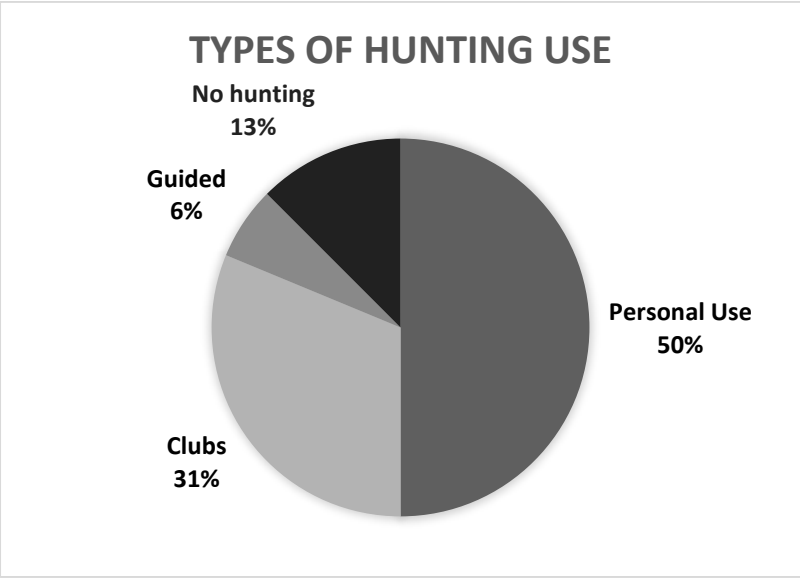
The questionnaire requested information from landowners about revenue earned from their property by various uses, and asked about management practices performed on the property for various business enterprises. Interviews revealed that most landowners did not keep detailed accounting records on the operations of their property. As such, information about net revenue of the property was difficult to assess primarily due to challenges of quantifying expenses such as labor, fuel or depreciation associated with the ranch operation. As such, interviews did not reveal whether land management efforts resulted in net revenue improvements in coming years. The estimates cited in this manuscript are assessments of gross revenue estimates unless expenses are specifically noted. The interview guide also requested information on common rangeland management practices such as planting food plots (usually agricultural crops planted specifically for wildlife to feed on), predator control, brush management, game species introductions, and habitat improvement practices (such as planting oaks or maintaining riparian zones). For a condensed list of questions asked during interviews, please refer to the following questionnaire.

3.3 Results

3.3.1 Economics of recreational use and the type of recreational activities pursued

In general, the interviews found that much of the recreational activity on private lands greater than 404 ha (1000 ac) was for hunting, although many of those properties, in particular those less than 2023 ha (5000 ac) oftentimes only allowed friends and family on the property for hunting. Overall, 87 percent of randomly sampled landowners had some recreational hunting on their property, with half of those landowners reserving hunting for personal or family use, 31 percent having hunting clubs, and 6 percent with guided operations (fig. 1).

Figure 1—Types of hunting use (or no hunting) on 16 randomly selected properties greater than 404 ha (1000 ac) in the Central Coast of California.



Results from interviews found large variation in earnings from hunting. Several landowners were primarily oriented towards livestock operations and had hunting clubs that earned marginal amounts of money (oftentimes just enough to pay for annual hay expenses or property taxes). Two properties were oriented towards hunting as a primary use, with relatively small or no income from livestock. One operation ran a simple hunting club with yearlong access, and earned gross income of $10/0.4 ha (1 ac), with minimal expenses that resulted in profitable operations, while the other maintained employees and facilities for the hunting club, which resulted in higher gross income on the property, but also substantial costs which led this property to operate at a loss. One property in particular operated both livestock and hunting operations at high efficiency, using a guide service for hunting, and as a result appeared to earn higher levels of revenue than any other properties. I did not encounter properties earning money from fishing or wildlife watching, although one large property was oriented primarily towards personal horseback-riding recreation.

The availability of habitat in the form of shrub cover, oak trees and grassland was an important need for the availability of hunting operations. Properties composed of almost entirely of grasslands had little wildlife availability for hunting operations, and generally viewed feral

pigs as pests that they sometimes would pay others to help manage. This suggests an interesting shift in perspective about pigs, with some landowners viewing them as a cost to a primarily livestock operation, while those seeking income from all available streams found a source of income from the presence of pigs.

In keeping with rangeland research (Huntsinger and others 1997), many landowners were not focused on earning income from their land and instead utilized the property for personal enjoyment. Others seemed content with their income as it stood from livestock operations alone and did not want to invest the time it would take to set up a hunting operation. The most common reason that landowners did not have hunting on the property was that they did not want hunters disturbing other aspects of their livestock operation.

Interviews confirmed that rangelands were marginal income generating operations when considering the cost of capital that is invested in the land. Operations trying to make income from the property grossed between $10 to $30 per 0.4 ha (1 ac), not including costs of labor and management. Land prices for rangelands in the Central Coast region oftentimes range from $1000 to $2000 per 0.4 ha (1 ac). Dividing annual gross income by land value only (the primary capital cost for these operations) results in estimated gross returns from operations in the range of 0.5 percent to 3 percent, not including the cost of labor and other expenses. These estimates should be viewed as rough approximations given that interviews did not gather detailed accounting records, but instead relied on landowner's general estimates at the time of the interview. Furthermore, these estimates do not include capital appreciation of land over time. Nonetheless, the results show the relatively small returns that can be expected in annual cash flow from rangelands in the central coast of California.

3.3.2 Are conservation practices incentivized by recreational use?

Interviews found that in many cases landowners were managing for wildlife-associated recreation, but at times utilized practices that could have some negative repercussions for other aspects of the ecosystem. Although some landowners utilized practices that conservation professionals recommend, such as leaving cover for wildlife and minimizing risks of invasive species, many others were interested in attaining benefits for desired wildlife species through more intensive management actions such as the planting of food plots or introducing or maintaining non-native game animals such as feral pigs and wild turkeys.

I found that most landowners maintained water resources, such as stock ponds and water troughs, which were utilized for both livestock and wildlife populations. It appeared as though the primary motivation for installing and maintaining water sources was for livestock operations, although several of these landowners had implemented wildlife escape ramps into troughs, which helped reduce mortality of wildlife that fell into troughs. At least one landowner implemented water sources at ground level, designed specifically for wildlife access.

Several landowners implemented practices to maintain habitat for wildlife, such as creating and maintaining brush piles for wildlife as well as leaving cover for wildlife by reducing grazing in certain areas. Several landowners conducted some level of brush management as a means of increasing grass and forb production; although, areas where brush was cleared tended to represent very small percentages of property ownership. There did not appear to be a sufficient

benefit to justify the expense of widespread clearing of brush. I did not encounter landowners who had implemented any planting of oaks, nor did I find many landowners managing grazing to protect riparian areas.

I found evidence that two landowners purposefully either introduced or maintained exotic game species to improve hunting opportunities as part of the recreational use of the property. In one case, landowners introduced wild turkey (*Meleagris gallopavo*) to their property to establish a population for hunting, and several other properties benefitted from the regular occurrence of wild pigs (*Sus scrofa*) on the property, sometimes taking actions such as planting food plots or limiting harvest to help maintain high populations for the hunting operation.

In several cases, the landowner planted agricultural crops as food plots for wildlife. The most common species planted was barley, which was oftentimes planted to support or enhance feral pig populations. In at least one case, areas that were previously planted had converted to dense infestations of the invasive species, yellow starthistle (*Centaurea solstitialis*), and one patch was planted over a small remnant patch of relatively rare native grassland. At the same time, one landowner was familiar with the risks of invasion of weeds following food plot planting, and as a result carefully selected locations for such plot and restricted these areas to previously cultivated areas.

Most landowners had a lack of knowledge of current harvest ecology recommendations for hunting, which oftentimes recommend the harvest of female deer to enhance production and increase the buck to doe ratio of the deer population. Doe harvest can result in increased deer production through increasing fawn survival due to reduction in competition by other does (known as a density-dependent response) (Macaulay and others 2013). The main exception to this trend was found in landowners and land managers of property enrolled in the Private Lands Management (PLM; see discussion) program, who had a much better understanding of harvest ecology, and at the same time implemented various habitat improvement practices that benefited wildlife. One PLM land manager safeguarded fawning areas from overly intensive grazing and implemented regular deer surveys on his property, allowing him to document changes in deer populations over the years as well as population parameters such as fawn survival rates. Others implemented re-vegetation of riparian zones, reduced grazing intensity on certain pastures, and removed old, unused barbed wire fences from the landscape.

Most landowners practiced some form of predator control, and several complained that government regulation had hampered their ability to control predators. Much concern was directed towards state laws implemented since 1972 that have significantly limited the ability of landowners to use lethal control of mountain lions, culminating in the passage of Proposition 117 in 1990 (CDFW 2007). Predator control can be a controversial practice, and landowners noted that many state Department of Fish and Wildlife biologists advocated that game species populations were only minimally affected by predation. However, at least one landowner performed regular November deer census surveys and documented an improved fawn survival rate from .28 fawns per doe to 1.1 fawns per doe as a result of an intensive 2-year effort to reduce coyote populations, which is consistent with studies of coyote control on white-tailed deer fawn survival (Beasom 1974). Although the practice of predator control of coyotes is opposed by various environmental groups (Project Coyote 2014), no landowners were able to eliminate coyote populations on their property despite years of intensive efforts in control. As such, it appears that the persistence of coyotes on the landscape is not threatened, and that control efforts serve to temporarily reduce the density of coyotes on the landscape.

3.4 Discussion and conclusion

Interviews with landowners and land managers in California, uncovered large variation in earnings from recreational hunting operations, which suggests that there is an inefficient market where the value of recreational hunting rights is not well established or known. Although some landowners were implementing habitat improvement practices on their property, I found that most did little habitat improvement and instead either harvested game that was already occurring on the property, or focused on planting food plots and performing predator control. This suggests a high potential for government efforts through Cooperative Extension to inform and educate landowners about the market potential as well as best practices for wildlife management for recreational hunting enterprises. Because the most often cited reason for not allowing hunting on the property was concern about hunters disturbing livestock operations, there appears to be a significant opportunity to educate hunters about how to minimize disturbances to ongoing ranching operations.

With the exception of properties enrolled in the PLM program, there appeared to be a lack of knowledge of many wildlife management practices that are currently recommended, such as deer census counts, estimation of fawn survival, and collection of information about age of harvested deer. The PLM program, administered by the California Department of Fish and Wildlife (CDFW), is a government program that has the potential to improve conservation outcomes associated with recreational use. This program was developed in the 1980s and allows landowners greater flexibility over seasons and harvest in return for improved conservation practices. Yet, despite the potential for benefits for both landowners and natural resources, it has relatively low adoption rates with 101 landowners currently enrolled in the program statewide, representing just over 404 686 ha (1 million acres) of private land (approximately 2 percent of private land in California) (CDFW 2014). Although participation has increased over the years, (in 1988, 54 landowners covering 284 494 ha (703,000 ac) were enrolled) there appears to be a large potential to expand this program, which appears to result in increased implementation of conservation practices. One concern for future expansion that may need to be addressed is the cap on elk tags, which is currently set at 250 tags per year. Expanded adoption of the PLM program could also result in increased ability of ranchers to maintain their livestock operation by diversifying income streams (Huntsinger and others, 2014)

In addition, results confirm the findings of Smith and Martin (1972), which showed that land prices for rangelands are far higher than would be expected given the income that can be made from ranching operations. This is likely due to the non-monetary enjoyment and benefits that landowners gain from the property, as well as anticipated capital appreciation in the value of the land in the future. The lack of detailed accounting by land managers of the expenses associated with management practices and the difficulty in attributing revenue benefits from particular management practices makes an economic analysis of management practices difficult. Furthermore, the time lag between the implementation of management practices and the potential for increased returns to recreation further complicates making direct connections between management practices and recreation benefits.

In conclusion, while landowners and managers implemented some conservation practices to enhance the recreational values on their properties, there appeared to be many opportunities to

enhance this work. There also appeared to be a lack of knowledge about the income-earning potential of hunting, as well as significant distrust of hunters by landowners. Ongoing outreach efforts could serve to improve relationships between hunters and landowners and increase the conservation practices performed on properties, which could improve the potential to earn revenue from recreational use.

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Chapter 4

Estimating Density and Structure of a Columbian Black-tail Deer Population Using Camera Traps

Abstract: Columbian black-tail deer (*Odocoileus hemionus columbianus*) are one of the most widespread and economically important wildlife species in the Western U.S. due to their value for both wildlife viewing and recreational hunting. Managing and recommending hunting harvests of these deer has been challenging due to difficulty in assessing population density and sex ratios of these deer using commonly practiced methods of estimating population parameters such as spotlight surveys. This study develops a model to estimate abundance, density, sex ratios, and home ranges of Columbian black-tail deer. The study also analyzes the effect of bait on the estimates to provide recommendations in how to best assess deer density with camera traps. Thirteen cameras were placed on all known water sources of a 2,500 acre private ranch and bait was provided approximately once every month at each camera location to increase detections and provide the ability to assess the impact of bait on estimates. Over 10,000 visits occurred between January 1, 2012, and January 21, 2013, and 50 deer were individually identified using ear notches or antler characteristics. This study uses Bayesian spatial mark-resight models to estimate abundance, density, and sex ratios of deer. This study estimated deer density at 9.6 deer/km^2 (26 acres/individual), with females occupying much smaller home ranges of 196 ha (484 acres) compared to males which were estimated to use 1,147 ha (2,834 acres). Males composed 10.3% of the population with the remainder made up of females and fawns without antlers. The use of bait increased baseline encounter rates from 5.4 photographs/week to 19.9 photographs/week. This study provides a modeling method and information on the use of bait that landowners can use to assess the population parameters of deer, which is crucial to formulate harvest strategies to properly manage deer populations for both economic return and ecological sustainability.

Keywords: Hunting, bait, Bayesian, private land, sex-ratio, population parameters

4.1 Introduction

Conducting a population inventory for hunted wildlife populations is a crucial first step in developing harvest guidelines to achieve both economic and ecological goals (Silvy 2012). The Columbian black-tailed deer is a major source of game meat for hunters as well as a revenue generating source for private landowners who benefit from selling private land access rights (Macaulay in revision at Land Use Policy). Furthermore state wildlife agencies earn a significant portion of their revenues from sales of hunting licenses and deer tag sales, which supports efforts such as research, habitat improvement, and enforcement of game laws. The geographic range of the Columbian black-tailed deer (*Odocoileus hemionus columbianus*) spans from southern British Columbia to Santa Barbara County in California, and as far east as the Cascade and Southern Sierra Nevada Mountains (Wallmo 1981). The Columb ian black-tailed deer is a

keystone species, serving as a major food source for large and medium-sized predators, and population fluctuations have the ability to influence vegetation characteristics of the landscape (Singer & Renkin 1995; Ballard et al. 2001).

One of the most critical pieces of information to manage hunting harvest of a game species is a good understanding of the species' population characteristics. Unfortunately, commonly practiced methods of estimating population parameters of deer such as road-count or spotlight surveys, are less efficacious in black-tail deer habitats, which oftentimes includes dense shrub or forest-dominated habitats (McCullough 1993; McCullough et al. 1994). Researchers have recommended an independent estimate of population size to guard against serious errors in herd composition count results (McCullough 1993).

Camera traps are a common and relatively inexpensive method for gaining an improved understanding of presence of deer, but models for estimating abundance, density and sex ratios for a partially marked or unmarked population have only recently been developed (Chandler & Royle 2013). Although an increasing number of studies are using this method to estimate population parameters, no studies have done so with the Columbian black-tailed deer. Furthermore, estimating population parameters of Columbian black-tail deer is a major difficulty for wildlife managers. For example, in California, the Department of Fish & Wildlife describes the need to improve population estimates as one of their primary goals for the future management and conservation of deer (CDFW 2015b). The development of this estimation method is important to improve the ability of wildlife managers to develop harvest quotas based on an improved understanding of black-tailed deer populations.

In performing camera trap studies, many managers will provide bait to attract the species of interest to camera locations to improve detection probability and reduce the amount of time needed to develop population assessments. While baiting undoubtedly increases the frequency of visits to camera locations, the question of how this baiting may affect encounter rates for black-tailed deer remains unknown. By better understanding the impact of bait on encounter probability, managers may be able to improve estimates of population parameters, use fewer cameras, or place cameras out for shorter time periods.

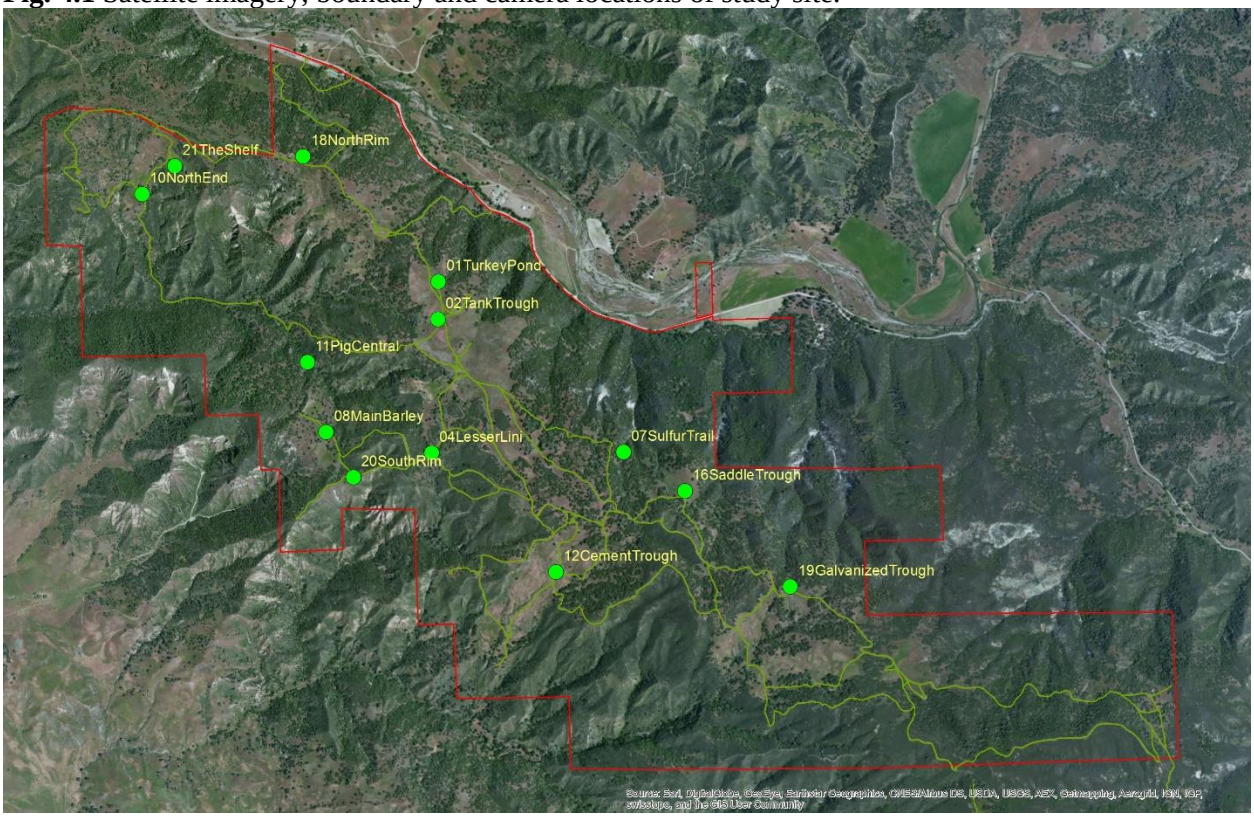
This study develops a model to estimate population parameters of Columbian black-tail deer from camera trap images and analyzes the effect of bait on the estimates. Specifically, the model estimates abundance, density, sex ratios, and home ranges of deer, which can be used to inform harvest criteria and to better understand the carrying capacity of the land for deer.

4.2 Methods

4.2.1 Study area

The study was conducted on a private ranch in southern San Benito County, CA (Figure 1). The property encompasses about 2,540 acres in the eastern foothills of the Central Coast Range of California and is characterized by a mosaic of grassland, chaparral, and woodland vegetation types. Elevation on the property ranges from about 1,700 feet to 3,600 feet, and is considered a Mediterranean semiarid climate, with cool wet winters and hot dry summers. Average annual precipitation is 41-46 cm (16-18 inches). The study site center is located at approximately 36°22'20" N, 120°55'26" W (Voelker 2010) .

Fig. 4.1 Satellite imagery, boundary and camera locations of study site.



4.2.2 Data

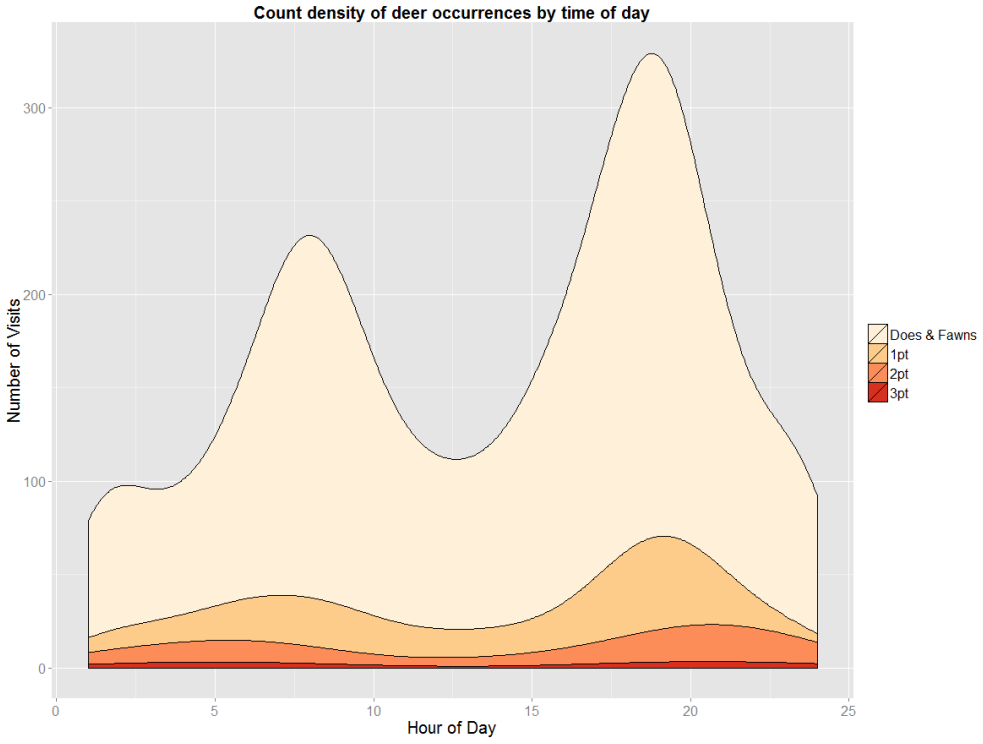
Thirteen motion-activated cameras were placed at known water sources (both natural and man-made) and on one well-traveled game trail. Bait was provided approximately once every month (on the dates of May 24, June 22, July 17, and August 23, 2012) at each camera location to increase detections and provide the ability to assess the impact of bait on estimates. RECONYX camera traps models PC900 were used (RECONYX, Inc. Holmen, Wisconsin 54636). All camera traps were placed at a height of approximately 2 meters and about 4 meters away from bait and water. The cameras were set to high sensitivity, no delay, continuous operation, and one photo per trigger, which captured one photo every other second if there was movement in front of the camera. The bait used was 30 pounds of commercial pelleted pig feed (16% protein). Each camera station also included a salt lick in front of it nearly continuously throughout the study. The bait would often last for 1-2 weeks before it was entirely consumed, resulting in approximately 250 trap days with bait present. Although the bait was only present for 18% of the trap days, 42% of all deer visits occurred while bait was present, resulting in much higher detection probability when bait was present (fig. 4.4).

The camera array has been in place on the study site since 2008, but this analysis is constrained to images obtained to the 123 day period between May 24, 2012 and September 24, 2012 due to an influx of male deer from outside the area after that date, which caused a substantial violation of the assumption of a closed population for the model.

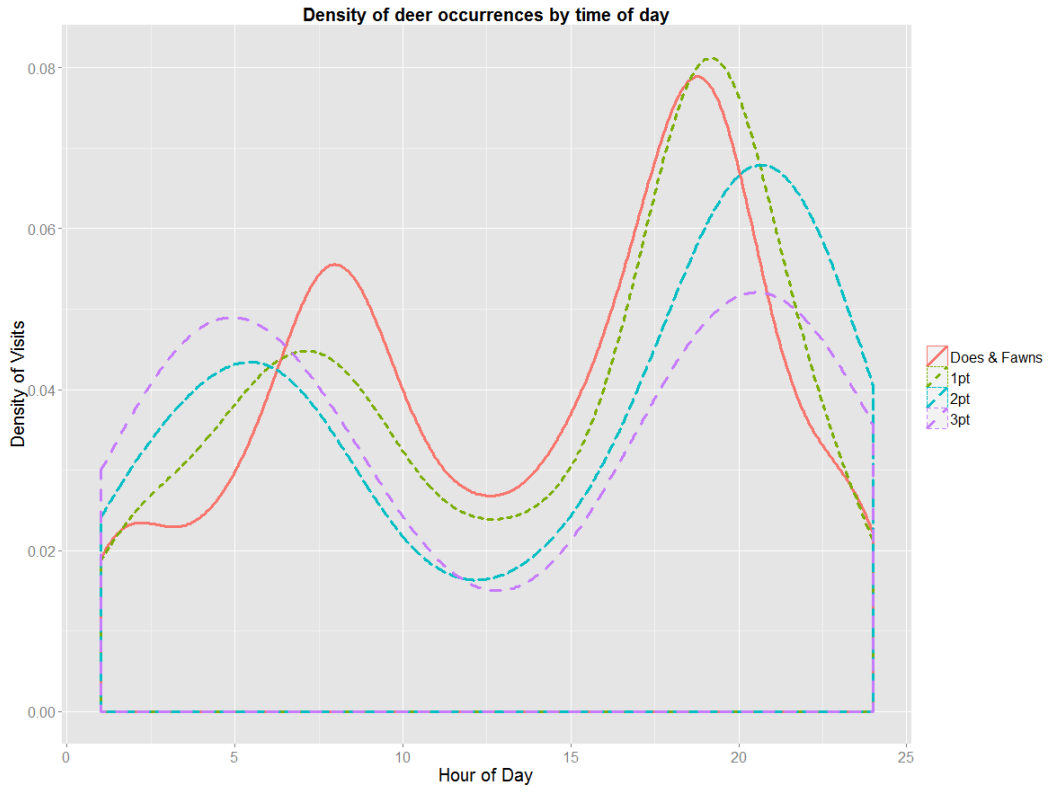
Photographs were analyzed using the concept of a visit, which was defined by any grouping of photographs of deer that had less than a six minute gap between photographs. This

six minute gap was determined based on extensive analysis of the deer photographs over several years which found that after this length of time without a photograph a different group of individuals was usually visiting. Images were subsetted to include the first and last photograph of a visit of deer, and excluding repeated images of deer that did not add more information to the visit data. This subsetting resulted in 41,826 photos of deer for review and data entry. During the study period, there were 5,451 visits by deer. Deer images were reviewed for identifiable characteristics usually from a combination of antler characteristics and/or naturally occurring ear notches, resulting in 19 identified bucks and 9 identified does during the 123-day sampling period (table 4.1). For the purposes of this study, fawns and does were treated as one group, and deer with visible antlers were treated as males in a separate group. Forty-one percent of visits included the occurrence of an identified deer, and most of the bucks identified had one point on each antler (table 4.1). The number of visits by marked individuals ranged from seven to 229, with an average of 94 visits recorded for each individually identified deer. The deer exhibited a bimodal crepuscular visitation pattern throughout the day, with does and fawns using more of the daylight hours than bucks. More mature bucks (as determined by antler points), shifted towards more nocturnal vistiation patterns (fig.4.2). Almost all of the individually identified deer were photographed on multiple camera traps, with the average marked deer being photographed at 4.6 sites. The thirteen cameras which operated over the 123 days of the study resulted in a total of 1,401 trap-days, accounting for a 12% failure rate, or 198 trap-days, in which cameras either malfunctioned or lost battery charge. Malfunctioning cameras were accounted for in the model by multiplying the probability of detection by the percentage of time that the camera was operational in a given week.

Fig. 4.2 a) Count density of deer occurrences by type of individual by hour of day; **b)** density of deer occurrences by type of individual by hour of day.



a)

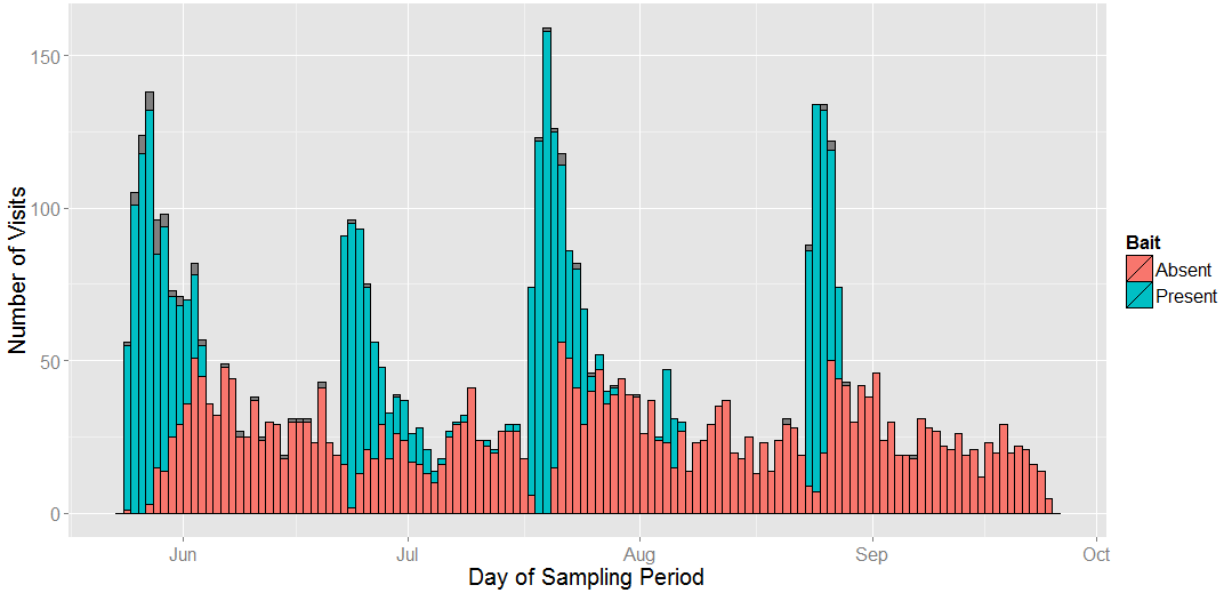


b)

Table 4.1 Visits to camera trap sites by identifiable and unidentifiable deer over a 123 day period in 2012 at the study site in San Benito County, California.

	Name of Deer	01TurkeyPond	02TankTrough	04LesserLini	07SulfurTrail	08MainBarley	10NorthEnd	11PigCentral	12CementTrou	16SaddleTroug	18NorthRim	19GalvanizedT	20SouthRim	21TheShelf
Does	subtle	34	9	0	0	0	0	9	0	0	51	0	0	0
	r	10	33	69	0	20	1	1	0	0	1	0	94	0
	lefty	1	0	0	1	0	0	0	2	0	0	62	0	0
	dot	0	0	0	0	0	45	0	0	0	0	0	0	13
	gama	0	0	0	0	0	4	0	0	0	68	0	0	96
	smlovely	0	0	12	0	26	0	4	0	0	0	0	15	0
	m	99	48	0	0	0	0	0	0	0	0	0	0	0
	chop	0	0	0	0	0	0	0	0	0	0	57	0	0
	lovely	4	1	0	0	0	0	0	0	0	173	0	0	0
1 point bucks	bele	1	0	0	0	0	20	14	0	3	14	0	0	7
	crooked	1	1	1	1	0	0	0	3	0	130	24	0	0
	fafa	1	2	0	21	0	1	0	40	13	0	14	0	0
	jacky	0	0	0	2	0	0	0	1	111	4	26	0	0
	lulu	2	0	0	0	0	0	0	0	0	0	5	0	0
	lumpy	1	6	3	0	0	1	1	0	0	30	0	3	31
	n	4	6	0	0	4	11	11	0	0	1	0	0	1
	nini	20	8	1	0	0	0	0	0	0	1	0	0	0
	shred	2	1	0	45	0	0	0	15	46	0	51	0	0
	smn	0	0	19	16	1	17	0	1	0	0	0	10	24
	zozo3	0	0	0	0	0	0	0	0	0	0	9	0	0
	zozobig	0	0	0	0	0	0	0	9	0	0	140	0	0
	zozosm	0	0	0	0	0	26	0	2	1	0	17	0	2
2 point bucks	jackson	0	1	0	25	0	0	0	32	18	1	17	0	0
	jaji	11	1	0	0	0	0	0	0	0	64	0	0	0
	kuku	0	3	0	5	0	0	0	10	0	32	1	0	2
	radar	1	1	0	0	0	60	2	0	0	9	0	0	6
	zuzu	4	3	0	2	0	4	0	1	0	58	0	0	18
3 point buck	mo	0	0	0	0	0	26	0	0	0	8	0	0	15
	Sum of named deer	196	124	105	118	51	216	42	116	192	645	423	122	215
	Unnamed does	506	507	302	153	133	626	262	251	191	254	754	285	405
	Unnamed fawns	53	82	314	14	83	359	103	82	176	264	315	119	358
	Unnamed bucks	2	2	0	15	1	8	0	7	1	18	5	0	2
	Total counts	758	715	720	299	268	1206	406	455	556	1174	1488	526	979

Figure 4.4 Number of deer visits per day during the sampling period color coded by presence or absence of bait. The pulses of bait additions are apparent on May 24, June 22, July 17, and August 23.



Population parameters were estimated using a spatial mark-resight model for a partially marked population (Chandler & Royle 2013; Royle et al. 2013). This model is implemented in a Bayesian framework and uses Metropolis-within-Gibbs Markov chain Monte Carlo (MCMC) sampling to update the latent encounter histories of unmarked animals. Parameters of the detection process are estimated using trap-specific encounter histories of individually identified deer at various camera stations. The model yields overall population estimates of sex ratio, density of deer, and home range size.

The model functions by assuming that each individual *i* has an activity center, s_i, and that each of these activity centers are distributed uniformly in the geographic state-space, *S*. *S* is the study area, and is sufficiently large to include all individuals that could have been photographed. As individual deer can be detected multiple times at each camera location, we assume that the number of detections of an individual *i* at trap *j* and trapping period *k*, y_{ijk}, is a Poisson random variable with mean encounter rate λ_{ijk}. The encounter rate, λ_{ijk}, is assumed to decrease as the distance, d_{ij}, from trap *j* to the individual's activity center, s_i increases. This decay of encounter rate is modeled using a half-normal function

$$\lambda_{ijk} = \lambda_0 \times \exp\left(-\frac{d_{ij}^2}{2\sigma^2}\right) \quad (4.1)$$

where λ_0 is the baseline trap encounter rate, or the expected number of times an individual *i* would encounter the camera if the camera was located at its activity center s_i, and σ is the scale parameter of the half-normal function, which is related to the radius of the animal's home range, i.e., as σ increases, the encounter probability, λ_{ijk}, increases at greater distances from the animal's activity center. For this study, there were 13 camera stations (*j*) and 18 trapping periods lasting one week each (*k*).

For unmarked individuals, we cannot observe individual encounter histories, but we only observe the summary counts of unmarked deer at each camera trap and sampling occasion, n_{jk}.

Assuming that the encounter process for marked and unmarked animals is governed by the same parameters, the unobserved encounter histories of unmarked animals, $\boldsymbol{y}_{ujk}$, can be described with the following multinomial distribution (Chandler & Royle 2013):

$$\mathbf{y}_{ujk} \sim \text{Multinomial}\left(n_{jk}, \frac{\boldsymbol{\lambda}_{ujk}}{\sum_{u=m+1}^{M} \boldsymbol{\lambda}_{ujk}}\right).$$

To estimate the number of unmarked animals, the model employs data augmentation (Tanner & Wong 1987; Royle et al. 2007; Royle & Dorazio 2012). The n_{jk} records of unmarked deer are assigned as described above to M hypothetical individuals, where M is chosen larger than the expected population size, so as to not truncate the estimate of the number of unmarked individuals in the population. The model then estimates a latent individual covariate, z_i, which is 1 if the animal is part of the population, and 0 otherwise using a Bernoulli distribution,

$$z_i \sim \text{Bernoulli}(\Psi),$$

where Ψ is the probability of an individual being part of the population.

Because natural markings were used to identify deer, the actual number of marked animals present in the area is also not known with certainty as a deer with distinguishing markings may have evaded detection on the camera trap array. As such, a similar data augmentation approach is also used to estimate the size of the marked population. The total population is estimated by summing the z_i for both the marked and unmarked populations. Density of deer is derived by dividing the estimated total population by the study area.

Under data augmentation, the observation model becomes

$$Y_{ijk} \mid z_i \sim \text{Poisson}(\lambda_{ijk} \times z_i),$$

so that the expected encounter rate of an individual with $z_i = 0$ is also 0 – that is, a non-existing individual cannot be observed.

Male deer and females/fawns are treated as two distinct populations due to differential proportions of identification (nearly all photographed males were identified, while only a partial sample of females were) and distinctive home ranges. Treating these two groups as separate populations allows for the calculation of the proportion of males in the deer population, which is calculated as the sum of z for the male population divided by the entire population.

To define the state-space S, we used a 1.5 km buffer to form a convex hull polygon surrounding the 13 camera trap locations. This distance appeared to be sufficient, considering model estimates of sigma and the recommendation to buffer the trap array by about 3xsigma (Royle et al. 2013). This buffered area is 2,281 ha (5,636 acres). We divided the total population estimate, N (the sum of all z_i), by this area to estimate density.

The presence and influence of bait on baseline encounter rate was modeled using a log-linear function

$$\lambda_0 = \exp(\alpha + \beta * \textit{Bait})$$

where *Bait* is a value between 0 and 1 denoting the percentage of the sampling period that bait was present.

The Gaussian encounter probability model (eq. 4.1), which is used to account for the decay of encounter probability given the distance from the animal's activity center, can be used to estimate home range. The implied model of space usage assumes a circular home range that follows a bivariate normal model. As such, we can estimate the radius that accounts for 95% of the area used by the animal from estimates of sigma (for details, see Royle et al. 2013). The home range area is then calculated using the formula for the area of a circle, πr^2.

This model assumes the following characteristics of the data: 1) demographic population closure, 2) marked individuals are a representative sample of the study population, 3) there is no mis-identification of marks, and 4) there is no loss of marks. Although it is almost certain that some deer emigrated, immigrated, or died during this sampling period, violating the closure assumption, truncating the sampling period to a four month period prior to the breeding season minimizes violation of this assumption. Because deer were marked from ear notch and antler characteristics, male deer with antlers were likely identified at a higher rate than female deer, which could only be identified if they had distinguishing ear notches. This marking method does not hold to the assumption of identifying animals that are representative of the population, however, this problem is addressed by treating males and females as two separate populations, allaying the concern about differential marking of males and females. At the same time, combining both data sets into a joint analysis allowed us to have male and female deer share certain parameters, such as the intercept alpha and the effect of bait, beta, of the log-linear baseline trap encounter rate model. Photographs were double checked for accuracy in various iterations of data cleaning. Difficult to identify deer were re-checked for accuracy. Although deer do lose their antlers annually, this occurs in February, not affecting this population, and deer ear notches can be added, but not removed. In all of the marked females, ear notches were identified within the first 9 days of the 123 day study period, so that we did not include animals as marked if they developed ear notches partway through the sampling period, or if a deer with ear notches appeared later in the study period. There was only one example of the former, where a fawn with fresh cuts in its ears was observed. There were no newly observed deer with ear notches partway through the sampling period.

The model was implemented using a Metropolis within Gibbs algorithm adapted from models contained developed in Royle et al. (2013) Sollmann et al. (2012), and Rutledge et al. (2014) in the software R 3.2 (R Core Team 2015; Royle et al. 2015). Six MCMC chains of the algorithm were run with 50,000 iterations each, with the first 20,000 iterations of each chain discarded as burn-in, resulting in 180,000 iterations used in the posterior distribution estimates.

Results

Black-tail deer abundance for the 1.5 km buffer surrounding the cameras was estimated to be 226 individuals (SE 20) with a density of 9.9 (SE 0.91) individuals/km^2 (10.16 ha/individual or 25.1 acres/individual). The model estimated that 10.5% (SE 0.01) of the population was composed of male deer with visible antlers, and 89.5% female or fawns without visible antlers. Baseline trap encounter rate λ_0 was influenced by the presence of bait. In the full presence of bait, the baseline encounter rate at the activity center of an animal was 18.8 photographs/week, while in the absence of bait, λ_0 dropped to 5.1 photographs/week. Home

ranges during this summer period were estimated to be 2 km^2 (494 acres) for female deer and fawns, while males were estimated to have a much larger home range of 11.47 km^2 (2,834 acres). Summary statistics and percentiles of posterior distribution of parameter estimates are contained in Table 4.2.

Table 4.2 Summary statistics and percentiles of posterior distribution of parameter estimates for black-tailed deer on the study site in San Benito County, California, in summer 2012. See text for definition of variables.

			Percentile		
Parameter	**Mean**	**(SE)**	**2.5**	**50**	**97.5**
σ (female/fawn)	0.33	(0.01)	0.31	0.33	0.34
σ (antlered male)	0.78	(0.02)	0.75	0.78	0.81
α	1.64	(0.05)	1.55	1.64	1.73
β	1.30	(0.04)	1.23	1.30	1.37
φ	0.11	(0.01)	0.09	0.10	0.13
Ψ (female/fawn)	0.75	(0.08)	0.59	0.75	0.92
Ψ (antlered male)	0.17	(0.07)	0.06	0.16	0.32
m (female/fawn)	13.97	(2.78)	10.00	14.00	20.00
m (antlered male)	19.00	(0.00)	19.00	19.00	19.00
N	226.34	(19.94)	188.00	226.00	267.00
Density (no./km^2)	9.93	(0.92)	9.34	9.95	11.71

The Gelman-Rubin statistic was calculated using the R package 'coda' to assess chain convergence, and found that all parameters except for σ for males achieved values below 1.11 indicating chain convergence (Plummer et al. 2006). The chains estimating σ for males had a Gelman-Rubin statistic of 1.8, which is likely due to greater variation among male deer home ranges leading to wider standard error estimates. Although these estimates did not reach a convergence value below 1.1, the difference in estimates of the most divergent chains was quite small: between 0.765 and 0.8.

Discussion

This study demonstrated a method for assessing deer density using camera traps that is not prone to the estimation pitfalls of methods such as spotlight or road-count surveys, or the expense and risks of collaring live animals. This represents an important next step in improving management of harvested game animals, as is evidenced by the high priority that the California Department of Fish & Wildlife is placing on improving estimation of game animals such as deer (CDFW 2015b). The model findings contribute to our understanding of black-tailed deer population parameters, especially of home range of male deer, which are often not evaluated in population studies. The sex ratios developed in this analysis confirms the relatively low numbers of males to females in these populations (CDFW 2015a).

The density estimates in this study (9.9 deer/km^2) are substantially lower than those in a recent study using fecal DNA to quantify density of deer (Lounsberry et al. 2015). The authors of that study note uncertainty involving calculating the effective area sampled, and note that their estimates are meaningful only within the context of similarly calculated estimates. Because this study explicitly incorporates spatial movement of individuals, as well as the area of the space in which animals are moving, our density estimates are not confounded by problems associated with the effective area sampled, and should be a more reliable result.

Female (and fawn) deer home range estimates of 2 km^2 (494 acres) also differed from a recent telemetry study of black-tailed deer, which estimated average seasonal home range size for female deer to be 0.71 km^2 (175.4 acres) (Wittmer et al. 2015). This is likely due to that study's use of a local convex hull method to calculate home range instead of an assumption of a bivariate normal distribution. The local convex hull method develops a finite region of use, accounting for features of the landscape that are not used, such as inaccessible areas like lakes or fenced areas. The bivariate normal distribution of use, used in this study, implies a circular home range with symmetric circular contours of use intensity, which could easily include a much greater area than is actually used by the individual.

This study found that use of bait increased detections by a factor of four. This suggests that the use of bait could shorten the sampling period needed to develop estimates, which could be useful in adhering more closely to the assumptions of closed systems that is required by many density estimation models. However, bait can also disproportionally affect the detection of different segments of the population, leading to over, or under-estimation, of particular segments of the study population. For example, if female deer and fawns preferentially use bait compared to mature bucks, the use of bait could bias results in favor of females. We do not have evidence of this occurring, however, it is important to consider tradeoffs such as these when utilizing bait in a study.

This method assumes a closed system, and that assumption was almost surely violated due to the immigration, emigration, and death of animals during the 123-day study period. We addressed this to the best of our ability by using a sampling period that was relatively short, and ended prior to the onset of the breeding season in October. One finding of note, is that bait could be a beneficial tool to help shorten the sampling period and adhere more closely with the closure assumption, although other challenges could arise if bait artificially attracts animals from outside the study area. This method also requires that camera are placed closely enough together so that the same individuals occur on multiple cameras, and although it can be used to develop estimates in the absence of marked individuals, markings help to increase the accuracy of the estimates.

The population density model developed here has great potential for future development and implementation in large-scale studies of wildlife populations. In addition to helping develop management goals for harvested wildlife, it can be used to identify significant declines or increases in wildlife populations over time. With the increasing sophistication and decreasing costs of camera traps, this method has wide applicability for use in the future.

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Appendices for Chapter 2

Appendix A, Differences between this analysis and U.S. Fish and Wildlife Survey reports:

Some similar statistics are reported in the National Survey of Fishing, Hunting, and Wildlife-Associated Recreation (National Survey) reports from 2001, 2006, and 2011. However, the analyses found in those reports and this article differ in three important ways:

1) the estimates in this manuscript are much more precise due to the pooling of three survey years,
2) the estimates in my article treat missing data by using listwise deletion method, which is generally more supported method of dealing with missing data (Allison 2002; Howell 2007), and
3) the estimates in this manuscript report standard error estimates that are not provided in those reports.

Furthermore, this manuscript contains numerous calculations that are not contained in the public reports. These include estimates of the per-acre payments for different recreation types and estimates showing contrasts between how different recreation types utilize properties of different size classes. This is very important for understanding how land-use may be affected by different types of recreation. Additionally, in making revisions to the manuscript, I have performed additional analyses of 1) the participation rate of recreational users who pay fees for different types of private land access, 2) the proportion of lessees or owners who form clubs to own or lease their land, and 3) the average number of people in those clubs, and 4) an analysis of regional variation in this land use. None of these estimates can be calculated from the publicly available reports.

To illustrate how this analysis is improving upon already published estimates, I provide a table (below) that compares (1) estimates published in USFWS National Survey reports (Table A.1.a), (2) estimates that I calculated on a single year survey basis with standard error estimates and treating missing data using listwise deletion (Table A.1.b), and (3) estimates that I report based on pooling the three survey years (Table A.1.c). The table illustrates how single year estimates have varied considerably, with an estimated 440 million acres leased for hunting in 2011, which is approximately double the estimate from 2006 and 2001. Similarly, fishing leases were estimated to cover 1.3 million acres in 2001, which is far below the estimates of more than 9 million acres in 2006 and 2011. Pooling these three years of data increases sample sizes and provides a far more accurate estimate of the land area used for wildlife-associated recreation than has been provided in the public reports. While Table A.1 provides an example of the improvements to the estimates that occur in acreage utilized, a similar improvement in precision occurs across all the estimates in this manuscript as a result of the increased sample size obtained by pooling three survey years.

Secondly, the estimates provided in the public reports in some cases treated missing values in a problematic way. In calculating acreage estimates, one must divide the number of acres reported by the number of individuals in the club that owns or leases that property to avoid

double counting of acreage. In several instances, there is data on the number of acres owned or leased, but missing data for the number of people in the club. The government calculations treated these instances as having a club of one individual, which likely overestimates the number of acres used, especially in cases where the missing data would have contained many members in a club. The analysis performed in this paper used a more traditional approach in dealing with missing values by using list-wise deletion and omitting these observations from calculations (Allison 2002; Howell 2007). The government's analysis method resulted in an estimation of the hunting lease acres in 2011 of nearly twice the amount of previous years, very much influenced by this treatment of missing data. Correctly omitting missing data yielded results that were in line with previous years' estimates, which can be seen in Table A.1.b.

Finally, the National Survey reports report only a few selected standard error estimates. Although the reports contain parameters for calculating approximate standard error for some high-level estimates, they do not include parameters to calculate error associated with the estimates that are provided in this manuscript.

Table A.1: Estimated acres owned & leased for wildlife-associated recreation (in thousands of acres). Estimates with large deviation from other years are in bold.

a) Nationally Reported Land Area Used for Wildlife-Associated Recreation (in millions of acres)

	Fishing Own	Hunting Own	Wildlife-Watching Own	Fishing Lease	Hunting Lease	Wildlife-Watching Lease
2001	24.0	117.8	44.1	**1.3**	225.3	10.5
SE	*unreported*	*unreported*	*unreported*	*unreported*	*unreported*	*unreported*
2006	33.4	134.3	38.3	9.6	216.8	11.9
SE	*unreported*	*unreported*	*unreported*	*unreported*	*unreported*	*unreported*
2011	25.2	155.2	39.4	10.0	**420.0**	3.6
SE	*unreported*	*unreported*	*unreported*	*unreported*	*unreported*	*unreported*

b) Author's estimation dropping missing values and reporting standard error

	Fishing Own	Hunting Own	Wildlife-Watching Own	Fishing Lease	Hunting Lease	Wildlife-Watching Lease
2001	24.0	117.8	44.6	**1.3**	224.8	10.5
SE	7.4	38.3	12.5	0.6	81.3	6.4
2006	33.4	134.3	39.6	9.3	201.4	11.9
SE	7.1	22.8	9.4	3.9	52.4	6.8
2011	25.2	155.2	39.4	9.7	234.2	3.6
SE	6.3	27.2	11.2	4.6	67.0	2.0

c) Author's estimation based on pooled data 2001-2011

	Fishing Own	Hunting Own	Wildlife-Watching Own	Fishing Lease	Hunting Lease	Wildlife-Watching Lease
Pooled Estimate	27.5	135.8	41.2	6.8	220.1	8.7
SE	4.0	17.4	6.4	2.0	39.2	3.2

References for Appendix A

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Appendix B, Ownership Data:

The survey questionnaire requested a single value for the amount spent to own land, which included mortgage payments, down payments, taxes and maintenance expenses. The inclusion of these various payments into a single amount aggregates different accounting concepts (capital expenditures, capital costs, and operating costs) into a single value. Down payments and the portion of mortgage payments that pay for the principle could be considered capital expenditures invested into a fixed asset (the land), or under some circumstances could be considered capital costs. Taxes, maintenance, and interest paid are generally considered operating costs, (although some may be considered capital costs). Additionally, some individuals who own land, may have inherited it, or already paid for it, thus reporting only tax and maintenance costs even though they may own very large properties. Finally, capital appreciation and the opportunity cost of capital are not included in this value. As a result the values for ownership are difficult to compare to other standard measures, and exhibit relatively large standard errors. I suggest an improvement to this question so that future analyses can create estimates comparable to general accounting measures. Nonetheless, these estimates provide valuable insight into the level of spending and investment in land for the primary purpose of wildlife-associated recreation.

Appendix C, Methods:

The eligible universe for the FHWAR is the civilian non-institutionalized and non-barrack military population living in the United States. The institutionalized population, which is excluded from the population universe, is composed primarily of the population in correctional institutions and nursing homes. Each survey was designed to provide state-level estimates of the number of participants in recreational hunting and fishing and in wildlife-watching activities (e.g., wildlife observation). Information was collected on the number of participants, where and how often they participated, the type of wildlife encountered, and the amounts of money spent on wildlife-related recreation. Additional details on the sampling procedure are contained in Appendix D of each of the annual reports (U.S. Department of the Interior, Fish & Wildlife Service & U.S. Department of Commerce, Census Bureau 2011).

Questions that were used to calculate the values in this research included the following:

In 20XX, did you own, in part or whole, property (*in the case of wildlife-watching:* "more than a mile from your home") in the United States PRIMARILY for the purpose of [hunting / fishing / "the purpose of observing, photographing, or feeding wildlife"]?
1 – Yes
0 – No

How many acres did you own? If less than one acre, enter 1.

Were you part of a club or group, including a family group, owning this land in 20XX?
1 - Yes
0 – No

How many others were members of this group or club?

What did you spend in 20XX for YOUR SHARE of the land which was owned PRIMARILY for hunting? Include mortgage, taxes, maintenance, and down payment cost if purchased in 20XX. Do not include the cost of a cabin if reported earlier.

When you were [hunting / fishing] in the U.S. chiefly for / in [big game / small game / migratory birds / other animals / saltwater / freshwater / great lakes] during 20XX, how much was spent for YOUR SHARE of private land-use or access fees? Do not include leases.

On your [wildlife-watching] trip(s) during 20XX in the U.S. at least a mile from home how much was spent for YOUR SHARE of private land-use or access fees? Do not include leases.

In estimating mean values of payments for ownership and leasing, I included zero values for the amount spent to own land due to the potential of individuals with inherited land that require no costs. However, when calculating means for leases I excluded zero values due to the definition of leasing involving a payment for access. Due to the nature of shared ownership of properties, calculations of total acreage requires the division of the reported acreage by the number of members in the group or club to avoid double counting acreage of club or group members. I could not utilize survey data prior to 2001 due to changes in several key questions between 1996 and 2001.

The questions in this survey were pre-tested and passed established U.S. Census Bureau quality control procedures (U.S. Census Bureau 2011). Nonetheless, the estimates in this manuscript could contain non-sampling error such as measurement error or non-response error. An example of potential measurement error that could affect the estimates is the possibility of interviewees misunderstanding questions. For example, one potential source of measurement error involves the case of individuals who own land primarily for recreation and at the same time lease out that same land to others for recreation, thus providing the same answer to questions about acres leased and owned. This would result in the same land being counted twice. To test the potential impact of such error, I examined the data for land owned or leased for hunting and found that only 29 out of 1,766 observations have the same acreage reported for ownership and leasing, which account for 6.7 million acres of land, a relatively small amount. Even if all of these cases were the result of measurement error as described (which is not necessarily the case) they account for less than 2% of the total estimate of lands leased or owned for hunting and is far smaller than the standard error estimate of 43.2 million acres. Other significant differences between leased land and owned land also suggest that this potential double counting does not

significantly affect the results. Despite the potential for upward bias of results due to measurement error such as this, acreage estimates about land utilized for wildlife-associated recreation remain conservative for the following reasons: 1) the survey did not collect information about the land area used for day-use fees, which could affect even greater areas of land, and 2) owned acreage was only accounted for if it was owned *primarily* for wildlife-associated recreation, meaning that lands owned with a secondary purpose of wildlife-associated recreation were not counted.

Regarding the interpretation of figure 2, there is the possibility for some cases of leases being composed of multiple, non-adjacent properties possibly owned by different individuals. However, literature examining fee hunting operations in the U.S., and interviews with landowners in ongoing research suggests that these cases are rare and would not substantially affect the results (Rasker et al. 1992; Butler & Workman 1993).

References for Appendix C

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Appendix D, Specific types of recreation:

The survey questionnaire does not directly gather information on particular sub-categories of hunting and fishing leases, but I was able to estimate these values for hunting leases and property owned primarily for hunting (days spent wildlife-watching and fishing on private land were not gathered) by using a ratio of the number of days spent in pursuit of the four sub-categories of hunting on private lands, i.e. big game, small game, migratory birds, and other animals. As illustrated by the table below, these estimates show similar proportions as seen in the day-use fee access calculations reported in Table 4.

Table D.1: Estimated Spending by Type of Hunting on Leased and Owned Properties on Private Land in the U.S.

Lease	**Total**	**SE**
Big Game	654	111
Small Game	68	13
Migratory Bird	93	15
Other Animals	46	13
Own		
Big Game	3,140	431
Small Game	700	205
Migratory Bird	473	129
Other Animals	159	36