Finding Correlations among Successful Reintroduction Programs: An Analysis and Review of Current and Past Mistakes

by

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Abstract

In the past half century the world has seen a dramatic decline in species. More and more species are being pushed to brink of extinction. In the past, there have been several methods utilized to mitigate these trends, however with the recent surge of local extinctions, reintroductions have become a growing conservation tool. Despite many disadvantages of developing a reintroduction plan, hundreds have been attempted over the past 40 years, with mixed outcomes. Some conservationists have studied the factors associated with success; however the criteria on which their assessments were based were flawed. I attempted to complete my own assessment of successful programs using detailed program information along with life history traits of focal species. My results illustrate the many obstacles faced by reintroduction biologists. Based on the limitations faced throughout this study, I conclude that conservationists must take a step back and address the many issues with current reintroduction protocols prior to attempting any further assessments. My recommended solutions to some of these issues include defining universal criteria for a reintroduction program to be considered successful; monitoring, logging, and disseminating standardized data; and collaborating with captive facilities that have the ability to offer additional support.

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Introduction

In the past two decades, the number of species listed as vulnerable, endangered, or critically endangered has nearly doubled; there are now estimated to be over 20,000 species at least vulnerable to extinction (IUCN, 2013). As this number rapidly increases, conservationists are faced with the task of determining how to delay the trend. Two of the possible methods for achieving this goal are protection and restoration. Historically, conservationists have created policies, set aside protected areas, and raised awareness to safeguard species and their natural habitats (Erwin, 1991; Fiedler & Jain, 1992). Because the decline of so many populations is typically not due to one pressure but many, the protection of species and their habitat has been the most commonly used conservation tool. Habitat protection is typically the most cost efficient means to save species (MacKinnon & MacKinnon, 1991), although setting aside protected lands does not account for the dispersal of animals outside of these areas and may not always prevent threats from harming endangered populations. Additionally, increasing the amount of protected areas conflicts with human habitation and land use, which could promote human-wildlife conflicts that are currently threatening these species. More recently, as environmental protection has become standard practice, governmental and non-governmental organizations have turned to restoration tactics where protection was not an option. Restoration is a less attainable goal because the more degraded an area is, the more effort and funding is required to completely restore it. Moreover, the outcomes of such projects have high uncertainty as to the area of a project that can successfully be restored as well as the productivity level of a restored area (Suding, 2011). The effectiveness of restoration and protection varies widely with managing practices however they have become the two most prevalent methods of saving habitats and the species within them. While these practices are crucial to conservation, they are no longer an option in some dire situations.

When a species is on the brink of becoming locally extirpated or has already become locally extinct, there is truly only one method remaining to save that species in that area. Once a population has fallen to a certain point, the only way to improve population growth, or to return that species back to its original habitat, is by reintroducing individuals. Reintroduction is considered the "intentional movement of an organism into a part of its native range from which it has disappeared or become extirpated in historic times" (IUCN, 1987). While this conservation tool is critical for species on the brink of extinction, it is a last-resort method that should only be employed under very certain circumstances. Exploiting reintroduction could potentially reduce its overall success as a conservation strategy; the more effort put forth to save species that are not good candidates reduces the likelihood that well-fitting candidates will receive the support they need.

An abundance of species have locally gone extinct prior to reintroduction (Stuart, 1991). In fact, at least 17 of the 27 programs used in this study did not have a local population at the time of release. Without captive breeding programs and recovery plans, these animals (such as the California condor, black-footed ferret, and Przewalski's horse, to name a few) would only exist in captivity and for an unknown amount of time. In addition to saving species from the brink of extinction, reintroductions provide many other benefits. Many of the species involved in these programs become archetypes for further conservation issues and efforts. They increase awareness about the fate of populations due to human activity and they promote the protection for key ecosystems (e.g., golden lion tamarin in Brazil and the gray wolf in the western United States). Along with their extinction, ecosystems in which these species are found could have adverse effects due to their absence. Many endangered species are apex predators or provide other critical services that allow their ecosystem to thrive (Ripple & Beschta, 2012; Ripple et al., 2014). Despite these benefits, many are opposed to reintroduction programs due to the stress incurred by the animals caused by handling and novel environments, as well as additional stressors in the wild that are not present in captivity (Beck et al., 1994). This has caused a heated debate among scientists and practitioners about the ethics of reintroducing captive-born animals into novel environments (Harrington et al., 2013). Additionally, the type of species being reintroduced has caused backlash from non-supportive locals. This presents one of many issues with using reintroduction as a standard practice, including a lack of commonality among protocols, definitions, and criteria for success.

Reintroducing a species into its original habitat is not a novel idea; however the concept has only become increasingly popular as a conservation strategy since the 1970s (Stuart, 1991). In the 40+ years of reintroductions as a conservation focus, there have been hundreds of attempts to reintroduce species to their natural habitat. Most have been small-scale "experimental" trials. Others have been complete recovery plans extending to habitat quality and post-release monitoring. The number of attempts and those that survived past experimentation is not as large as one would expect due to limitations of preconditions that must be met for a species to become a reintroduction candidate (Stuart, 1991; Seddon, et al., 2012). *First*, there must be suitable habitat available as a release site; it should include abundant food source, sufficient area for dispersing individuals, and limited or no competition for niche placement. *Second*, threats causing the initial population decline must be removed prior to release (Stuart, 1991). These threats are typically directly or indirectly caused by humans, such as habitat loss and poaching, but can also include predation, competition with other species, and disease. The sheer number of threats any one species faces at a given time demonstrates how difficult becoming a candidate can be. The need to meet these preconditions reduces the number of situations in which reintroduction becomes a viable option for conservation. Because humans are rapidly changing habitats, and threats are often difficult to eliminate or are progressively becoming worse, reintroductions are often an unsuitable strategy and have become a final recourse for conservation techniques. Due to the large array of taxa needing to be reintroduced and the variety of supporting organizations establishing the reintroductions, there have also been mixed results in terms of the success of such efforts.

The number of “successful” reintroductions to date is debatable. Indicating the level of success of a reintroduction program has shown itself to be a complicated and sometimes controversial concept. At present, there is no adequate universal definition of success (Gusset, 2009). Many refer to “successful” programs or those that have been “unsuccessful”, but seldom do reviews specifically define what those terms mean to their study. A scarce number of these programs have been scientifically assessed for tangible evidence of success and the majority that have not have resorted to making their own claims without offering credible criteria (Ounsted, 1991).

Very few projects have faced the task of determining reasons for success (Germano & Bishop, 2009; Griffith et al., 1989; Wolf et al., 1996, 1998). These studies used surveys to collect data from conservation project managers and zoo curators and defined success as “producing a viable, self-sustaining population in the wild” (Griffith et al., 1989). It is unclear whether the programs were categorized as successful or unsuccessful by survey takers or by the researchers. In any case, the definition used was not specific enough to overcome taxon-specific biases; characteristics of species that allow this to be attained sooner than others. These studies are also relatively outdated in a field that is quickly progressing towards alternative techniques. Another major assessment project, a series of publications by the Reintroduction Specialty Group (RSG) that is released every few years, also uses the managers of each reintroduction group to indicate program goals and indicators of success. They then rate the programs in terms of level of success based on the number of goals met. The variation in the number of goals set by the program as well as the level to which they are created (e.g., the difference between simply releasing a population and having a confirmed increase in reintroduction rates) has the potential to be very high, causing the evaluation to lose merit. Only recently, as shown by Germano and Bishop (2008), has the definition of success been more narrowly defined in use with project assessments. In their study, the traditional definition was broken down and success was determined by multiple criteria that took both short-term and long-term results into consideration. To date, this this study along with King et al. (2012) seem to be the only two studies in which a thorough definition is used.

The obsolete data and conclusions provided by the majority of assessments offer little help to present and future reintroduction projects. This lack of modified, unified criteria presents an enormous problem when determining aspects that create a successful reintroduction, and in developing new recovery plans. Determining universal criteria for success is difficult because the assessment of success is often dependent upon the goals of the program (Kleiman, 1989). I was able to find two types of definitions used to measure the success of programs, all of which were vague definitions at best. For example, the RSG determined success based on the overall goals of the recovery plan (RSG, 2010). Depending on the number of goals a resulting program meets, the program is classified on a scale from failure to success. This completely changes the interpretation of the term "successful" from one program to the next. Others (Griffith et al., 1989; Fischer & Lindenmayer, 2000) define success as "establishing a self-sustaining population". Of course, a self-sustaining population can mean many things for different taxa. Also, the amount of time needed to establish a self-sustaining population is vastly different between species depending on their life history traits. Many criteria have been offered for defining success (Gipps, 1991); however those that are being used in practice do not take into consideration these differences in life history traits and provide little knowledge to organizations making decisions about future reintroduction protocols and candidates.

Species' life history traits provide a number of reasons to develop distinct definitions of success that maintain usage across groups that share those characteristics. Using life history characteristics to assess success levels has been introduced in small-scale assessments (Olney, 1994). In the early 1990s it became clear that differences among species should mean differences in reintroduction protocols and assessment criteria (Kleiman, 1989, Beck et al., 1994; Kleiman et al., 1994, Wolf et al., 1996, 1998; Reading et al., 1997; Fischer & Lindenmayer, 2000). Life span, reproduction, and dispersal are three of the attributes most argued to be vitally important to measuring the success of a program (Kleiman et al., 1991; Stanley Price, 1989). Numerous correlated traits such as generation time, age at first reproduction, and home range affect population growth rates, and thus provide differing appraisals for success; the evaluation of a program being successful could change with changing values of these traits across taxa. Using more of these variables in assessments of reintroduction programs may reduce the questionability of results.

While the variety of life history traits among reintroduction candidates provides a necessity for somewhat customized criteria of success, there can be a way to compare successful programs among species with similar life histories in order to determine procedures and criteria to apply to future reintroductions. Determining key factors to programs involving species with a certain life history profile may shed light on other species with a matching profile. The similarities can be used to not only develop new recovery plans, but also a new set of criteria that should be used if a species shares those characteristics. This need, along with the underwhelming amount of data providing evidence of “success” and the growing number of years since those data were published, provide an opportunity for a small-scale meta-analysis to be done on several old and new reintroductions spanning the globe.

In this review, I chose 27 reintroduction programs dating as far back as 1984 for use in a comparative study looking at the similarities of reintroduction protocols as well as life history characteristics among several “successful” programs (see Appendix I). The purpose of this study is to determine commonalities among successful programs in order to specify characteristics that should be the primary focus of future plans to reintroduce animals categorized in similar groups. I chose to use a revised definition of programs as reintroductions, as opposed to translocations, which were the subject of many former studies (Nielson & Brown, 1988; Wolf et al., 1998).

Materials and Methods

Data Sources

For this study, twenty seven reintroduction programs representing release areas all over the globe were chosen to be used in a comparative analysis. More studies were not used because so few programs provided the set of data needed for this particular study. Only mammals and birds were used due to vast life history differences among taxa and due to the publication bias that severely limits published data on amphibians, reptiles, and fish. Additionally, all programs used were previously deemed successful by the assessment from which they were taken. Unsuccessful programs were not used due to the large variation in definitions of such, and due to the lack of appropriate data available for a comparison between successful and unsuccessful programs. Data collected for the purpose of this study were found in published literature and information posted on program websites. These publications were reviewed and specific data related to a previously determined set of variables were extracted (see Appendix II). In order to compare similarities among successful programs, three editions of the Reintroduction Specialist Group (RSG) books of assessed reintroduction programs were used. These sources were chosen based on the limited availability of numerous programs assessed within the same study and for the purpose of using consistent criteria of success. The RSG publications are highly regarded as authoritative assessments of reintroduction programs and are to date the only known extensive assessments of large numbers of programs.

The majority of programs used in the study were included in these sources, although amidst further researching the programs, it became clear that many were inappropriately deemed successful or did not fit the definition of reintroduction used in this study. For example, a program repatriating Bongos (*Tragelaphus eurycerus)* from North America to a captive site in Kenya was treated as a reintroduction program and classed as successful. Additionally, a pair of Rock Hyraxes (*Procavia* *capensis)* translocated in South Africa were assessed (Soorae, 2010). These are among several species that were inadequately assessed according to the definitions and expectations of this study. To avoid using too many programs that were inappropriately classified, I chose programs within these assessments that had similar goals or indicators of success, and alternative programs were chosen from lists of known programs highlighted by the IUCN and Association of Zoos and Aquariums websites and publications. Indication of success for the supplemental programs was determined using similar criteria as RSG, but with added criteria set by myself, influenced by studies of success (Coonan & Schwemm, 2009; Mee & Hall, 2007).

Information collected from multiple publications was only taken if the same program during the same dates were noted. In some cases, not all information sought was provided, and thus was supplemented by former or more recent reintroductions associated with the same program. Most programs that were used in the study had multiple release sites. To avoid redundancy, only the release site with either the longest running time or that provided the most information was used. In some cases, distinguishing separate projects for one species was difficult. For example, the Peregrine Falcon (*Falco peregrinus*) has been introduced throughout the continental United States, however these efforts were not described separately and were unusable for this study.

Definitions/Criteria

As per Beck et al. (1994), programs were considered to be "an administratively distinct reintroduction", however this study included captive-bred animals as well as wild-borne animals that were rehabilitated or translocated. Based on this definition, there are multiple programs per species and thus there are more programs than there are species in the study.

Variables were determined based on importance to release protocol and compatibility between program details and life history characteristics. Variables used were finalized based upon available data.

Definitions of variables:

1. **Program Begin Date**

The date listed as the “program begin date” is considered to be the date upon which the first were released. For many programs, the specific date or even month was not provided, in which case only the year was documented.

2. **Program end date**

The date listed as the “program end date” is considered to be the year upon which either the supporting organization discontinued post-release monitoring, whether it were due to full confidence in success or to a program being stopped.

3. **Habitat**

Habitat was classified by the type of land cover most dominant in the release area.

4. **Release Site Area**

The release site area is the amount of land set aside specifically for the reintroduction. Some reintroductions were released into large national parks in which area was not explicitly expressed. In these instances, the area of the park was recorded.

5. **Distance from historical range** -

Distance from historical range was determined based on how far away the release area was from the species' historically known location.

6. Fenced/Unfenced

Fenced/unfenced refers to whether the release site had a man-made barrier surrounding it at the time of release. This does not take into consideration acclimation areas or islands. If a population was initially released into a fenced area then released outside the fenced area, this was recorded as whichever was the case during the most critical population growth, for the general purpose of population restrictions.

7. Proximity to urban areas

Proximity was determined by the distance to the nearest town or city. Urban areas used for proximity were selected based on availability of population data.

8. Population of closest urban area

Populations of towns were used from available census data. The dates provided were the most recent population sizes, ranging from 2004-2012.

9. Species source

Individuals were considered captive if born in captivity and wild if born in the wild. Young wild-borne individuals that were taken into captivity were typically released before adulthood and were considered wild.

10. Released Population Size

Released population size is the number of individuals in the very first release.

11. Total individuals released

Total individuals released were recorded as the most recent population count of released individuals and their offspring. This number was not provided by some programs, and only the total wild population was given. In these cases, this information was left blank.

12. Carrying capacity

Carrying capacity was calculated based on the size of the release area and typical population density of the species. In some instances, carrying capacity was already given but needed to be calculated for most programs.

13. Type of release

Programs were determined to be soft release if the animals were put into acclimation pens/cages prior to release and were then provisioned food and/or water for any amount of time after release. Acclimation pens were defined as an enclosure within or near the release site. Medium release was considered to be animals that were put into acclimation pens/cages prior to release but were not provisioned post-release. Hard releases were animals that were released without first being acclimated, but may have been provisioned immediately after release.

14. Sociality

Species were depicted as either solitary or gregarious. Animals living only in breeding pairs were considered solitary.

15. Social Structure

Social structure refers to the type of group a species lives in, if gregarious. This includes social groups, family groups, breeding pairs, and harems

16. Age at first reproduction

Refers to the age at which the female can first reproduce. It should be noted that some sources provided the average age at which most females actually first reproduce, which is slightly different.

17. Average interbreeding interval

Number of days typically spent between parturition/egg laying events.

18. Average weaning period

Number of days until young has been weaned or fledged.

19. Average number of offspring per reproductive event

Self-explanatory.

20. Average number of reproductively active pairs per group

This factor was measured as either one pair per group or more than one. This does not apply to animals living in breeding pairs.

21. Time to sub-adult phase

This factor was measured as time (in days) to independence, when offspring are no longer considered “young”.

22. Estrous frequency

Estrous frequency was determined as the number of times per year that the female goes into estrous.

23. Mating system

Mating system was determined as the number of mates an individual acquires during a breeding season.

24. Diet

Diet was categorized as carnivorous, omnivorous, or herbivorous. The diet categories were determined using the same criteria as Wolf et al. (1996).

25. Food obtainment method

The food obtainment method was set to be slightly more informative than diet category. This includes predators, scavengers, grazers, and foragers.

26. Number of food sources

Number of food sources was categorized into type (animal source, plant source, both).

27. Generalist/Specialist

Within the number of food sources, a species may rely on for instance multiple species of animals or just one.

28. Lifespan in captivity

Lifespan in captivity was determined as the average number of years surviving in captivity.

29. Lifespan in the wild

Lifespan in the wild was determined as the average number of years surviving in the wild.

30. Sedentary/nomadic

Species were considered sedentary if they typically stay in one area at a time, as opposed to species that are constantly moving in search for food.

31. Maximum home-range size for male

The average home-range size was not always available, so data was collected on the maximum home-range size needed for the male of the species.

32. Territoriality

Species' territoriality was measured as high, medium, or low. High territoriality is considered to be when animals actively mark territories. Medium territoriality was assigned for species that do not actively mark territories but will fend off threatening intruders. Species that have low territoriality do not fend off others using the same area.

Data analysis

Because I was unable to compare successful and unsuccessful programs, I chose to test for significant differences in variables of successful programs between categories. All species were stratified by taxon and trophic level: as bird or mammal, and also as carnivore, herbivore, or omnivore.

The two sets of categories were tested independently and together. Variables were tested for differences using SPSS 20.0 (IBM, 2012). Due to the mixture of continuous, dichotomous, and multinomial values, ANOVA's, generalized linear model, and nominal regression functions were used. The outcomes of these functions were used to determine the differences of each variable with respect to each category (significance was determined based on a p-value <0.05).

Results

The results of this analysis were limited at best. Most variables were found to have no significant differences between the taxa or between diets. Two variables were found to have informative differences. Average offspring per reproductive event was shown to be significantly different among diet (ANOVA: $F(2, 1) = 6.423$, $p = .006$). Tukey post-hoc comparisons further indicated higher average offspring for herbivores over carnivores (M = 3.01, 95% CI [1.10, 4.92], p = .002) and a higher average offspring for omnivores over herbivores (M = 2.18, 95% CI [.41, 3.96], p = .014). This result provides implications for the population growth of herbivores as a group and can be used in conjunction with a life table to influence reintroduction protocols. The second factor that showed a significant difference is average lifespan between taxa (ANOVA: $F(1, 13) = 4.879$, $p = .046$). The drastic difference in average lifespan between birds and mammals may affect other life history traits and the amount of time needed for post-release monitoring. Release site area is a factor that had marginal significance among taxa (ANOVA: $F(1, 23) = 4.125$, $p = 0.054$. An independent t-test confirmed the significance of this difference (M = 148560986, 95% CI [341566189, 44444217], p

= .001. It should be noted that although some avian species have wide range potential and thus may be released into an open area, the high mean difference value was likely due to the California condor having been released into an area expanding millions of kilometers squared.

The number of results that had significance was disappointing. Due to the sheer lack of available data, particularly biological values, there were many omissions throughout the database, which likely played a large role in the reduced significance of the results. Had more data been available, there perhaps would have been more significant results. While these factors seem to be the only ones with significance with respect to diet and taxa, there are several more that are potentially important to the process of reintroduction biology. For instance, release site area was expected to have a difference among diet; with carnivores having significantly higher areas of release. Also, sedentary versus nomadic, or the difference between species that stay in one area/territory versus those that migrate, is one factor I had expected to be significantly different between diets. The same result was expected for territoriality. All three of these factors would be vitally important to reintroduction programs, for setting aside the amount of space needed by a species is among the first priorities preceding release.

Discussion

Although the results of this study were not able to support or dismiss the findings of previous assessments (Griffith et al., 1989; Wolf et al., 1996, 1998), they are able to provide two pieces of knowledge for future reintroduction programs and assessments. First, the significant differences that were found among taxa offer an indication that life history traits have an incredible range that could be affecting the outcome of recovery plans. These differences affect species' abilities to reproduce, acclimate, and survive in novel environments. This study along with future assessments may help to determine which factors are important for certain species, and develop a list of traits on which to focus. Finding priority factors could eliminate experimental trials and reduce failure rates. Second, the limited findings of the analysis reveal the absence of a structure that enables the ability to assess the outcomes. Despite the limitless potential for useful evaluations, this study was unable to adequately analyze the factors correlated with “successful” reintroductions. While these results seemingly show little significance among life history characteristics and program protocols, there are many challenges that were faced while collecting this data.

Defining Success

The largest obstacle in reintroduction biology is the lack of a clear definition for success. Upon searching for “successful” and “unsuccessful” programs to compare, few provided criteria for why their program was classified as such. When criteria are not specified within an assessment, it is hard to infer that the process is credible and uncertainty about their conclusions becomes an issue. Success seems to be mentioned as small aspects of a program rather than as a whole. For example, a number of programs discussed certain aspects of the release (e.g., individuals successfully finding a den/nesting site) that had a huge impact on survivability rates or reproduction, but they rarely remarked on the program as a whole being “successful” or not (RSG, 2011). Among those that provided a definition, it was extremely vague and provided no measures to use in comparison with others. Griffith et al., Wolf et al., and Boitani et al. (1989; 1996, 1998; 2004) all defined success as the “establishment of a viable population”. This definition arguably offers the hardest criteria to meet. Firstly, what does it mean for a population to be established? Typically to establish something implies some sort of permanence. At what point is a population deemed permanent? “Establish” is yet another term that is in need of a strict definition. Additionally, the amount of time it takes for a population to reach a set size is vastly different across taxa and typically spans beyond the scope of any trial reintroduction and for some species, exceeds the scope of any conservationist attempting to assess a program (Kleiman et al., 1991). The use of a definition as broad as this raises more questions than are answered and eliminates a standard process of determining success.

In spite of previous assessments using obscure definitions of success, I argue that this should be avoided to prevent programs from indiscriminately being defined as successful or unsuccessful. The IUCN Reintroduction Guidelines, a brief document followed by most reintroduction programs, states that the identification of short-term and long-term success indicators should be determined, and later notes that “re-establishment implies success”; however they fail to provide a list of appropriate indicators or offer ideal measures of success (IUCN/SSC, 1998). This document was updated in 2013 (IUCN/SSC, 2013); but even so, the same problems remain. Indeed, there is still a huge problem with defining what it is that reintroduction biology seeks to study.

When one thinks of the term “success” in reference to a reintroduction, there are certain things that would typically come to mind. Primarily, one would immediately think of an individual retaining its fitness after release. This would consist of the released individual surviving to reproduce and at least some of its offspring surviving to reproduce. Also, one may think of a level of genetic variation being maintained throughout a population. Ideally, a population would grow to have enough members and genetic variation among them for a fission event to occur. Lastly, the term “success” should imply that a prolonged amount of time has passed prior to a program being regarded as such. This implication is based on the idea that a population's reproductive abilities may require many years for it to reach an ideal size. It is argued that a population must have a high probability of persistence with minimal or no intervention to be deemed successful (Seddon, 1999), and in order for this to happen, sufficient amount of time should be given. These criteria should be inherent; however these do not seem to be employed by conservationists.

Reintroduction biology has been praised for the progress the field has made (Seddon et al., 2007, 2012; Stanley Price, 1991). The IUCN and the RSG (a group created solely for the purpose of assessing and influencing reintroduction programs) have been cited in plenty of publications as the forefront of reintroduction biology advancement (Ewen et al., 2012; Armstrong & Seddon, 2008). It has come to my attention, however, that many of the authors and scientists making these claims are all associated with each other, either via the Reintroduction Specialty Group, or through program affiliations. Based on this close association of reintroduction biologists it seems that there are few outsiders assessing these projects and determining faults and virtues. While the field of reintroduction biology has come a long way since the 1970s, it is a relatively new field and many of the problems faced decades ago still persist. This field is not yet known for its depth of scholarship and; reintroduction, post-release monitoring, and assessment techniques are still being studied.

Another problem made noticeable within the RSG assessments is that of programs prematurely being labeled as successful and/or releasing a small number of individuals. One example is of a reintroduction of twenty black rhinos released into the North Luangwa National Park in Zambia over a three year period. Small time increments were used to reference indicators for success, such as a 90% survival rate after one year (van der Westhuizen et al., 2010). Because rhinos have long gestation times and long interbreeding intervals, one year does not provide any indication of success. The cases in which programs are prematurely labeled as successful further showcase the inadequacies of the goals being used to indicate success. Furthermore, there are programs that have been highly regarded as successful but now seem to be struggling. After 13 years, the Wisconsin Whooping Crane recovery program still has extremely high chick mortality rates; maintaining the need for intervention. In the case of the red wolf, the large population of unwanted coyotes living within the same territories and recent coyote hunting permission has led to humans mistakenly killing red wolves. Despite the length of post-monitoring being one year or five, population growth varies by taxon, so success should not be concluded until that species has had the appropriate amount of time needed to reach a low-risk size.

Vague definitions or not defining the criteria for which a program is deemed successful reduces the ability to make comparisons among programs. Rather than throwing out vague definitions that only few programs use or using outdated assessment methods that falsely capture information (such as the Griffith, 1989 methods that have been reproduced), recovery plans should focus on conservation ecology and determinative life history traits that can better gauge success for each group of species sharing similar characteristics. Success should also take into consideration the resilience of a program. There are many examples of programs that initially were unsuccessful, but after appropriate changes to release protocol or post-release monitoring, they were able to improve their outcomes (e.g., golden lion tamarin and the black-footed ferret). This would require a lengthy amount of time prior to concluding whether or not a program is successful.

Data Acquisition

In addition to a paucity of defining success, a second obstacle I faced during this review was data acquisition, as emphasized by my results. Many of the programs used have been discussed in numerous books and papers about conservation, and some have published a great deal of raw data about released individuals and release protocol (Coonan & Schwemm, 2009; King & Gurnell, 2005; Fritts et al., 1997). However, rather disappointingly, the majority of programs provide very few data about reintroduction specifics. Among these, many offer vague descriptions of how many individuals were released within a decade or longer time-scale. They also frequently provided most recent population number in place of total number of individuals released (Spalton et al., 1999; Zafar-ul Islam, 2011). This number tells us little about reproductive rates and survivability of released individuals compared to wild-borne. Furthermore, mortality rates, reproductive rates, and age structure of released individuals were not included in published data. Lastly, even basic biological data was unknown for some species, such as lifespan, average age at first reproduction (as opposed to age at which an animal can reproduce), and average age at weaning. These variables, among many others that had to be removed from this study are critical for determining success. Because of the inability to find many pieces of information, my database for which the assessment was based had several omissions. The ideal type of analysis that would have been used for this study was compromised and thus, a simpler analysis was performed which likely affected the results.

From an adaptive management perspective, offering such obscure information mitigates population study efforts and hinders our ability to modify programs. There are perhaps two reasons why detailed information was not published. In some cases, providing vague information about the release of endangered animals may be the result of poaching pressures on those populations. Over-sharing knowledge of numbers and gender of released populations with those who may exploit that information can cause major concern. Another possible reason is that programs may not know or have the resources to collect certain data. If this is the case, there is even more support for the arguments presented. To reiterate, a level of specification is needed in order for scientific reviews to be conducted and provide assessments and recommendations for future releases. To streamline the reintroduction process as well as open up the ability to study and monitor their outcomes, there needs to be a universal list of characteristics that monitoring efforts should log (see Appendix II). Griffith, Wolf, and their respective colleagues collected data by surveying reintroduction practitioners. Although this method allows for more data to be collected, it has the potential to introduce subjective opinions that may have influenced classification of the programs. In comparison to the data acquisition method chosen for this study, a survey may have been superior despite its faults, for many data points were excluded due to their absence in published literature.

On top of information being omitted from published works, there are some cases in which the information is either unknown or argued amongst the scientific community. Being that many of these animals have dwindling populations and are typically protected, there are also few opportunities to study their biology. Only in the cases of species that have been thoroughly studied prior to their decrease; or if a species is held at a research-based facility, is knowledge of their reproduction and lifespan well-documented. For species such as the Przewalski's horse, Chimpanzee, and California condor, this information is well-known and not commonly disputed (Boyd & Houpt, 1994; Nowak, 1999; Mountfort & Arlott, 1988). In contrast, the Asian houbara bustard, a species that was almost hunted to extinction, has been reintroduced despite having not been studied until recently. This resulted in the program facing hurdles related to ignorance about the bird's natural predators and their sensitivity to unknown landscapes (Seddon et al., 1995). Strides have been made to understand captive breeding, however there is still much to learn about their general biology. Another key biological factor that is not well understood is that of home range, density, and thus carrying capacity (Prato, 2009). Carrying capacity depicts how many individuals an area can support prior to a population decline; something of which all programs should be aware. While this information is known of some species, for others such as predators or species with large home ranges the number often varies or is unknown. In many cases, the only densities provided are that of current research on populations in other areas. Endangered species rarely have populations in other areas, so scientists are unable to estimate their densities. This also causes disagreements among biologists and creates difficulties for those of us attempting to relate such factors to success. Knowing the biology of a species is the primary step a recovery plan should take prior to habitat selection or release, therefore more effort should be put into solidifying knowledge of endangered and expected to become endangered species so as to improve potential for success.

In addition to a deficiency of explicit data, there are biases throughout the reintroduction biology literature. First, there is a taxonomic bias in which most publications are studies of vertebrates; particularly mammals and birds (Bajomi et al., 2010). This bias is likely due to national priorities, unequal support, and to assessments of conservation status primarily focusing on birds and mammals (Seddon et al., 2005). While this bias is not uncommon throughout biology and conservation, it reduces the ability to understand reintroduction biology more thoroughly. There is also a bias as to which programs are published. Accounts of “unsuccessful” programs are rare to say the least. What was originally to be a study focusing on contrasting characteristics between successful and unsuccessful programs was modified because published data only focused on those that seemingly prospered. Without data on failed attempts to reintroduce species, conservationists are unable to determine aspects of release protocol and species demographics that should be avoided. Moreover, the majority of information published is from large programs that have considerable support from the government or influential organizations, and thus have the funding available to publish annual reports and maintain websites that consistently offer updates. While there may be a correlation between financial support and success rate (although this too was omitted from this study because of a lack of published data), it does create a bias that limits the scope of assessments. It also prevents information about smaller reintroductions of the same species from being included. For example, the reintroduction of the peregrine falcon across the continental United States is commonly recognized as a successful reintroduction story; however only limited data on any particular release area are available. This is unfortunately not the only program for which this is the case (Heinrich, 2009; Green & Ramsden, 2001).

The future of reintroduction programs

Reintroduction programs will inevitably become a go-to solution for charismatic species and those on the brink of extinction. As of yet, a majority of reintroductions have been for mammals or birds; charismatic mega-fauna that are relatively large and whose success is easy to track and monitor. Reptiles, amphibians, fish, and invertebrates have had significantly less support using the reintroduction method due to the challenges they introduce to the process; either limited knowledge of a species or difficulties with release and monitoring procedures. With the growing number of species requiring special attention, prioritization is necessary, as creating customized recovery plans for the 17,000 species already in need is not feasible. Not all of these species will be lucky enough to have a recovery plan implemented and of those that do, only some will be successful. Because reintroduction programs are not one-size-fits-all and should only be implemented in dire situations, there are many considerations to be made prior to assuming that a recovery plan is feasible for a particular species. While the U.S. Fish and Wildlife Service has created criteria to use when prioritizing which species to recover (Martin, 1994), those criteria are not necessarily used by other organizations and do not cover some key factors. For this reason, I propose the following questions for organizations to ask when determining whether or not to implement a recovery plan:

15. **Is the species extinct in the wild?** *Or is there a certain location where the species has gone locally extinct?*
16. **If not, how likely is the wild population to go extinct?** *For how much longer is the population expected to persist? These two assess whether or not a reintroduction program is actually needed*
17. **What else has been done to save the species?** *Are they legally protected? Has anything been done to conserve their habitat? Is there local support for the presence of the species? What else could to be done before resorting to reintroducing the species?*
18. **Is there suitable habitat?** *How wide of a range does it cover? Are there threats that would make that habitat unsuitable in the near future?*
19. **Is the historic range still suitable?** *If the population needs to be released outside of the species' historic range, how would that affect native species in that area?*
20. **How much genetic diversity is there in the founder population?** *Is it worth the risk of losing highly valuable individuals for an experimental reintroduction program?*
21. **Is there a sufficient number of animals left in captivity to provide supplementary individuals if the program is a failure?** *How large is the captive population and are there enough individuals to create an ideal sized-population (or more) in the wild*
22. **How much is known about the biology of this species?** *Important to understand age-structure for a release, how reliant a species will be, population dynamics, etc.*
23. **What is the largest threat to the species?** *How manageable are these threats? Is it possible to reduce these threats prior to release? It is possible to manage them during recovery? How likely is each threat to demolish a population (disease, human-wildlife*

conflict, etc.)?

The first two questions assess whether a reintroduction program is necessary. Since reintroductions are used to recover a population, the status of a population and its risk of significantly declining is an indicator of whether a recovery plan is needed. The last seven questions assess whether a reintroduction program is feasible. These questions present more complex factors that each should be assessed prior to the development and implementation of a recovery plan. The level of complexity presented showcases the numerous reasons as to why reintroductions should only be used under dire circumstances. The criteria species should meet in order to be considered a candidate are copious and some are difficult to meet.

In addition to characteristics about the animal, there are characteristics about the habitat or potential release area that are more variable, the status of which can drastically change over time. As mentioned earlier, for a habitat to be suitable the immediate threats that initially drove the species out must be removed. While this challenge can be met at times, the certainty of keeping threats at bay is low. In the case of many unwanted predators being reintroduced, humans risk the outcome of a reintroduction by purposefully killing individuals or forming complaints about their presence, which can lead to the discontinuation of the program. In other situations, humans pose threats by poaching or using the resources within a release site. In both of these situations, local participation and support remains crucial to the success of their recovery (Zuccotti, 1995; Bright & Manfredo, 1996), but can be extremely difficult to attain. There are numerous other threats including predation by native and non-native predators, competition for food sources, and disease. The prevention and continued mitigation of these threats again remains uncertain. This situation also introduces another challenge: that a species' historical habitat is not necessarily suitable at present-day. Furthermore, a habitat that is currently suitable will not necessarily remain suitable for the foreseeable future (Osborne & Seddon, 2012). With climate change threatening climatic ranges, animals living in habitats with particularly harsh conditions, or small ranges in temperature or rainfall, will face obstacles after being reintroduced. In the reintroduction of the Arabian oryx and Przewalski's horse, some sites faced severe droughts that killed off many individuals (Stanley Price, 1991; Kaczensky et al., 2011a). Due to their resilience, the populations were able to overcome depressions, but these species and others may not have the same luck in the future. For instance, the Polar Bear would likely not be a selected candidate for reintroduction due to the dire prognosis of its habitat, among other reasons.

Almost as importantly, programs customized for one species, in one release area, are extremely costly. While the total expense of a project is not often published, an ongoing recovery can easily require millions of dollars in support (Simon et al., 2012; USFWS, 2013). In order to receive this funding, programs need governmental or non-profit organizations with the resources to support them. In the case that a reintroduction is managed by a small organization or a governmental branch with limited funding, the success of that program is threatened (Sarrazin & Barabault, 1996). Typically, programs begin with an experimental stage; that is to say that funding of a long-term reintroduction is contingent upon the success of the first few years. If a program does not meet a level of success within the first few years, funding is discontinued and any remaining individuals are taken back into captivity. This of course creates a problem because it may take several years for a population to gain momentum. These experimental reintroductions are notably very risky, for they consume a large amount of funding in the development and implementation of the program, as well as have low assurance of a successful outcome. The number of new reintroduction programs makes determining their success difficult, especially because so many end prematurely due to insufficient support or funding.

These challenges create reason for conservationists to seek alternative ways to protect the species that have not yet reached the need for a recovery plan. This would mean better protection or restoration techniques, or methods that have not received much attention. For those that conservationists have not been able to protect, a more organized and inclusive reintroduction process will need to be developed. For instance, rather than focusing the efforts of each reintroduction program on one species, they should incorporate multiple species, a move that could help restore the ecosystem in which other species could flourish. Multi-species reintroductions have been carried out and show strong potential to be a successful alternative to the single-species track that conservationists are currently on (Finlayson et al., 2008). Research has suggested that prioritizing reintroductions for the most endangered species will not reduce the number of extinctions in the long run and instead, focusing on biodiversity should be considered (Wilson et al., 2011). Additionally, recovering multiple species at one time could provide economic and political benefits; saving funding on separate implementations and providing stronger support for policies to be created for an area or region (Loomis & White, 1996).

Adaptive management is one method that can minimize the variability of reintroduction programs, using an iterative process to make optimal decisions throughout a management practice. These decisions are based on objectives, current status, and knowledge about the system's response (Nichols & Armstrong, 2012). By using adaptive management techniques, program managers would not have to rely upon initial estimates for release protocols. Instead, they would have the ability to assess population dynamics, response to variable abiotic and biotic factors, and could then react to any changes. Until very recently, most reintroduction programs were structured on a trial and error system. A group of individuals were released under the most optimal conditions at the time of release. Managers would essentially wait to see what happened with that population, and restructure future releases based on the outcome. This method is highly ineffective and reduces the reaction time for management decisions to be made. Adaptive management requires a plan prior to release on how to respond to varying situations or outcomes (McCarthy et al., 2012). The use of adaptive management should also be used in conjunction with long-term ecological monitoring. For some species, established monitoring programs prior to a species decline have enabled the program to quickly act prior to the population reaching extinction. In the case of the Channel island fox (*Urocyon lattoralis*), monitoring by the National Park Service observed severe declines in the population and were able to hastily determine its cause, remove the threat, and reverse the trend through reintroductions (Coonan & Guglielmino, 2012).

One cause of the limited success seen in the first reintroduction years was the minimal level of post-release monitoring (IUCN 1987, 1998; Lyles & May, 1987). Fortunately, this has become a new focus with more recent programs and has reduced the number of reintroductions that end prematurely (Soorae, 2008). Post-release monitoring is the continued monitoring of released individuals for up to several years following a release. This allows conservationists to track the growth or decay of a population and gives them an opportunity to alter management practices in order to maximize the outcome for the released population (Wakamiya & Roy, 2009). As a form of adaptive management, this added stage utilizes the knowledge learned from the initial stage of the release to adapt to any unforeseen challenges. Modifying management practices reduces the need for more intensive interventions. The only difficulty introduced by post-release monitoring is added cost and a need for adequate knowledge of the species in order to fully understand their status.

Although there are plenty of reasons why conservationists should not rely on reintroductions as a guaranteed option, there are some species for which reintroduction is a must. In these cases, the species has been completely extirpated (as was the case with the gray wolf and black-footed ferret). These species are necessary for the proper functioning of ecological processes within their ecosystem. The need for a local population outweighed the risks of reintroducing the two in their respective areas, and so, reintroduction was necessary (Smith & Ferguson, 2006; USFWS, 2013). Another case in which reintroduction is a must is when a species has the characteristics needed to prosper but has had the disadvantage of an event or a series of events that drastically reduced or eliminated the population. Species such as the California condor, golden-lion tamarin, and Przewalski's horse have one or multiple characteristics that allow them to take advantage of their environment, but have formerly faced declines in the population. The potential to which these populations could thrive indicates that reintroduction could be an efficient method with low likelihood of needed interventions later on, given that the initial threat is removed. These situations create a need for reintroductions but if conservationists must rely on reintroduction, the greatest amount of effort should be invested.

Life-history traits

Life-history traits are clearly a critical component to developing and assessing a reintroduction program. The number of important characteristics disregarded by publications showcases the immediate need for modifications of reintroduction protocol, data collection, and assessment techniques. Life history traits can drastically influence the outcome of a reintroduction program, especially if those of the focal species are not well-understood. These characteristics have so much variation across taxa that they inevitably should be considered when addressing the final outcome of a species' reintroduction.

The age at which a species should be reintroduced is largely dependent on their life history characteristics (Sarrazin & Legendre, 2000). Studies of soon to be released individuals have shown that the age at which an animal is released can significantly impact their post-release survival (Jalme et al., 1996; Saltz, 1996). Reintroducing mothers and their offspring may have a different outcome than releasing all juveniles. Throughout a lifetime, there are several life history traits to evaluate. At the beginning of an individual's life, birth weight, growth rate, age at weaning, and age at sexual maturity all determine the length of the juvenile stage. Once an animal becomes reproductive the estrous cycle, mating system, gestation period, litter size, and interbreeding interval are all characteristics that severely affect the life table of that species and rate at which a population can grow. Additionally, the number of years reproductively active and lifespan can shed light on the total number of offspring a particular individual can contribute to a population.

Understanding all of these characteristics is important to the success of the reintroduction and/or management of a species. What is equally as important is understanding the variation of these traits among species and even among genders within species. For instance, the American bison (*Bison bison*) is a ruminant that exhibits gender segregation with respect to diet (Bowyer, 2004). Males, the larger of the two, have longer digestive tracts, enabling them to digest lower quality vegetation. In contrast, females have smaller digestive tracts and are thus more restricted to higher quality vegetation that does not require longer periods of fermentation. Because of this difference, the type of vegetation within a herd's home range must be a mixture of high and low quality. This introduces a management problem that would need to be solved prior to any reintroduction of this species.

Health /Disease risk

In addition to the numerous life history characteristics to consider when developing a recovery plan, the overall health of the animals is an issue that should be assessed while in captivity and monitored closely in the time following their release. The stress of transportation and release into a novel environment can jeopardize an animal's ability to properly acclimate to the area in which they now must survive (Letty et al., 2000). Also, features that were unknown in captivity could present a source of illness or mortality for naïve or young individuals. For example, some programs had at least one released individual die of drowning; large water bodies were likely not experienced in captivity (Tutin et al., 2001; Phillips et al., 2003). The type of release used in the reintroduction protocol can greatly minimize the effects of these factors. Of the programs used in this study, more than half used either a soft-release technique in which the animals were acclimated to the release site prior to being released, or provisioned released animals with food and/or water. These methods are known to reduce mortality risk by allowing animals to become familiar with their surroundings and preventing them from starving while they learned how to find and acquire their own food (Bright & Morris, 1994; Mitchell et al., 2007). In relation to health, disease risk is also a factor that should be addressed. Any time an individual is introduced to a new location, they are bound to be susceptible to pathogens. Moreover, the animals already living in the release area are prone to any disease introduced by the released population (Ewen et al., 2012). This risk needs to be carefully considered prior to any release, and preventative measures should be as conservative as possible. For instance, Black-footed ferrets were immunized for all of the commonly known diseases to affect mustelids prior to being released in Wyoming. What was not known was that this species is also susceptible to the Sylvatic plague, a disease prominent in prairie dog populations; the primary food source for these animals. This ignorance caused the initial released population to suffer a significant loss and threatened the entire program (Williams et al., 1994).

Genetics

Another issue that could affect or be affected by the reintroduction of a species whose population is in decline or all but gone is that of genetic consequences. In order for a small population to persist without intervention, they must be able to adapt to environmental change (Keller et al., 2012). Without genetic variation, the number of phenotypes represented in a population is diminished, resulting in a smaller chance that the phenotypes present are suitable for an environment. Small populations are more at risk for certain genetic effects, which in turn could reduce their ability to adapt (Lande, 1988). Of the major genetic effects that can affect a population, the two most relevant to endangered species are inbreeding depression and genetic drift. Both have the ability to affect the long-term viability of reintroduced populations because of their naturally small sizes.

The first concerning consequence is genetic drift; a change in allele frequencies over generations. Typical to many bottlenecks, once a population is reduced to a small number of individuals, the genetic variation for that population is also likely reduced. By chance, alleles may have been lost from the population leaving a small percentage of the original alleles remaining. Loss in genetic variation due to genetic drift can cause certain alleles to become fixed; reducing heterozygosity and altogether removing what could be adaptive phenotypes from a population. This narrows the population's plasticity and inhibits its evolutionary potential. Genetic drift is often seen in endangered species, however its severity is highly variable (Keller et al., 2012). In cases in which a species has multiple populations, if more than one population experiences declines and thus genetic drift, this could also lead to genetic divergence between them (Gaggiotti & Couvet, 2004). With the goal of recovery plans being to reintroduce populations of a species, the potential for genetic divergence is not an insignificant matter.

Inbreeding depression is a genetic effect that can respond to the effects of genetic drift, or affect a population on its own. Inbreeding depression is the reduced reproductive success, survival, disease resistance, etc. of an individual resulting from the mating of two relatives (Keller et al., 2012). The combination of similar alleles increases the appearance of deleterious alleles, weakening an individual's ability to survive the stressors of the novel environment in a reintroduction. In cases in which the last remaining individuals are captured, there is a higher risk of inbreeding depression, especially if those individuals came from one known population (Narayan et al., 2008). Inbreeding depression can affect several life history characteristics such as fertility, birth weight, and recruitment (Keller & Waller, 2002); traits that can reduce reproductive rates and inhibit population growth. In order to improve reproduction, breeding must be carefully assigned for the purpose of optimizing the amount of variation in following generations. There is some evidence that the effects of inbreeding depression may be more severe in situations of high environmental stress (Miller, 1994; Kristensen et al., 2003). Due to the nature of reintroductions, this could mean that the effects seen in captive animals may become more severe after reintroduction. It also introduces the question of how much more severe this genetic effect might be in response to the ever-changing habitats in which we are releasing animals. Because of the stress induced by novel environments and environmental change, it is argued that genetic variation of traits involved with stress response are critical for the long-term survival of reintroduced species (Parsons 1989; Hoffmann & Parsons, 1991).

Genetic and phenotypic variation

Among the loss of phenotypic variation due to genetic drift is the loss of adaptations in long-term captive animals. The environment created by captive facilities rarely provides adequate enrichment or settings that mimic a species' natural habitat (Menzel & Beck, 2000). Preventing the exposure to natural conditions can often relax or modify selective pressures acting on behavioral and morphological traits (Price, 1984; O'Regan & Kitchener, 2005). The resulting changes to these traits may prove fatal to captive-bred animals released back into a native habitat full of stressors to which they are no longer fully adapted. In fact, it has been shown that translocated animals are significantly higher post-release survival rates than those that are captive-bred (Jule et al., 2008). This difference is likely due to the inexperience and loss of adaptations caused by captivity. For example, a reintroduced species in which anti-predatory behavior was diminished during captivity is the Asian houbara bustard (Islam et al., 2011). This species has been reintroduced into Saudi Arabia since the early 1990s, but initial releases faced close to 40% mortality rates caused by predation within days of their release (Combreau & Smith, 1998).

Studies have shown that anti-predator avoidance is a natural response that can be reduced or lost in captive populations over successive generations (McPhee et al., 2003; Hakansson & Jensen, 2005). Both studies measured behavioral responses before and after simulated predator attacks and found insignificant differences. The results of these studies along with further investigation of the genetic structure of captive-bred populations indicate that future reintroduction programs should take into account the number of generations an individual or group of individuals have been captive as well as the amount of genetic diversity represented but that group to minimize genetic adaptation to captivity (Frankham, 2008). Additionally, it is concluded that a greater number of individuals would need to be released to counteract the effects of captivity (McPhee & Silverman, 2004). Given that reintroduction programs are already faced with limited numbers of individuals that have under-represented genes, the ability to breed a sufficient release population becomes more difficult.

The topic of genetic adaptation to captivity has promoted a discussion surrounding the process under which this phenomenon is occurring. One argument is the possibility that zoos and captive facilities in general could unintentionally be causing temperament shifts due to breeding programs and animal husbandry practices (Frankham et al., 1986; McDougal et al., 2006). Differences in temperament cause certain individuals to be either easier to breed or more successful at breeding; meaning that perhaps captive facilities are only successfully breeding individuals that have reduced stress and increased cooperation. For instance, an extensive study of the effects of captive environments on black rhinoceros found that captive-bred individuals were much friendlier than those born in the wild. Additionally, males with reduced dominance and females with reduced chasing had higher breeding success (Carlstead et al., 1999). Although species' breeding programs are managed by species survival plans, taxon advisory groups and others, a mating will only be successful with the cooperation of the animals themselves as well as their emotional and physical health (Carlstead, 1999). Those with higher levels of aggression or fear may be predisposed to pair incompatibility or reduced overall breeding success. Overall, captive facilities could be unconsciously selecting for “tamer” traits through breeding practices and the way that animals are reared and interacted with by keepers. These problems could be an aspect of population genetics and animal husbandry that zoos or other captive facilities can alleviate by modifying the environments of animals designated for future release.

The Role of zoos

The aforementioned problems seen before and after populations are reintroduced into the wild provide a major role for captive facilities to modify and improve their techniques. With respect to the suppression of natural instincts, there are several things that zoos can improve upon. Animal enrichment is one skill-developing experience provided to animals that mimic behaviors that would be used in the wild (Hoy et al., 2010; Shepherdson, 1994). Increasing the types of skills animals use within their exhibit can continue the selective forces that cause adaptations to the wild. The benefits of skill-training in general have been debated among scientists for years. A recent study by Reading et al. (2013) found that enrichment improved the locomotion, foraging skills, and physical condition of some animals. However studies at the National Zoo did not form the same conclusions. Prior to release, the golden-lion tamarin population at the National Zoo was placed in an outdoor woodland habitat that allowed the monkeys to experience locomotion and foraging under natural conditions. After release, the survival of these individuals was compared to those that did not have this enrichment. No significant difference was found, implying that the enrichment did not increase survival (Stoinski & Beck, 2004; Stoinski et al., 2003). The differences found among these studies imply that enrichment may be a beneficial tool for some but not all species.

Anti-predator training is another tool used in captivity that has promising effects on post-release survival (Alonso et al., 2011). Evidence suggests that pairing the presence of a predator with a negative experience can elicit and train a natural response to predatory species (Griffin et al., 2001). In the case of the Asian houbara bustards' loss of anti-predator response, research has suggested that an anti-predator training method exposing future release individuals to live predators has a significant positive effect on their post-release survival (van Heezik et al., 1999). Understanding the problems that captive-bred individuals face after release and adapting enrichment programs to the skills that they lack should be employed when avoiding long-term captive lineages is infeasible.

Genetic management is another area in which captive facilities can play their part to increase the success of reintroduction programs. Although a majority of facilities providing the animals for release do not directly participate in their reintroduction, they control the breeding programs that produce the animals and thus can have a large influence on their survival. The steps taken to optimize genetic variation and reproductive success are crucial to the future generations of released populations. Zoos and others have used breeding plans such as studbooks and species survival plans that organize the matings of underrepresented DNA, and should continue to work towards this goal. One improvement that may be useful is to begin integrating the genes of wild animals into the captive population. The artificial insemination of captive females by sperm collected from wild males has been an experimental method of improving the genetic variation of captive populations (Philippart, 1995; Rodriguez-Clark & Sanchez-Mercado, 2006). Zoos are currently transporting sperm from males all over the world but can take the next step to make arrangements with wildlife organizations to collect semen from wild-borne or first generation captive animals for the purpose of expanding the genes being represented. This expansive breeding technique could also prove to reduce the effects of adaptations lost in long-term captive populations.

Despite the supportive role zoos play in the field of reintroduction biology, not many go beyond indirect involvement (Conway, 2003). Zoo scientists hold a great deal of biological and behavioral knowledge on endangered animals that are not particularly well-known by the governmental organizations that typically organize and manage reintroduction programs. The research, educational, and funding opportunities available to zoos can be used to support recovery plans managed by them; likely increasing their potential for success (Chivers, 1991). Some well-known reintroduction programs with direct zoo involvement, such as the golden-lion tamarin, California condor, and black-footed ferret, have released populations that are to this day doing well (Kierulff et al., 2012; Mee & Hall, 2007; Miller et al., 1994). All three programs used the experience of multiple organizations to assess initial problems, modify protocols to remove those problems, and create post-release monitoring plans to effectively track the released population's growth. These cases show the potential for collaborations between wildlife organizations and zoological societies; both bringing their own knowledge and connections to the table. All three of these areas indicate a significant amount of influence captive facilities have on the processes involved in reintroductions. Ideally in the future, reintroduction biology will become an inter-organizational effort to use a broad range of knowledge and practices to maximize the outcomes of reintroduction programs.

Concluding remarks

As shown by the numerous complications encountered throughout the process of this study, there is a need for the field of reintroduction biology to take a step back and resolve the problems disabling the ability to effectively assess past and current reintroduction programs. Without standardized criteria the credibility of assessments is reduced, offering little to work on when developing future guidelines and protocols. For this reason, I offer my own recommendations to the field.

To define what a successful reintroduction program is entails more than a simple, all-encompassing one-liner. Because of the vast differences in life history traits among taxa, the "definition" of success should instead be a set of criteria from which success is deduced; taking into account key characteristics of the species in question. For instance, the reproductive rate may be more important to success than overall survival rates for species with shorter lifespans and overall high fecundity rates (Kleiman et al., 1991). A common goal of reintroduction is to establish a population that will eventually require little to no intervention (Seddon, 1999). This end result should be the focus of determining criteria for success; however this goal is attained at different rates by different species. Accordingly, the time line of the program should be considered. The outcome of a program after two years can be drastically different than if it were evaluated following ten years, especially when comparing between taxa and species with varying generation times. Accordingly, short-term and long-term criteria must be implemented so as to understand the status of the program over time.

In order for the designation of success to be meaningful, the protocols under which the programs were implemented must also be somewhat standardized. To reiterate from before, certain organizations such as the United States Fish and Wildlife Service and the IUCN have created guidelines and protocols attempting to make the procedure more mainstream, however they are not extensive or used outside certain regions of the world (Martin, 1994; IUCN, 1998). Although I have expressed how critical the differences among life histories are among taxa, the process can still account for these differences while remaining normalized. In addition to its normalization, release criteria must also focus on acquiring expansive data on the status of individuals throughout their lifetime after a release. For some programs this would require prolonged monitoring; a task not easily done without proper funding. The outcome of future programs depends on our ability to properly assess current and past programs, so this goal must be prioritized.

There has been tremendous progress in the field of reintroduction biology since its start in the 1970s. Nonetheless, looking at RSG among other publications (Germano & Bishop, 2009; Griffith et al., 1989; Wolf et al., 1996, 1998), we find no consistency in defining success or providing needed data and criteria. Considering the Re-introduction Specialist Group is viewed as the authority on international reintroduction programs, the modifications needed to improve should start at their level. I argue that the RSG need re-organization or better criteria for their assessments of programs. This group is highly necessary for reintroduction science, but its progress to date has not met its potential. In terms of assessments, this study would have been much more informative if unsuccessful programs were available to the comparison. Without those, not much information was able to be extracted from the limited number of successful programs that were compared. For this reason, it can be hypothesized that life history traits likely play a significant role in the outcome of reintroductions; however this hypothesis cannot be elaborately studied without changes to the problems discussed.

Appendices

Appendix I: Reintroduction Programs Used in this Study

Species	Num. of Programs	References
Golden Lion Tamarin *Leontopithecus rosalia*	1	Baker et al., 2002; Kierulff et al., 2012; Kleiman et al., 1991; Menzel & Beck, 2000; Rambaldi et al., 2008; Ruiz-Miranda et al., 2010; Stoinski et al., 2003; Stoinski & Beck, 2004
Arabian Oryx *Oryx leucoryx*	3	Al Jahdami et al., 2011; Al Zaidadeen & Al Hasaseen, 2008; Harding et al., 2007; Mesochina et al., 2003; Spalton et al, 1999; Stanley Price, 1989; Stanley Price, 2012; Strauss, 2008; Wronski et al., 2011; Zafar-ul Islam et al., 2010, 2011
Red Wolf *Canis rufus*	1	Bartel & Rabon, Jr., 2013; Ginsberg & McDonald, 1990; Hedrick & Fredrickson, 2008; Macdonald & Sillero-Zubiri, 2004; Phillips et al., 2003; Venters, 1989; Waddell & Rabon, Jr., 2012
Gray Wolf *Canis lupus*	1	Bangs & Smith, 2008; Bangs et al., 1998; Bangs et al., 2001; Bright & Manfredo, 1996; Fritts et al., 1997; Macdonald & Sillero-Zubiri, 2004; Scullery, 2003; Smith & Ferguson, 2006; Zuccotti & Andrew, 1995
Mexican Wolf *Canis lupus baileyi*	2	Brown & Parsons, 2001; Hedrick & Fredrickson, 2008; Parsons, 1998; USFWS, 1996b
Black-footed Ferret *Mustela nigripes*	1	Dobson & Lyles, 2000; Jachowski & Lockhart, 2009; Livieri, 2011; Miller et al., 1994; USFWS, 2013; Marinari, 2012
Przewalski's Horse *Equus ferus przewalskii*	2	Boyd & Bandi, 2002; Boyd & Houpt, 1994; Kaczensky et al., 2007, 2011a, 2011b, 2013; King & Gurnell, 2005; Slotta-Bachmayr et al., 2004; van Dierendonck & Wallis de Vries, 1996; van Dierendonck et al., 1996; Walzeer et al., 2012
California Condor *Gymnogyps californianus*	1	Burnham & Whaley, 2013; Mee & Hall, 2007; The Peregrine Fund, 2013; USFWS, 1996a; Wallace, 2012
Yellow-footed Rock Wallaby *Petrogale x. xanthopus*	1	Andrews et al., 2010; Lapidge, 2001, 2005, 2009; Lapidge & Munn, 2012; Lim et al., 1980, 1992; Sharp, 2009
Sumatran Orangutan *Pongo abelii*	1	Kelle et al., 2013; Kuze et al., 2012; Riedler et al., 2010; Singleton et al., 2004; Trayford et al., 2010; Wich et al., 2004; Wich et al., 2009
Iberian Lynx *Lynx pardinus*	1	Junta de Andalucia, 2009; Lopez-Parra et al., 2012; Palomares et al., 2005; Palomares et al., 2011; Simon et al., 2012, 2013;

		Vargas et al., 2008; Delibes et al., 2000
Whooping Crane *Grus americana*	1	Animal Diversity Web; CWS & USFWS, 2005; Keller & Hartup, 2013; Travsky & Beauvais, 2004; WDNR, 1996
Chimpanzee ***Pan troglodytes***	1	Cawthon Lang, 1996; Farmer et al., 2010; Goossens et al., 2005; Marsden et al, 2006; Thompson, 2013; Tutin et al., 2001
Asian Houbara Bustard *Chlamydotis undulata macqueenii*	2	Combreau & Smith, 1998; Islam et al., 2011; Islam et al., 2013; Jalme et al., 1996; Judas et al., 2006; Seddon et al., 1995; van Heezik et al., 1999
Malherbe's Parakeet *Cyanoramphus malherbi*	2	Ewen et al., 2013; Kearvell et al., 2002; Ortiz-Catedral & Brunton, 2009; Ortiz-Catedral et al., 2009, 2010, 2012
Helmeted Honeyeater Lichenostomus melanops cassidix	1	Franklin et al., 1995; Menkhorst, 2008; Menkhorst & Middleton, 1991; Menkhorst et al., 1999; Mountfort, 1988; Moyser, 1997; Smales et al., 2000
Barn Owl *Tyto alba*	1	Barn Owl Trust, 2010; Green & Ramsden, 2001; Meek et al., 2003; Warburton, 1984
Pygmy Hog *Porcula salvania*	1	Deb, 1995; Meijaard et al., 2011; Narayan, 2004; Narayan et al., 2008, 2009. 2010
Channel Island Fox *Urocyon littoralis*	2	Clifford et al., 2006; Coonan & Guglielmino, 2012; Coonan & Schwemm, 2010; Garcelon et al., 1992; Kohmann et al., 2005; Macdonald & Sillero-Zubiri, 2004

Appendix II: Table of Recommended Characteristics to Monitor/Log

Program Characteristics							
Location	Release Site Area	Fenced or Unfenced	Proximity to Urban Areas	Released Population Size	Total Individuals Released Since Start Date	# Individuals Surviving to Reproduce	Current Size of Population
Social Structure of Release	Carrying Capacity of Area	Total in Captivity	Type of Release	Supplemental Food/Water?	Average Cost per Individual	Founder population size	Largest Threat to Success

Life History Traits						
Lifespan (Captivity; Wild)	Age at First Reproduction (female)	Known Viruses in Captive/ Wild Populations	Average Weaning Period	Average Number of Offspring per Reproductive Event	Territoriality	Estrous Frequency
Breeding System	Average Interbreeding interval	Degree of Diet Specialization	Food Source	Average # of Reproductively Active Pairs per Group	Home Range Size	Social Structure

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