

THREE ESSAYS ON WILDLIFE CONSERVATION PROGRAMS: VALUE, TIMING, AND LOCATION

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Recovery efforts are critical in saving species by reducing the risk of extinction. The first essay proposes a method of measuring this reduction in risk, which applies a stochastic population model as discussed by Dennis, Munholland and Scott (1991). The value of recovery efforts or a conservation program can be calculated as the expected present value of a species' longer survival. The value can be calculated in two parts. In the first part, the reduction in extinction risk can be measured by the probability difference between the extinction first arrival time before and after a conservation program starts, for all future periods. In the second part, the existence value of an endangered species might be estimated either from stated preference surveys (contingent valuation) or from collective revealed preferences, as measured by the public's actual payments on recovery programs. Considering both the reduction in risk and the existence value of an endangered species gives a monetary measure of the benefits from recovery actions. A numerical example of the California condor is used to illustrate the method for calculating the monetary value of recovery programs.

The second essay studies the optimal timing to start a conservation program when a species' population is declining and risks extinction in the future if no actions are taken. Two methods are proposed. In the static approach, the timing depends on hazard rates of extinction of all future periods. Two types of hazard rate functions are explored. One is derived from an extinction first ar-

rival time probability distribution, which is an inverse Gaussian distribution. This probability can be estimated from the historical population size data of a species and by using the same stochastic population model as in the first essay. The other hazard rate function is a Weibull distribution. First order and second order conditions are derived to calculate the optimal timing to start a conservation program. The dynamic approach is a state contingent model, in which the optimal timing depends on the current population level of the species itself. In addition, the California condor and the whooping crane are studied as two examples in this essay to illustrate the calculation method.

The use of captive breeding and subsequent release of endangered species is a valuable conservation practice in saving species from extinction. When releasing captively bred endangered species back into the wild, the US conservation agency often requires the establishment of two or three separate wild populations to avoid simultaneous local extinctions. However, when the sizes of separate populations are spatially correlated because of regional stochasticity, the distance between release sites influences the overall setup and management cost as well as the probability of joint extinction. Thus the distance between release sites create a trade-off between the expected loss should extinction occur and the setup and management cost. The third essay examines (1) the relationship between the distance and the probability of joint extinction; (2) the trade-off created by that distance; (3) comparative statics between the optimal distance and the spatial correlation, and between the optimal distance and the setup and management cost; and (4) a comparison between the outcomes of one, two and three release sites. In addition, an example of the California condor is used to discuss the importance of the trade-off created by the distance between multiple conservation sites.

BIOGRAPHICAL SKETCH

Yu Ye (Susan) started her PhD study in 2012 at the Department of Economics at Cornell University. Her main research field is environmental and resource economics. At Cornell University, she has worked on topics including wildlife conservation, international environmental agreements, invasive species management, and renewable energy. Economics analysis in her research includes dynamic optimization, hazard rates, dynamic game, non-parametric econometrics and geo-spatial modeling. Besides her research, Susan also supported undergraduate level courses in both micro- and macroeconomics as a teaching assistant at Department of Economics at Cornell University.

Prior to graduate school, Susan received her Bachelor of Commerce degree with a major in economics from University of Canterbury, where she developed her interest in environmental and resource economics. During her third year in college, Susan studied at University of California San Diego on an exchange program. She completed a research project on European Monetary Union using a computable general equilibrium approach as an exchange student.

Susan called Wuhan, China her hometown, where she grew up and completed her primary and secondary education before setting off to study in New Zealand with a full scholarship.

This document is dedicated to all Cornell graduate students.

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CHAPTER 1

INTRODUCTION

1.1 Endangered Species and Endangered Species Act (ESA)

1.1.1 A History of the Endangered Species Act of 1973

Introduced by an Iowa Congressman and made into law in 1900, the Lacey Act of 1900 opened the first chapter of a long history of efforts by the U.S. government to preserve wildlife (United States Department of Agriculture, 2017). According to the USDA (2017) the original purpose of this Act was to prohibit illegal hunting and trade of animals and plants. Currently, however, its main objective is to prevent the spread of invasive species (US Fish & Wildlife Services, n.d.).

Later, several species specific legislations were made. For example the Migratory Bird Conservation Act of 1929 regulates the prices of land or water to be purchased or rented for birds' habitats (USFWS, 2017); the International Convention for the Regulation of Whaling of 1937 prohibits the hunting of right and gray whales (International Whaling Commission, 2017); and the Bald Eagle Protection Act of 1940 protects the two species of eagles which are the symbol of the United States (USFWS, 2016d).

The Endangered Species Preservation Act of 1966, on which today's Endangered Species Act of 1973 is based, allows some native U.S. animal species to be listed as endangered. This Act has put more emphasis on habitat preservation. For example, besides requiring federal land agencies to protect natural habitats

on their lands, this Act also financially encouraged the United States Fish and Wildlife Service to expand listed species' habitat area through land acquisition (USFWS, 2011).

The Endangered Species Act of 1969 has three main amendments from the Endangered Species Preservation Act of 1966. First, it protects a broader range of species, for example, "crustaceans and mollusks...mammals, fish, birds, and amphibians" (USFWS, 2013a); second, it extends protection to foreign endangered species as well by prohibiting their importation into the United States; third, in order to strengthen the enforcement, this amendment increased the fine and the length of jail time for illegal commercial hunting and trading (USFWS, 2011).

1970s saw a global cooperation when it comes to endangered species conservation. The Convention on International Trade of Endangered Species of Wild Fauna and Flora, which is a comprehensive multilateral treaty, was signed in February, 1973 (USFWS, 2011).

In the US, the Endangered Species Act of 1973 was created. It is jointly administered by the U.S. Fish and Wildlife Service (USFWS) and the Commerce Department's National Marine Fisheries Service (NMFS) (USFWS, 2016c). The main responsibility of the USFWS is terrestrial and freshwater organisms, while the main responsibility of the NMFS is marine mammals, such as whales and anadromous fish (Endangered Species Act of 1973). All species of plants and animals, except for pest insects, are eligible for listing as endangered or threatened under the ESA, but must meet one of the five following criteria:

1. There is the present or threatened destruction, modification, or curtailment

of its habitat or range.

2. An over utilization for commercial, recreational, scientific, or educational purposes.
3. The species is declining due to disease or predation.
4. There is an inadequacy of existing regulatory mechanisms.
5. There are other natural or manmade factors affecting its continued existence (Endangered Species Act of 1973).

According to this Act, a species can be listed either directly by the USFWS or the NMFS through its candidate assessment program, or by an individual or organizational petition. In order for a species to be listed, this Act requires a 90-day initial screening followed by a within-12-months comprehensive evaluation of the species' biological and ecological status, based on scientific data and analysis. The various status and codes of listed species are described by USFWS (2012), which are also presented in Appendix A.

As of January 2016, the Endangered Species Act listed 1,125 endangered species, among which 493 species are endangered animals and 732 species are endangered plants. There are 365 species listed as threatened in the United States, including 200 threatened animals and 165 threatened plants (US. Fish & Wildlife Services, 2016c). According to Greenwald, Suckling, & Taylor (2006), the number of species added to the list as "threatened" or "endangered" has been increasing except during George W. Bush administration, when there are only 8 new listings per year. During the Ford administration, there are 15 new listings per year, 32 per year during Carter and Reagan, 58 per year during George H. W. Bush, and 65 per year during Clinton (Greenwald et al., 2006).

When a wildlife species is listed as endangered or threatened, besides giving

a public notice, the ESA also requires the development of a recovery plan within three years to specify the criteria, actions required, estimated costs, and timeline needed for recovery (Defenders of Wildlife, 2013). The ultimate goal is to help the endangered species re-establish a self-sustaining population in the wild.

The original Act specifies that during the listing process, economic factors cannot be considered to determine the status of species. The 1978 amendment, however, allows considerations of the economic impact in the provision on critical habitat designation (USFWS, 2011). However, including economic considerations during the listing process has unwanted impacts. For example, the 1978 amendment which linked the listing procedure with critical habitat designation and economic considerations, almost prevented 2,000 species from consideration (The Stanford Environmental Law Society, 2001). For this reason, the 1982 amendment excluded economic considerations again (USFWS, 2011). Although not appropriate criteria for listing, when it comes to making recovery efforts and plans, economic considerations are crucial. This dissertation looks at three economic issues associated with the recovery efforts from three different perspectives: value, timing and location.

1.1.2 The Value of Wildlife

The esthetic, ecological, educational, recreational, and scientific value of plants, wildlife and fish has long been recognized by the public (USFWS, 2015a). According to the classic categorization by McNeely, Miller, Reid, Mittermeier & Werner (1990), wildlife has both direct and indirect values. They characterize direct values as the immediate enjoyment or satisfaction from biological resources.

They also define direct values to include consumptive use values and productive use values. Consumption values concern direct consumption of nature's products which are not traded in a market (McNeely et al., 1990). If the nature's products are commercially harvested for trading in markets, then they have productive use values (McNeely et al., 1990). For example, McNeely et al. (1990) pointed out that taking firewood from the woods for energy needs is a consumption use, while purchasing timber for making furniture is a productive use. The indirect values, on the other hand, are characterized as the functions or environmental services of ecosystem, which include non-consumptive use values such as scientific research, and the option value, which is maintaining options available for the future, as well as the existence value, which is ethical feelings of existence of wildlife (McNeely et al., 1990).

A rather more pragmatic approach categorizes the values of wildlife into the economic importance, the nutritional value, the ecological role and the socio-cultural significance (Chardonnet, Clers, Fischer, Gerhold, Jori & Lamarque, 2002).

There are two general categories of the economic values, the consumptive and non-consumptive uses. These authors define the non-consumptive uses of wildlife as the aesthetic value of wildlife, for example, the tourism industry. They define the consumptive uses of wildlife as the value obtained by removing a certain amount of the resources, for example, taking an animal for food, clothes, or furniture. The consumption uses of wildlife are important to many developing countries, while most developed countries rely on domesticated animals for consumption instead (Chardonnet et al., (2002)).

Wildlife was historically an important source of food for human, but its share

in the human diet has gradually diminished (Chardonnet et al., 2002). However, as Chardonnet et al. (2002) pointed out, currently there are still some hunter-gatherer groups and forest ethnic groups who see the wildlife as the main source of their food. These authors also drew attention to some developed countries like Sweden, which supplies a large amount of moose meat.

Biological diversity itself has important value. Wildlife has important role both in natural habitats and in animal communities. For example, elephants in African savannahs keep open habitats from turning into forests, wild rabbits in south-east France help reduce the number of forest fires by keeping the garrigue open (Chardonnet et al., 2002). Interactions within wildlife itself help maintaining its diversity.

The social-cultural significance of wildlife in developing countries includes its role as a resource, a property, a cultural taboo, a symbol of political power and economic power, ceremonial signs (Chardonnet et al., 2002). In developed countries, Chardonnet et al. (2002) state that the social-cultural significance is mainly in wildlife-associated recreational activities.

1.1.3 Reasons of Extinction

The fact that a variety of species in the United States went extinct or are in danger of becoming extinct has been referred to as the “sixth mass extinction” (Kaplan, 2015). Besides natural factors, economic growth and urban development have also contributed to habitat loss, pollution, poaching, etc, which are the main factors for wildlife species extinction.

According to the National Wildlife Federation (n.d.), damage to, or destruction of a species' habitat is the number one reason for wildlife endangerment. The National Wildlife Federation categorizes habitat loss into three main types—destruction, fragmentation and degradation. Paving over wetlands for streets and parking lots, converting woods into farms, and cutting down forests to get lumber are examples of habitat destruction. Road development, dams constructions and water diversions can fragment habitats that are crucial for immigratory species. Urban and industrial pollutants such as oil spills, petroleum products, pesticides, and lead poisoning are an important reason for habitat degradation. The National Wildlife Federation (n.d.) also points out that the main drivers for habitat loss are agriculture, land conversion for development, water development, and global warming.

Poaching refers to illegal hunting. For example, poachers are killing elephants for their teeth, wild tigers for their skin and bones, and rhinos for their horns. According to the World Wildlife Fund (n.d.), 1,004 rhinos were illegally killed in South Africa in 2013 and at least 2,500 elephants were poached in 2011. In Asia, tiger skins are symbols of power and wealth; tiger bone wine is a well-known herbal wine and has been believed to cure arthritis in China; rhino horn is considered to cure cancer in Vietnam. The high profit margins given by high prices of these wildlife products, especially in Asian, have driven up poaching activities and illegal trades (World Wildlife Fund, n.d.). On the other hand, it is very difficult to stop poaching because poachers have developed sophisticated methods to avoid being caught by the authorities.

In addition to human factors, natural factors can also harm a species' survival. For example, sudden climate changes, volcanic eruptions, earthquakes,

droughts, diseases, hurricanes and wildfires are all known to have impacts on wildlife population sizes. Chapter 4 of this dissertation is mainly concerned with the natural factors and how to choose conservation sites that take into account of their impacts.

1.2 Conservation Efforts

When a species is listed under the Endangered Species Act, a recovery plan is required to specify the actions needed for recovery. Recovery actions include habitat protection, hunting ban, education, outreach programs, captive breeding, reintroduction programs, etc.

As habitat loss is the primary threat to most imperiled species National Wildlife Federation (n.d.), the 1978 amendment to the Endangered Species Act of 1973 requires the identification, designation and protection of critical habitats, whether they are lands, water or air for all listed endangered species. Critical habitats are areas that are essential to the species' survival and need special management and protection (USFWS, 2017b). Critical habitats may be on private or public lands. Critical habitats that are publicly owned lands are prohibited from destroy or adversely modification (USFWS, 2017b). Critical habitats that are privately owned lands are not subject to such prohibition, but large-scale development, logging and mining projects are subject to habitat conservation regulations (Endangered Species Act of 1973). Since 1978 the USFWS has regularly designated critical habitats, except for a period of interruption between 1986 and the late 1990s, due to a controversial regulation issued by the Reagan Administration which significantly limited USFWS's capacity to design-

nate critical habitats (Center for Biological Diversity, 2003). In the late 1990s, the Reagan regulation was suspended and regular critical habitat designations resumed. Since 2005, the USFWS has tried to speed up the critical habitat designation process. However, the level of critical habitat protection is significantly impacted by the political environments. For example, during the Bush Administration the USFWS had a difficult time obtaining sufficient funding and designating sufficient areas as critical habitats, compared to the Clinton Administration (Center for Biological Diversity, 2003). An interesting note is that while the listing decisions of a species as an endangered or threatened do not take into account economic considerations, the USFWS is required to consider economic impacts when designating critical habitats (USFWS, 2017b).

The use of captive breeding and reintroduction is a popular conservation action in the United States. A captive breeding program is bringing species from the wild to a captive breeding facility such as zoos. With the care and protection from biologists and veterinarians inside the facility, the reproduction rates and survival rates are likely higher than in the wild. When there are enough captively bred individuals, a reintroduction program will begin to release a selection of suitable individuals back to the wild.

Conservation efforts have achieved positive results. According to the Delisted Species Report by the USFWS Environmental Conservation Online System (n.d.), over the four decades after the Endangered Species Act of 1973, around 50 species have been delisted due to effective recovery. Even before reaching delisting criteria, recovery plans and actions have already helped reverse the decline of many endangered wildlife populations. Examples include the whooping crane, the red-cockaded woodpecker, the peregrine falcon, the

red wolf, and the Florida panther. All of these species were endangered during mid-1990s for various reasons. Although they still face many challenges, these species are recovering due to recovery actions and efforts. The whooping crane wild population increased from 16 birds in 1941 to 442 in early 2015 (USFWS, 2015c). The number of active clusters of red-cockaded woodpeckers increased from 4,694 in 1993 to 6,105 in 2006 (USFWS, 2015b).¹ American peregrine falcons increased from 324 breeding pairs in 1975 to 2,000~3,000 breeding pairs in Canada, Mexico, and the United States in 2013 (USFWS, 2013b). In 1980 the red wolf (*Canis rufus*) was extinct in the wild. With captive breeding, a population of 50-75 individuals has re-established in its native habitat in eastern North Carolina (USFWS, 2016b). Additionally, according to the same report, there are around 200 individuals in captive breeding facilities in the United States. The Florida panther population recovered and includes approximately 200 adults and sub-adults today, from approximately 20 adults in the early 1970s (USFWS, 2016a).

The California condor recovery efforts, for example, have achieved surprising success and are used as a case study in this dissertation. The first California Condor Recovery Plan was created in 1975, followed by 3 revisions, in 1979, 1984 and 1996. The program is led by the U.S. Fish and Wildlife Service, and partnered by the Los Angeles Zoo, the San Diego Wild Animal Park, The Peregrine Fund, the Ventana Wildlife Society, the Pinnacles National Monument (National Park Service), and the Oregon Zoo (USFWS, 2017c).

The USFWS leads the program and operates release sites in southern California (USFWS, 2017c). A comprehensive status report by Walters, Derrickson, Fry, Haig, Marzluff & Wunderle (2010) summarizes the roles and contributions

¹Active clusters are potential breeding groups.

of each partner. For example, besides being a captive breeding facility, the Los Angeles Zoo also provides veterinary staff and keepers for field support at the southern and central California release sites, and provides the zoo as a treatment center for birds that need medical care. The Ventana Wildlife Society and the National Park Service run the release sites in central California. The San Diego Wild Animal Park is also a captive breeding facility, and it partly operates the Baja California release site. The Peregrine Fund operates a captive breeding facility in Boise, Idaho and the release site in Arizona. The Oregon Zoo provides a captive breeding facility as well. These major partners of the California Condor Recovery Program and their financial contributions are summarized in Appendix C, which is from the status report by Walters et al. (2010).

Some other partners are also involved, for example, federal agencies such as the Bureau of Land Management and the U.S. Forest Service; state governments such as the Arizona Game and Fish Department (AGFD), the Utah Division of Wildlife Resources (UDWR), the California Department of Fish and Game (CDFG) and the Oregon Department of Fish and Wildlife (ODFW); private and non-profit contributors such as the San Diego Zoo, the Santa Barbara Zoo, the Institute for Wildlife Studies, and local business communities (USFWS Pacific Southwest Region, 2013).

The 5 year review provided by USFWS Pacific Southwest Region (2013) summarizes the contributions of these partners in the Pacific Southwest Region. The Bureau of Land Management (BLM) provides a feeding site near Pinnacles National Park, transportation for condors from breeding facilities to release sites, funds for monitoring equipment and trash removal in specific areas. The AGFD provides a full time condor biologist to work with the TPF biologists on day-

to-day management. It also leads a public education program and the free non-lead ammunition program. The UDWR provides two biologists and two outreach specialists to support the program. It sponsors TPF for GPS transmitters purchases, reimburses hunters for purchases of non-lead ammunition as well. Grand Canyon NP also provides additional support through on-site field monitoring. The CDFG provides a full-time condor biologist. The ODFW is investigating the potential for a release site in the Pacific Northwest. The Santa Barbara Zoo helps with outreach, breeding research and also field monitoring. The Institute for Wildlife Studies also organizes lead awareness campaigns in central and southern California. A private ranch in Baja California contributes to operations at the release site there. In southern California, the Tejon Ranch recently signed an agreement with several conservation organizations to set aside nearly 100,000 ha of habitat for condors. At Big Sur, Pacific Gas and Electric has spent hundreds of thousands of dollars to reduce condor deaths caused by collisions with power lines in this region. These partners' financial contributions are summarized in Appendix D, the numbers of which are reported in the 5 year review provided by USFWS Pacific Southwest Region (2013).

With an enormous amount of attention received, the California condor conservation program is a case study throughout this dissertation in Chapter 2, 3 and 4.

1.3 Literature Review

This dissertation is based on three strands of literature. First is the population viability analysis. Second, the optimal timing literature. Third, the conservation

spatial patterns literature.

1.3.1 Population Viability Analysis

An important goal of wildlife conservation is to prevent extinction. Effective conservation management of endangered species needs solid assessment of extinction risk. In endangered species conservation literature, the population viability analysis (PVA) is a cornerstone risk assessment method. Statistical techniques are used to calculate the probability that a population will go extinct within a given time frame, or to calculate the minimum population size needed for a population to be a viable one. It can also shed light on the urgency of taking recovery actions, identify key life stages that should be the focus of recovery efforts, compare proposed management options, and assess existing recovery efforts.

The first PVA was developed by Shaffer (1981, 1983), which incorporated random variability and calculated extinction probabilities and minimum viable population size for the grizzly bears in the Yellowstone National Park. This study helped the Yellowstone National Park design and implement management plans. Before Shaffer, population demographic analyses were deterministic (Beissinger & Westphal, 1998). Gilpin & Soulé (1986) extended the method by modeling many other factors that affect the dynamics of a population, such as genetics. Recently, PVA has also been extended with Bayesian estimation (e.g., Evans, Holsinger & Menges, 2010), which accounts for parameter uncertainties.

It is now frequently used by various federal conservation agencies to make recovery plans. For example, the Fender's blue butterfly (Schultz & Hammond,

2003), the island fox (Kohlmann, Schmidt & Garcelon, 2005), the Puritan tiger beetle (Gowan & Knisley, 2010), the red wolf (Faust, Simonis, Harrison, Waddell & Long, 2016), etc.

Among the various PVA approaches to estimate population viability, time-series and demographic PVAs are the most commonly used (Gerber & Gonzalez-Surez, 2010).

Time-series PVA, such as Dennis, Munholland & Scott (1991), is used throughout the three essays in this dissertation. It is the simplest PVA because it uses only estimates of the total number of individuals in a population over time, treating each individual as identical (Gerber et al., 2010). Population growth and variance can be estimated and can be used to calculate the probability of extinction for future periods. Dennis et al (1991) provides illustrative analyses of data on the whooping crane, grizzly bear, Kirkland's warbler, California condor, Puerto Rican Parrot, Palila, and Laysan Finch.

Demographic PVA, such as Gilpin et al. (1986), incorporates age- and stage-specific survival and reproduction rates. Compared to time-series PVAs, considerably more data will be needed (Gerber et al., 2010). However, with these extra details, the demographic PVA can be used to identify key life stages that need more attention for protection. In addition, it can also be used to compare different management plans.

Other PVA methods include those based on patch occupancy data and complex spatial models, which are reviewed separately in Section 1.3.3.

PVAs have some drawbacks:

1. They are species-specific (Taylor & Ralls, 1997);

2. Model parameterization is usually difficult because for threatened or endangered species, there is generally a lack of some important population dynamic data such as density dependence, stochasticity and spatial structure. In many cases there are substantial errors in estimated abundance. In addition, the results usually depend sensitively upon estimated parameters (Beissinger et al., 1998);
3. Risks that are difficult to estimate or detect are often omitted, and risk classification is often difficult as well (Taylor et al., 1997);
4. Population viability forecasting is only reliable if one is not looking at a far future period (Taylor et al., 1997). Short time series or poor fits of the model to data lead to wide confidence intervals for the probability of extinction, sometimes making the estimates become meaningless.

In spite of the drawbacks, PVA is one of the most valuable approaches to estimate the relative risk of extinction and to explore different conservation management scenarios.

1.3.2 Optimal Timing

In the field of resource and environmental economics, several authors have studied the timing using real option approach. For example, Conrad (2000) evaluates the sequence and timing of wilderness preservation, irreversible resource extraction, and irreversible development. When the wilderness is intact, there is a stochastically evolved threshold for the resource price and for the return of development, above which the resource extraction and the development

options should be exercised and under which the wilderness should be preserved (Conrad, 2000).

Pindyck (2000) studies the timing of environmental policy, where he argues that the standard static cost and benefit framework in environmental policy evaluation is inappropriate. Economic uncertainties and ecological uncertainties are explored. Pindyck (2000) also explores different assumptions, such as linear reduction cost, convex reduction cost and partial reduction in emissions, convex benefit function and gradual emission reductions. The main argument of this paper is that the existence of uncertainty leads to a higher benefit or pollution threshold for policy adoption (Pindyck 2000).

Insley (2002) studies the optimal cutting of a stand of trees when lumber prices are assumed to follow some known stochastic process. Saphores (2003) studies the harvesting of a renewable resource, subject to that the renewable resource has an extinction threshold. The decisions are when to harvest and how much to harvest, when the renewable resource follows a stochastic process (Saphores, 2003).

Kassar & Lasserre (2004) evaluates biodiversity in a real options framework. The resource in use is substitutable by other resource that are not in use, of which the future use are uncertain Kassar et al. (2004). Kassar et al. (2004) argues that each unused species has an option value because it derives value from the fact that it might, in the future, be in a better position to provide the same product or service as the one currently in use. In their model, the decisions are choosing the dates when each least valuable species can be allowed to go extinction to maximize the expected total value of all existing species.

Abdallah & Lasserre (2012) studies the optimal timing to stop or resume logging in forest upon which an endangered species relies for survival. Habit area, which is the state variable in their model, is assumed to follow an Ornstein-Uhlenback mean-reverting diffusion process. Compared to entry and exit decision models, Abdallah et al. (2012) allows exercise decisions to impact the diffusion process of the state variable. In addition, they also explicitly imposes an extinction threshold. These two model specifications are also present in the state contingent problem in Chapter 3 of this dissertation. The difference between Abdallah et al. (2012) and the state contingent problem of this dissertation is that here in this dissertation, the format of the optimal policy is not assumed to be a pair of critical thresholds before the optimization problem is developed. Although both Saphores (2003) and Abdallah et al. (2012) formulated their optimization problems assuming a threshold policy is known beforehand, neither of them provides a proof why this ad hoc assumption is reasonable.

A recently submitted paper to the Ecological Economics by Conrad uses a real option approach to study the conservation of endangered species. Conrad introduces an anxiety function that is linked to the population size of an endangered species. The state contingent model of this dissertation is similar to Conrad's real option model, except for that instead of using an anxiety function, the state contingent model in this dissertation uses the loss cost of an endangered species directly. More importantly, an explicit positive extinction threshold is imposed in this dissertation, while Conrad's model has the "natural" absorbing barrier of a Geometric Brownian motion, which is zero. There are two main reasons to impose a positive extinction threshold. First, the population dynamic model used is developed by Dennis et al. (1991) and in their paper, the extinction thresholds were chosen at positive population levels. Second, Geometric

Brownian motion is essentially an exponential growth model, which actually will not reach zero starting from a positive initial value. Thus, having a positive absorbing barrier allows a positive probability of extinction. Thus, the concept of extinction is absent in Conrad's paper while it is explicitly modelled in this dissertation.

Besides the real option approach, hazard rates can also be used to study timing questions. Hazard rates measure the probability of an event's failure given that the failure hasn't happened yet. It is a commonly used component for risk analysis, especially when the underlying process is a stochastic event. It has been widely used in engineering, economics and sociology. Khan & Stinchcombe (2015) discusses hazard rates and studies the optimal waiting time in a stochastic environment that changes at random points in time. Chapter 3 also uses the concept of hazard rates to study a static decision problem of a optimal timing question.

1.3.3 Spatial Patterns

Wildlife conservation literature contains a strand of studies that discuss spatial characteristics of conservation sites. For example, there is a debate about whether a large reserve will be better than multiple small reserves, where the multiple small reserves are formed due to habitat fragmentation brought by urban development. Simberloff & Abele (1976) argued that several small reserves may contain more species than a single large reserve, because extinction of a population in one location may be rescued by a population in another location, if the species is capable of dispersal. They also provided some other reasons for

why several reserves are preferred than one large reserve, such as catastrophes and mutually exclusive competitors. However, Diamond, Terborgh, Whitcomb, Lynch, Opler, Robbins, Simberloff & Abele (1976) challenged some of their arguments by pointing out that not all species are capable or willing to immigrate among isolated habitats and species have different values so a greater number of species isn't necessarily more valuable than saving several important species.

Wright & Hubble (1983) uses a stochastic logistic model for one species and found that when the reserve is closed to outside immigrants, one large reserve is preferable to two small reserves, no matter whether the two small reserves have inter-reserve immigration; when the reserve is open to outside immigrants, then one large reserve and two small reserves are comparable in terms of population viability. They also point out that for endangered species, the populations are restricted solely to reserves and thus there is no outside immigration. In this case, one large reserve is preferable to two small reserves. One thing to note about the study of Wright et al. (1983) is that it only takes into account extinction caused by demographic stochasticity. Thus, when adding environmental stochasticity into the model, conclusions might differ.

Similarly, Gonzalez-Suarez, McClunney, Aurioles & Gerber (2006) compared the extinction probability under three spatial scenarios of the California sea lions: unlimited movement between all sites (one single large population), limited movements (several independent median size populations), and no movement at all (multiple independent small size populations). Similar to this dissertation, they also adopted the population dynamic model of Dennis et al. (1991). After calculating a risk matrix, cumulative probability of quasi-extinction, probabilities of 80%, 50% and 30% of population reduction within a certain period

of time, they concluded that the one single large population has the highest population viability while the multiple independent small size populations scenario has the lowest population viability, while their confidence intervals are quite broad. When Gonzalez-Suarez et al. (2006) calculates the risk of extinction for the several independent median size population and the multiple small size population scenario, they used the mean value of infinitesimal means and the pooled value of infinitesimal variances. However, as Chapter 4 of this dissertation shows, using this approach won't give an accurate estimation of extinction risks for multiple independent populations. When the infinitesimal means and variances are small enough, using the mean and pooled value approach will underestimate the risk of extinction, while when the infinitesimal means and variances are large enough, using the mean the pooled value approach will overestimate the risk of extinction.

Recently, the effects of spatially correlated disturbances on wildlife extinction have also been explored. For example, McCarthy & Lindenmayer (2000) studies the importance of considering spatially-correlated extinction in metapopulation viability analysis using a case study of the Leadbeater's Possum. In their case case, spatially-correlated fires can cause local extinction of the Possum. They conclude that incorporating correlated local extinctions can increase the predicted risk of joint extinction and thus it is important to take it into account.

Kallimanis, Kunin, Halley & Sgardelis (2005) also confirmed the conclusion from McCarthy et al. (2000). They compared the importance among species dispersal properties, landscapes' spatial structure, and disturbances' spatial pattern and concluded that disturbance pattern significantly impacts the extinction

risk of a metapopulation, almost at the same level as habitat availability.

Albers, Busby, Hamaide, Ando & Polasky (2016) developed a reserve site selection framework and compared the results under spatially-correlated fires and spatially-independent fires. Their goal is to maximize the number of surviving species and they found that results depend on the characteristics of the species distribution. For a species distribution without hotspots (where more than one species is present), location does not matter in the selection of the optimal reserve design, while for the species distribution with one or more hotspots, the risk scenario will make a difference when it comes to reserve selection (Albers et al., (2016)).

However, these studies that consider spatially-correlated disturbances did not explicitly model the population dynamics at each location. Instead, the status of each location is either occupied or not occupied. In the third essay of this dissertation, spatially correlated natural disturbances are modelled in a framework where the population dynamics at each location are explicitly modelled using Geometric Brownian Motions.

In addition, the current wildlife conservation literature that incorporated spatial patterns has very little discussion about the economic considerations. The only discussions available are either the Species Set Covering Problem (SSCP) (Underhill, 1994), which selects the minimum number of land sites so that each species is covered, or the Maximal Covering Species Problem (MCSP) (Church, Stoms, & Davis, 1996), which selects reserves to protect as many species as possible with a given budget. One interesting paper in this literature is Possingham, Ball, & Andelman (2000), which incorporates a minimum separation distance into the model requirements. The third essay of this dis-

sertation will explicitly model the economic trade-off between higher extinction risk and higher management cost.

CHAPTER 2

ON THE VALUE OF REDUCING THE RISK OF EXTINCTION

2.1 Research Question

As discussed in Section 1.2, the main benefits associated with recovery efforts are reducing the risk of extinction through increasing a species' population abundance. This study proposes a method of measuring this reduction in risk using a stochastic process as a model of population abundance as discussed by Dennis et al. (1991). The economic value of reducing the risk of extinction can be calculated as the expected present value of longer survival. In addition, the existence value of an endangered species might be either estimated from stated preference surveys (contingent valuation), or from "collective revealed preferences", as measured by the public's willingness to pay for recovery programs.

Combining the risk reduction and the existence value of an endangered species gives a measure for the benefits of recovery actions. Conservation efforts may have many side benefits (positive externalities), in addition to reducing the risk of extinction for a specific endangered species. For example, recovery actions might improve habitats for other species and increase the value of an area to bird watching and outdoor recreation. These positive externalities may not be reflected in stated willingness to pay or in the total expenditures on a specific species by government and conservation organizations. If this is the case, our model will underestimate the value of recovery efforts for endangered species.

The rest of this essay is organized as follows. Section 2.2 presents a model for calculating the risk of extinction based on Dennis et al. (1991). Section 2.3 devel-

ops an equation for calculating the expected present value of reducing the risk of extinction. Section 2.4 provides an analysis of the California condor captive breeding program and the reintroduction of the California condor to selected locations in the U.S. and Mexico. Section 2.5 concludes.

2.2 Modeling the Risk of Extinction for Endangered Species

2.2.1 Infinitesimal mean and variance, and modal time to extinction

The statistical methods for estimating various growth- and extinction-related measures are discussed in Dennis et al. (1991). These measures might be developed from a two-parameter stochastic process based on a stochastic exponential growth model. The two parameters are the infinitesimal mean and infinitesimal variance for a species population. For example, all the following measures—the finite rate of growth, the geometric finite rate of growth, the probability of reaching a lower threshold population size,¹ the mean, median, and the modal time to extinction,² and the projected population size—can be calculated from the estimated infinitesimal mean and the infinitesimal variance of a population. In this study, the modal time to extinction is taken as one of the key measures to calculate the value of conservation efforts.

According to Dennis et al. (1991), an endangered species' population dynamics can be modelled as a geometric Brownian motion with an absorption

¹The lower threshold is associated with extinction or local extirpation.

²The most likely time of attaining the extinction threshold.

barrier at a pre-specified extinction threshold. To be specific, before an endangered species becomes extinct, its population size $N = N(t)$ at instant t , can be adequately approximated by a stochastic exponential growth model given that the population is small compared to its carrying capacity, as is the case with endangered and threatened species. Thus the population size N is the solution to a stochastic differential equation (SDE),

$$dN = rNdt + \sigma Ndz, \quad (2.1)$$

where r is the intrinsic growth rate, $\sigma > 0$, is the standard deviation rate and dz is the increment of a Weiner-drift Process. In addition, dz is normally distributed, $dz \sim \mathcal{N}(0, dt)$. The population size $N(t)$ will be log-normally distributed with an expected value at $t > 0$ of $\mathbb{E}[N(t)] = N(0)e^{rt}$ and a variance of $\mathbb{V}[N(t)] = N(0)^2 e^{2rt}(e^{\sigma^2 t} - 1)$.

From Itô's Lemma, the natural logarithm of the population size $x = \ln N$ satisfies

$$dx = (r - \sigma^2/2)dt + \sigma dz = \mu dt + \sigma dz, \quad (2.2)$$

where $\mu = r - \sigma^2/2$. Before hitting a pre-specified extinction threshold x_e , the natural logarithm of the population has a probability density function (PDF) corresponding to a normal distribution, i.e., $x(t) \sim \mathcal{N}(x_0 + \mu t, \sigma^2 t)$ where μ is the infinitesimal mean of the process and σ^2 is the infinitesimal variance. The initial population size N_0 and $x_0 = \ln N_0$ are assumed given. Note that this Brownian motion with drift by itself is not sufficient to model the population dynamics. There is an absorption barrier such that once the population hits a pre-specified

extinction threshold, it will be “trapped” at the threshold forever. The extinction threshold can be specified at various values according to the research needs. For example, some studies assume that $N(t) = 1$ implies extinction, i.e. $x_e = (1) = 0$, while some studies assume x_e at some positive number to imply that once a population is below a certain positive, the population won’t have a chance of surviving because the population size is not large enough to support breeding and growing. In this dissertation extinction threshold is specified at $x_e = (1) = 0$.

The probability distribution of a Brownian motion with a drift and a barrier at zero is derived in Ingersoll (1987). The CDF and PDF of a strictly positive population level $x > 0$,

$$F_a(x) = \Phi\left(\frac{x - x_0 - \mu t}{\sigma \sqrt{t}}\right) + e^{\frac{-2\mu x_0}{\sigma^2}} \Phi\left(\frac{-x - x_0 + \mu t}{\sigma \sqrt{t}}\right), \quad (2.3)$$

$$f_a(x) = \phi\left(\frac{x - x_0 - \mu t}{\sigma \sqrt{t}}\right) - e^{\frac{-2\mu x_0}{\sigma^2}} \phi\left(\frac{x + x_0 - \mu t}{\sigma \sqrt{t}}\right), \quad (2.4)$$

where $\Phi(\cdot)$ and $\phi(\cdot)$ are the CDF and PDF of a standard normal distribution respectively.

This model is one type of the frequently studied first-passage problems. Land and Orzack (1988), and Tuljapurkar (1989) have shown that this model can provide a good statistical approximation to the behavior of the stochastic Lewis-Leslie model when age-specific fecundity and survival rates are drawn from an independent multivariate distribution.

Figure 2.1 shows sample population paths generated from several underlying Geometric Brownian Motions, starting with a population of size $N_0 = 50$.

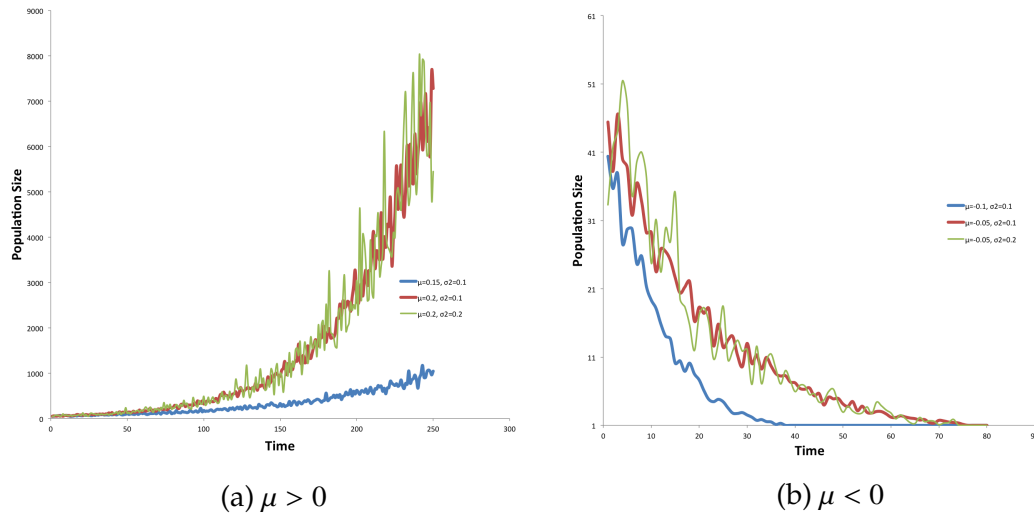
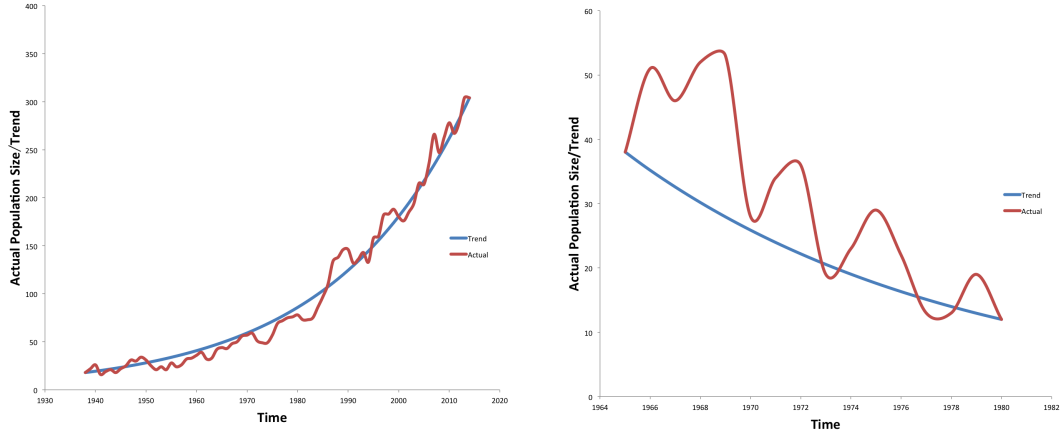


Figure 2.1: Population Dynamics Generated from Several Geometric Brownian Motions

In Figure 2.1(a), the underlying Geometric Brownian Motions have positive infinitesimal means. In Figure 2.1(b), the underlying Geometric Brownian Motions have negative infinitesimal means. Figure 2.2 displays the actual whooping crane population and California condor sizes and the estimated means of the processes. It has been shown by Dennis et al (1991) that this model fits the whooping crane and the California condor data well.

For an endangered species population, if there is a significant trend change, then two Geometric Brownian Motions will be needed to model the population dynamic. For example, Figure 2.4 in Section 2.4.1 shows the trend change after the captive breeding program started. Thus, two Geometric Brownian Motions are needed to fit the two parts of the population respectively.

As in Dennis et al. (1991), estimates of μ and σ^2 can be obtained by observing a population's sizes $n_0, n_1, n_2, \dots, n_q$ at times $0, t_1, t_2, \dots, t_q$ with an equal time interval τ between periods. A recommended way of fitting the stochastic exponential growth model to such data is maximum likelihood (ML) estimation. The



(a) Whooping Crane

(b) California Condor

Figure 2.2: Actual Population Sizes and Population Trends

ML estimates for the infinitesimal mean and variance are

$$\hat{\mu} = [\log(n_q/n_0)]/t_q \quad (2.5)$$

$$\hat{\sigma}^2 = 1/q \sum_{i=1}^q (1/\tau_i) [\log(n_i/n_{i-1}) - \hat{\mu}\tau_i]^2. \quad (2.6)$$

However, according to standard results from statistics, the ML estimator for variance σ^2 is biased, $\mathbb{E}[\hat{\sigma}^2] = \frac{q-1}{q}\sigma^2$. An unbiased estimate is thus

$$\tilde{\sigma}^2 = 1/(q-1) \sum_{i=1}^q (1/\tau_i) [\log(n_i/n_{i-1}) - \hat{\mu}\tau_i]^2. \quad (2.7)$$

The $100(1 - \alpha)\%$ confidence intervals for μ and σ^2 are

$$(\hat{\mu} - t_{\alpha/2, q-1} \sqrt{\tilde{\sigma}^2/t_q}, \hat{\mu} + t_{\alpha/2, q-1} \sqrt{\tilde{\sigma}^2/t_q}) \quad (2.8)$$

and

$$((q-1)\tilde{\sigma}^2/\chi_{\alpha/2, q-1}^2, (q-1)\tilde{\sigma}^2/\chi_{1-\alpha/2, q-1}^2) \quad (2.9)$$

respectively.

Under the continuing unpredictable fluctuations of the Wiener-drift model, the population $N(t)$ could possibly cross any lower threshold size n_e . The amount of time, T , elapsing before the threshold is first reached is a positive, real-valued random variable with a continuous probability distribution. The cumulative distribution function (CDF) of T can be written in terms of a standard normal CDF

$$P(t) = \Phi\left(\frac{-x_d - \mu t}{\sigma \sqrt{t}}\right) + e^{-2x_d\mu/\sigma^2} \Phi\left(\frac{-x_d + \mu t}{\sigma \sqrt{t}}\right), 0 < t < \infty, \quad (2.10)$$

where $x_d = x_q - x_e$ measures the difference between the natural log of the population sizes at the most recent period x_q and the pre-specified extinction threshold x_e . In addition, $\Phi(\cdot)$ is the CDF of a standard normal distribution. The two population parameters μ and σ^2 are the infinitesimal mean and infinitesimal variance defined earlier. The PDF of T is the derivative of $P(t)$ with respect to t

$$p(t) = x_d(2\pi\sigma^2 t^3)^{-1/2} e^{-(x_d + \mu t)^2/2\sigma^2 t}. \quad (2.11)$$

The CDF and PDF expressions above are standard from the first passage time literature (Ingersoll, 1987). This distribution is known as the inverse Gaussian distribution. When the infinitesimal growth rate is non-positive, i.e., $\mu \leq 0$, the process will hit the extinction threshold within a finite time with probability 1.

When $\mu > 0$, the probability that the process will ever attain the threshold is $e^{\frac{-2\mu x_d}{\sigma^2}}$, i.e., the probability distribution has a mass point of $1 - e^{\frac{-2\mu x_d}{\sigma^2}}$ at positive infinity.

The mode of the distribution is the most likely time of hitting a threshold n_e for the first time, or the value of t maximizing the PDF. It is calculated by equation (2.12) below

$$\hat{t}^m = \frac{x_d}{|\hat{\mu}|} \left[\left(1 + \frac{9}{4\hat{v}^2}\right)^{\frac{1}{2}} - \frac{3}{2\hat{v}} \right] \quad (2.12)$$

where $\hat{v} = x_d |\hat{\mu}| / \tilde{\sigma}^2$ and $x_d = \log(n_q/n_e)$. Thus, by plugging in the estimates for μ and σ from population time series data of a species, one can obtain a prediction of the modal time of extinction. Since \hat{t}^m is a function of $\hat{\mu}$ and $\tilde{\sigma}^2$, the delta method can be used to give the approximate distribution and variance of \hat{t}^m . An approximate $100(1 - \alpha)\%$ confidence interval for t^m is

$$(\hat{t}^m - z_{\alpha/2, q-1} \sqrt{\hat{Var}(\hat{t}^m)}, \hat{t}^m + z_{\alpha/2, q-1} \sqrt{\hat{Var}(\hat{t}^m)}), \quad (2.13)$$

where the variance of \hat{t}^m is

$$Var(\hat{t}^m) \approx (\sigma^2/t_q)(\partial t^m/\partial \mu)^2 + 2(q-1)(\sigma^2/q)^2(\partial t^m/\partial \sigma^2)^2. \quad (2.14)$$

The modal time to extinction, instead of the mean time to extinction, is a very popular measure to indicate the risk of extinction in the wildlife conservation literature, because the distribution of the first-passage time is positively skewed and has a heavy right tail. However, when it comes to the value of reducing the

risk of extinction, the modal time to extinction won't be an appropriate measure for risk, as discussed in Section 2.2.2 below.

2.2.2 Modal Time to Extinction Comparative statics

The first order derivatives of the modal time to extinction, t^m , with respect to the infinitesimal mean μ and the infinitesimal variance σ^2 are

$$\frac{\partial t^m}{\partial \mu} = -\frac{1}{2} \left(\frac{x_d^2}{\mu^2} + \frac{9\sigma^4}{4\mu^4} \right)^{-\frac{1}{2}} \left(\frac{2x_d^2}{\mu^3} + \frac{9\sigma^4}{\mu^5} \right) + \frac{3\sigma^2}{\mu^3} \quad (2.15)$$

$$\frac{\partial t^m}{\partial \sigma^2} = \frac{1}{2} \left(\frac{x_d^2}{\mu^2} + \frac{9\sigma^4}{4\mu^4} \right)^{-\frac{1}{2}} \frac{9\sigma^2}{2\mu^4} - \frac{3}{2\mu^2}. \quad (2.16)$$

Equation (2.16) can be shown to imply that $\frac{\partial t^m}{\partial \sigma^2} < 0$. Thus the bigger the infinitesimal variance is, the sooner will be the population's modal time to extinction. The sign of $\frac{\partial t^m}{\partial \mu}$ may be positive or negative. Figure 2.3 (a) (b) plot the values of $\frac{\partial t^m}{\partial \mu}$ as the value of μ changes positively or negatively. The two values for the infinitesimal variance σ^2 are fixed at the levels estimated from the California condor annual population data in 1965-1980 and 1995-2014, respectively, which are discussed in details in section 2.4.1.

From Figure 2.3 one can see that when $\mu < 0$, as μ becomes less negative, the modal time to extinction increases at an increasing rate until a certain threshold and then increases at a decreasing rate.³ When $\mu > 0$, as μ becomes more positive, the modal time to extinction decreases at an increasing rate until a certain threshold and then decreases at a decreasing rate.⁴ This implies that when $\mu > 0$,

³the value of the threshold depends on the data set and is close to $\mu = 0$

⁴the value of the threshold depends on the data set and is close to $\mu = 0$

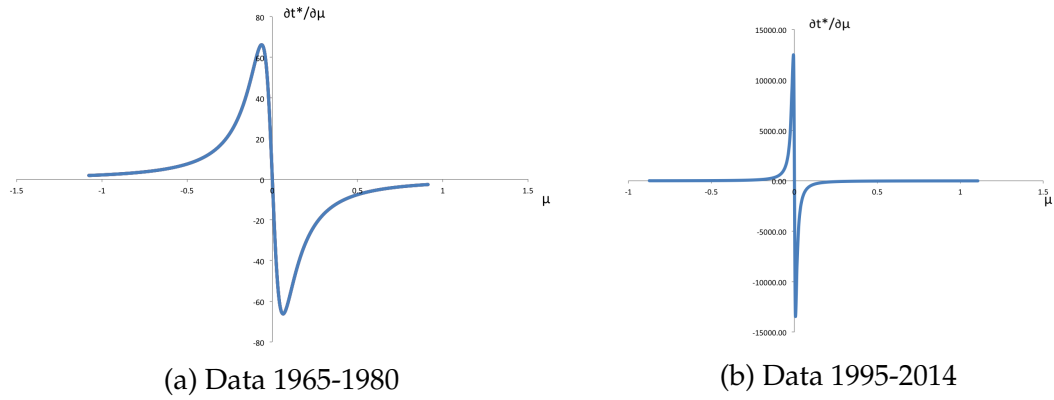


Figure 2.3: Modal Time Changes with Infinitesimal Mean

a conservation program that increases μ will actually fasten the modal time to extinction, although it will reduce the probability of extinction, i.e. both the PDF and CDF will be reduced for any t . For this reason, although the modal time is a commonly used indicator for risks, it won't be a good measure when it comes to risk comparison.

2.3 The Benefits of Reducing the Risk of Extinction through Captive Breeding and Recovery Programs

Suppose population data of a species in periods $t_0, t_1, \dots, t_q, t_{q+1}, \dots, t_n$ are observed, where the captive breeding program/recovery program starts from period t_{q+1} . If the conservation program significantly altered the population dynamic parameters μ and σ^2 , it may also significantly alter the risk of extinction. Using the population data in $t_0 \leq t \leq t_q$ one can estimate $\hat{\mu}_0$ and $\tilde{\sigma}_0^2$. Similarly, using the population data in $t_{q+1} \leq t \leq t_n$ one can estimate $\hat{\mu}_1$ and $\tilde{\sigma}_1^2$. These two sets of parameter values $(\hat{\mu}_0, \tilde{\sigma}_0^2, x_q)$ and $(\hat{\mu}_1, \tilde{\sigma}_1^2, x_n)$ will give two different projectories of the future risk of extinction, $p(t; \mu_0, \sigma_0, x_q)$ starting from period t_{q+1} and

$p(t; \mu_1, \sigma_1, x_n)$ starting from period t_{n+1} .

The benefits of the captive breeding and recovery program for a species might be measured by

$$B = \left(\int_{t=0}^{\infty} e^{-\delta t} p(t; \mu_0, \sigma_0, x_q) dt - \int_{t=0}^{\infty} e^{-\delta t} p(t; \mu_1, \sigma_1, x_n) dt \right) L \quad (2.17)$$

where δ is a discount rate and L represents the existence value of a specific endangered species. The first term $L \int_{t=0}^{\infty} e^{-\delta t} p(t; \mu_0, \sigma_0, x_q) dt$ measures the expected loss should extinction occur for all future periods when there is no conservation program. The second term $L \int_{t=0}^{\infty} e^{-\delta t} p(t; \mu_1, \sigma_1, x_n) dt$ measures the expected loss of extinction for all future periods when human interventions have already started and altered the growth parameters. The difference between these two values of expected loss should extinction occur gives a measure for the value of reducing the risk of extinction through a conservation program.

The existence value of a wildlife species L is usually estimated based on society's willingness to pay (WTP) to avoid its extinction. Although Weitzman (1992) also proposed a diversity function where each species' existence value can be measured by its genetic or character distance from the collection of existing species, it won't give a monetary measure for the species' existence value. Thus, this essay uses the WTP to calculate the monetary value of reducing the risk of extinction. However, the diversity function by Weitzman (1992) is still summarized in Section 2.3.1 below for curious readers.

2.3.1 Diversity Function by Weitzman (1992)

The genetic similarity of two species within a genus might be measured by a symmetric distance-dissimilarity coefficient, which, in turn, can be derived as a hedonic weighted sum of distances between phenotypic or genetic micro-characteristics. For example, the distance coefficient may represent the weighted number of observable “character-state differences” between two species (Weitzman, 1992).

Weitzman (1992) proposes a value-of-diversity function where the loss of diversity when a species goes extinct depends on the species’ distance from its closest relative. He then shows that the “value of diversity” of a species can be recursively generated from more fundamental information about the dissimilarity-distance between any pair of species in the set. The value of a particular species depends on the realized pattern of extinction. If genetically close relatives have gone extinct, the value of this species would be larger.

This measure of diversity value and the diversity function are a key ingredient for selecting and prioritizing species to protect given a limited amount of resources. However, it doesn’t give a monetary estimate for the existence value of a species. Thus, WTP is a more appropriate measure for L . In addition, when measuring the WTP, the diversity value proposed by Weitzman (1992) is probably already taken into consideration. For example, people are willing to pay more to visit and see a more rare and unusual species than a species that has close relatives in existence.

2.3.2 Society's Willingness to Pay to Avoid Extinction

In economics, willingness to pay (WTP) is the maximum amount an individual is willing to pay to acquire a good or an attribute. If the individual currently possesses the good or attribute, willingness to accept (WTA) would be the minimum compensation required to maintain the same level of utility without the good or attribute. Randall and Stoll (1980) demonstrated that WTP and WTA for changes in environmental amenities should not differ greatly unless there are unusual income effects. Hanemann (1991) further argued that not only the income effect but also the substitution effect between a public good and privately marketed commodities matter for the difference between WTP and WTA. They showed that holding income effects constant, the smaller the substitution effect, the greater the difference between WTP and WTA, which explains most empirical results.

In this dissertation it is assumed that the society has an amount of total WTP to save an endangered species. The amount can be estimated by stated preference methods, which uses a survey to measure household WTP, or can be implicitly revealed from the actual amount paid for a conservation program, which is called revealed preference methods.

The contingent valuation method (CVM) is one of the common stated preference approaches and has been applied to estimate the WTP per person for many threatened or endangered species. For example, Bowker and Stoll (1988) estimated the annual WTP for whooping cranes to be within the \$5 to \$149 range depending on which estimation method was chosen. Kotchen and Reiling (1998) estimated the mean WTP for the peregrine falcon to be around \$29 and shortnose sturgeon to be around \$23. Chambers and Whitehead (2003) es-

estimated the median WTP for a wolf management plan and a wolf damage plan in Minnesota to be \$4 – \$21. Jakobsson and Dragun (1996) applied the CVM to the conservation of endangered species in the State of Victoria, Australia. A CVM survey includes a hypothetical market where a respondent can state his/her WTP for an endangered species. Usually the surveys include three elements: (1) background information and status about the species; (2) payments methods and frequency for the specific species; and (3) the form of the WTP question, whether it is an open question or it is a choice question etc. Loomis and White (1996) uses a regression analysis to identify the variables which explain the variation in WTP values. They found that the percentage change in the species' population size proposed in the survey, the frequency of payments, whether the respondent is a visitor or not, and the ecological category of the species are influencing factors for WTP.

CVM is approved and widely used by federal agencies to perform benefit-cost analysis and natural resource damages estimation. Carefully designed CVM studies have been found to be reliable in test-retest reliability studies (Loomis, 1990). However, its reliability is continually debated (Kotchen and Reilling, 2000). For example, due to the hypothetical nature of the surveys, the stated value might be overestimated. In addition, CVM is relying on human-centered preference but the knowledge is often incomplete to make the best evaluation of a species (Loomis et al., 1996). National Oceanic and Atmospheric Administration (NOAA) suggested that in order for CVM to produce reliable estimates, several guidelines need to be followed. In addition to these guidelines, Kotchen and Reilling (1999) emphasized the importance of examining underlying motivations when studying the CV responses. They found that ethical motives for species protection are associated with environmental attitudes, and

environmental attitudes influence the CVM reliability. For example, they found that respondents with stronger pro-environmental attitudes are more likely to provide legitimate responses and higher WTP, while those with weaker attitudes are more likely to protest hypothetical CVM scenarios. Loomis et al. (1996) argued for valuation of multiple species inhabiting the same ecosystems instead of species-by-species valuation, because one species' survival also relies on other species in the entire ecosystem.

Revealed preference methods are based on observations of individual choices related to an ecosystem service. The two common revealed preference approaches include travel cost models and hedonic price models. Travel cost measures the time and money people spend getting to an environmental good, such as forests, mountains, bird-seeing sites. Hedonic pricing assumes that the price of a good is a function of its characteristics, and the environmental attributes are an important characteristic of the good. Thus the value of the environmental attributes can be calculated from comparing goods with and without such attributes. A more crude revealed preference approach is just adding up the actual expenditures on an environmental good. However, the value estimated by this crude method won't be able to perform cost-benefit analysis.

2.4 An Example of the California Condor

The California condor (*Gymnogyps californianus*) is known for its large size. It has the widest wingspan of any North American bird,⁵ and the heaviest mass among native North American birds.⁶ However, Koford (1953) reported that

⁵Around 10 feet

⁶Around 20-22 pounds

in 1950, California condors were only found in six counties just north of Los Angeles, California. Although it was historically widely distributed in North America, the population had declined to an estimated size of 50–60 birds in 1968 and 25–35 birds in 1978 (Walters et al., 2010). It became protected as an endangered species by federal law in 1967 and by California state law in 1971 (USFWS Pacific Southwest Region, 2013). However, the population continued to decline and it was at the brink of extinction in the 1980s, when it received an enormous amount of attention from the public and the US government. To prevent the extinction of the California condor, a captive breeding program was initiated in 1982. Eggs and several birds were removed from the wild and taken to captive breeding facilities at zoos in San Diego and Los Angeles for hatching and rearing. In the next few years all of the remaining wild condors were brought into captive breeding facilities (Jurek, 2014). After the Los Angeles Zoo and San Diego Wild Animal Park, the Peregrine Fund joined the partnership and opened a captive breeding facility in Boise, Idaho for the California condor in 1993. The Oregon Zoo in Portland joined as a fourth captive breeding facility in 2003. Reintroduction began in 1992 when the first eight captive-reared birds were released in Southern California (USFWS Pacific Southwest Region, 2013). Later on, Arizona, central coastal California, northern Baja California, Mexico, and Pinnacles National Monument were added as additional reintroduction locations (USFWS, 2013). Since then, the total condor population has experienced steady growth over the last two decades (USFWS, 2013).

While the main objective of the California Condor Recovery Program is to prevent extinction, it has also generated a multitude of side benefits, including the development of new techniques for captive breeding, releases, genetic analysis, and the conservation of habitat that supports other species (USFWS

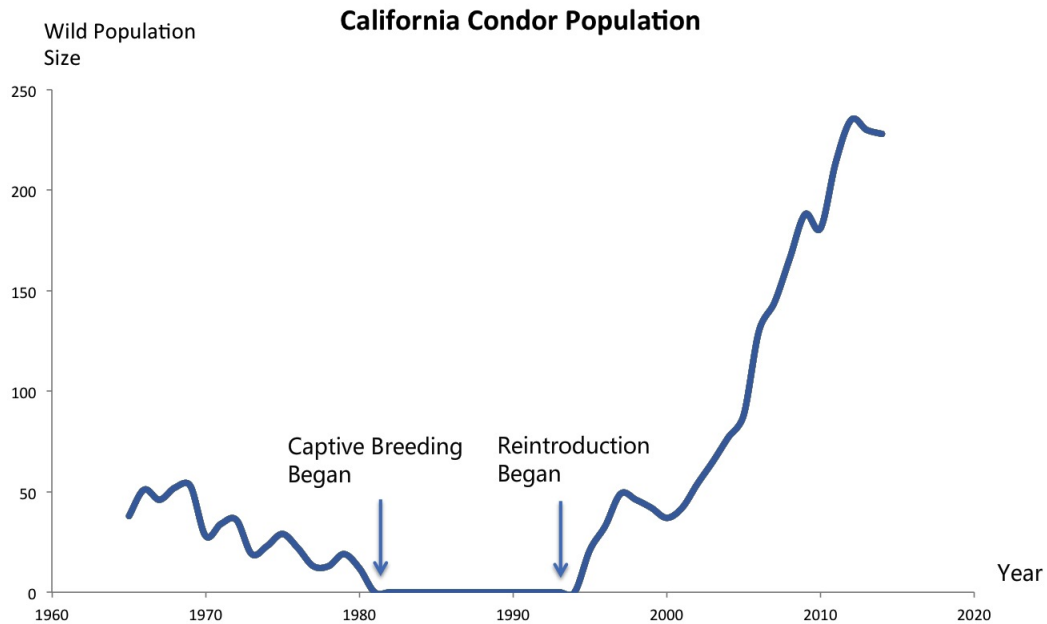


Figure 2.4: California Condor Population

Hopper Mountain National Wildlife Refuge Complex, 2007).

To estimate the benefits of reducing the risk of extinction through the California condor captive breeding and recovery program, equation (2.17) in section 2.3 is used in the empirical analysis in this study.

2.4.1 Population, Modal Time to Extinction and Extinction Risk

Estimates of the California condor population before the captive breeding and recovery program are available from 1965 to 1980 in the October surveys (Walters et al., 2010). These counts may not be strictly accurate because no thorough count was undertaken until the 1980s (Snyder & Johnson, 1985). However, the available estimates do show a continuing decline of the wild population during that period, see Figure 2.4.

Using the pre-program time series data, the parameters $\hat{\mu}_0$ and $\hat{\sigma}_0^2$ and their 95% confidence intervals are estimated and reported in the column (0) of Table 2.1. With extinction defined as only 1 bird left in the wild, the most likely time to reach this threshold from 1980 was 14.6 years.

Wild population data after the captive breeding and recovery program is available from several different reports. However, the reports reveal mild differences in the estimated wild population sizes, due to probably different observation times and collection methods.⁷ The available estimates do show a continuing increase of the wild population after the captive breeding and recovery program, see Figure 2.4. In this subsection, results of estimations from different data samples are presented and discussed.

The California Condor Population and Distribution Monthly Reports (US-FWS Pacific Southwest Region, n.d.) contain monthly population data from December of 2002 to December of 2014. Out of the 144 months, there are 20 months in which population data are unavailable. In this study, these missing entries are filled in with the population average of the month immediately before and the month after each missing entry or each set of consecutive missing entries.

Using the monthly data from 2002 to 2004, the estimates of μ_1 , σ_1^2 and t_1^m (in years) are reported in Table 2.1 Column (1). The time interval between 2 observations is 1 month. The result shows that the predicted modal time to extinction for the California condor is 60.7 years from the end of 2014.

In Column (2) the time interval between 2 observations is 1 year instead of 1 month. Data sets are the same monthly reports as used by Column (1), but

⁷There is not enough information to tell the exact reasons.

Column (2) uses only the end of each year's (December 31st) wild population numbers. The predicted modal time to extinction from year 2014 is 60.6 years. Comparing Column (1) and (2), both the estimates of t_1^m and the 95% confidence intervals are very close. The greater sample size of Column (1) brings higher precision, which gives a narrower 95% confidence interval than Column (2). The reason to display the Column (2) estimation results in addition to Column (1) is for comparison purposes, since the results of Column (3)-(5) are all based on yearly data.

One can also utilize the California condor population data before 2003. Each of the monthly reports does not only contain the population size estimated for the current month, some of them also report yearly time series data of the California condor wild population from 1995 until the years which those reports were produced in. However, unlike the data from 2003 to 2014, the population estimates between 1995 and 2002 have larger variations among reports. The numbers reported in these time series exhibit different population estimates.

These 1995-2002 time series data can be grouped into two sets according to the sizes of monthly variations. The first group of 1995-2002 yearly time series data are reported in each of the monthly reports of 2013 to 2014. The second group of 1995-2002 time series data are reported in each of the monthly reports of 2003 to 2012. Compared to the second group, the first group has a more optimistic estimation of the population between 1995 and 1998 and a less optimistic estimation between 1999 and 2005. Because the time series data also has some mild variations within each group itself, a range of $\hat{\mu}_1$ and $\tilde{\sigma}_1^2$ and a range of their 95% confidence intervals are reported in Column (3) for group 1 and in Column (4) for group 2. The prediction of the modal time to extinction

is between 40.887 and 40.928 years with group 1 data, and between 25.006 and 26.552 years with group 2 data. The predictions are quite different between these two groups.

Column (5) displays the estimates using the yearly time series between 1995 and 2012 reported in the California Condor 5-Year Review (USFWS Pacific Southwest Region, 2013). This data predicts a modal time to extinction of 21.468 years, which is very close to the prediction of group 2 data.

The population data between 1981 and 1994 is not used for estimation because the captive breeding program began in 1982 and the California condor was extinct in the wild from 1982 to 1992. Furthermore, the wild population had another interruption in 1994 when all the released birds were brought back into the captive breeding facilities. In 1995, they were released again. The wild population began to increase thereafter.

To summarize, Column (0) displays the estimations from yearly data of 1965-1980 before the California condor captive breeding program. Column (1) displays the estimations from monthly data of 2003-2014 and Column (2) from yearly data of 2003-2014. Both Column (3) and (4) display estimations from yearly data of 1995-2014. However, the population data of 1995-2003 for Column (3) is from monthly reports in 2013 and 2014 while for Column (4) it is from monthly reports in 2003-2012. Column (5) is based on the yearly population data of 1995-2012 reported in the California Condor 5-Year Review (US Fish Wildlife Services Pacific Southwest Region, 2013). Although Column (1) is based on monthly data thus with more observation points, it only uses the information of 2003-2014. On the other hand, Column (3) and (4) also include

Table 2.1: Estimated Growth Parameters and Modal Time to Extinction (in years) for the California Condor. (Numbers in parentheses are 95% confidence interval bounds)

	Estimation From Population Data		
	(0)	(1)	(2)
$\hat{\mu}_0, \hat{\mu}_1$	-0.0768 (-0.269) (0.115)	0.00727 (0.00305) (0.0115)	0.0873 (0.0273) (0.147)
$\hat{\sigma}_0^2, \hat{\sigma}_1^2$	0.120 (0.0643) (0.298)	0.000659 (0.000529) (0.000843)	0.00890 (0.00446) (0.0256)
\hat{t}_0^m, \hat{t}_1^m	14.626 (1.804) (27.447)	60.672 (26.695) (94.669)	60.623 (26.027) (95.220)
	(3)	(4)	(5)
$\hat{\mu}_0, \hat{\mu}_1$	0.126 (0.0463~ 0.0470) (0.204~ 0.205)	0.192 (0.0450~ 0.0856) (0.297~ 0.338)	0.216 (0.0868) (0.345)
$\hat{\sigma}_0^2, \hat{\sigma}_1^2$	0.0265~ 0.0270 (0.0152~ 0.0154) (0.0580~ 0.0591)	0.0482~ 0.0923 (0.0275~ 0.0527) (0.105~ 0.202)	0.0629 (0.0349) (0.146)
\hat{t}_0^m, \hat{t}_1^m	40.887~ 40.928 (18.722~ 18.918) (62.937~ 63.051)	25.006~ 26.552 (9.693~ 14.057) (39.047~ 40.320)	21.468 (11.773) (35.162)

the observations of 1995-2002 and the estimations for t_1^m are significantly lower compared to Column (1) and (2). Thus it seems that the estimations are sensitive to the observations' time length. In addition, the fact that estimations of Column (3) and (4) are quite different shows that the results are quite sensitive to the 1995-2002 population data variances.

The PDFs of the extinction arrival time before and after the captive breeding

and reintroduction programs are calculated and displayed in Figure 2.5. The discrete time interval is 1 year and the time horizon is 220 years.⁸ No matter which time series is used, there is an obvious decrease of the extinction risks before and after the recovery program in Figure 2.5(a)-(d). The expected risk reduction $\int_{t=0}^{\infty} e^{-\delta t} p(t; \mu_0, \sigma_0, x_q) dt - \int_{t=0}^{\infty} e^{-\delta t} p(t; \mu_1, \sigma_1, x_n) dt$ are almost the same across the five data groups, as shown in the first row in Table 2.2. Their confidence intervals are obtained using a Bootstrap method. This risk reduction, together with the willingness to pay to prevent extinction, are used to calculate the benefits of the captive breeding and recovery program that started in the 1980s.

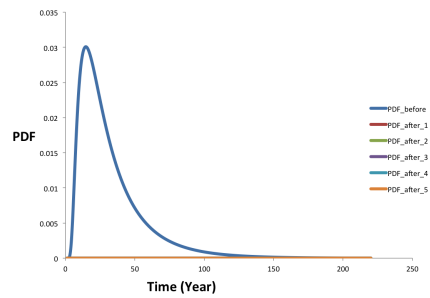
2.4.2 Willingness to Pay to Prevent Extinction

So far there is only one study that estimates the WTP for the California condor and it is based on stated preference surveys (Redden, 2008). It estimated the WTP to be \$99.2 per person. Aggregating this amount by the population in Southern California gives \$2 billion US dollars.

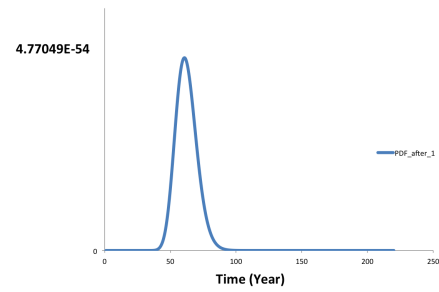
The existence value L can also be approximated using the revealed preference approach, by the total costs that have been and will be spent on the captive breeding and recovery plan. As discussed in Section 1.2 in Chapter 1, the California Condor Recovery Program is led by the U.S. Fish and Wildlife Service and is partnered with other federal agencies, state governments, non-governmental organizations and private communities. The funding of the recovery program comes from the USFWS, the partners, and private donors.

The annual financial expenditures by the USFWS and the 6 major partners

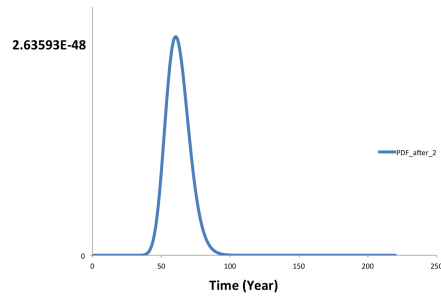
⁸After 220 periods the results converge at 10^{-5} level.



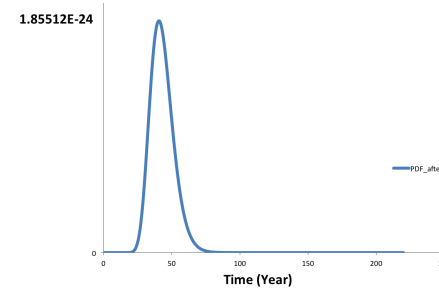
(a) Before the Conservation Program



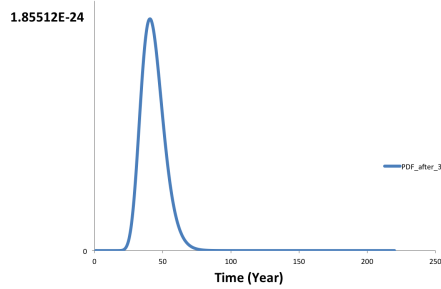
(b) After (with Column (1) estimation)



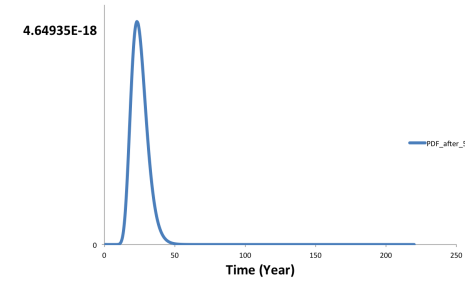
(c) After (with Column (2) estimation)



(d) After (with Column (3) estimation)



(e) After (with Column (4) estimation)



(f) After (with Column (5) estimation)

Figure 2.5: Extinction Arrival Time PDF Before and After the California Condor Captive Breeding and Reintroduction Programs

in 2007 are reported in Walters et al (2010) and is shown in Appendix. However, their annual expenditures in years other than 2007 are not available except for TPF, whose expenditures between 1996 and 2007 are reported in the three 5-year reviews by the California Condor Recovery Southwest Working Group. In their 5-year reviews, other partners in the southwest region also reported their annual expenditures between 1996 and 2007.

Although no detailed cost data is available, the amount of money spent on the condor program over the past 2-3 decades is estimated to be tens of millions of dollars. Since 2007, the sum of expenditures spent on the California condor by all program participants is estimated to be 5 million dollars annually (Walters et al., 2010). However, the data of the annual expenditure is not available for other years.

The actual expenditures, however, are much higher than the projected costs listed in the 1996 recovery plan (Kiff, Mesta & Wallace, 1996). The five actions needed for recovery are described below and the estimated costs associated with each action for the 16 years between 1995 and 2010 listed in Appendix B:

1. Establish a captive breeding program to preserve the gene pool.
2. Reintroduce California condors to the wild.
3. Minimize mortality factors in the natural environment.
4. Maintain habitat for condor recovery.
5. Implement condor information and education programs (Kiff et al., 1996).

After taking into account of the discount factor, the total projected expenditures of the five actions over the 16 years for the California condor are \$24.920 million.

With the limited cost information available, here four estimates for L are proposed. The first uses the total expenditures spent on the program up until 2007, which was around \$35 million (USFWS Hopper Mountain National Wildlife

Refuge Complex, 2007). The second uses the 2007 annual spending on the program, which was around \$5 million, as an estimate of the average annual cost for an infinite horizon and calculates the present value of the discounted sum of all future program costs. The third uses the WTP from the contingent valuation survey from Redden, 2008. The fourth uses the total expenditures predicted by the 1996 Recovery Plan for the California Condor (Kiff et al., 1996).

Thus, the four estimates of existence value are $L_1 = \$35$ million, $L_2 = \$5,000,000 \sum_{t=0}^{\infty} \rho^t = \$5,000,000 \frac{1}{1-\rho} \approx \255 million, where $\rho = \frac{1}{1+\delta}$ and $\delta = 0.02$, $L_3 = \$2$ billion, and $L_4 = \$24.920$ million.

The five columns in Table 2.2 display the results with the five different μ_1 and σ_1 estimated. However, the expected risk reduction $\int_{t=0}^{\infty} e^{-\delta t} p(t; \mu_0, \sigma_0, x_q) dt - \int_{t=0}^{\infty} e^{-\delta t} p(t; \mu_1, \sigma_1, x_n) dt$ are almost the same across the five data groups.

The values for L_1 and L_2 , which are based on actual spending, might be viewed as a more reasonable measure that provides an upper and a lower bound estimate for the value of the California condor captive breeding and recovery program. For $L_2 = \$255$ million, it is assumed that the public is willing to pay \$5 million each year, ad infinitum. Thus equation (2.17) with L_2 would give an upper bound estimate of the benefits of the risk reduction through a captive breeding and recovery program. If instead, only the spending up to 2007 is used as a conservative estimate of the California condor's existence value, one might argue that equation (2.17) with L_1 provides a lower bound estimate of the value.⁹ The monetary benefits of the program are displayed in Table 2.2. The PDFs of the extinction first arrival time are calculated by equation (2.11) discussed earlier.

⁹Since the cost data is only available up to 2007, one can use the total costs spent up until 2007 as a conservative estimation of the total costs spent on the program so far.

To summarize, using equation (2.17) as the measure for the monetary value of reducing the extinction risk of California condors, it is \$20.167 million when the existence value is \$35 million; \$146.934 million when the existence value is \$255 million. Its value is around \$1152.426 when the existence value is \$2 billion; \$5.371 when the existence value is \$9.321 million. The population data variations will only bring negligible variations on the estimated conservation program value.

2.5 Conclusions

Efforts to reduce the extinction risk of an endangered species are valuable. This study proposes an economic measure for this value and the calculation involves two steps. The first step is to estimate the PDF of the extinction first arrival time before and after a captive breeding and recovery program is initiated. Alternatively, one can calculate the expected risk reduction. A statistical method from Dennis et al. (1991) is used. The second step is to estimate the existence value of a species, which might be either estimated from contingent valuation methods or revealed from actual or budgeted expenditures on the captive breeding and recovery program.

The California condor captive breeding and recovery program is examined to illustrate this two-step approach. The benefits of risk reduction are estimated to be in a range between \$5.371 and \$1152.426 millions.

Besides the successful story of the California condor, many other endangered species have benefited from captive breeding and recovery programs. One of the first successful example is that the Phoenix Zoo started a captive breed-

ing program in 1962 for Arabian oryx and that the first Arabian oryx was reintroduced in Oman in 1982 (“The loneliest animals”, 2009). This elegant white ungulate went extinct in the wild in 1972 due to over-hunting. More recently, captive breeding programs have been successful in recovering the black-footed ferret, the golden lion tamarin, and the red wolf (“The loneliest animals”, 2009). The calculation method discussed in this study can also be applied to estimate the risk reduction benefits of these other captive breeding and recovery programs.

Table 2.2: Estimated Benefits of the California Condor Captive Breeding and Recovery program, in millions US dollars. (Numbers in parentheses are 95% confidence interval bounds)

	Estimates		
	(1)	(2)	(3)
Expected Risk Reduction	0.576 (0.505) (0.645)	0.576 (0.505) (0.645)	0.576 (0.505) (0.645)
\hat{B}_1 (with $L_1 = \$35$ million)	20.167 (17.675) (22.575)	20.167 (17.675) (22.575)	20.167 (17.675) (22.575)
\hat{B}_2 (with $L_2 = \$255$ million)	146.934 (128.775) (164.475)	146.934 (128.775) (164.475)	146.934 (128.775) (164.475)
\hat{B}_3 (with $L_3 = \$2$ billion)	1152.426 (1010.373) (1290.477)	1152.426 (1010.373) (1290.477)	1152.426 (1010.373) (1290.477)
\hat{B}_4 (with $L_4 = \$9.321$ million)	5.371 (4.707) (3.464)	5.371 (4.707) (3.464)	5.371 (4.707) (3.464)
	(4)	(5)	
Expected Risk Reduction	0.576 (0.505) (0.645)	0.576 (0.505) (0.645)	
\hat{B}_1 (with $L_1 = \$35$ million)	20.167 (17.675) (22.575)	20.167 (17.675) (22.575)	
\hat{B}_2 (with $L_2 = \$255$ million)	146.934 (128.775) (164.475)	146.934 (128.775) (164.475)	
\hat{B}_3 (with $L_3 = \$2$ billion)	1152.426 (1010.373) (1290.477)	1152.426 (1010.373) (1290.477)	
\hat{B}_4 (with $L_4 = \$9.321$ million)	5.371 (4.707) (3.464)	5.371 (4.707) (3.464)	

CHAPTER 3
ON THE TIMING OF ACTIONS TO SAVE A WILDLIFE SPECIES

3.1 Research Question

If successful, recovery efforts may reduce the risk of extinction of a wildlife species. However, when should costly recovery actions be taken given that the population of a wildlife species is declining? For an emergency endangered species, the answer should probably be right now, because the risk of extinction is very high. For a declining species which might or might not become extinct in the foreseeable future, it might be worth it to wait for more information (Khan et al., 2015), or simply because the existence of fixed cost. This study proposes two methods for calculating the optimal waiting time. First is a static problem where the future risks can all be calculated, and the optimal timing is simply to wait when the risk of extinction is high enough to justify the fixed cost. The second is a dynamic problem where the optimal timing is state contingent, i.e. it depends on the current population size.

The population is again modeled as a stochastic process discussed by Dennis et al. (1991). For the static approach, hazard rates of extinction for all future periods can be calculated from the population model and it is a key element in calculating the optimal waiting time before taking costly conservation actions (Khan et al., 2015). The optimal timing depends on the costs of recovery actions, the existence value of the species, the effectiveness of the recovery programs in terms of reducing the risk of extinction, and the population size characteristics. The second approach is similar to an entry and exit model (Dixit, 1989), where the conservation agency is deciding whether to intervene (entry), or terminate

(exit) an existing conservation program.

The rest of this essay is organized as follows. Section 3.2-3.6 studies the static problem. Section 3.2 derives the inverse Gaussian hazard rate of extinction from the stochastic population model by Dennis et al (1991). Section 3.3 discusses a hazard rate function derived from a Weibull distribution. Section 3.4 derives the first and second order conditions for the optimal waiting time. Section 3.5 studies a numerical example of the California condor. Section 3.6 studies an example of the whooping crane. Section 3.7 presents and numerically solves the dynamic optimal timing problem and Section 3.8 concludes.

3.2 The Hazard of Extinction

The first approach is a static model that involves the hazard rate. The hazard rate, also known as the failure rate, refers to the probability of failure for an event at a given time, conditional on that the event hasn't failed yet. The hazard rate for any given time can be calculated using the following general form of equation

$$h(t) = p(t)/(1 - P(t)), \quad (3.1)$$

where $p(t)$ is the probability density function of the failure's arrival time and $1 - P(t)$ is the probability that the failure hasn't arrived yet at time t .

For a wildlife species whose population size is way lower than its natural carrying capacity, its population dynamics can be adequately approximated by the same stochastic exponential growth model as in Chapter 2 Section 2.2.1, i.e. the population size N is the solution to a stochastic differential equation (SDE)

$$dN = rNdt + \sigma Ndz, \quad (3.2)$$

where r is the intrinsic growth rate, $\sigma > 0$, is the standard deviation rate and dz is the increment of a Weiner-drift Process. The adequacy of this model is discussed in detail in Chapter 2 and in Dennis et al (1991). The hazard rates of extinction for future periods can be calculated from the species' historical population data as well.

The amount of time, T , elapsing before a pre-specified extinction threshold is first reached is a positive, real-valued random variable with a continuous probability distribution. The value of the pre-specified threshold is decided according to the needs of each study. The cumulative distribution function (CDF) of T can be written in terms of a standard normal CDF

$$P(t) = \Phi\left(\frac{-x_d - \mu t}{\sigma \sqrt{t}}\right) + e^{-2x_d\mu/\sigma^2} \Phi\left(\frac{-x_d + \mu t}{\sigma \sqrt{t}}\right), 0 < t < \infty, \quad (3.3)$$

where $x_d = x_q - x_e$ measures the difference between the natural log of the population sizes at the most recent period x_q and the pre-specified extinction threshold x_e . In addition, $\Phi(\cdot)$ is the CDF of a standard normal distribution. The two population parameters μ and σ^2 are the infinitesimal mean and infinitesimal variance defined earlier. The PDF of T is the derivative of $P(t)$ with respect to t

$$p(t) = x_d(2\pi\sigma^2 t^3)^{-1/2} e^{-(x_d + \mu t)^2/2\sigma^2 t}. \quad (3.4)$$

The CDF and PDF expressions above are standard from the first passage time literature (Ingersoll, 1987). This distribution is known as the inverse Gaussian

distribution. When the infinitesimal growth rate is non-positive, i.e., $\mu \leq 0$, the process will hit the extinction threshold within a finite time with probability 1. When $\mu > 0$, the probability that the process will ever attain the threshold is $e^{-\frac{2\mu x_d}{\sigma^2}}$, i.e., the probability distribution has a mass point of $1 - e^{-\frac{2\mu x_d}{\sigma^2}}$ at positive infinity.

The hazard rate can be calculated as

$$h(t) \equiv \frac{p(t)}{1 - P(t)} = \frac{x_d(2\pi\sigma^2 t^3)^{-1/2} e^{-(x_d + \mu t)^2 / 2\sigma^2 t}}{\Phi\left(\frac{x_d + \mu t}{\sigma\sqrt{t}}\right) - e^{2x_d\mu/\sigma^2} \Phi\left(\frac{-x_d + \mu t}{\sigma\sqrt{t}}\right)}. \quad (3.5)$$

The expression for $h(t)$ is rather complicated but it is not difficult to compute for any given values of parameters x_d, μ and σ^2 . The shape of a hazard rate function is relevant in determining the optimal timing, as discussed in Section 3.4. The inverse Gaussian hazard rate function here in general is non-monotonic. Chhikara & Folks (1977) proved that this hazard rate is increasing then decreasing with a unique maximum attained at t^{IG} , where t^{IG} is the solution to the equation

$$h(t) = \frac{\mu^2}{2\sigma^2} + \frac{3}{2t} - \frac{x_d^2}{2\sigma^2 t^2}. \quad (3.6)$$

It is also proved by Chhikara et al (1977) that $t^{IG} \in (t^m, \frac{2x_d^2}{3\sigma^2})$, where t^m represents the modal time to extinction and

$$\hat{t}^m = \frac{x_d}{|\hat{\mu}|} \left[\left(1 + \frac{9}{4\hat{\nu}^2}\right)^{\frac{1}{2}} - \frac{3}{2\hat{\nu}} \right] \quad (3.7)$$

where $\hat{\nu} = x_d |\hat{\mu}| / \hat{\sigma}^2$, as discussed in Section 2.2.1 in Chapter 2.

3.3 The Weibull Distribution and Hazard Rates

As opposed to the inverse Gaussian hazard rate function discussed in the previous section, another interesting probability distribution for hazard rates is the Weibull distribution because it has been often used in risk analysis. In this section the optimal timing question is analyzed again for Weibull distribution hazard rate, for curious readers.

Assume that extinction arrival time, T , has continuous densities p on $(0, \infty)$ and possibly an atom at ∞ . If T has an atom at ∞ , it is called an incomplete distribution, corresponding, e.g., the species will never become extinct. A Weibull distribution is of the form $T = X^\gamma$, where $\gamma > 0$ and X is a negative exponentially distributed random variable with parameters $\lambda > 0$, $0 \leq q < 1$. Strictly positive q implies that the Weibull distribution is incomplete, i.e., there is a chance that extinction will never happen. And $q = 0$ implies that the Weibull distribution is complete, i.e., extinction will happen within a finite time for sure. The CDF of the Weibull distribution is

$$P(t) = (1 - q)(1 - e^{-\lambda t^{1/\gamma}}). \quad (3.8)$$

The PDF is

$$p(t) = (1 - q) \frac{\lambda}{\gamma} t^{\frac{1-\gamma}{\gamma}} e^{-\lambda t^{1/\gamma}}. \quad (3.9)$$

Thus, the hazard rate is

$$h(t) = \frac{\lambda}{\gamma} t^{\frac{1-\gamma}{\gamma}} [(q/(1 - q))e^{\lambda t^{1/\gamma}} + 1]^{-1}. \quad (3.10)$$

The shape of the hazard rate function has the following cases depending on γ and q

(a) If $\gamma > 1$, then $h(0_+) = \infty$, and the hazard rate strictly decreases to 0 whether or not $q > 0$.

(b) If $\gamma < 1$, then $h(0) = 0$. If $q > 0$, the hazard rate is hump-shaped, first increasing then decreasing back to 0. If $q = 0$, the hazard rate is strictly increasing, and $\lim_{t \rightarrow \infty} h(t) = \infty$.

(c) If $\gamma = 1$, the Weibull distribution is the same as the negative exponential distribution. If $q = 0$, the hazard rate is constant over time. If $q > 0$, the hazard rate strictly decreases to 0.

The infinitesimal mean μ is negative before an intervention. So $q = 0$ i.e., the extinction arrival time T has a complete Weibull distribution before an intervention. Thus the PDF is

$$p(t) = (\lambda/\gamma)e^{-\lambda t^{1/\gamma}} t^{(1-\gamma)/\gamma}, \quad (3.11)$$

and the CDF is

$$P(t) = 1 - e^{-\lambda t^{1/\gamma}}, \quad (3.12)$$

and the hazard rate

$$h(t) = (\lambda/\gamma)t^{(1-\gamma)/\gamma}. \quad (3.13)$$

3.4 FOCs and SOC

Conservation efforts can change the risk of extinction. There are several ways to model this risk reduction. In Chapter 2 and Section 3.7 of this chapter, for

example, it is modelled as a change in the population parameters μ and σ before and after an intervention. In Khan et al. (2015), it is modelled as a proportional downward shift of the disaster arrival time probabilities. This approach is used in this section. Specifically, assume that conservation actions can reduce the extinction arrival time probability at time t by changing the probability density from $p(t)$ to

$$p_{\theta}(t; t_1) = \begin{cases} p(t) & \text{if } t < t_1 \\ (1 - \theta)p(t) & \text{if } t \geq t_1 \end{cases}, \quad (3.14)$$

where t_1 is the time when conservation actions are started and $0 < \theta < 1$ is an efficacy measure of the conservation actions. The optimization problem is

$$\text{Min}_{t^*} L \int_0^{t^*} p(t)e^{-\delta t} dt + L \int_{t^*}^{\infty} (1 - \theta)p(t)e^{-\delta t} dt + e^{-\delta t^*} K[1 - P(t^*)] + \int_{t^*}^{\infty} \left(\int_{t^*}^t ce^{-\delta t} dt \right) (1 - \theta)p(t) dt \quad (3.15)$$

The first two terms represent the expected loss before and after intervention. The third term is the discounted fixed cost conditional on that extinction hasn't arrived yet at time t^* . The fourth term represents the discounted sum of flow costs, which will be incurred, as long as the species is still in existence after t^* .

First Order Condition are derived

$$p(t^*)e^{-\delta t^*} L - (1 - \theta)p(t^*)e^{-\delta t^*} L - e^{-\delta t^*} Kp(t^*) - \delta e^{-\delta t^*} K[1 - P(t^*)] - ce^{-\delta t^*} (1 - \theta)p(t^*) - \int_{t^*}^{\infty} (1 - \theta)p(t)ce^{-\delta t} dt = 0$$

$$\theta p(t^*)e^{-\delta t^*} L - e^{-\delta t^*} Kp(t^*) - \delta e^{-\delta t^*} K[1 - P(t^*)] - ce^{-\delta t^*} (1 - \theta)p(t^*) - (1 - \theta)ce^{-\delta t^*} [1 - P(t^*)] = 0$$

$$(\theta L - K - (1 - \theta)c)p(t^*) = (\delta K - (1 - \theta)c)[1 - P(t^*)]$$

Thus, the FOC in terms of hazard rate is

$$h(t^{FOC}) = \frac{p(t^*)}{1 - P(t^*)} = \frac{\delta K - (1 - \theta)c}{\theta L - K - (1 - \theta)c} \quad (3.16)$$

The Second Order Condition requires $h'(t^{FOC}) > 0$, because it only makes sense to postpone the fixed cost K and flow cost c if they outweigh the hazard of losing the benefits of the continued existence of the species. It might seem reasonable that the optimal time to start conservation actions is right now. However, given the economic costs associated with it, the conservation agency should wait until the extinction risk is high enough and becoming more so.

The optimal timing for several different hazard rate functions is

(a) When the hazard rate is everywhere increasing, the SOC is satisfied. Thus the optimal time to start conservation programs should be $t^* = t^{FOC}$. For example, a Weibull distribution with $0 < \gamma < 1$ and $\lambda > 0$ has a hazard rate that is everywhere increasing.

(b) If the hazard rate is everywhere decreasing, and if $h(0) \geq h(t^{FOC})$, the optimal waiting time is $t^* = 0$, immediately launching conservation programs. If the hazard rate is everywhere decreasing, and if $h(0) < h(t^{FOC})$, the optimal waiting time is $t^* = \infty$, never launching the conservation programs.

(c) If the hazard rate is hump-shaped (such as in the case of stochastic exponential growth model in Section 3.2), then either $t^* = t^{FOC}$ or $t^* = \infty$ is optimal. If

there is a t^{FOC} where the hazard rate is increasing, $t^* = t^{FOC}$ is optimal. Otherwise, it is optimal to never launch the conservation program.

The **efficacy parameter** θ differs among species and among programs. For example, the captive breeding and reintroduction programs for the California Condor appear to have been more effective than the captive breeding and reintroduction programs for the whooping crane in terms of hedging and reintroducing birds that are self-sustaining in the wild. Thus the California condor would have a higher efficacy parameter θ for reducing the probability of extinction. A reasonable range of values for θ should be decided by biologists and ecologists. From a retrospective, however, with the available historical population data for those endangered species whose conservation programs have already been started, there is a plausible candidate for the efficacy parameter θ —the ratio of the number of captively bred individuals that survive in the wild (didn't die or return to the facility) and the total number of individuals that are released into the wild.

This measure can be calculated only when the recovery program has already been launched so that the data before and after the recovery programs became available. However, the question of optimal waiting time is meaningful only in the cases where the recovery actions haven't started yet. Thus, the efficacy parameter θ value ultimately has to be decided by field experts' judgments and predictions. The two examples in the following sections are two retrospective cases to illustrate the calculation method and they are not used to determine or suggest whether their conservation programs were started too early or too late.

3.5 California Condor & Captive Breeding Program

Section 2.4 in Chapter 2 gives the background information of the California condor, which is an endangered bird, and its recovery program. This section illustrates how the equations from Section 3.2-3.4 can be used to calculate the optimal time to start the conservation programs for the California condor, if the conservation programs hadn't been started in 1980 yet. Note that due to the lack of detailed cost data, this example is only used to illustrate the calculation method.

Gaussian Distribution

With the 1965-1980 California Condor population data, which is before the captive breeding program started, and the stochastic exponential growth model, both the infinitesimal mean μ and the infinitesimal variance σ^2 can be estimated. Plugging in these two parameters into equation (3.5) and assuming 1980 was the year when conservation commencement decisions had to be made, the hazard rates of extinction from 1980 onwards can be forecasted. The hazard rates of extinction for the next 100 years after 1980 are shown in Figure 3.1. The PDF of the extinction arrival time has an inverse Gaussian distribution.

If the efficacy parameter θ is calculated as the proportion of the total released California condors that are still surviving. From the 3rd 5 year review of the southwest region (Austin, Day, Gatto, Humphrey, Parish, Rodgers, Sieg, Smith, Sullivan & Young, 2012), 64 of the 134 released individuals in the Southwest region between 1996 and 2011 were still surviving in 2012. Thus the conservation program efficacy parameter can be calculated as $\theta = 0.478$. In this section the hazard rate formula represented by equation (3.5) will be used and a discount factor $\delta = 0.02$, which is the current inflation rate, is assumed.

From the 1996 California Condor Recovery Plan (Kiff et al., 1996), the following five recovery actions are needed and their estimated costs are also given over the 15 years from 1996 to 2010.

1. Establish a captive breeding program to preserve the gene pool.
2. Reintroduce California condors to the wild.
3. Minimize mortality factors in the natural environment.
4. Maintain habitat for condor recovery.
5. Implement condor information and education programs (Kiff et al., 1996).

However, these budgeted costs are much lower than the amount that is actually spent. According to the 5 year review report (Walters et al., 2010), \$5 million is spent on the California condor conservation programs in 2007. Because more detailed breakdown cost information is not available, the value of the program's fixed cost is unknown. Therefore, a range of arbitrary values for fixed cost $K \in (\$5, \$30)$ million is used. Fixed cost includes initial land acquisition, setting up captive breeding facilities, etc. Flow cost c is set at \$5 million, which is estimated from the 5 year review report (Walters et al., 2010). The existence value of the California condor is difficult to estimate. The only study estimating the willingness to pay (WTP) for the continuing existence of the California condor (Redden, 2008) gives the WTP as \$99.2 per person. Aggregating this amount by the population in Southern California gives \$2 billion.

Weibull Distribution

It might be interesting to find the optimal waiting time when the extinction arrival time has a Weibull distribution. The approximated Weibull distribution

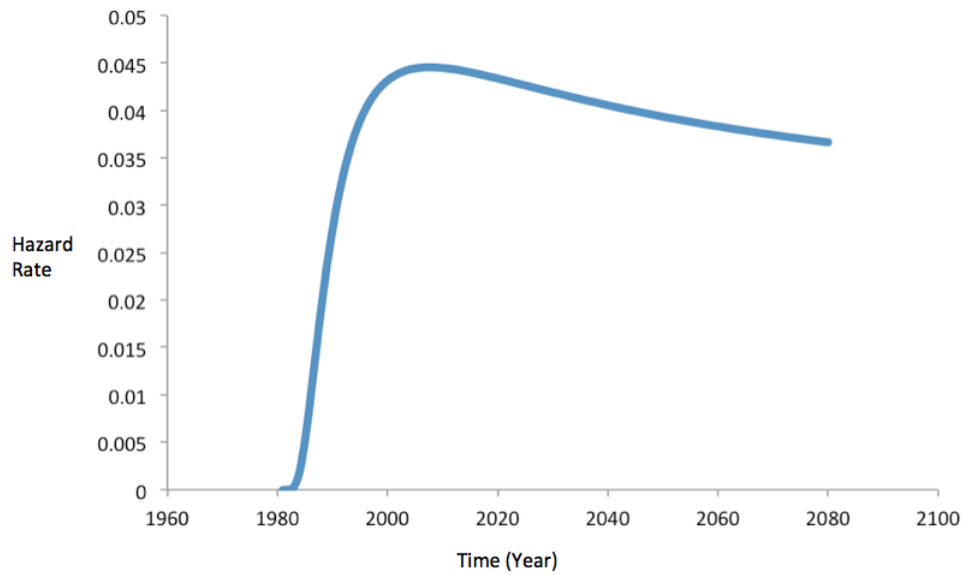


Figure 3.1: Hazard Rate of an Inverse Gaussian Distribution from California Condor Population 1965-1980

is obtained by taking n probability observations p_1, p_2, \dots, p_n over 50 years after 1980 from the inverse Gaussian CDF, given by equation (3.3). These points are used to estimate the most fitted parameters λ and γ for the Weibull distribution by minimizing the sum of squared residuals. The CDF of the Gaussian Distribution and the CDF of the fitted Weibull Distribution are shown in Figure 3.2, where the fitted parameters for the Weibull distribution are $\gamma = 0.625$, $\lambda = 0.00375$.

The hazard rates derived from the fitted Weibull distribution of the California Condor are calculated for 50 years after 1980, as shown in Figure 3.3.

No matter which hazard rate function is assumed, with the cost parameter values and the discount factor value listed above, it is never optimal to wait to start a conservation program unless the fixed cost K is above \$131 million. Table 3.1 is an example of the optimal timing when the fixed cost is \$200 million.

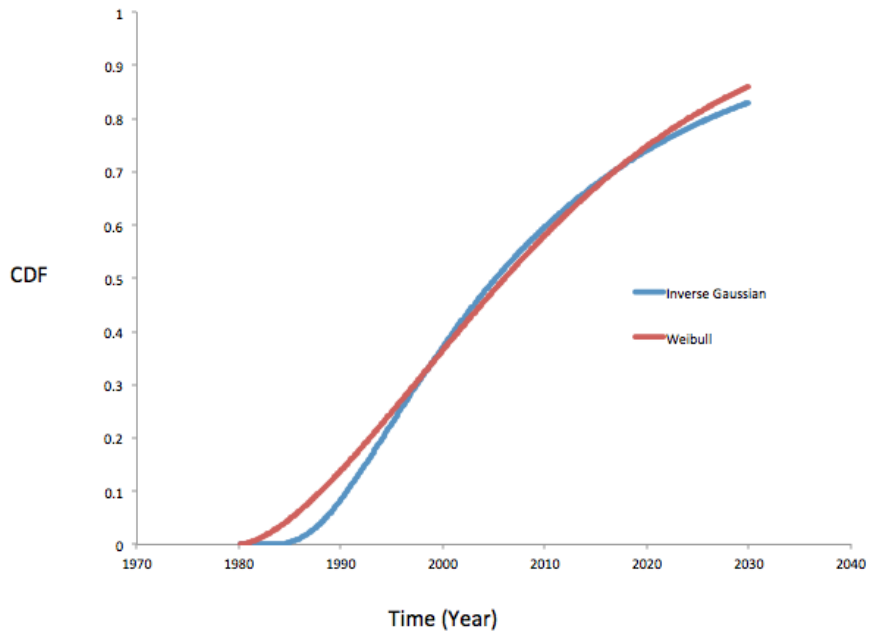


Figure 3.2: CDF of Gaussian Distribution vs CDF of fitted Weibull Distribution from California Condor Population 1965-1980

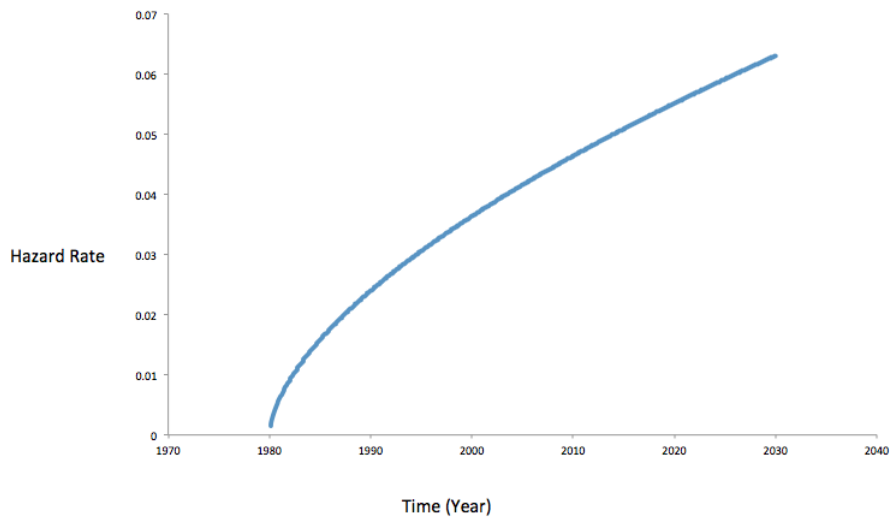


Figure 3.3: Hazard Rate of fitted Weibull Distribution from California Condor Population 1965-1980

Table 3.1: California Condor Captive Breeding Program Optimal Timing

	Optimal Waiting Time (Years)		Threshold $h(t^{FOC})$
	Inverse Gaussian	Weibull	
$\theta = 0.478$	4.0	0.1	0.00184

3.6 Whooping Crane & Captive Breeding Program

3.6.1 Whooping crane

The whooping crane, named after its whooping sound, is a large and elegant white bird (Operation Immigration, n.d.-b). Before human interference, there were believed to be 15,000-20,000 whooping cranes, which fell to around 1400 in 1860 and then became at the brink of extinction of only 16 birds in 1941 (USFWS, 2015c). The decline in population is mainly resulted from habitat loss, hunting, construction of additional power lines, degradation of coastal ecosystems, and threats of chemical spills in Texas.

The 16 surviving whooping cranes all belonged to one flock that migrated between Wood Buffalo National Park in Canada and the Aransas National Wildlife Refuge in Texas. The recovery goal is to delist the whooping crane from the endangered list through establishing multiple self-sustaining populations of whooping cranes in the wild. Self-sustaining populations are the ones that do not require any human help. According to the recovery plan prepared by the Canadian Wildlife Service and USFWS (2007), given that the establishment of second and third wild self-sustaining populations hasn't yet been successful, biologists believe the Aransas-Wood Buffalo (AWB) population must have at

least 1,000 individual birds and 250 reproductive pairs before the species can be considered for down-listing under the Endangered Species Act.¹

To achieve this goal, conservation strategies include habitat protection, maintenance of a captive breeding flock; reintroduction of self-sustaining wild flocks (Canadian Wildlife Service & USFWS, 2007). With the conservation efforts, the AWB population reached 57 by 1970 and 304 by 2015 (Didrickson, 2015). Today, there are two migratory populations and two non-migratory populations of whooping cranes, which are listed below and are shown in the figure 3.4 (Didrickson, 2015).

1. Natural Migratory Flock (AWB Population): The largest flock is also the only natural flock. It spends winters in Aransas National Wildlife Refuge in Texas and breeds in Wood Buffalo National Park in Canada.

2. Non-Natural Migratory Flock (WCE population): This flock winters at the Chassahowitzka National Wildlife Refuge in Florida and breeds in the Necedah National Wildlife Refuge in Wisconsin. It was initially reintroduced in 2001, and it reached 94 birds at the end of 2015.

3. Two Non-Migratory Flocks: One of the non-migratory flock was formed in Florida as a reintroduction program. They live near Kissimmee in Florida year-round. After its initial reintroduction in 1993, this population reached 87 birds in 2001 but it dropped to only 8 birds in 2015, due to high mortality rate. Another non-migratory flock was formed in the coastal area in Louisiana. The

¹However, if two additional wild self-sustaining populations can be successful, the delisting criteria are maintaining a minimum of 40 productive pairs in the AWB for at least 10 years, while establishing a minimum of 25 productive pairs at each of the two additional locations. Population targets are 160 in the AWB population, and 100 each in the other two populations. If only one additional wild self-sustaining population is successful, then the delisting criteria are at least 400 individuals (i.e. 100 productive pairs) for AWB population, and the additional population must remain above 120 individuals (i.e. 30 productive pairs).

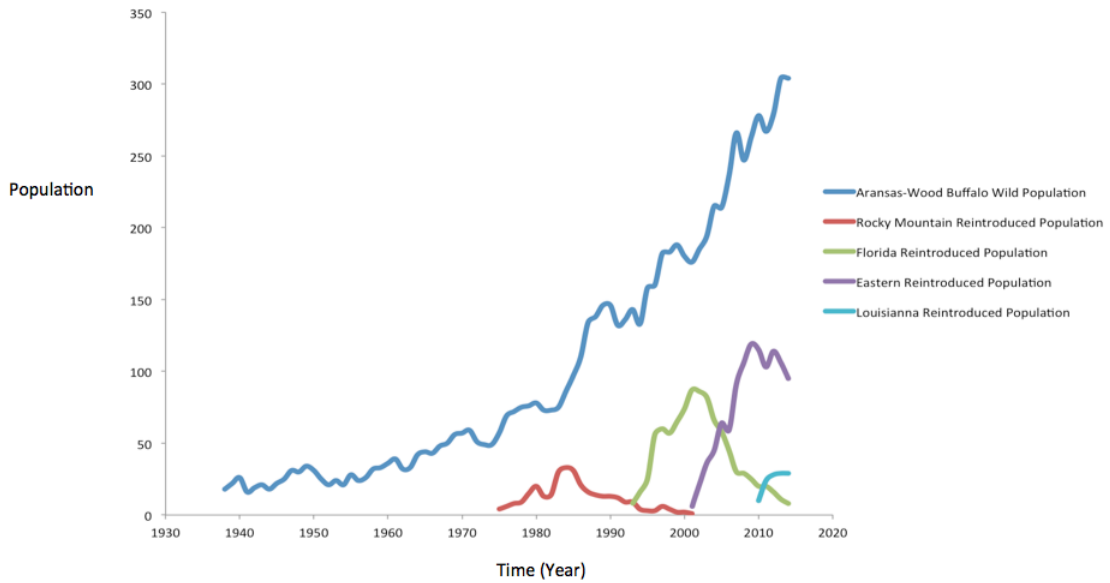


Figure 3.4: The Population of Multiple Whooping Crane Flocks

Louisiana non-migratory population increased to 29 birds in 2015 since its first reintroduction in 2010 (Operation Immigration, n.d.-b).

3.6.2 The Efficacy Parameter θ

The survival rate of released whooping cranes in WCE population can be obtained from 2014 Condensed Annual Report (Whooping Crane Eastern Partnership, 2014). According to this report, since the reintroduction into the WCE Population began in 2001, a total of 239 whooping cranes have been released as juveniles by the end of 2014. In addition, seven wild hatched fledglings (one in 2006, two in 2010, two in 2012, one in 2013, one in 2014) resulted in a grand total of 246 reintroduced individuals. Currently 100 may have survived in the free-ranging WCE population. Thus the survival rate is around 40.6%. For the Louisiana flock, the survival rate is 58%. These two values can be used as the

effectiveness parameter θ of the whooping crane captive breeding and reintroduction program.

3.6.3 Cost Data

Program Costs

The only publicly available data on the recovery program costs is from the 2007 Whooping Crane Recovery Plan (Canadian Wildlife Service & USFWS, 2007). The budget for whooping crane recovery program in year 2007 is estimated to be \$6.1 million, at just over \$30 million through 2010 and nearly \$126 million through 2035. These costs are broken down into 5 actions.

1. Maintain the AWB population through habitat protection.
2. Maintain and increase productivity of 4 captive facilities in the United States and 1 in Canada.
3. Establish, protect and manage 2 or more additional self-sustaining wild populations.
4. Continue and improve genetic research in conservation biology.
5. Maintain an outreach program (Canadian Wildlife Service & USFWS, 2007).

However, the breakdown information on the expenditures is only available for the program in Louisiana of the fiscal year 2013-2014. Total is \$573,800, among which \$250,000 is for salaries and fringe benefits, \$61,000 is for university overhead, \$10,000 for travel, \$500 for holding pens, \$16,300 for supplies, \$56,000 for acquisitions, \$10,000 for major repairs, \$170,000 for contractual services, such as contract aircraft, satellite transmission data acquisition, veterinary

services, and outreach/education (Harrell, 2014).

Due to the lack of breakdown costs information, an arbitrary range of the fixed cost $K \in (\$5, \$30)$ million is used for the calculation. The flow cost c is set at the annual expenditures of \$6.1 million reported in the 2007 recovery plan.

Loss Cost L

Several efforts were made to estimate the WTP for saving the whooping crane. With the major ranges between \$20 and \$70 per person (Bowker & Still 1988). Although aggregating this number by the human population size and discounting it over time are difficult, they are not the focus of this essay. This essay simply multiplies the WTP by the whole population in the US (318.9 million in 2014) and gives $L \in (\$6.3, \$22.3)$ billion.

3.6.4 Optimal Time to Wait

Making a Decision in 1966

The whooping crane captive breeding program began in 1966 at the Patuxent Wildlife Research Center and the first reintroduction program began in 1975 at the Rocky Mountain. The Florida population, WCE population and the Louisiana population were reintroduced in 1993, 2001 and 2010 respectively. Suppose that the captive breeding program hadn't start in 1966. Instead, in 1966, a decision of when to start the captive breeding program needed to be made. Using the annual population data of the whooping crane between 1939 and 1966, the infinitesimal mean and the infinitesimal variance can be estimated to be $\hat{\mu} = 0.0311$ and $\tilde{\sigma}^2 = 0.0316$. With these two parameters and given the range

of the existence value $L \in (\$6.3, \$22.3)$, the hazard rates of extinction over the next 100 years after 1966 would actually be too small to justify a captive breeding program, no matter how effective the program is or how little the fixed cost is.

One plausible explanation is that since 1941, recovery efforts such as habitat protection have been made for the AWB population to revert the population from declining to growing. Thus, with a rather significant positive infinitesimal growth rate, the hazard rate of extinction is too small to justify a captive breeding program. Notice that this hypothetical calculation is only for illustrating how the methods can be used. It doesn't make any conclusion or suggestion on whether the USFWS has made the optimal timing decision or not. As a matter of fact, USFWS was making the captive breeding program decision way earlier than 1966, when the "new" information about the population prior to 1966 was not available. In addition, even if it was known beforehand that the AWB population would be growing steadily and the hazard rates would be extremely low, there is a concern that a single catastrophe event such as drought, hurricane, or disease can wipe out the whooping crane population if all of the birds were living only at one location. Thus, USFWS started the captive breeding program in order to reintroduce one or two separate whooping crane populations. Chapter 4 of this dissertation will study this consideration of reintroducing multiple populations at different conservation sites.

Another result is that with the cost parameters and the discount factor, it is never optimal to wait to intervene unless the fixed cost K is above \$182 when the efficacy parameter is $\theta = 0.406$ and above \$129 when the efficacy parameter is $\theta = 0.580$.

Making a Decision in 1900

In 1941 there were only 16 whooping cranes left in the wild. With this extremely low number, starting the captive breeding program in 1966 seems to be very late. Another interesting retrospective question is if the decision needed to be made in 1900, when would the optimal time be to start the captive breeding program.

The reason to choose 1900 as the retrospective year is because it is halfway between 1860 and 1941, and it is known that there were around 1400 whooping cranes in 1860 and 16 in 1941. More detailed population information between 1860 and 1941 is not available². Thus, if assuming that in 1900 there are 708 whooping cranes, which is the average between the population sizes in 1860 and 1941, then the infinitesimal mean is $\hat{\mu} = -0.0168$, while the infinitesimal variance is unknown.

The magnitude of the infinitesimal variance $\tilde{\sigma}^2$ influences the hazard rates. Instead of asking when was the optimal time to start the captive breeding program, this part calculates the range of $\tilde{\sigma}^2$ that would have made 1966 optimal, given the same cost values, efficacy parameter values and the discount factor as in the previous part. As an example, the fixed cost is arbitrarily assumed at $K = \$200$ million. The results are summarized in table 3.2 and table 3.3 gives the threshold hazards rate associated with the optimal timing. Compared to the infinitesimal variance value estimated from the AWB population, it seems that starting the captive breeding program in 1966 was actually not too late.

²The annual population data of the whooping crane is available from 1939

Table 3.2: Whooping Crane Captive Breeding Program Optimal Timing

	Range of Infinitesimal Variance σ^2	
	L=\$6.3 billion	L=\$22.3 billion
$\theta = 0.406$	0.0454	0.0351
$\theta = 0.580$	0.0579	0.0422

Table 3.3: Threshold Hazard Rates Associated with Optimal Timing

	Threshold Hazard Rates $h(t^{FOC})$	
	L=\$6.3 billion	L=\$22.3 billion
$\theta = 0.406$	0.000160	0.0000426
$\theta = 0.580$	0.000417	0.000113

3.7 State Contingent Approach

Instead of calculating for future risks of extinction and making a static decision, the conservation agency can also make state contingent decisions. For example, facing a declining population of the whooping crane, instead of making a set recovery plan into the future, the USFWS can decide the circumstances when an intervention is needed. Which approach is more reasonable can be argued and supported by different institutional needs and it is beyond the concern of this essay. Here two dynamic problems are studied. The first assumes that the conservation program has no stopping option once started. The only chance that the program will be stopped is when the species becomes extinct. The second assumes that a conservation agency can initiate a conservation program and also terminate it if the threat of extinction is small enough.

A discrete time version is studied to avoid first passage time complications.

The discrete time version of the Geometric Brownian motion population dynamics can be represented as

$$\begin{aligned} \frac{x' - x}{\Delta t = 1} = \mu_0 + \sigma_0 z \quad \text{or} \quad \frac{x' - x}{\Delta t = 1} = \mu_1 + \sigma_1 z \\ x' = x + \mu_0 + \sigma_0 z \quad \text{or} \quad x' = x + \mu_1 + \sigma_1 z \end{aligned} \quad (3.17)$$

where z is a discretized Weiner process $z \sim N(0, 1)$. Natural log of the current period's population size is x , next period's is x' . The infinitesimal mean and variance of the process are μ_0 and σ_0^2 before an intervention, and μ_1 and σ_1^2 after an intervention. According to the population dynamics above, the natural log of next period's population size conditioned on current period's x and population parameters, has a normal distribution $x' \sim N(x + \mu_0, \sigma_0^2)$ or $x' \sim N(x + \mu_1, \sigma_1^2)$ with an absorption state at 0.

3.7.1 No Termination

Assume the conservation program has no stopping option once started. The only chance that the program will be stopped is when the species becomes extinct. As in entry and exit model, if the cost associated with termination is too large, termination might not be an option. For environmental polices, this is often the case.

Let V_0 and V_1 denote the optimized expected value of an endangered species when a conservation program is absent and present in current period. The value function V_0 with state variable x is

$$V_0(x) = \text{Max} \left\{ \underbrace{\frac{1}{1+\delta} \left[\int_{x'>0}^{\infty} V_0(x') f(x'; x, \mu_0, \sigma_0) dx' - L \int_{-\infty}^0 f(x'; x, \mu_0, \sigma_0) dx' \right]}_{\text{next period's value if not to intervene}} \right. \\ \left. - \underbrace{K}_{\text{fixed cost}} + \underbrace{V_1(x)}_{\text{value function after intervention}} \right\}. \quad (3.18)$$

The value function V_1 with state variable x is

$$V_1(x) = - \underbrace{c}_{\text{flow cost}} + \underbrace{\frac{1}{1+\delta} \left[\int_{x'>0}^{\infty} V_1(x') f(x'; x, \mu_1, \sigma_1) dx' - L \int_{-\infty}^0 f(x'; x, \mu_1, \sigma_1) dx' \right]}_{\text{next period's value}}. \quad (3.19)$$

The state variable $x > 0$ is the natural log of the current population size. The value function $V_0(\cdot)$ and $V_1(\cdot)$ are only defined for a state variable in the range $(0, \infty)$, i.e. neither $V_0(0)$ or $V_1(0)$ are defined. Variable c is the flow cost and K is the fixed cost of a conservation program. When a species becomes extinct, a one-time loss cost of L will be incurred immediately.

Let probability density functions $f(x'; x, \mu_0, \sigma_0) = \frac{1}{\sqrt{2\pi\sigma_0^2}} e^{-\frac{x'-x-\mu_0}{\sigma_0\sqrt{2}}}$ and $f(x'; x, \mu_1, \sigma_1) = \frac{1}{\sqrt{2\pi\sigma_1^2}} e^{-\frac{x'-x-\mu_1}{\sigma_1\sqrt{2}}}$ denote the normal probability distributions of $x' \in (-\infty, \infty)$, when an absorption state is absent. Thus, the population will become extinction in the next period with probability $\int_{-\infty}^0 f_0(x'; x, \mu_0, \sigma_0) dx'$ if a conservation program is absent or $\int_{-\infty}^0 f_1(x'; x, \mu_1, \sigma_1) dx'$ if a conservation program is in operation. These two are the probabilities that the population will hit the absorption state in the next period, given that the species is still in existence in the current period $x > 0$. If extinction happens, a loss of L will be incurred immediately. Thus the terms $L \int_{-\infty}^0 f(x'; x, \mu_0, \sigma_0) dx'$ and $L \int_{-\infty}^0 f(x'; x, \mu_1, \sigma_1) dx'$ represent the expected loss should extinction occur. If the next period's population size hasn't hit the extinction threshold yet, then the next period's

value function $V_0(x')$ and $V_1(x')$ will be used for calculation. Thus the terms $\int_{x'>0}^{\infty} V_0(x')f(x'; x, \mu_0, \sigma_0)dx'$ and $\int_{x'>0}^{\infty} V_1(x')f(x'; x, \mu_1, \sigma_1)dx'$ represent the expected value of next period if the species survives into the next period.

Unique Existence of the Value Functions Similar to the proof in Section 3.7.2, it can be easily proven that under the condition that $c \leq \frac{\delta}{1+\delta}L$, both V_0 and V_1 are bounded by zero and $-L$. It can also be easily verified that both V_0 and V_1 satisfy Blackwell's sufficient conditions for a contraction. Thus the contraction mapping theorem applies, which guarantees the unique existence of V_0 and V_1 . Please refer to Section 3.7.2 for detailed proof.

Optimal Policy This essay is not able to analytically prove that the optimal policy is a thresholds x^* . Instead, numerical simulations are run to find the optimal policy.

3.7.2 Initiation and Termination

Alternatively, assume that a conservation agency can initiate a conservation program with a fixed cost and also terminate it without a termination cost. Note that the model can easily incorporate a termination cost as well, if necessary. Let V_0 and V_1 denote the optimized expected value of an endangered species when there isn't and there is a conservation program.

$$\begin{aligned}
V_0(x) = \text{Max} \left\{ \underbrace{\frac{1}{1+\delta} \left[\int_{x'>0}^{\infty} V_0(x') f(x'; x, \mu_0, \sigma_0) dx' - L \int_{-\infty}^0 f(x'; x, \mu_0, \sigma_0) dx' \right]}_{\text{next period's value if not to intervene}}, \right. \\
\left. -K - c + \underbrace{\frac{1}{1+\delta} \left[\int_{x'>0}^{\infty} V_1(x') f(x'; x, \mu_1, \sigma_1) dx' - L \int_{-\infty}^0 f(x'; x, \mu_1, \sigma_1) dx' \right]}_{\text{next period's value after intervene}} \right\} \quad (3.20)
\end{aligned}$$

$$\begin{aligned}
V_1(x) = \text{Max} \left\{ \underbrace{\frac{1}{1+\delta} \left[\int_{x'>0}^{\infty} V_0(x') f(x'; x, \mu_0, \sigma_0) dx' - L \int_{-\infty}^0 f(x'; x, \mu_0, \sigma_0) dx' \right]}_{\text{next period's value if terminate}}, \right. \\
\left. - \underbrace{c}_{\text{flow cost}} + \underbrace{\frac{1}{1+\delta} \left[\int_{x'>0}^{\infty} V_1(x') f(x'; x, \mu_1, \sigma_1) dx' - L \int_{-\infty}^0 f(x'; x, \mu_1, \sigma_1) dx' \right]}_{\text{next period's value if continue}} \right\} \quad (3.21)
\end{aligned}$$

When there is no conservation program in the current period, the optimized value is $V_0(x)$ given the natural log of the current population size is $x > 0$. The conservation agency is choosing between intervene and not to intervene. The expected payoff without intervention is $\frac{1}{1+\delta} \left[\int_{x'>0}^{\infty} V_0(x') f(x'; x, \mu_0, \sigma_0) dx' - L \int_{-\infty}^0 f(x'; x, \mu_0, \sigma_0) dx' \right]$, which says that no cost will be incurred in the current period. However, if next period's population size x' is positive, then next period the value function will be $V_0(x')$. If the natural log of population hits the extinction threshold $x_e = 0$, then a loss of L will occur. The expected payoff with intervention is $-K - c + \frac{1}{1+\delta} \left[\int_{x'>0}^{\infty} V_1(x') f(x'; x, \mu_1, \sigma_1) dx' - L \int_{-\infty}^0 f(x'; x, \mu_1, \sigma_1) dx' \right]$, which says that a fixed cost K and one period of flow cost c will be incurred in the current period. Next period, however, if the natural log of population size x' is positive, then next period the value function will be $V_1(x')$. If next period the population hits the extinction threshold $x_e = 0$, then a loss of L will occur. In addition, the conservation program will shift the growth parameters μ_0 and σ_0^2 to $\mu_1 > \mu_0$ and $\sigma_1^2 \leq \sigma_0^2$, which impacts the population transition probabilities $f(\cdot)$.

When there is a conservation program existing in the current period, the optimized value is $V_1(x)$ given the natural log of the current population size is $x > 0$. The conservation agency is choosing between continuing to intervene and terminate the conservation program. The expected payoff of termination is $\frac{1}{1+\delta}[\int_{x'>0}^{\infty} V_0(x')f(x'; x, \mu_0, \sigma_0)dx' - L \int_{-\infty}^0 f(x'; x, \mu_0, \sigma_0)dx']$. Notice that the growth parameters will shift back to μ_0 and σ_0^2 once the program is terminated. Alternatively, it can be assumed that when the conservation program is terminated, the growth parameters will remain at an intermediate level (μ_2, σ_2^2) , which can be easily incorporated. The expected payoff with continuation of intervention is $-c + \frac{1}{1+\delta}[\int_{x'>0}^{\infty} V_1(x')f(x'; x, \mu_1, \sigma_1)dx' - L \int_{-\infty}^0 f(x'; x, \mu_1, \sigma_1)dx']$.

The main difference between this model and the traditional entry and exit models is that the extinction threshold is explicitly written inside the equation. In entry and exit models, 0 is a natural boundary, which is automatically satisfied given the state variable is following a Geometric Brownian motion. In the biological literature, however, the absorption state is $N_e = 1$, which has to be explicitly specified in the model. One might argue to use 0 as the extinction threshold. The reason for not using 0 as the extinction threshold is because x_e would become $-\infty$ which won't be a mathematically meaningful absorption point.

Unique Existence of the Value Functions Both V_0 and V_1 have unique existence. To see this, define operator T_0 and T_1 as

$$\begin{aligned}
(T_0 V_0)(x) = \text{Max} \left\{ \underbrace{\frac{1}{1+\delta} \left[\int_{x'>0}^{\infty} V_0(x') f(x'; x, \mu_0, \sigma_0) dx' - L \int_{-\infty}^0 f(x'; x, \mu_0, \sigma_0) dx' \right]}_{\text{next period's value if not to intervene}}, \right. \\
\left. -K - c + \underbrace{\frac{1}{1+\delta} \left[\int_{x'>0}^{\infty} V_1(x') f(x'; x, \mu_1, \sigma_1) dx' - L \int_{-\infty}^0 f(x'; x, \mu_1, \sigma_1) dx' \right]}_{\text{next period's value after intervene}} \right\}
\end{aligned} \tag{3.22}$$

$$\begin{aligned}
(T_1 V_1)(x) = \text{Max} \left\{ \underbrace{\frac{1}{1+\delta} \left[\int_{x'>0}^{\infty} V_0(x') f(x'; x, \mu_0, \sigma_0) dx' - L \int_{-\infty}^0 f(x'; x, \mu_0, \sigma_0) dx' \right]}_{\text{next period's value if terminate}}, \right. \\
\left. - \underbrace{c}_{\text{flow cost}} + \underbrace{\frac{1}{1+\delta} \left[\int_{x'>0}^{\infty} V_1(x') f(x'; x, \mu_1, \sigma_1) dx' - L \int_{-\infty}^0 f(x'; x, \mu_1, \sigma_1) dx' \right]}_{\text{next period's value if continue}} \right\}
\end{aligned} \tag{3.23}$$

Both T_0 and T_1 are contraction mapping with modulus $\frac{1}{1+\delta}$.

Let $X = \mathbf{R}_+$ and let $B(X)$ be the set of bounded functions $V : X \rightarrow \mathbf{R}$ with the sup norm, $\|V\| = \sup_{x \in X} |V(x)|$. Assume $V_0(x')$ and $V_1(x')$ are bounded between 0 and $-L$. Then after one operation, $T_0 V_0$ and $T_1 V_1$ will also be bounded between 0 and $-L$.

Proof The assumption that $V_0(x') < 0$ and $V_1(x') < 0$ implies that $T_0 V_0 < 0$ and $T_1 V_1 < 0$. The first term on the right hand side of Equation (3.22) is $\frac{1}{1+\delta} \left[\int_{x'>0}^{\infty} V_0(x') f(x'; x, \mu_0, \sigma_0) dx' - L \int_{-\infty}^0 f(x'; x, \mu_0, \sigma_0) dx' \right] = \frac{1}{1+\delta} \left[\int_{x'>0}^{\infty} (V_0(x') + L) f(x'; x, \mu_0, \sigma_0) dx' - L \right] \geq -\frac{1}{1+\delta} L > -L$, because $\left[\int_{x'>0}^{\infty} (V_0(x') + L) f(x'; x, \mu_0, \sigma_0) dx' > 0 \right]$. Thus $T_0 V_0 > -L$. The same proof applies to $T_0 V_0 > -L$. Q.E.D.

Thus the operators $T_0 : B(X) \rightarrow B(X)$ and $T_1 : B(X) \rightarrow B(X)$ are function

mappings of $B(X)$ into itself. Operators T_0 and T_1 satisfy Blackwell's sufficient conditions for a contraction.

Proof For any $V_0^m, V_0^n \in B(X)$ and $V_0^m(x) \leq V_0^n(x)$, for all $x \in X$, it is obvious that $\int_{x'>0}^{\infty} V_0^m(x')f(x'; x, \mu_0, \sigma_0)dx' \leq \int_{x'>0}^{\infty} V_0^n(x')f(x'; x, \mu_0, \sigma_0)dx'$. Thus $(T_0V_0^m)(x) \leq (T_0V_0^n)(x)$, for all $x \in X$. In addition, $\frac{1}{1+\delta} \int_{x'>0}^{\infty} (V_0(x') + a)f(x'; x, \mu_0, \sigma_0)dx' \leq \frac{1}{1+\delta} \int_{x'>0}^{\infty} V_0(x')f(x'; x, \mu_0, \sigma_0)dx' + \frac{1}{1+\delta}a$, which implies that $T_0(V_0+a)(x) \leq T_0(V_0)(x) + \frac{1}{1+\delta}a$, for all $V_0 \in B(X)$, all $a \geq 0$ and all $x \in X$. Similarly, it can be easily proved that T_1 also satisfies these two conditions. Q.E.D.

Since $B(X)$ with the sup norm is a complete metric space, the contraction mapping theorem applies, which guarantees the unique existence of V_0 and V_1 .

Optimal Policy

This essay is not able to analytically prove that the optimal policy is a pair of thresholds (x_0^*, x_1^*) . In the literature of entry and exit, although it is commonly acknowledged that the optimal policy is a pair of thresholds, there is no formal proof. In this essay, numerical simulations are run to find the optimal policy.

Figure 3.5 shows the value function and the optimal policy given the parameter values $K = 10$, $c = 5$, $L = 2,000$, $\delta = 0.02$, $\mu_0 = -0.0768$, $\sigma_0^2 = 0.1199$, $\mu_1 = 0.216$, $\sigma_1^2 = 0.0629$. The state variable x is discretized into a grid of 0.1 between 0.1 and 100. Function iteration is used with a convergence level specified at 0.01. The numerical results give a pair of thresholds $x_0^* = 1.1$ and $x_1^* = 2.4$, which says when there is no conservation program currently in operation, once the population size drops below $e^{x_0^*}$, the conservation agency should start to intervene. When the conservation agency is currently intervening, then once

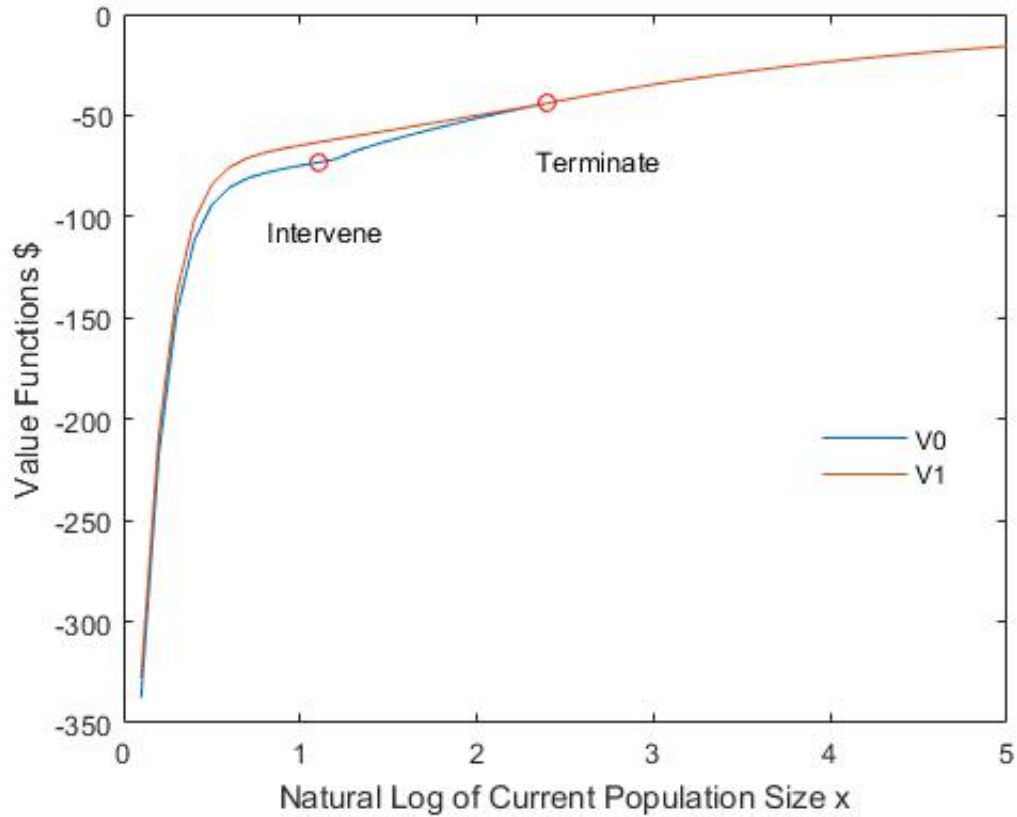


Figure 3.5: Value Functions V_0 and V_1 , in millions

the population size increases above $e^{x_1^*}$, then the conservation program can be terminated.

Table 3.4 and 3.5 below are the simulation results for the California condor and the whooping crane. They display the threshold x_0^* to intervene and x_1^* to terminate. For the California condor, the five columns represent the results from using the five different estimates of μ_1 and σ_1 obtained in Chapter 2. Two values of fixed cost $K = 5$ million and $K = 30$ million are used for simulations. The flow cost is set at $c = 5$ million as discussed in Section 3.5. For the whooping crane, the $\mu_0 = -0.0168$ is calculated using the population sizes in 1900 and 1941, which is before any conservation efforts. Since the infinitesimal variance is unknown

for that period, the same values as of after conservation programs are used for the simulations. Simulations are run on each of the four separate populations, with two different values of the fixed cost $K = 5$ million, $K = 30$ million and two different values of the loss cost $L = 6.3$ billion, $L = 22.3$ billion. The flow cost is set at $c = 6.1$ million as discussed in Section 3.6.3.

Table 3.4: Optimal Policy for California Condors

	Thresholds (x_0^*, x_1^*)		
	(1)	(2)	(3)
$K = 5$	(1.0, 1.4)	(1.1, 1.6)	(1.1, 1.7)
$K = 30$	(0.9, 2.1)	(1.0, 2.6)	(1.0, 2.8)
	(4)	(5)	
$K = 5$	(1.1~ 1.4, 1.9 ~ 2.2)	(1.2, 2.0)	
$K = 30$	(1.0~ 1.2, 3.1 ~ 3.4)	(1.1, 3.2)	

Table 3.5: Optimal Policy for Whooping Cranes

		Thresholds (x_0^*, x_1^*)	
		L=\$6.3 billion	L=\$22.3 billion
AWB	$K = 5$	(0.5, 0.6)	(0.6, 0.7)
	$K = 30$	(0.5, 0.7)	(0.6, 0.8)
Florida	$K = 5$	(2.2, 2.6)	(3.2, 3.7)
	$K = 30$	(1.8, 2.9)	(2.9, 4.0)
WCEP	$K = 5$	(2.0, 2.5)	(2.4, 2.9)
	$K = 30$	(1.8, 3.1)	(2.1, 3.5)
Louisiana	$K = 5$	(1.9, 2.5)	(2.3, 2.8)
	$K = 30$	(1.7, 3.2)	(2.0, 3.5)

3.8 Conclusion

When a species' population is declining, there is a threshold timing of starting conservation actions. This threshold depends on its population data, its existence value, conservation cost data, the discount factor and the effectiveness of the conservation actions. This essay proposes two methods of calculating such thresholds. The two examples studied in this essay illustrate how to use these methods. In the California condor example, using the hazard rate static approach, the optimal waiting time to start its conservation program is calculated to be 4.0 years, if using an inverse Gaussian hazard rate; and 0.1 year, if using a Weibull hazard rate, from a 1980 perspective. In addition, if the fixed cost is below \$131 million, it is never optimal to wait.

In the whooping crane example, it's never optimal to start a captive breeding program, from a 1966 perspective. However, from a 1900 perspective, it's only optimal to start the captive breeding program in 1966 if the infinitesimal variance of the population was between 0.0351 and 0.0579. In addition, if the fixed cost is below \$129 (for efficacy rate of 0.580) or below \$182 (for efficacy rate of 0.406), it is never optimal to wait.

Using a state contingent approach, there is a threshold of the population below which it is optimal to intervene and there is a threshold of the population below which it is optimal to terminate the existing conservation program. For the California condor, the threshold of triggering a conservation program is between $\lceil e^{0.9} \rceil = 3$ and $\lceil e^{1.4} \rceil = 5$. The threshold of triggering a termination is between $\lceil e^{1.4} \rceil = 5$ and $\lceil e^{3.4} \rceil = 30$, depending on the fixed cost and population data. The threshold of triggering a conservation program seems to be really low.

For comparison, another simulation is run with fixed cost $K = \$1.1$ million and flow cost $c = \$1.024$ million, which are the budgeted estimations from the 1984 California Condor Recovery Plan (Harlow, Carrier, Jurek, Kiff, Kimple, Ogden, Risser, Snyder, Thomas & Verner, 1984). With these much lower costs, however, the threshold of triggering a conservation program is still low at 5 birds and the threshold of triggering a termination is at 10 birds. For the whooping crane, the threshold of triggering a conservation program is between $\lceil e^{0.5} \rceil = 2$ and $\lceil e^{3.2} \rceil = 25$. The threshold of triggering a termination is between $\lceil e^{0.6} \rceil = 2$ and $\lceil e^{4.0} \rceil = 55$, depending on the fixed cost, population data and the loss cost.

It can be concluded that with the same parameter values, using a state contingent approach makes it wait for longer than using a static hazard rate approach. The rational behind this result is that for a static problem, the only benefit of waiting is to delay the fixed cost. For a dynamic problem, there is an additional benefit of waiting because one can obtain more information through waiting.

However, there are two caveats with these examples. First, both the California condor and the whooping crane are after-the-fact examples. It might be more relevant to study a species whose conservation actions haven't been started yet. Second, the conservation cost data are not detailed enough to give an accurate answer.

CHAPTER 4
ON THE LOCATIONS OF ENDANGERED SPECIES REINTRODUCTION
PROGRAMS

4.1 Research Question

Captive breeding and reintroduction/releasing programs are two of the main types of recovery actions, and they are complementary to each other. Successfully captive bred individuals will be reintroduced/released into the wild. One of the most successful implementations of these techniques is exemplified in the recovery of the California condor, which is discussed in detail in Section 3. For a successfully captively bred species which is effectively released into the wild, there still lies the threat that a single event, such as a hurricane, drought or disease could wipe out the newly re-established population. For this reason, the recovery plan for the whooping crane seeks to establish three separate and self-sustaining wild populations (Louisiana Department of Wildlife and Fisheries, n.d.). Similarly, the recovery plan for the California condor requires two separate wild populations (Kiff et al., 1996).¹ For the Florida panther, the species will be considered for delisting when there are three separate populations of sufficient size, as determined by the USFWS (2016a). Similar requirements are present in recovery plans for other endangered species to avoid “putting all your eggs in one basket”.

The selection of an additional release site involves multiple criteria. For example, Louisiana was chosen for a new whooping crane population because it was formerly a habitat for the whooping crane and is also a suitable habi-

¹Actually there are three separate wild California condor populations at the moment

tat. Therefore, there is a good chance that a reintroduction there will turn into a self-sustaining population. In addition, this state has a tradition of making commitments and giving support to environmental conservation efforts (Louisiana Department of Wildlife and Fisheries, n.d.).

Habitat suitability is a key criteria for the selection of multiple release sites. Similarly, to ensure the greatest chance for survival, the distance between multiple release sites also requires careful consideration, particularly when there exists regional stochasticity — the risk that diseases, weather catastrophes, forest fires, etc. can simultaneously wipe-out re-established populations located near each other. Several ecological conservation studies have already examined regional stochasticity. For example, in the literature on the Species Set Covering Problem (SSCP) (Underhill, 1994) and the Maximal Covering Species Problem (MCSP) (Church et al., 1996),² Possingham et al. (2000) incorporates a minimum separation distance into the model requirements. An alternative class of models – metapopulation models³ – has also been used for studying spatial-temporal conservation decisions.

In addition to the importance of regional stochasticity in the release site selection, this paper will introduce an economic consideration—the impact of distance on the cost of setup and management for reintroduction/releasing programs. Two re-established wild populations close to each other often incur lower costs than two re-established wild populations that are more distant from each other because they may require less travel time and fewer total conservation staff.

²The SSCP model selects the minimum number of land sites so that each species is covered. The MCSP model aims at protecting as many species as possible for a given number of land sites selected or for a given budget.

³It is a collection of relatively isolated, spatially distributed, local populations bound together by occasional dispersal between populations.

Given that the distance between multiple release sites is important for both the regional stochasticity and the management cost, there a trade-off created by the distance in the selection of release sites. Closer re-established wild populations require lower management and setup costs, but exhibit higher joint extinction risks. The optimal distances between multiple release sites where captively bred individuals should be reintroduced/released is the main query of this study.

Similar to natural reserve design studies which focus on regional persistence, a difficult issue here is the determination of the probability of an extinction arrival time, especially when there is regional stochasticity. In this paper, each individual population is modeled as a stochastic process, as discussed in Dennis et al. (1991). Regional stochasticity is incorporated by assuming that stochastic terms are spatially correlated. The probability of an extinction arrival time must be numerically simulated, because there is no analytical expression for it. Section 4.2 presents this model and Section 4.3 discusses the case study of the California condor. Comparative statics are then discussed followed by conclusions in Section 4.4.

4.2 The Model

4.2.1 Single Population Model

According to Dennis et al. (1991), an endangered species of population size $N = N(t)$ at instant t , can be adequately approximated by a stochastic exponential growth model given that the population is small compared to its carrying

capacity, as is the case with endangered and threatened species. This single population model is discussed in details in Section 2.2 in Chapter 2.

An extinction threshold can be defined as $x_e = \ln(1) = 0$. This corresponds to the case of a single individual left in the population which is unable to breed. Then, the probability $p(t)$ that the population first hits the extinction threshold at time t is given by

$$p(t) = \frac{x_d}{\sqrt{\sigma^2 t^3}} \phi\left(\frac{x_d + \mu t}{\sigma \sqrt{t}}\right), \quad 0 < t < \infty, \quad (4.1)$$

where $\phi(\cdot)$ is the PDF of a standard normal distribution (Ingersoll, 1987). This probability function is the PDF of the extinction arrival time. When the infinitesimal growth rate is non-positive, i.e., $\mu \leq 0$, the process will hit the extinction threshold within a finite time with probability 1. When $\mu > 0$, the probability that the process will ever attain the threshold is $e^{-\frac{2\mu x_d}{\sigma^2}}$, i.e., the probability distribution has a mass point of $1 - e^{-\frac{2\mu x_d}{\sigma^2}}$ at positive infinity. The cumulative distribution probability (CDF) of the extinction arrival time is given as

$$P(t) = \Phi\left(-\frac{x_d + \mu t}{\sigma \sqrt{t}}\right) + e^{-\frac{2\mu x_d}{\sigma^2}} \Phi\left(\frac{-x_d + \mu t}{\sigma \sqrt{t}}\right), \quad 0 < t < \infty, \quad (4.2)$$

where $\Phi(\cdot)$ is the CDF of a standard normal distribution.

4.2.2 Two Sites Model

Suppose that a conservation agency has already chosen the first location for releasing captively bred individuals of an endangered species, but a second lo-

cation needs to be decided. Also suppose that ecologists and biologists have found a number of equally suitable locations, but they are at different distances from the first location. Once a population is reintroduced to the second location, each of the two populations can be described by the stochastic population model presented in Section 4.2.1, except that the stochastic terms are spatially correlated between the two populations, i.e., $\text{corr}(dz_1, dz_2) = \rho(D_{12})$. The correlation $\rho(D_{12})$ is a decreasing function of D_{12} , which is the distance between locations 1 and 2.

When the conservation agency is looking for the location for reintroducing a second wild population, how close should it be from the first location, given a number of equally suitable candidates?⁴ The trade off is that the closer the two populations are, the lower the setup and management cost will be, but the higher the probability of joint extinction will be — the risk that a regional shock will wipe out both populations. The two populations can be modelled as

$$dx_1(t) = \mu_1 dt + \sigma_1 dz_1 \quad (4.3)$$

$$dx_2(t) = \mu_2 dt + \sigma_2 dz_2 \quad (4.4)$$

$$dz_2 = \rho(D_{12})dz_1 + \sqrt{1 - \rho(D_{12})^2}dz' \quad (4.5)$$

In equation (4.5), dz_2 is a weighted sum of dz_1 and dz' , where dz' is also an increment of a Wiener-drift process. It is normally distributed, $dz' \sim \mathcal{N}(0, dt)$ and it is independent of dz_1 . In this model, the stochastic terms dz_1 and dz_2 are both normally distributed and their joint distribution is bivariate normal. Their correlation coefficient is $\rho(D_{12}) \in [0, 1]$, where $\rho(0) = 1$ and $\rho(\infty) = 0$. Differences

⁴For simplicity, it is assumed that there is at least one equally suitable habitat at each possible distance D_{12} .

in habitat suitability between location 1 and 2 can be characterized by different values of μ_1 and μ_2 . It can be proven that $(x_1(t), x_2(t))$ is subject to a bivariate normal distribution with the mean vector $(x_0 + \mu_1 t, x_0 + \mu_2 t)$, and the covariance matrix

$$\begin{bmatrix} \sigma_1^2 & \rho(D_{12})\sigma_1\sigma_2 \\ \rho(D_{12})\sigma_1\sigma_2 & \sigma_2^2 \end{bmatrix} \mathbf{t}.$$

It can also be proven that the correlation between the two populations' sizes $x_1(t) = \ln N_1(t)$ and $x_2(t) = \ln N_2(t)$ is also $\rho(D_{12})$.

The model also assumes that the cost of establishing two release sites is $C(D_{12}) \cdot (c_1 + c_2)$ where $C(D_{12}) > 0$ is an increasing function of D_{12} . If the two sites are managed separately and independently, c_1 and c_2 are the setup and management cost at locations 1 and 2 respectively. This assumption implies that the total management and setup cost increases with the distance between the two locations.

The discount rate r , the loss value/existence value of the species L , the management and setup cost at locations 1 and 2 c_1, c_2 , the cost-distance relation function $C(D_{12})$, and the correlation function $\rho(D_{12})$ are assumed given. The objective is to minimize the total management and setup cost plus the expected loss should extinction occur, i.e.,

$$\text{Minimize}_{D_{12}} \quad \pi = C(D_{12}) \cdot (c_1 + c_2) + L \int_0^{\infty} p(t)e^{-rt} dt. \quad (4.6)$$

Larger distance D_{12} gives a larger management and setup cost $C(D_{12}) \cdot (c_1 + c_2)$, but it also gives lower expected loss $L \int_0^{\infty} p(t)e^{-rt} dt$ should extinction occur.

Joint extinction occurs when both populations have hit the extinction threshold, although not necessarily at the same time. Let τ_1 and τ_2 denote the extinction arrival time for each of the two populations. Thus, the joint extinction arrival time is defined as $\tau = \max(\tau_1, \tau_2)$. With two correlated Brownian motions with drift, the PDF and CDF of the extinction arrival time, $p(t) = \text{Prob}(\tau = t)$ and $P(t) = \text{Prob}(\tau \leq t)$, don't have an analytical form. In this essay, $p(t)$ and $P(t)$ in Section 4.3 are obtained from numerical simulations based on the California condor population data available.

4.2.3 Three or More Sites Model

According to USFWS recovery plans, having three separate populations are often one of the requirements for delisting an endangered species. A three sites model will be used to study the example of the California condor in Section 4.3. In this section, a more generalized model of reintroducing captive bred individuals of an endangered species at N different locations is presented. Instead of choosing only one distance D_{12} in the two sites model in Section 4.2.2, now there are $\frac{N(N-1)}{2}$ pairwise distances among the populations to be decided. Assume that the cost-distance relation function $C(\cdot)$ is increasing of any of the pairwise distances $\{D_{jk}\}$, $\forall 1 \leq j < k \leq N$. The overall setup and management cost is proportional to the sum of individual setup and management costs across all locations $\sum_{1 \leq i \leq N} c_i$.⁵ The objective function is

⁵The individual setup and management cost is under the condition that each location is managed separately and independently

$$\underset{\{D_{jk}, \forall 1 \leq j < k \leq N\}}{\text{Minimize}} \quad \pi = C(\{D_{jk}, \forall 1 \leq j < k \leq N\}) \cdot \left(\sum_{1 \leq i \leq N} c_i \right) + L \int_0^{\infty} p(t) e^{-rt} dt. \quad (4.7)$$

If the location decisions are sequential, i.e., when $N-1$ locations have already been chosen and the conservation agency is deciding on the N^{th} location for an additional reintroduced population, then the minimization problem is

$$\underset{\{D_{jN}, \forall 1 \leq j \leq N-1\}}{\text{Minimize}} \quad \pi = C(\{D_{jk}, \forall 1 \leq j < k \leq N\}) \cdot \left(\sum_{1 \leq i \leq N} c_i \right) + L \int_0^{\infty} p(t) e^{-rt} dt. \quad (4.8)$$

The pairwise distances should be subject to two dimensional geometry constraints. In this more generalized model, the N populations at the N chosen locations can be modelled by

$$dx_1(t) = \mu_1 dt + \sigma_1 dz_1 \quad (4.9)$$

$$dx_2(t) = \mu_2 dt + \sigma_2 dz_2 \quad (4.10)$$

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$$dx_N(t) = \mu_N dt + \sigma_N dz_N \quad (4.11)$$

Also assume that dz_1, dz_2, \dots, dz_n are from a multivariate normal distribution with a mean vector of zeros and a covariance matrix below

$$\text{Covariance}(\mathbf{dz}) = \begin{bmatrix} 1 & \rho(D_{12}) & \rho(D_{13}) & \dots & \rho(D_{1N}) \\ \rho(D_{12}) & 1 & \rho(D_{23}) & \dots & \rho(D_{2N}) \\ \vdots & \vdots & \vdots & \ddots & \vdots \\ \rho(D_{1N}) & \rho(D_{2N}) & \rho(D_{3N}) & \dots & 1 \end{bmatrix} dt.$$

It can be proven that $(x_1(t), x_2(t), \dots, x_N(t))$ is subject to a multivariate normal distribution with a mean vector $(x_0 + \mu_1 t, x_0 + \mu_2 t, \dots, x_0 + \mu_N t)$, and a covariance matrix

$$\text{Covariance}(\mathbf{x}(t)) = \begin{bmatrix} \sigma_1^2 & \rho(D_{12})\sigma_1\sigma_2 & \rho(D_{13})\sigma_1\sigma_3 & \dots & \rho(D_{1N})\sigma_1\sigma_N \\ \rho(D_{12})\sigma_1\sigma_2 & \sigma_2^2 & \rho(D_{23})\sigma_2\sigma_3 & \dots & \rho(D_{2N})\sigma_2\sigma_N \\ \vdots & \vdots & \vdots & \ddots & \vdots \\ \rho(D_{1N})\sigma_1\sigma_N & \rho(D_{2N})\sigma_2\sigma_N & \rho(D_{3N})\sigma_3\sigma_N & \dots & \sigma_N^2 \end{bmatrix} t.$$

Joint extinction occurs when all populations have hit the extinction threshold. Let τ_i denote the first passage time when population i hits the extinction threshold, $1 \leq i \leq N$. Thus, the joint extinction arrival time is defined as $\tau = \max(\{\tau_i, 1 \leq i \leq N\})$. With N correlated Brownian motions with drift, the PDF and CDF of the extinction arrival time, $p(t) = \text{Prob}(\tau = t)$ and $P(t) = \text{Prob}(\tau \leq t)$, don't have an analytical form. In order to simulate the extinction arrival time probability, the continuous time model needs to be modified into a discrete time model, i.e.,

$$\underset{\{D_{jk}, \forall 1 \leq j \leq N-1\}}{\text{Minimize}} \quad \pi = C(\{D_{jk}\}, \forall 1 \leq j \leq k \leq N-1) \cdot \left(\sum_{1 \leq i \leq N} c_i \right) + L \sum_{t=0}^T p(t) \delta^t, \quad (4.12)$$

where $\delta = \frac{1}{1+r}$ is a discount factor and T is a sufficiently large terminal time.

4.3 A Numerical Example of the California Condor

In this section the three location model from Section 4.2.3 is applied to the population of the California condor. Although the data availability and accuracy are limited, this example is useful to illustrate the relevance of spatial correlation and the economic trade-off created by distances.

4.3.1 California Condor

Section 2.4 in Chapter 2 gives a background information of the California condor and its recovery program. This chapter focuses on its reintroduction program. Although there are five release sites for the California condor,⁶ they can be grouped into three separate wild populations according to geographic locations: California, Arizona and Baja California, Mexico. The distance between Bitter Creek, California and Vermilion Cliffs, Arizona is approximately 722 km.⁷ The distance between Bitter Creek National Wildlife Refuge, California and Sierra De San Pedro Mirtir, Mexico is approximately 608 km. The distance between Sierra de San Pedro Mirtir, Mexico and Vermilion Cliffs, Arizona is approximately 630 km. These numbers are obtained from an online distance calculator developed by Infoplease (n.d.). The release locations are shown in figure 4.1 below. For simulation simplification, the distances are rounded to 700 km, 600 km and 600 km.

⁶Three in California, one in Arizona and one in Baja, Mexico

⁷According to Infoplease (n.d.), “names and locations in this distance calculator are from the Geographic Names Information System (GNIS). Distance calculations are based on the WGS84 ellipsoid using geod. The computation is for the greatest circle distance between points, and does not account for differences in elevation”.

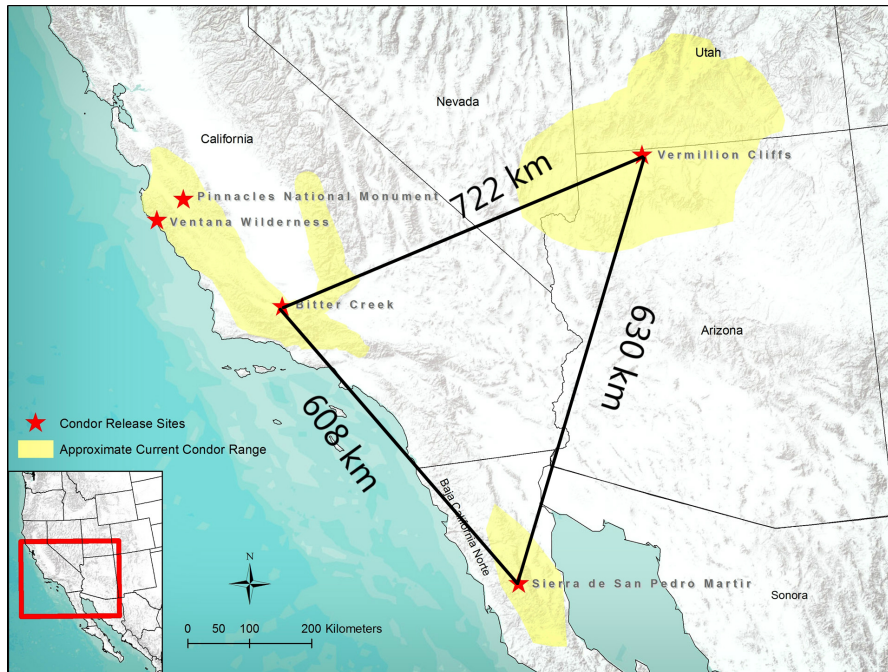


Figure 4.1: Release sites of California Condors (Adapted from USFWS, 2017d)

In Southern California, release sites at Hopper Mountain and Bitter Creek National Wildlife Refuges are managed by the USFWS. In Central California, release sites at Big Sur are operated by the Ventana Wildlife Society and release sites at Pinnacles National Monument are operated by the National Park Service (NPS). In Northern Arizona and Southern Utah, The Peregrine Fund (TPF) operates both the captive breeding facility in Boise and the release site at Vermilion Cliffs Monument, in partnership with federal agencies and state agencies. San Diego Wild Animal Park operates both a captive breeding facility at the zoo and the Baja California release site in collaboration with the Instituto Nacional De Ecologia in Mexico (Walters et al., 2010).

4.3.2 Parameters

The California Condor Population and Distribution Monthly Reports (USFWS Pacific Southwest Region, n.d.) contain annual population data from December 1995 to December 2015 in California, from December 1997 to December 2015 in Arizona, and from December 2002 to December 2015 in Baja Mexico. In the first year of the release programs 6, 13 and 3 birds were released at each location. Using the maximum likelihood method, estimated infinitesimal growth rates and infinitesimal standard deviation are $\mu_1 = 0.163$ and $\sigma_1 = 0.256$ for the California wild population, $\mu_2 = 0.101$ and $\sigma_2 = 0.118$ for the Arizona wild population, $\mu_3 = 0.184$ and $\sigma_3 = 0.278$ for the Baja, Mexico wild population. One thing to note is that each year there are new California condors being released into the three populations. Thus the infinitesimal growth rates and infinitesimal variances estimated from the population data are not the actual infinitesimal growth rates and infinitesimal variances. Ideally, only the initially released California condors and their wild born and fledged descendants should be used to estimate μ_i and σ_i . However, this information is not available among the data reported. It might be reasonable to assume that the estimated μ_i and σ_i are what the USFWS wishes to achieve in the long run. With this set of infinitesimal growth rates and variances, simulations show that the trade-off created by the distance is not relevant, because the risk of joint extinction is almost zero. Thus, in Section 4.3.3-4.3.5, estimated from 1965-1980 before the reintroduction program, $\mu = -0.0768$ and $\sigma^2 = 0.1199$ are used as the baseline values to show the trade-off created by the distance.

In this numerical example, the correlation function is specified as $\rho(D_{ij}) = \frac{1}{1+\gamma D_{ij}}$, $\forall 1 \leq i < j \leq N$, where γ is a correlation parameter. A larger γ gives a

weaker correlation. From the population data available, three different values of γ can be estimated. However, as discussed above, the observed spatial correlation is highly influenced by the ongoing release actions to the three populations. Thus the estimated γ is not good candidates for spatial correlation calculation. Instead, the baseline γ is chosen to be 0.01, so that it would give weak spatial correlations of 0.12 ~ 0.14 among the three California condor populations.⁸

While the actual expenditures are not broken down by captive breeding and reintroduction programs, the estimated cost of recovery stated in 1996 Recovery Plan for the California Condor by the USFWS shows that the total cost of the reintroduction program is \$9,321,000 for all three locations for 25 years after discounting (Kiff & Mesta, 1996). Therefore, the overall setup and management cost is $C(D_{12} = 700, D_{13} = 600, D_{23} = 600) \cdot (c_1 + c_2 + c_3) = C_1 + C_2 + C_3 = \$9,321,000$, where $C_1, C_2,$ and C_3 denote the setup and management cost for each of the three locations at current distances. Due to lack of more detailed data, the setup and management cost at each location is assumed to be $C_1 = C_2 = C_3 = \frac{\$9,321,000}{3} = \$3,107,000$. The total setup and management cost is normalized around this cost number and the given distances, so $C(D_{12} = 700, D_{13} = 600, D_{23} = 600) \cdot \frac{c_1+c_2+c_3}{C_1+C_2+C_3} = 1$. One possible candidate for the normalized cost-distance relationship function is

$$C(D_{12}, D_{13}, D_{23}) \cdot \frac{c_1 + c_2 + c_3}{C_1 + C_2 + C_3} = \alpha + (1 - \alpha) \cdot \frac{D_{12} + D_{13} + D_{23}}{700 + 600 + 600}, \quad \frac{1}{3} < \alpha < 1. \quad (4.13)$$

This function specifies that increasing any pairwise distance would increase the total management and setup cost. It also specifies that if California condors were released at only one location, the conservation agency would need to pay

⁸An accurate estimated value for spatial correlations is not available in the literature so far. It can be an interesting project for future studies.

more than a third of the current overall management and setup cost,⁹ due to the economics of scale. In addition, for simplicity, the effect of distances on total setup and management cost is assumed to be linear.

The loss cost or the existence value of California condors is difficult to estimate. The only study estimating the individual willingness to pay (WTP) for the continuing existence of the California condor (Redden, 2008) gives the WTP as \$99.2 per person. Aggregating this amount by the population in Southern California gives \$2 billion. In addition, the Time horizon for simulations is chosen at 100 periods because the probability of extinction decreases to zero after 50 periods. A million population paths were drawn at each location to calculate the extinction arrival time probability. Parameter values for the California condor are listed in the Table 4.1.

4.3.3 Distance

This section discusses the trade-off created by distance in the example of the California condor. The baseline parameter values are specified in Table 4.1. Assume that the distances between populations in Arizona and Baja California, and between populations in California and Baja California are fixed at 600 km. Figure 4.2 below shows that a greater distance (from 0 km to 100 km to 900 km) between the populations in California and Arizona would delay extinction arrival time. Although the CDFs for the three different distances are close in value, due to the large value of loss cost/existence value L , the expected loss should extinction occur $L \int_0^{\infty} p(t)e^{-rt} dt$ can vary greatly.

⁹Due to lack of detailed cost information, it is arbitrarily chosen to be a half in the simulations.

Table 4.1: Parameter values for California Condor

Parameters	Parameter Values	Descriptions
μ_i	-0.0768	The infinitesimal growth rate for each population
σ_i^2	0.1199	The infinitesimal variance for each population
$x_1(0)$	$\ln(6)$	Natural log of the initial released population size in California
$x_2(0)$	$\ln(13)$	Natural log of the initial released population size in Arizona
$x_3(0)$	$\ln(3)$	Natural log of the initial released population size in Baja Mexico
D_{12}	700km	Distance between California and Arizona/Utah populations
D_{13}	600km	Distance between California and Baja Mexico populations
D_{23}	600km	Distance between Arizona/Utah and Baja Mexico populations
γ	0.01	A parameter affecting spatial correlation
δ	0.9	Discount rate
t	100	Time horizon
L	\$2,000,000,000	Loss cost estimated from willingness to pay
C_1, C_2, C_3	\$3,107,000	Setup and management cost of each reintroduced population
α	$\frac{1}{2}$	A parameter in the cost-distance relationship function

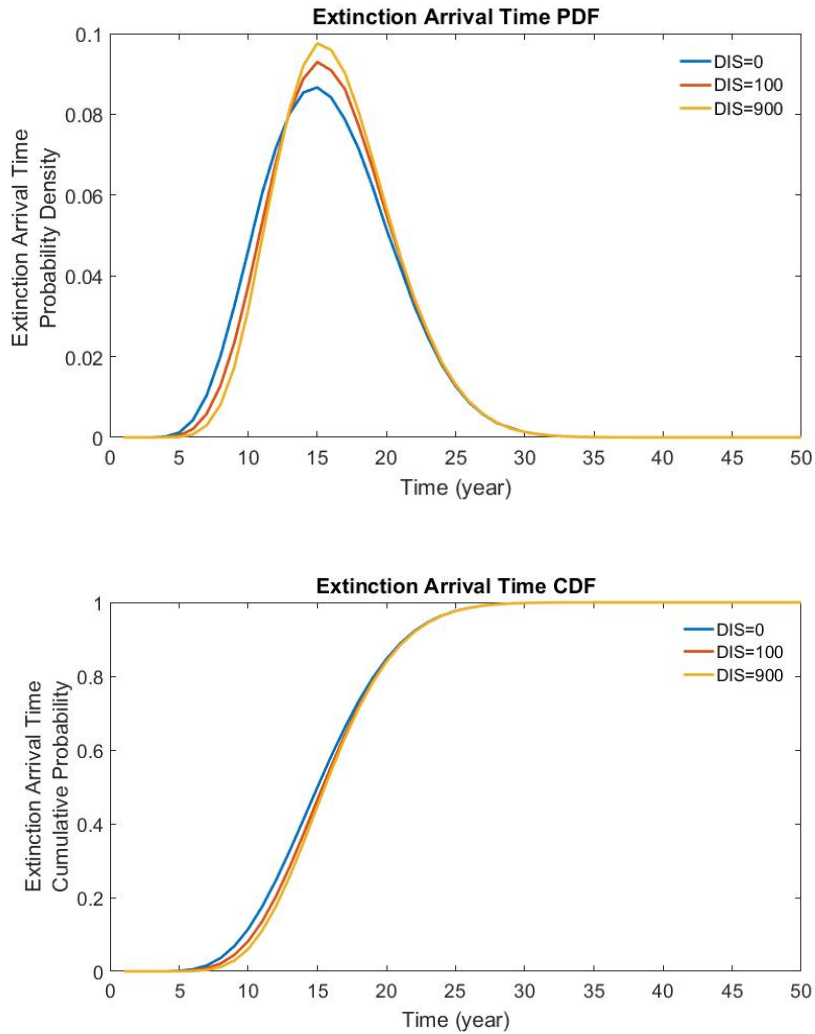


Figure 4.2: PDF and CDF with Different Distances between California and Arizona Populations

The expected total cost (setup and management cost plus expected loss cost) is shown in figure 4.3 below. When the distance between populations in California and Arizona is increasing within the range of 0 ~ 800 km, reduction in extinction probability dominates the increase of the overall management and setup cost. When the distance between populations in California and Arizona increases beyond 800 km, the increase of the overall management and setup cost dominates

the reduction in extinction probability. The optimal distance between California and Arizona of 800 km gives the minimum expected total cost.

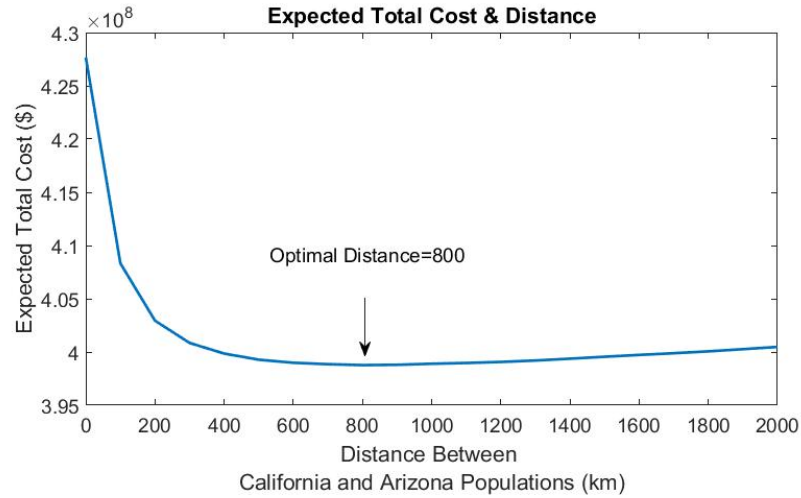


Figure 4.3: Expected Total Cost and Distance between California and Arizona Populations

4.3.4 Comparative Statics

Correlation Parameter With the assumed form of the correlation coefficient being $\rho(D_{jk}) = \frac{1}{1+\gamma D_{jk}}$, larger γ gives weaker correlations between any two locations. In the example of the California condor, figure 4.4 shows that as γ increases, the optimal distance between populations in California and Arizona decreases, holding the other two distances constant at 600 km.

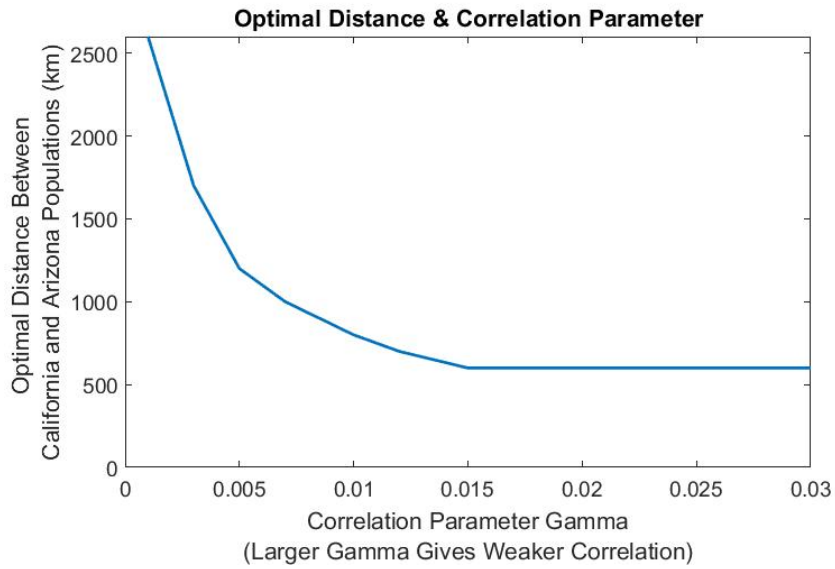


Figure 4.4: Optimal Distance and Correlation Parameter γ

Setup and Management Cost In the example of the California condor, figure 4.5 below shows that the optimal distance between populations in California and Arizona decreases when the setup and management cost at the three locations increases simultaneously.

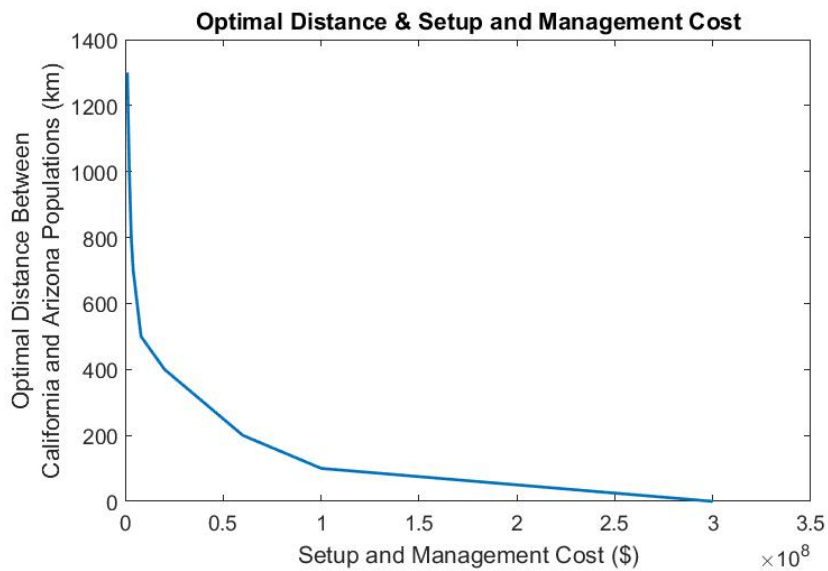


Figure 4.5: Optimal Distance and Setup and Management Cost

4.3.5 One Location, Two Locations or Three Locations

In this section, the outcomes of releasing California condors in only one location with 18 birds, two locations with 9 birds at each location, and three locations with 6 birds at each location are compared. The total amount of initially released birds, infinitesimal growth rate and variance are assumed to be the same across all locations and across these three scenarios. Other parameters are set at the baseline values in Table 4.1. Figure 4.6 shows that having more populations doesn't necessarily delay the extinction arrival time, and the expected total cost would increase from $\$4.10 \times 10^8$ to $\$4.22 \times 10^8$ and $\$4.96 \times 10^8$ when increasing the number of locations from one to two and three. This interesting outcome is against the intuition of avoiding "putting all your eggs in one basket." A reasonable explanation lies in the definition of the extinction threshold, which is $N = 1$, i.e., where there is only one bird left. Joint extinction of two populations occurs when each population only has one bird left. However, if these two birds are put together as one population, then this single population hasn't hit the extinction threshold yet, because the two birds might be able to reproduce.

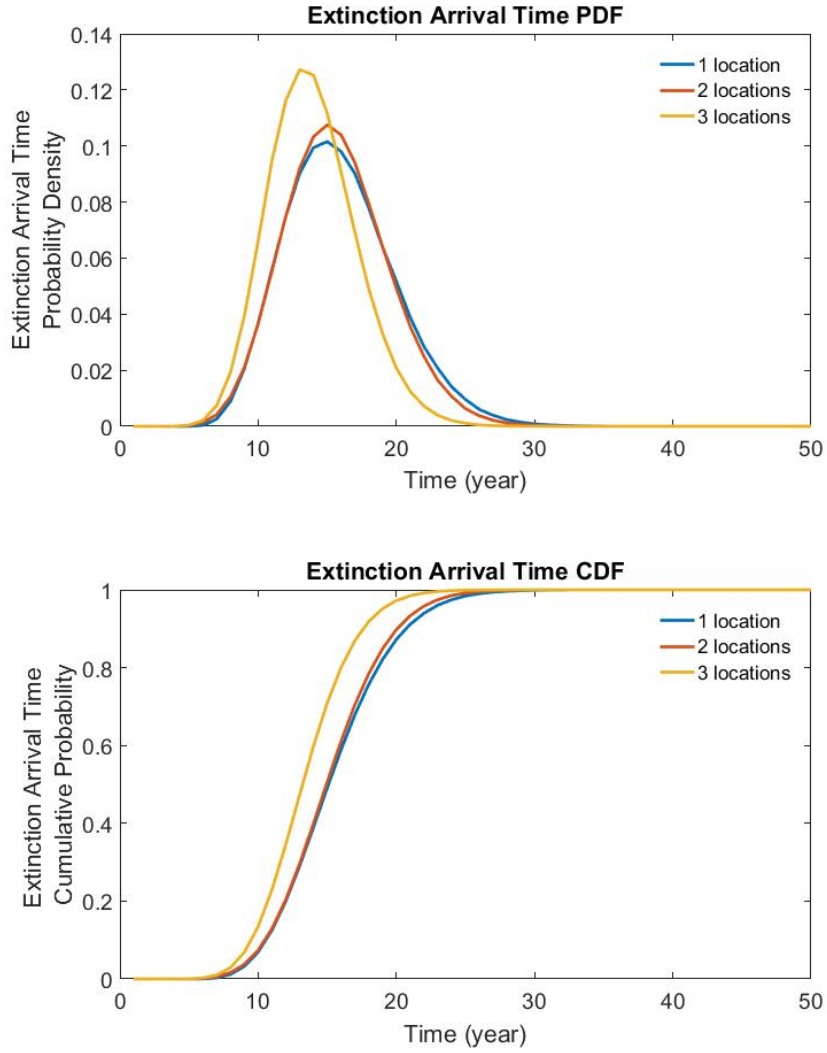


Figure 4.6: Extinction Arrival Time PDF and CDF of one, two, and three locations

As listed in Table 4.2, with the baseline value $\mu = -0.0768$, the expected loss should extinction occur is lower under the two locations scenario than the one location scenario (lower under the three locations scenario than the two locations scenario), if σ^2 is greater than 0.127 (1.94). In addition, the expected total cost is lower under the two locations scenario than the one location scenario (lower under the three locations scenario than the two locations scenario) if σ^2 is greater than 0.131 (2.02). Similarly, with the baseline value $\sigma^2 = 0.1199$, the

expected loss should extinction occur is lower under the two locations scenario than the one location scenario (lower under the three locations scenario than the two locations scenario), if μ is greater than -0.066 (2.00). In addition, the expected total cost is lower under the two locations scenario than the one location scenario, if μ is greater than -0.059. However, the expected total cost under the three locations scenario will always exceed the two locations scenario.

Table 4.2: One, Two or Three Locations

μ and σ^2	$\mu = -0.0768$	$\sigma^2 = 0.1199$
Expected loss should extinction occur 2 locations < 1 location	$\sigma^2 > 0.127$	$\mu > -0.066$
Expected loss should extinction occur 3 locations < 2 locations	$\sigma^2 > 1.94$	$\mu > 2.0$
Expected total cost 2 locations < 1 location	$\sigma^2 > 0.131$	$\mu > -0.059$
Expected total cost 3 locations < 2 locations	$\sigma^2 > 2.02$	None

4.4 Conclusion and Discussion

This study investigates the trade-off between management cost and the expected loss should extinction occur, created by the distances between multiple reintroduced populations of an endangered species. However, with two or three correlated Brownian motions with drift, there is no analytical form for the joint extinction arrival time probability distribution. Thus, this study relies on numerical simulations to calculate the extinction arrival time probability, using the example of the California condor. These results show that the expected loss should extinction occur decreases with distance, and the setup and manage-

ment cost increases with distance. Comparative statics show that the optimal distance would decrease when spatial correlation weakens, and when the setup and management cost increases.

This study also discusses the outcomes under one, two and three reintroduced population(s) scenarios. Whether having more release sites will reduce the expected loss should extinction occur depends on parameter values.

Due to lack of detailed cost information, the cost-distance relationship function specified in the California condor example may not realistically reflect the relationship between management costs and distance. For the purpose of faster computation, the distance between Arizona and Baja California, as well as the distance between California and Baja California are held constant when the trade-off and comparative statics were analyzed. To gain a better quantitative understanding of the trade-off and comparative statics, one of the other two distances, or even both, should be endogenously chosen. The simulation algorithm for the probability of an extinction arrival time is of low efficiency and can only be applied to discrete time models. A more efficient simulation algorithm to estimate extinction arrival time when there are three correlated Brownian motions with drift would allow deeper analysis.

CHAPTER 5

SUMMARY

This dissertation contains three essays on the topic of wildlife conservation programs. The first essay proposed a method to calculate the value of a conservation program through reducing the extinction risk. The second essay uses a static approach and a dynamic approach to calculate the optimal timing to start a conservation program. The third essay considers a spatial-economic factor and studies the economic trade-off of the distance between two conservation sites.

The value of a conservation program for an endangered species can be measured as the expected present value of longer survival. The longer survival can be predicted using historical population data. In the example of the California condor, the economic value of the captive breeding program is estimated to be \$20.167 million, if the existence value of the California condor is \$35 million; \$146.934 million, if the existence value of the California condor is \$255 million; \$1152.426 million, if the existence value of the California condor is \$2 billion; and \$5.371 million, if the existence value of the California condor is \$9.321 million. The variation of the values is due to differences in several data-sets of historical California condor population sizes.

The optimal waiting time before starting a conservation program can be calculated using either a static approach or a state contingent approach. Given the fixed cost, the flow cost, the efficacy value of the conservation program, the existence value of a species, and the discount factor, one can derive the first and second order conditions for the optimal timing. The hazard rates can be predicted using historical population size data of a species. In the example of the

California condor, if the decision was made in 1980, then the optimal waiting time was either 0.1 or 4.0 years, depending on the hazard rate function. In the example of the whooping crane, if the decision was made in 1966, then it was optimal to never commence the captive breeding program. However, if the decision was made in 1900, then in order to justify the commencement in 1966, the infinitesimal variance of the population needs to be between 0.0351 and 0.0579 depending on the efficacy parameter value and the whooping crane existence value. In addition, given the infinitesimal variance estimated from the AWB population data, it was not too late to start the program in 1966. In addition, in order to be optimal to wait, given the flow cost data, existence value, efficacy parameter values, and the discount factor, the fixed cost has to be high enough. The state contingent approach found that there is a pair of thresholds of the population size that trigger the commencement and the termination of a conservation program. For the California condor, the threshold of triggering a conservation program is between 3 and 5. The threshold of triggering a termination is between 5 and 30, depending on the fixed cost and population data. For the whooping crane, the threshold of triggering a conservation program is between 2 and 25. The threshold of triggering a termination is between 2 and 55, depending on the fixed cost, population data and the loss cost.

The distance between multiple conservation sites for the same species creates an economic trade-off between the management cost and the risk of joint extinction. The multiple populations can be modelled as multiple spatially correlated Geometric Brownian Motions. The probability of joint extinction can be predicted by simulations. In the example of the California condor, without habitat suitability considerations, the optimal distance between two conservation sites is calculated to be 800km. When the management cost is higher, or the

spatial correlation is weaker, a conservation agency should consider choosing two conservation sites that are closer to each other. In addition, birds are not eggs. Putting all birds in one basket is not necessarily a bad thing.

APPENDIX A
SPECIES STATUS CODES (USFWS, 2012)

Table A.1: Listing Codes and Status

Codes	Status	Descriptions
E	endangered	in danger of extinction throughout all or a significant portion of its range
T	threatened	likely to become endangered within the foreseeable future throughout all or a significant portion of its range
C	candidate	under consideration for official listing; there is sufficient information to support listing
SAE, E(S/A)	endangered due to similarity of appearance	endangered due to similarity of appearance with another listed species and is listed for its protection. Species listed as E(S/A) are not biologically endangered or threatened
SAT, T(S/A)	threatened due to similarity of appearance	threatened due to similarity of appearance with another listed species and is listed for its protection. Species listed as T(S/A) are not biologically endangered or threatened
EXPE, XE	experimental essential population	listed as experimental and essential

Codes	Status		Descriptions
EXPN, XN	experimental non-essential population		listed as experimental and non-essential. Experimental, nonessential populations of endangered species (e.g., red wolf) are treated as threatened species on public land, for consultation purposes, and as species proposed for listing on private land
PE	proposed endangered	en-	proposed for official listing as endangered
PT	proposed threatened		proposed for official listing as threatened
PEXPE, PXE	proposed experimental population, essential	ex-	proposed for official listing as experimental and essential
PEXPN, PXN	proposed experimental population, non-essential	ex-	proposed for official listing as experimental and non-essential
PSAE, PE (S/A)	proposed dangered, to similarity of appearance	en- due species	proposed for official listing as endangered due to similarity of appearance with another listed species

Codes	Status	Descriptions
PSAT, PT (S/A)	proposed threatened, due to similarity of appearance	proposed for official listing as threatened due to similarity of appearance with another listed species
EE	emergency endangered	a temporary (240 days) listing for emergency purposes when species is at significant, immediate risk
D	delisted	has been removed from the list due to recovery, original data in error, or extinction
SC	species of concern	have not been petitioned or been given E, T, or C status but have been identified as important to monitor
RT	resolved Taxon	have been petitioned for listing and for which a Not Warranted 12 month finding or Not Substantial 90-day finding has been published in the Federal Register. Also includes species that have been removed from the candidate list
UR	under review	species that have been petitioned for listing and for which a 90 day finding has not been published or for which a 90 day substantial has been published but a 12 Month finding have not yet been published in the Federal Register. Also includes species that are being reviewed through the candidate process, but the CNOR has not yet been signed

APPENDIX B

**CALIFORNIA CONDOR TOTAL ESTIMATED COST OF RECOVERY, AS
IN 1996 RECOVERY PLAN FOR THE CALIFORNIA CONDOR**

Costs (\$1,000's)

<u>Year</u>	<u>Need 1</u>	<u>Need 2</u>	<u>Need 3</u>	<u>Need 4</u>	<u>Need 5</u>
1995	658.0	582.0	83.0	128.0	37.0
1996	643.0	590.0	83.0	193.0	37.0
1997	643.0	895.0	83.0	218.0	237.0
1998	643.0	890.0	83.0	218.0	237.0
1999	643.0	870.0	83.0	148.0	237.0
2000	650.0	850.0	85.0	150.0	50.0
2001	650.0	850.0	85.0	150.0	50.0
2002	650.0	850.0	85.0	150.0	50.0
2003	650.0	850.0	85.0	150.0	50.0
2004	650.0	850.0	85.0	150.0	50.0
2005	650.0	850.0	85.0	150.0	50.0
2006	650.0	850.0	85.0	150.0	50.0
2007	650.0	850.0	85.0	150.0	50.0
2008	650.0	850.0	85.0	150.0	50.0
2009	650.0	850.0	85.0	150.0	50.0
2010	650.0	850.0	85.0	150.0	50.0
<u>Total Costs</u>	10,380.0	13,177.0	1,350.0	2,555.0	1,335.0

APPENDIX C

**ANNUAL FINANCIAL CONTRIBUTIONS TO THE CALIFORNIA
CONDOR RECOVERY PROGRAM BY MAJOR PARTNERS IN 2007,
(WALTER ET AL,2010)**

Partner	Annual expenditure	Rearing facility	Release site
U.S. Fish and Wildlife Service	\$857,000	No	Bitter Creek
Los Angeles Zoo	\$573,000	Yes	Nope
San Diego Wild Animal Park	\$1,479,000	Yes	Baja
The Peregrine Fund	\$1,520,000 ¹	Yes	Arizona
Ventana Wildlife Society	\$244,000	No	Big Sur
Pinnacles National Monument	\$500,000	No	Pinnacles
Oregon Zoo	\$172,000	Yes	None

1. Includes \$394,000 from USFWS.

APPENDIX D

**EXPENDITURES BY PARTNERS IN THE SOUTHWEST REGION (USFWS
PACIFIC SOUTHWEST REGION, 2013)**

Partners	1996~ 2001	2002~ 2006	2007~ 2011
The Peregrine Fund	\$4,486,242 ¹	\$6,163,827 ²	\$6,186,255 ³
Arizona Game and Fish Department	\$267,206 ⁴	\$584,100 ⁵	\$845,800 ⁶
Utah Division of Wildlife Resources	N/A	\$36,250 ⁷	\$65,000 ⁸
Grand Canyon National Park	\$78,000 ⁹	N/A	\$310,425 ¹⁰
Zion National Park	N/A	N/A	\$19,000 ¹¹
Bureau of Land Management-Arizona Strip District	\$164,650 ¹²	\$99,405 ¹³	\$150,000 ¹⁴
Kaibab National Forest	N/A	\$8,500 ¹⁵	\$25,500 ¹⁶
Glen Canyon National Recreation Area	N/A	\$1,600 ¹⁷	N/A
Arizona Ecological Services	N/A	\$55,000 ¹⁸	N/A
Jacob Lake Inn	\$2,700 ¹⁹	N/A	N/A
Arizona Public Service utility company	\$32,939 ²⁰	N/A	N/A

1. Approximately \$2,817,000 was from the USFWS.
2. Approximately \$1,984,939 was from the USFWS and \$140,000 was from AGFD.
3. Approximately \$2,395,954 was from the USFWS,\$150,000 from the BLM, \$100,000 from the National Fish and Wildlife Foundation in 2007, \$15,000 from

AGFD, \$13,800 from UDWR in 2009, and \$10,000 from the Kaibab NF in 2010.

In addition, its annual expenditures between 2007 and 2011 are available:

2007 \$1,337,139

2008 \$1,406,411

2009 \$1,293,861

2010 \$1,044,664

2011 \$1,104,18

4. The usage breakdown is available:

Condor coordinator supported by Section 6 (75%) and AZ match (25%)
\$189,506

Condor coordinator supported by Heritage Fund \$12,000

Nongame specialist supported by Heritage Fund (total of 0.93 FTE) \$40,700

Nongame birds program manager supported by Pittman-Robertson funds
\$10,000

Chief of nongame and endangered wildlife supported by Arizona Nongame
Wildlife Checkoff Fund \$5,000

Other Department personnel (e.g., law enforcement and public outreach) sup-
ported by State Game and Fish funds and Heritage Fund \$10,000

5. Its annual expenditures are available during this review period:

2002 Condor biologist operating costs \$51,800

2003 Condor biologist operating costs \$62,200

2004 Condor biologist operating costs \$70,300

2004 6 satellite transmitters and data download \$25,000

2005 Condor biologist operating costs \$86,700
2005 15 satellite transmitters and data download \$54,500
2005 X-ray machine and developer; veterinary lab equipment, trailer to haul
calf carcasses, two chest freezers to hold carcasses, three telemetry receivers, 11
Personal Data Assistants and field data entry system \$40,500
2006 Condor biologist operating costs \$68,200
2006 Satellite transmitter data download \$8,500
2006 10 spotting scopes and tripods, field lead test equipment, video equipment,
lab equipment, and telemetry receiver \$11,500
2006 Free non-lead ammunition program \$104,900

6. Its annual expenditures are available during this review period:

2007 \$168,400 (including \$85,100 condor program operating costs and \$83,300
free non-lead ammunition program cost)
2008 \$209,200 (including \$96,000 condor program operating costs and \$113,200
free non-lead ammunition program cost)
2009 \$226,200 (including \$96,100 condor program operating costs and \$130,100
free non-lead ammunition program cost)
2010 \$163,700 (including \$78,500 condor program operating costs and \$85,200
free non-lead ammunition program cost)
2011 \$156,900 (including \$78,300 condor program operating costs and \$78,600
free non-lead ammunition program cost)

7. Annually, it has committed approximately 0.1 FTE (\$6,500) and other expenditures of approximately \$750.00.

8. Its breakdown expenditures during this review period are available:

Support of biological and outreach staff \$39,000

A grant to TPF for purchase of GPS transmitters \$13,800

Reimbursing hunters for purchase of non-lead ammunition \$4,225

Travel, publication of educational information, and other miscellaneous expenses \$8,200

9. Its breakdown expenditures during this review period are available:

Condor technician supported by Grand Canyon National Park Foundation and Grand Canyon Association funds \$39,000

Trailer rental space for The Peregrine Fund supported by Grand Canyon National Park Fee Demo (20%) funds \$3,000

Travel for a certified radiation officer to assist in affixing deterrents to the Orphan Mine tower structure \$3,000

Wildlife biologist and wildlife program manager \$28,000

10. No breakdown information available, but the expenditures included one to three condor biologists and technicians, operating costs, and volunteer-based condor monitoring program costs.

11. Its annual expenditures on personnel are available:

2009 \$4,000

2010 \$5,000

2011 \$10,000

12. Its breakdown expenditures are available during this review period:

Wildlife biologist (5 years @ 20% FTE per year) \$50,000

Transport of birds from captive rearing facilities (\$5,000/yr) \$25,000

BLM aircraft from the National Interagency Fire Center for transport NFR Travel attending meetings and workshops \$10,000

Ceremony for first release \$10,000

Installation of informational kiosks \$8,500

Condor brochures \$2,500

Radios (three) for The Peregrine Fund \$2,800

Installation of Bird Balls in water tanks \$10,850

Installation of two Remote Automated Weather Stations (RAWS) atop Vermilion Cliffs \$30,000

Annual maintenance of two RAWS weather stations \$15,000

13. Its breakdown expenditures are available during this review period:

Annual transportation of condors from the breeding facility in Boise to the release site \$6,000

The ASDO condor lead biologists time in 2006 \$5,881

Construction of a new viewing area below the release site that will include parking, a new shelter, restroom, and fence around the site \$40,000

14. Its breakdown expenditures are available during this review period:

Annual transportation of condors from the breeding facility in Boise to the release site \$6,000

Vehicles and personnel to get the condors from the viewing area to the release pens \$53,411

Staff time for condor work 2007 \$11,520

Staff time for condor work 2008 \$11,221

Staff time for condor work 2010 \$26,296

Staff time for condor work 2011 \$17,552

15 Expenditures on meetings, consultations, and outreach with the public and USFS personnel.

16. Expenditures on meetings, consultations, and outreach with the public and USFS personnel.

17. It spent 40 hours at \$40 per hour (\$1,600) on labor for section 7 consultations over the last five years.

18. Provided approximately a 0.15 FTE each year from 2002 through 2006 at an annual cost of approximately \$11,000. That total represents condor-related activity including participation in the SCWG, recovery actions, section 7 consultations, and outreach.

19. Its breakdown expenditures are available during this review period:

Horse killed when a news helicopter panicked it during early publicity \$2,200

Travel to testify in favor of reintroduction \$500

20. Installation of raptor protection devices on utility lines and poles

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