U.S. conservation translocations: Over a century of intended consequences

Ben J. Novak | Ryan Phelan | Michele Weber

Revive & Restore, Sausalito, California

Abstract
Conservation translocations (reintroductions, reinforcements, ecological replacements, and assisted colonization) have played a vital and necessary role in conserving endangered species and ecosystems. Yet concerns over potential unintended ecological consequences frequently hinder the progress of translocation activities. We reviewed the history of U.S. translocations to ask: how often were intended benefits the result versus harmful unintended consequences? We found that translocations played a key role in recovery for 30% (14 of 47) of U.S. delisted taxa. Translocations have been performed, are planned, or are part of continuing recovery actions for 70% (1,112 of 1,580) of listed threatened and endangered taxa. Of the 1,014 total taxa we found with recorded conservation translocations spanning 125 years, we found only one restricted instance that caused a loss of biodiversity. All other reports of negative consequences were caused by translocations performed for economic and cultural interests in the absence of conservation-based governance. Examples included fish stocking for sport and biological control programs for agricultural pests. We included biological control programs in this analysis because they can be and often are used as conservation tools, to directly benefit ecosystems. In addition, they are often raised as examples of harmful unintended results during the conservation planning process. However, only 1.4% (42) of 3,014 biological control agents released globally have caused ecosystem-level deleterious impacts. All of these were initially released before the 1980s and conservation-based practice and governance in recent decades have reduced off-target impacts from biological control practice. Two themes emerged from our review: (a) conservation translocations routinely yielded their intended benefits without producing unintended harm, and (b) when ecological damage did occur, it was in the absence of conservation practice and regulation. This evidence shows that well-planned translocation efforts produce ecosystem benefits, which should be weighed against the costs of inaction when deliberating conservation strategies.
1 | INTRODUCTION

The practice of translocation involves moving individuals from a source population to a target population or habitat. Conservation translocations, specifically, are the intentional and planned movement of organisms for conservation purposes (e.g., to benefit the survival of populations, species, and the restoration, persistence, or enhancement of biodiversity, ecological processes, and habitat). Conservation translocation efforts in the U.S. began at least as early as 1895; the first documented example we found in our review of the literature was Edward Avery McIlhenny’s successful translocation of snowy egrets (Egretta thula) to Avery Island, Louisiana. The Avery Island population was the source for McIlhenny’s later translocation efforts in Florida, credited heavily for the species restoration (Seddon & Armstrong, 2016).

Various conservation translocation practices fall under three primary categories: (a) reintroductions, (b) reinforcements, and (c) conservation introductions (in some cases introductions and reintroductions are arguably interchangeable given debates over “indigenous range”, see Box 1 A and B). Although the release of biological control agents to manage invasive or disruptive species is not included under current conservation translocation guidelines and definitions (IUCN/SSC, 2013), many biological control programs for conservation purposes have been performed successfully (Fowler, 2004; Heimpel & Cock, 2018; Van Driesche et al., 2010). The release of biological control agents for conservation purposes meets the IUCN/SSC definition of a conservation translocation in that: (a) “the primary objective is a conservation benefit,” and (b) “the beneficiaries should be the populations of the translocated species or the ecosystem that [the translocated population] occupies” (IUCN/SSC, 2013). For this reason, we considered biological control strategies an effective tool of conservation practice and positioned them within the conservation translocation hierarchy to emphasize that they can be used to accomplish conservation goals (Figure 1).

Conservationists recognize that translocations are increasingly necessary to save species and ecosystems (Seddon & Armstrong, 2016; Swan, Lloyd, & Moehrenschlager, 2018; Swan, McPherson, Seddon, & Moehrenschlager, 2016), especially as threats to biodiversity intensify due to rapid anthropogenic climate change, habitat degradation, and the spread of invasive species and diseases along transportation routes in an increasingly interconnected global economy. Yet, proposals for new translocations are often met with opposition from stakeholders as well as conservation biologists (Aldred, 2016), whose concerns influence regulators, stall decisions, and adversely affect the timeliness of translocations. Although translocations are widespread, having been performed in all 50 U.S. states (Brichieri-Colombi & Moehrenschlager, 2016), popular opinion opposes most ecological interventions because of a fear of unintended consequences (Müller & Eriksson, 2013; Nogués-Bravo, Simberloff, Rahbek, & Sanders, 2016; Ricciardi & Simberloff, 2009; Simberloff, 2012).

Translocations may result in a variety of intended and unintended consequences. Translocations alter the bio-abundance and biodiversity of ecosystems as well as the genetic diversity of source and target populations (Furlan et al., 2020; Box 1 C). These alterations can be short-term or systemically minor to systemically transformative. While intended consequences are designed to have positive outcomes, unintended consequences can be complex and impact ecosystems along a continuum from harmful to benign to beneficial (Box 2). Common perception conflates “unintended outcomes” with “negative outcomes.” Combining all translocations, regardless of motivation, has obscured analyses of conservation translocations. Translocations performed for economic gains under lax environmental regulations and protocols have been erroneously treated as translocations performed for conservation. For example, Ricciardi and Simberloff (2009) presented unintended consequences for native island birds in the case of the government-sanctioned release of non-native red squirrels (Tamiasciurus hudsonicus) to Newfoundland Island. However, careful examination of this case reveals that this was not a conservation translocation. Red squirrels were released in 1963 by unsanctioned individuals for their aesthetic appeal and as a source of game for fur trappers (Goudie, 1978). After the initial introduction, the government endorsed releases the following year, hoping that pine marten populations would learn to predate the red squirrels and benefit from an increase in prey as a byproduct of the economically motivated introductions (Goudie, 1978; Mahoney, 2004).

We reviewed the history of U.S. translocations. We compared conservation and non-conservation translocations and examined the prevalence of unintended negative consequences. To do so, we first comprehensively reviewed translocations of U.S. endangered taxa to
A. Restoration and Range Expansion

Approximately 10 million wild turkeys (*Meleagris gallopavo*) ranged throughout 39 states in the 15th century (orange dashed line). Due to unregulated harvesting and habitat degradation, fewer than 200,000 survived in fragmented populations over 21 states by the 1930s (dark green). Reintroductions (light green) as well as introductions (brown) to new localities both played a role in recovering the species to its present-day numbers, which exceed six million, with populations now thriving within all 48 contiguous states as a result of this conservation effort (McRoberts, Wallace, & Eaton, 2014; National Wild Turkey Foundation, 2017).

B. Shifting baselines

The lines between some reintroductions and introductions begin to blur when different baselines are used. The majority of translocations categorized as reintroductions have occurred only years or decades after a population’s extirpation in a given region, leading to little contention. However, when a taxon has been absent within a region for centuries or longer, the chosen baseline for “indigenousness” can be very important for regulatory approvals to perform translocations (Morrison, 2014). It is commonly accepted from a regulatory standpoint that any species present in a region when Europeans began colonizing the Americas in the 15th century is indigenous. Nevertheless, there are instances of conservation introductions that qualify as reintroductions when the baseline of “former range” is shifted. Today, thanks to conservation introductions, California condors (*Gymnogyps californianus*) live and breed in Arizona, where condors have not lived residentially for ~10,000 years (Finkelstein, Snyder, & Schmitt, 2015). Wild Turkeys are considered introduced in most western states; however, Pleistocene records exist in Idaho and California (McRoberts et al., 2014).

C. Altered population structure

Distances between capture and release of individuals for reintroductions and reinforcements are typically tens to hundreds of kilometers, sometimes less than 1 km for taxa with low dispersal rates and restricted home ranges. However, there have been reintroductions and reinforcements of immense undertaking, transporting individuals >1,000 km between source populations and release sites. Bighorn sheep (*Ovis canadensis*) is one species which has been subject to extensive long-distance translocations. Pneumonia (*Mycoplasma ovipneumoniae*) outbreaks, introduced by domestic sheep (*Ovis aries*), led to major declines and local extinctions of bighorn populations starting in the late 1800s (Cassirer et al., 2018). This prompted reintroductions and reinforcements to conserve the species—activities that continue today to combat pneumonia outbreaks. Over 15,000 bighorn sheep have been translocated since the 1920s throughout hundreds of sites in western North America (Wild Sheep Working Group, 2015). For bighorn sheep, these translocations have extended throughout the species’ entire indigenous range, crossing ecotypic and subspecies boundaries. The population genetics of bighorn sheep are undoubtedly significantly different today from that of populations before the 1920s. The ecotypic plasticity of bighorn sheep has resulted in high rates of successful post-release establishment.
**Conservation Translocation**

The intentional, planned, and authorized movement and release of a living organism where the primary objective is a conservation benefit.

**Synonomous terms:** transplantation, relocation, stocking

### Reintroduction

The release of organisms to an environment within the former indigenous range from which the population has become extirpated.

**Synonomous terms:** outplanting, re-establishment

### Reinforcement

The intentional movement and release of an organism into an existing population of conspecifics.

**Synonomous terms:** augmentation, supplementation, restocking

Includes: Genetic Rescue (Ingvarsson 2001), Targeted-Gene Flow (Kelly and Philips 2016), Assisted Population Migration (Stanturf et al. 2014)

### Conservation Introduction

The release of organisms outside their indigenous range.

**Synonomous terms:** benign introduction

Includes: Ecological Replacement and Assisted Colonization

**Classical Biological Control**

The release of non-indigenous predators, parasitoids, or pathogens co-evolved with a target of invasive species for permanent suppression or complete eradication (Eilenberg, Hajek, and Lomer 2001).

| Intended Consequences | Unintended Consequences |
|-----------------------|-------------------------|
| Establishment of translocated individuals | Expanding population range/size |
| Preventing extinction of populations, subspecies, and species | Establishing refuge populations |
| Restoring, enhancement, or balancing of ecological processes | |
| Attaining conservation independence/low maintenance management states for populations by enhancing population viability | |
| Increasing total population size or proportion of specific demographic groups (genotypes, ecotypes, sex, various life stages) | |
| Reversing population decline | |
| Increasing genetic diversity or facilitating gene flow to improve fitness | |
| Introducing adaptive variability | |
| Suppressing irruptions | |
| Eradicating/suppressing invasives | |
| Failure of individuals to establish (persist and breed/interbreed) | |
| Post-establishment dispersal | |
| Off-target interactions | |
| Overpopulation | |
| Disease transmission | |
| Population becomes invasive | |
| Conservation reliance (a state characterized by the need for perpetual follow-up translocations or other interventions) | |
| Outbreeding depression | |
| Genetic swamping | |

**Figure 1**  Translocation hierarchy, definitions, and intended/unintended consequences. Translocation definitions are taken from IUCN/SSC (2013) and Seddon, Strauss, and Innes (2012) unless otherwise referenced. Currently, the IUCN does not recognize conventional biological control releases as conservation translocations. We proposed the inclusion of biological controls as a conservation translocation, and placed them as a subcategory of conservation introduction, alongside ecological replacements and assisted colonization. Potential intended and unintended consequences of translocations are overlaid according to the type(s) of translocations from which they may result. Intended consequences are ordered at the top in green. Consequences color-coded in orange are unintended outcomes that can range from negative to benign to beneficial. Post-establishment dispersal can also be an intended consequence. Outcomes color-coded red (at bottom) are negative unintended consequences that can lead to ecological damage; all of these consequences can be prevented, mitigated, or reversed.
Positive unintended consequences
According to our data, interventions demonstrate that conservation practitioners effectively reach their goals, but in a few cases, outcomes strengthened ecological interactions in unanticipated ways. For instance, the target goal of biological control agents is the complete control of invasive species, a goal that has been achieved many times. In a review of 70 cases of classical biological control for the conservation of natural ecosystems, complete control was found for 26 programs, while partial control still generated ecological benefits for an additional 40 programs (Van Dreische, 2010). But unintended off-target interactions of biological controls can also generate conservation gains.

The Eurasian hawk moth species *Hyles euphorbiae* has been introduced at multiple sites throughout the western U.S. to control the invasive noxious weed leafy spurge (*Euphorbia esula*). After establishing at release sites in Montana and Canada, *H. euphorbiae* spread to eastern North Dakota, reaching the Sheyenne National Grasslands, home to the only remaining populations of the endangered western prairie fringed orchids (*Platanthera praeclara*) in the state. The western prairie fringed orchid has an obligatory mutualistic pollinator relationship co-evolved with indigenous hawk moths, but despite the Eurasian origins of *H. euphorbiae*, it has become the primary pollinator of western prairie fringed orchids in North Dakota (Fox et al., 2013). North Dakotan orchid populations are presently stable owing to the pollination of *H. euphorbiae*.

Evaluate the extent to which translocations are used for conservation efforts and identified cases of negative unintended outcomes. We then reviewed U.S. translocations with known negative ecological consequences, independent of original motivations. While our focus was on conservation translocations, many translocations of species were performed for other reasons including cultural and economic purposes to create hunting or fishing opportunities, to serve as biological control agents of agricultural pests, or to provide specific ecosystem services (e.g., soil stabilization, aesthetic beauty, and so on). Non-conservation translocations were a large category with mixed-motivations and mixed-results that included well-documented introductions that became invasive species, such as cane toads and mongooses. These mistakes, which produced well-documented unintended consequences, are often cited during debates over proposed conservation translocation initiatives. When non-conservation introductions are misconstrued as conservation efforts, such as the case of Newfoundland’s red squirrels, their negative consequences distort the clear record of U.S. conservation translocation successes.

2 | Assessing Conservation Translocations of U.S. Threatened and Endangered Species

Though conservation translocations are not restricted to endangered taxa, we chose to focus on endangered taxa because U.S. regulations include recovery plans for 80% of listed taxa. Recovery plans included past, proposed, and in-progress actions as well as annual updates. These documents allowed us to identify the number of species for which past translocations have been performed. We also estimated the number of species for which future translocations will be necessary for recovery. We obtained the list of 2,416 endangered species, subspecies, and distinct population segments (referred to generally as taxa herein) from the United States Fish and Wildlife Service (USFWS) Environmental Conservation Online System (ECOS) database. We restricted this assessment to the 1,668 taxa residing within U.S. states and territories. We added the 47 taxa designated as delisted due to recovery on USFWS ECOS to this list and subtracted 88 currently listed taxa that are extinct or likely extinct (Greenwald, Suckling, Hartl, & Mehrhoff, 2019). We evaluated the resulting 1,627 taxa.

We searched the USFWS ECOS Recovery Plan Ad Hoc Report database for the following terms: translocated, translocate, translocation, transplanted, transplant, transplantation, outplanted, outplant, reintroduced, reintroduce, reintroduction, reinforced, reinforce, reinforcement, augmented, augment, augmentation, assisted colonization, assisted migration. We chose these queries to capture both records of translocation attempts and future translocations recommended within recovery plans. We cross referenced the remaining taxa for which definitive translocation records were not found in ECOS against the North American Translocation Database (Brichieri-Colombi & Moebrenschlager, 2016), Global Marine Translocation Database (Swan et al., 2016), Avian Reintroduction and Translocation Database (Lincoln Park Zoo, n.d.), and IUCN reintroduction case studies (Soorae, 2008, 2010, 2011, 2013, 2016, 2018).

We searched peer-reviewed and grey-literature for each listed taxon without database records. We recorded...
documentation of translocation efforts in peer-reviewed papers, federal registry documents, endangered species act bulletins, state agency level recovery documents, non-governmental organization (NGO) plans/reports, newspaper articles, and federal/state agency and NGO web pages. We also obtained unpublished translocation records directly from the directors of Hawaii’s Plant Extinction Prevention Program and Snail Extinction Prevention Programs. All sources for our records are listed in Supporting Information.

For each of the 1,627 taxa, translocation progress was coded as such:

1. Translocation performed: We restricted records of past translocations to the release of individuals for which establishing a population or integrating new individuals into an existing population was the explicit goal. These included both successful and attempted translocations, but excluded experimental translocations performed strictly for research purposes.

2. Translocation planned: We considered several types of records as indicative of explicit or implicit future translocation activities. This included translocation activities stated to be in active planning or translocation activities at proposal stages. We also included taxa in this category if translocations were listed as a necessary requirement for downlisting/delisting in recovery plans, regardless of current stages of translocation planning. We also included experimental releases in which the intention was not to establish a permanent population but was to collect data critical for subsequent translocation efforts in this category.

3. Controlled propagation: The end-goal of controlled propagation (ex situ breeding) is to release individuals to the wild as needed to maintain viable populations or re-establish populations at such times that it is safe to do so (Wilson & Price, 1994). Therefore, for taxa in which no records for performed or planned translocations were found, we considered controlled propagation programs to indicate that future translocations can reasonably be anticipated. We obtained records of controlled propagations directly from the USFWS internal database.

4. No records of translocation found: This label in our dataset does not mean that no translocations have ever been performed, but simply that we could not find documentation of past or planned translocations in our review, despite our exhaustive efforts.

2.1 | Results: Translocations are integral to the recovery of threatened and endangered species

Translocations have played and will play a vital and necessary role in conserving 70% of U.S. endangered species (Figure 2a). Translocations have been performed for at least 42% (660) of the 1,580 currently listed taxa. For 16% (261) of the listed taxa, translocations: (a) are planned, (b) are in preparation, (c) have been proposed, or (d) are necessary for future downlisting/delisting. We identified an additional 12% (191) of taxa currently held and bred in controlled propagation facilities. Although we did not find evidence of translocation records or proposals for these taxa, active captive breeding programs indicate that future translocations will be part of recovery efforts. Of the 47 delisted taxa that have recovered, translocations were instrumental to the recovery of 30% (14) (Figure 2b). The list of results by species/subspecies/distinct population segment can be found in Table S1.

2.2 | Beyond endangered species

Translocations of nonlisted taxa are often required for ecological restoration or to prevent taxa from becoming endangered. Translocation efforts saved many taxa before the
Endangered Species Act (ESA) was ratified in 1973, precluding the listing and delisting of those taxa (such as wild turkey, Gallapavo meleagris, Box 1(a)). An additional 340 nonlisted indigenous North American and U.S. territory species/subspecies—for which reintroductions, reinforcements, and ecological replacements were performed—were listed or referenced in the databases and literature we reviewed when researching ESA listed taxa. Including the 660 listed and 14 recovered taxa, a total of 1,014 documented species and subspecies were subjected to conservation translocations in North America. All nonlisted taxa are catalogued in Table S2. The total number of indigenous translocated taxa in North America is likely much higher, particularly for nonlisted plant taxa, which are planted by the thousands during habitat restoration efforts. A complete catalog of U.S. taxa subject to conservation translocations would require extensive research into ecological restoration literature.

3 | INTENDED AND UNINTENDED CONSEQUENCES OF CONSERVATION TRANSLOCATIONS

We defined “consequences” as secondary net changes in biodiversity, bioabundance, or ecological function caused by a translocation event. Intended consequences were scored when either primary or secondary effects of the intervention achieved the intended result (e.g., successful establishment resulting from reintroductions and introductions or integration, positive population growth stemming from reinforcements, or desirable ecosystem engineering and trophic cascades). Any impacts that were not originally planned, anticipated, or within the parameters desired by the practitioners that performed the translocation were considered unintended consequences. Unintended consequences were considered negative, or damaging, when population growth, expansion, or off-target impacts of the translocated taxon induced undesirable net losses of biodiversity, bioabundance or ecological function within ecosystems. When these impacts were the result of a conservation introduction the introduced taxon can be qualified as “invasive”, a term that is commonly restricted to ecologically damaging non-indigenous taxa (CBD, 2020; Clinton, 1999; Global Invasive Species Database, 2020b; NAISA, 2003). We also searched for reported impacts to source populations and their ecosystems, as the removal of individuals from source populations could have negative impacts resulting from the reduction of population size and genetic variability (Furlan et al., 2020).

We used these defining criteria to evaluate translocation outcomes documented in both peer-reviewed and grey literature. We restricted our review of non-conservation introductions, which were numerous, to designated invasive species. We mined the IUCN Invasive Species Specialist Group’s Global Invasive Species Database (GISD) to identify species intentionally released in the U.S. We determined how many were authorized for release by governing agencies and for what purposes.

3.1 | Conservation translocation practice has overwhelmingly led to beneficial intended consequences

After reviewing the conservation science literature, grey literature, and the GISD we found a single conservation translocation event that caused damaging unintended consequences according to our criteria. The conservation translocation was an effort to save the endangered watercress darter (Etheostoma nuchale) from extinction. The effort saved E. nuchale but at the expense of another lineage of indigenous darter. In 1988, USFWS translocated 200 watercress darters to Tapawingo Spring to establish a safe-harbor population outside the taxon’s restricted native range. The population thrived and by 2001, the growing population had outcompeted and caused the unintended local extirpation of the indigenous Tapawingo darters (George et al., 2009). At the time of translocation, the indigenous Tapawingo darters were morphologically determined to be E. parvipinne, a taxon common throughout the southeast. The potential replacement of a local, subset population of a widespread taxon of darters was not considered a problematic consequence when weighed against the benefit of saving E. nuchale, especially since the presence of a functional darter ecotype in the spring, regardless of origin, did not cause a net loss of ecological function or bioabundance in the system. The Tapawingo darters were eventually formally described as a divergent taxon, E. phytophilum (Bart Jr. & Taylor, 1999). Like E. nuchale, E. phytophilum was a highly restricted taxon. It only occurred in three Alabama counties; therefore, the extirpation of the Tapawingo population represented a significant net loss. The other E. phytophilum populations have persisted, but various threats at other localities have led to its listing as an endangered species. Although many cryptic taxa remain undescribed, we did not find other examples of population displacements.

Conservation translocations have predominantly achieved intended goals over the past 125 years. For hundreds of the 1,014 taxa for which conservation translocations have been performed, these interventions prevented extinctions and facilitated recovery from population bottlenecks and range contractions, as exemplified by the genetic rescue of the Florida panther (Puma concolor coryi) (Johnson et al., 2010). The goals of restoring and enhancing ecosystem processes, such as the cascading effects of wolf
reintroductions in Yellowstone National Park (Smith & Peterson, 2021), have also been accomplished.

None of the 608 total invasive species established in the U.S. and its territories as reported in the GISD resulted from conservation translocations. The GISD did not attribute the establishment of any of its 869 inventoried global invasive species to conservation translocations. We found the vast majority were unintended—the result of accidental transport, escape from economic propagation, or negligent release (e.g., dumping aquaria waste or disposing of live bait in natural habitats; see Table S3). Only 13% (81) were intentionally or possibly intentionally released to establish populations. The GISD identifies biological control, soil/dune stabilization, fishing, hunting, and “flora and fauna improvement” (landscape aesthetics), as purposes for intentional release; which are economic or cultural incentives. Almost all intentional releases were performed before environmental regulations were enacted or strongly enforced. However, the intentional propagation and spread of harmful invasive plant species, historically established for landscape aesthetics, has continued into recent decades, despite recognition of negative impacts (Reichard & White, 2001; Webster, Jenkins, & Jose, 2006).

3.2 Conservation-based risk assessment is key to avoiding unintended consequences: The example of biological control programs

The most well-known negative and off-target impacts of translocations stem from classical biological controls. Biological control programs of the 1800s and early 1900s resulted in some of the world’s worst invasive populations, such as cane toads (Rhinella marina) in Australia. In the U.S., 697 biological control agents successfully established stable populations (Table S4). However, only 3% (21) resulted in negative unintended consequences (Global Invasive Species Database, 2020a; Lynch & Thomas, 2000; Schwarzländer, Hinz, Winston, & Day, 2018). Only two, the rosy wolf snail (Euglandina rosea) and mosquitofish (Gambusia affinis), caused extinctions within the U.S. while the mongoose (Herpestes javanicus) caused extinctions elsewhere (Figure 3, Miller et al., 1989; Hays & Conant, 2007; Régnier et al., 2009). The global percentage of biological control agents that caused significant off-target impacts to ecosystems was 1.4% (42 out of 3,014, Table S5). The global total was lower because the U.S. was the major pioneer in biological control where lessons were learned. As a pioneer in the discipline, many of the biological control agents resulting in harm, the majority of which were non-insect species, were first released in U.S. states or territories (Table S5). As a result of these problematic early programs, the practice of biological control shifted towards agents with high host-specificity, typically predatory and parasitoid insects, parasitic invertebrates, or pathogens. Of the 697 established biological control agents in the U.S., 97% (678) were insects, other parasitic invertebrates, or pathogens.

Harmful biological control agents were most often released to control agricultural pests. In several cases private enterprises proceeded to release biological controls without authorization by governing agencies (Table S3). For example, the mongoose was released by sugarcane farmers in Hawaii (Hawaiian Invasive Species Council, n.d.). The unintended consequences of early biological control programs became evident in the 1970s and 1980s, prompting a paradigm shift in regulation and practice. By 1990, an environmentally-conscious paradigm replaced the prior era of economic “tunnel vision.” The new paradigm was characterized by better conservation-based pre-release risk assessment and higher specificity between control agent and target. (Cock et al., 2016; Heimpel & Cock, 2018; Van Driesche et al., 2010). No newly initiated biological control programs in the U.S. have resulted in ecosystem-level damage since 1965 (Figure 3). Biological controls have since become increasingly used in U.S. conservation. As of 2010, completed and ongoing programs targeting 40 invasive species of conservation concern were completed or underway (Van Driesche et al., 2010).

Pre-release testing and increased host-specificity of biological control agents in recent decades resulted in fewer off-target impacts (Cock et al., 2016). Modern programs deploy predatory arthropods, parasitoids, and pathogens as biological control agents; gone are the days when vertebrates and risky predatory invertebrates were authorized for biological control purposes in the U.S. Unfortunately, this paradigm shift has not been adopted globally. The rosy wolf snail was introduced as a biological control to Samoa as recently as 1992 (Global Invasive Species Database, 2020a), decades after it proved devastating to terrestrial mollusk biodiversity on other islands.

3.3 Potential negative consequences: Overpopulation, outbreeding depression, genetic swamping, and disease transmission

Several concerns have been raised as potential risks when translocating wildlife. These included overpopulation, outbreeding depression, genetic swamping, and disease transmission. To date in the U.S., these outcomes have been
extremely rare and manageable. In no case have conservation translocations caused significant harmful consequences to whole ecosystems. Despite their rarity in past translocations, such potential outcomes should still be considered; as for some taxa and habitats these outcomes pose probable risks. However, potential negative consequences should be weighted in accordance with expected likelihoods to produce accurate risk assessments and design appropriate mitigation strategies. There is no reason that these outcomes should ever pose serious negative consequences, as they can be anticipated and mitigated.

Overpopulation is a potential risk for translocated taxa, particularly when predators are absent. However, we found no records of conservation translocations resulting in overpopulation in the U.S. In a few foreign examples, reintroduced populations reached ecologically concerning numbers within fenced reserves (Druce, Mackey, & Slotow, 2011; Moseby, Lollback, & Lynch, 2018); however, mitigation strategies were successfully deployed, preventing damaging consequences (Butler, Paton, & Moseby, 2019; Druce et al., 2011; West, Tilley, & Moseby, 2019). The constraints of fenced reserves do not apply to expansive and connected open habitats, rendering overpopulation a low-risk outcome when individuals have the ability to expand their range or emigrate to other populations.

For the overwhelming majority of translocations designed to reverse inbreeding depression, translocations have been successful and have increased genetic diversity and population growth (Frankham, 2015; Frankham et al., 2017). While outbreeding depression has been observed in newly established populations and reinforced populations (Huff, Miller, Chizinski, & Vondracek, 2011; Marshall & Spalton, 2000), we found no instances in which prolonged impacts caused the extirpation of the population, nor declines below ecologically functional population sizes. In general, outbreeding depression is likely only of significant concern for self-fertilizing populations, and not sexually reproducing species (Edmands, 2007).
Genetic swamping is a potential negative consequence specific to reinforcements. We found no reports of genetic swamping adversely impacting reinforced populations, though it was only explicitly monitored in a few cases (Bouzat et al., 2009; Hogg, Forbes, Steele, & Luikart, 2006; Jahner et al., 2019; Olson, Whittaker, & Rhodes, 2012; Zimmerman, Aldridge, Apa, & Oyler-McCance, 2019). It is general practice to perform reinforcements of small numbers to avoid swamping. Observations from natural immigration events have shown that even a single immigrant can rescue a population (Vilà et al., 2003). Genetic swamping is likely only when large numbers of individuals are released into a target population, as was the case of restoration efforts for the threatened Kern River golden trout (Oncorhynchus mykiss whitei). Captive reared trout swamped wild trout, leading to populations with low genetic diversity. This swamping was not irreversible because the five genetically discrete translocated populations could be managed to facilitate gene flow and increase diversity within each population (Lasardi, Stephens, Moyle, McGuire, & Hull, 2015).

Fortunately, although pre-release disease screening is neither common practice nor enforced, there have been few critical disease outbreaks. Several cases of disease transmission associated with translocations have been observed (Chipman, Slate, Ruprecht, & Mendoza, 2008; Edwards, 2014; Kock, Woodford, & Rossiter, 2010), but have not resulted in significant consequences. As long as disease is rare in a population, disease-free individuals are more likely to be selected for release than infected individuals. Introduced diseases are a leading cause of biodiversity loss, exemplified by the accidental introduction of mosquitos to Hawaii, resulting in dozens of avian extinctions due to mosquito-borne diseases (Warner, 1968). Given the extreme impact that introduced diseases can have within ecological communities, disease transmission should be the most serious risk considered and mitigated when planning translocations. Disease screening and/or quarantine periods for translocated individuals should be universally implemented.

4 | IMPROVING UPON THE PAST

4.1 | Historic challenges

4.1.1 | Knowledge gaps and insufficient funding underlie low efficiency rates

The biggest challenge in translocation practice is the low rate of establishment and persistence. Our review suggested that >50% of translocation releases failed to establish; multiple releases were typically performed before success was achieved. Researchers broadly agreed that these failure rates were due to a lack of baseline biological and ecological knowledge and a lack of appropriate post-release monitoring (Bubac, Johnson, Fox, & Cullingham, 2019; Campbell et al., 2002; Seddon & Armstrong, 2016). The crucial components to designing successful programs were (a) eliminating the original factors that caused populations to decline, (b) ensuring appropriate habitat requirements, and (c) building the sociopolitical and cultural support needed for recovery.

Translocations are planned to achieve intended outcomes, but they have been rarely funded or equipped to adequately monitor the progress and occurrence of those outcomes. Without ongoing monitoring, it was not possible to adjust variables which could be important to both avoid unintended consequences and improve future translocation strategies (Campbell et al., 2002). The success rate of conservation programs has been directly correlated to funding (Evans et al., 2016; Luther & Gentry, 2019; Luther, Skelton, Fernandez, & Walters, 2016). These correlations provided evidence that knowledge gaps and insufficient monitoring were due to funding constraints, rather than problematic conservation practice.

4.1.2 | Existing knowledge is difficult to access

The field has not yet analyzed comprehensive global patterns in translocations. The largest impediment to extrapolating useful information from existing data is the lack of continually updated and accessible databases. Researchers must repeatedly review the incomplete, and sometimes inaccessible, peer-review and gray literature. Lack of an open-source database has hindered efforts to pull together the valuable insights from various analyses (Berger-Tal, Blumstein, & Swaisgood, 2020; Brichieri-Colombi, Lloyd, McPherson, & Swaisgood, 2019; Brichieri-Colombi & Moehrensclager, 2016; Campbell et al., 2002; Fischer & Lindenmayer, 2000; Luther & Gentry, 2019; Swan et al., 2016, 2018; Taylor et al., 2017; Thévenin, Mouchet, Robert, Kerbiriou, & Sarrazin, 2018) limiting broad meta-analyses despite a growing knowledge base.

4.1.3 | Conservation-based risk assessment is still lacking for some translocation activities

Two trends for U.S. translocations were clear: (a) harmful impacts resulting from conservation translocations were virtually non-existent, and (b) harmful impacts resulting from intentional non-conservation introductions were rare. Although rare, the impacts of problematic non-conservation intended introductions can be severe and
have resulted in irreversible evolutionary biodiversity losses. Translocation regulation and planning/implanta-
tion strategies have been revised over time in response to
damaging outcomes. Unintended consequences have
drastically declined in recent decades as a result. The
majority of historic translocations that did result in nega-
tive consequences were cases where impacts to natural
ecosystems were not considered prior to release.

Stronger regulation is needed to protect ecosystems
and ensure intended consequences for projects in which
the primary motivation is human interest (e.g., control-
loring agricultural pests and soil erosion for human land
use, generating hunting and fishing resources, and
altering landscape aesthetics). While the regulation of
biological control programs has become environmentally
conscious in the U.S., the translocation of non-
indigenous and invasive plants for ornamental and land
management use is poorly regulated, leaving opportuni-
ties open for potentially devastating plant invasions
(Reichard & White, 2001).

Fisheries management remains motivated by human
economic incentives. While there have been shifts to pri-
oritize indigenous fish taxa over non-indigenous taxa,
this has been driven by changes in sport-fishing values
(Pister, 2001). Decisions to protect or stock indigenous
fishes over non-indigenous fishes have not been univer-
sally concerned with whether non-indigenous fishes are
ecologically harmful, benign, or beneficial. The National
Park Service (NPS) banned fish stocking in the Northern
Cascades National Park Complex in 2009, after 10 years
of research demonstrated that stocking fish in historically
fishless lakes negatively impacted indigenous taxa. In
2014, driven by economic incentives, Congress passed a
bill reversing the NPS ban (Patterson, 2015). This reversal
showcased another problem inherent to environmental
management decisions: the agencies charged with con-
serving ecosystems do not have final authority.

4.2 | Opportunities and recommendations

Basic ecological research and post-release monitoring are
two factors that will significantly improve translocation
outcomes, by increasing rates of successful establishment
and ensuring long-term intended consequences. The suc-
cesses of intended consequences stemming from agricul-
tural biological control programs have been the result of
rigorous careful planning, pre-release experimental risk
assessments, and comprehensive post-release monitoring.
Of 3,014 biological control agents released globally, 1.4%
(42) have caused ecosystem-level negative consequences.
The majority of impacts have been manageable or revers-
ible and only 0.17% (5) contributed to global extinctions.

Though only recently adopted as a conservation man-
agement strategy, biological control agents have already
achieved intended outcomes, reducing the impacts of
invasive species and even saving species from imminent
extinction (Fowler, 2004; Van Driesche et al., 2010). We
recommend that conservation agencies consider biologi-
cal control as a powerful early response for invasive spe-
cies management. We also urge the inclusion of
biological controls in the IUCN’s reintroduction and
translocation guidelines, as their use for conservation
aligns with the overarching definition and applications of
conservation translocations.

There are limitations with extending the biological
control model to other translocation efforts. It is difficult
to conduct experimental pre-release risk studies for many
taxa, increasing the burden of generating accurate model
simulations. It is also unlikely that government agencies,
which oversee the majority of translocation efforts, can
fund both basic ecological research and comprehensive
post-release monitoring. To overcome these limitations,
we can: (a) innovate low-cost, high-yield application-
oriented research methods, and (b) optimize interdisci-
plinary research partnerships between specialized scien-
tists and conservationists working in academia,
government agencies, NGOs, and for-profit research
institutions.

4.2.1 | Enhancing translocation practice
with innovative tools and techniques

Tools and techniques that will enhance translocations
are being tested in conservation by early adopters.
With adequate ecological knowledge, computational
modeling methods, such as population viability ana-
lyses and species distribution models, can provide pre-
dictive datasets for translocation planning. Forecasting
is valuable because it can inform proactive conserva-
tion actions for reinforcements or assisted colonization
and climate change management (Ferrarini et al.,
2016; Hardy, Hull, & Zuckerberg, 2018; Molloy,
Burbidge, Comer, & Davis, 2020; Mozelewski and
Scheller, 2021). Other tools, including remote monitor-
ing technologies and genetic insights, can provide data
to strengthen predictive models. More sophisticated
models will better inform translocation planning and
enable conservationists to achieve historically
unattainable goals.

Drones (Hodgson et al., 2018), GPS transmitters
(Fischer et al., 2018), acoustic-monitoring (Yan et al.,
2019), and environmental DNA (eDNA) sequencing
(Hunter, Hoban, Bruford, Segelbacher, & Bernatchez,
2018), are among the many new technologies
augmenting remote, non-invasive monitoring. Genetic
insights from eDNA, paleogenomics (ancient DNA), population genomics, and functional genomics, offer robust data to enhance translocation planning and implementation ranging from identifying suitable source populations to opportunities to facilitate adaptation to threats such as introduced disease (Box 3).

**BOX 3  Genetic insight can enhance translocation success**

A premier example of applied genetic insight contributing to translocation success is the case of sturgeon restoration to the Baltic Sea region. Across the Atlantic in Europe, overharvesting decimated European sturgeon (*Accipenser sturio*); the Baltic Sea population was extirpated by the 1960s. With too few *A. sturio* left to restore the Baltic region, *A. baerii* and *A. gueldenstaedtii* from eastern Eurasia were introduced, but translocations failed long-term (Purvina & Medne, 2018). Ancient DNA sequencing of historic Baltic Sea sturgeons showed that the extirpated populations were not *A. sturio*, but were *A. oxyhynchus*, previously thought to be restricted to North America (Ludwig et al., 2002, 2008). In light of these data, *A. oxyhynchus* were translocated from Canada to the Baltic Sea, in 2004 (Kolman, Kapusta, Duda, & Wiszniewski, 2011), resulting in successfully established populations. While no reproduction has yet been confirmed (Elvira, Leal, Doadrio, & Almodóvar, 2015), nor expected given the source population’s age of sexual maturation (10–15 years, Balazik, 2012), introduced individuals appear to be thriving (Purvina & Medne, 2018).

Genetic insight may also be used to develop proactive gene flow management and even facilitate adaptation. Population genomics has been key to developing tools that can benefit future mountain lion translocations (Saremi et al., 2019; van de Kerk, Onorato, Hostetler, Bolker, & Oli, 2019). Comparative genomics of functional immune system genes of the ochre sea star (*Pisaster ochraceus*) has identified sea star wasting disease (SSWD) resistance alleles (Schiebelhut, Puritz, & Dawson, 2018). Gene screening for resistant alleles can enable captive breeding programs to selectively breed resistant individuals without incurring genetic diversity losses inherent to phenotypic selective breeding (Hollenbeck & Johnston, 2018); such a strategy can aid in the restoration of the species abundance, and potentially other sea stars species devastated by SSWD, such as the sunflower star (*Pycnopodia helianthoides*).
### 4.2.2 Improving conservation outcomes via interdisciplinary collaboration

Academic conservation research addresses eco-evolutionary questions, although much of this research is not designed or published in ways that yield useful and accessible information applicable to conservation decisions and actions (Habel et al., 2013; Williams, Balmford, & Wilcove, 2020). The formal scientific publication process delays the useful dissemination of data. Also, journals are often reluctant to publish null or negative results. The lessons learned from failures are critical to advancing disciplines. Publishing failed protocols, neutral outcomes, and negative consequences, will prevent conservationists from spending labor, time, and precious funding on methods that have already proven ineffective. We encourage early, open access, application-oriented dissemination of data. These reports should be independent of peer-reviewed publication timelines, in order to facilitate shared insights between conservation efforts as quickly as possible.

There is a need to connect conservation practitioners and conservation scientists with academic specialists. Genomicists and theoretical ecologists could contribute their expertise to translocation efforts, but open access is not sufficient to guarantee the application of relevant results. Often, valuable insights that could inform conservation efforts are published with the passive hope that they will be noticed and used by conservation managers. We recommend that academic researchers actively seek out conservation managers and include them collaboratively to ensure that their research produces relevant data for timely application. These interdisciplinary groups should work together at every stage from program design, data collection, interpretation of results, application of resulting insights, through to interpreting the data of post-translocation monitoring to correlate final outcomes with early stage steps to discern effective versus ineffective components of programs.

### 4.2.3 Consolidating and adjusting regulatory oversight to meet the increased demand for translocations effectively and safely

All translocation programs in the U.S. intending to release organisms on public lands must navigate regulations by multiple state and/or federal agencies. The USFWS, U.S. Geological Survey, National Park Service, Bureau of Land Management, and USDA and their state-level equivalents are among the agencies that conduct and regulate translocations. The complexity of agencies regulating translocations is an issue for both the timeliness of translocations and avoiding unintended consequences. For instance, decisions to reintroduce or not to reintroduce wolves to Colorado were delayed for over two decades due to uncertainty as to which regulatory agency has decisive authority (Tejeda, 2020).

Regulatory procedures vary from state to state, complicating interstate translocation efforts and creating inconsistencies for large-scale coordinated restoration efforts with single species. Nearly identical translocation efforts concerning the same taxon will be regulated differently if performed within different states. For example, gaining approval to translocate elk (*Cervus Canadensis*) has varied from several years to decades depending upon the state within which the translocation was performed (personal communication, Karie Decker, Director of Habitat Stewardship, the Rocky Mountain Elk Foundation). Regulatory consistency would improve efficiency and likelihood of success but could only happen under a shared, unified framework. This will require interagency coordination, mission alignment, as well as a consolidation of authority. It is also imperative that appropriate checks and balances be integrated into the governance of conservation decisions to prevent political or private interests from interfering with environmental decisions made by agencies charged with upholding environmental policies and managing conservation activities.

A fit-for-purpose regulatory framework would be adaptive. It would integrate new evidence and consider precedent in order to expedite and streamline the regulation of low-risk programs. Here again, the history of biological control programs is an illuminating model. The shift in regulation to focus on preventing negative unintended consequences has indeed increased safety but it has accordingly reduced the number of releases. This trend is problematic in light of the increasing pressures of invasive species (Cock et al., 2016; Heimpel & Cock, 2018). Regulation is often influenced by the Precautionary Principle. The Precautionary Principle heavily weights all potential risks and encourages inaction until 100% of potential risks, regardless of likelihood or severity, can be mitigated. This philosophy is meant to protect ecosystems from haphazard intervention and it certainly contributes to careful planning. However, the delays incurred by stringent interpretation of the precautionary principle may pose even greater risks to ecosystems, including the potential for extinction. A call for a regulatory paradigm that weighs benefits against risks has been proposed for biological controls (Heimpel & Cock, 2018). This would be similarly appropriate for conservation translocations, which also have an extremely low risk to benefit ratio. Currently the U.S. does not have its own translocation practice guidelines that could serve to
frame considerations for regulatory revisions. In lieu, the IUCN Guidelines for Reintroductions and Other Conservation Translocations, the Scottish Code for Conservation Translocations (National Species Reintroduction Forum, 2014), and the agreed best practice documents used in New Zealand (e.g., DOC Lizard Technical Advisory Group, 2018; Gummer, 2013) could serve as useful documents for framing U.S. regulations.

5 | THE ABSENCE OF REPORTED NEGATIVE IMPACTS IS REFLECTIVE OF AN ACTUAL LACK OF NEGATIVE IMPACTS WITHIN ECOSYSTEMS

We considered several alternative reasons that could explain a lack of reported negative impacts. Post-translocation monitoring is often insufficient (Bubac et al., 2019). Lack of monitoring could explain these results because projects with limited resources focus on the population viability of the translocated taxon and may neglect to measure population dynamics of directly and indirectly interacting taxa. Alternatively, if the duration of monitoring was too short, delayed negative outcomes would not appear. It can take years, decades, or even centuries for established populations to reach densities or expand ranges to the point of inducing negative impacts (Essl et al., 2011; Gallardo, Clavero, Sánchez, & Vilà, 2015; Kowarik, 1995). Negative impacts occurring during the period of post-release monitoring may not reach measurable thresholds during a project’s duration, precluding observation. Finally, even with intensive long-term monitoring, it can be difficult to attribute negative impacts to translocated populations because of the complexity of ecosystems (Didham, Tylianakis, Hutchinson, Ewers, & Gemmell, 2005), especially those observed long after the initial translocation occurred. The consequences of translocations may be misattributed if new variables complicate post-translocation patterns.

Though insufficient monitoring may plausibly explain the lack of documented negative impacts resulting from conservation translocations, ongoing environmental and wildlife management monitoring activities in the U.S., which are extensive and diverse (Olsen et al., 1999), supported our interpretation that the vast majority of conservation translocations have not caused damage to ecosystems. For example, state and federal wildlife agencies routinely monitor game species (Heffelfinger, Geist, & Wishart, 2013), which are present in almost every mainland ecological community. This monitoring activity compensates, in some regions to great effect, for observation biases of conservation translocation programs. Not a single U.S. game species has gone extinct since the enforcement of regulated management in the early 1930s (Table S6, Musgrave, Flynn-O’Brien, Lambert, Smith, & Marinakis, 1998). This is because population declines were detected early and allowed for timely adjustments in conservation management before populations reached critical levels, such as reduction or cessation of harvest, reinforcement translocations to reverse population decline, or providing full protection under ESA listing. The monitoring of game species serves as a proxy for un-related conservation translocations because game species are nearly ubiquitous in the U.S. Therefore, if translocations were causing negative impacts to off-target populations, the cascading effects would eventually impact game populations. This evidence supported the conclusion that conservationists have thus far avoided unintended consequences, likely as a result of careful planning and implementation. The absence of evidence for negative outcomes, in this case, is indeed evidence of absence.

6 | CONCLUSIONS

Translocations can be performed to improve the resilience and adaptability of ecosystems as well as prevent extinctions and reverse biodiversity losses. If all translocation activities in the U.S. adopted a common set of core principles and implement conservation-conscious planning, practitioners of all translocation disciplines would build trust and confidence with regulators and stakeholders. The basic tenet of conservation translocations, that interventions should be to benefit indigenous biodiversity without causing harm, should be the starting point to reshape non-conservation translocation practices. Conservation translocations, specifically, should work to restore ecological dynamics with indigenous taxa whenever feasible, but should not disregard the benefits that non-indigenous taxa may offer conservation goals. Ultimately, conservation practitioners should select indigenous or non-indigenous taxa that maximize evolutionary and functional biodiversity within ecosystems.

The geographic distributions of released individuals of the 1,014 conservation translocated taxa and 676 beneficial biological control agents in our dataset revealed that translocated organisms have constructively shaped every ecological community in the contiguous U.S. and Hawai’i, as well as many Alaskan ecosystems. Conservation translocations of dozens of taxa are presently in active planning, with hundreds more slated long-term, a trend that appears consistent in other countries (Carter, Foster, & Lock, 2017; Cromarty & Alderson, 2013; Swan et al., 2018; Zimmer, Auld, Cuneo, Offord, & Commander, 2019). Conservationists have avoided damaging unintended
consequences because of careful risk assessment and mitigation planning; however, the risks incurred by delaying beneficial translocations fall on the shoulders of regulators and decision-makers. Prolonged inaction risks significant costs to biodiversity. The widespread benefits and paucity of negative impacts stemming from conservation translocations are a signal to regulators, decision-makers, and stakeholders that conservationists can be entrusted with the safe and timely use of translocations. The history of conservation translocations documented here should encourage support of future proposed interventions.

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The authors have no conflicts of interest.

AUTHOR CONTRIBUTIONS
Ben J. Novak, Michele Weber, and Ryan Phelan equally conceptualized this study. Ben J. Novak conducted literature review, compiled data, created figures, and wrote this manuscript. Michele Weber assisted in literature review and significantly edited each draft of the manuscript. Ryan Phelan edited each draft of the manuscript.

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ORCID
Ben J. Novak https://orcid.org/0000-0003-0699-634X
Michele Weber https://orcid.org/0000-0003-1583-7367

REFERENCES
Aldred, J. (2016). More than 1,000 species have been moved due to human impact. Retrieved from https://www.theguardian.com/environment/2016/apr/20/more-than-1000-species-have-been-moved-due-to-human-impact.

Balazik, M. T. (2012). Life history analysis of James River Atlantic Sturgeon (Acipenser oxyrinchus oxyrinchus) with implications for management and recovery of the species (p. 112). Richmond, VA: Virginia Commonwealth University.

Bart, H. L., Jr., & Taylor, M. S. (1999). Systematic review of subgenus Fusciatulum of Etheostoma with description of a new species from the upper Black Warrior River System, Alabama. Tulane Studies in Zoology and Botany, 31, 23–50.

Berger-Tal, O., Blumstein, D. T., & Swaisgood, R. R. (2020). Conservation translocations: A review of common difficulties and promising directions. Animal Conservation, 23, 121–131.

Bouzat, J. L., Johnson, J. A., Toepfer, J. E., Simpson, S. A., Esker, T. L., & Westemeier, R. L. (2009). Beyond the beneficial effects of translocations as an effective tool for the genetic restoration of isolated populations. Conservation Genetics, 10, 191–201.

Bricieri-Colombi, T. A., Lloyd, N. A., McPherson, J. M., & Moehrensclager, A. (2019). Limited contributions of released animals from zoos to North American conservation translocations. Conservation Biology, 33, 33–39.

Bricieri-Colombi, T. A., & Moehrensclager, A. (2016). Alignment of threat, effort, and perceived success in North American conservation translocations. Conservation Biology, 30, 1159–1172.

Bubac, C. M., Johnson, A. C., Fox, J. A., & Cullingham, C. I. (2019). Conservation translocations and post-release monitoring: Identifying trends in failures, biases, and challenges from around the world. Biological Conservation, 238, 108239.

Butler, K., Paton, D., & Moseby, K. (2019). One-way gates successfully facilitate the movement of burrowing bettongs (Bettongia lesueur) through exclusion fences around reserve. Austral Ecology, 44, 199–208.

Campbell, S. P., Alan Clark, J., Crampton, L. H., Guerry, A. D., Hatch, L. T., Hosseinia, P. R., ... O’Connor, R. J. (2002). An assessment of monitoring efforts in endangered species recovery plans. Ecological Applications, 12, 674–681.

Carter, I., Foster, J., & Lock, L. (2017). The role of animal translocations in conserving British wildlife: An overview of recent work and prospects for the future. EcoHealth, 14, 7–15.

Cassirer, E. F., Manlove, K. R., Almberg, E. S., Kamath, P. L., Cox, M., Wolff, P., ... Besser, T. E. (2018). Pneumonia in big-horn sheep: Risk and resilience. Journal of Wildlife Management, 82, 32–45.

Chipman, R., Slate, D., Rupprecht, C., & Mendoza, M. (2008). Downsides risk of wildlife translocation. Developments in Biologicals, 131, 223–232.

Clinton, W. J. (1999). Executive order 13112. Invasive species. Washington, DC: The White House Retrieved from https://www.federalregister.gov/documents/1999/02/08/99-3184/invasive-species

Cock, M. J. W., Murphy, S. T., Kairo, M. T. K., Thompson, E., Murphy, R. J., & Francis, A. W. (2016). Trends in the classical biological control of insect pests by insects: An update of the BIOCAT database. BioControl, 61, 349–363.

Convention on Biological Diversity. (2020). Invasive alien species. Retrieved from https://www.cbd.int/invasive/.
Cromarty, P. L., & Alderson, S. L. (2013). Translocation statistics (2002–2010), and the revised Department of Conservation translocation process. *Notornis*, 60, 55–62.

Didham, R. K., Tylianakis, J. M., Hutchinson, M. A., Ewers, R. M., & Gemmell, N. J. (2005). Are invasive species the drivers of ecological change? *Trends in Ecology and Evolution*, 20(9), 470–474.

DOC Lizard Technical Advisory Group. (2018). *Guidelines for conservation-related translocations of New Zealand lizards* (p. 26). Wellington, New Zealand: New Zealand Department of Conservation.

Druce, H. C., Mackey, R. L., & Slotow, R. (2011). How immunococontraction can contribute to elephant management in small, enclosed reserves: Munyawana population as a case study. *PLoS ONE*, 6, e27952. https://doi.org/10.1371/journal.pone.0027952

Edmonds, S. (2007). Between a rock and a hard place: Evaluating the relative risks of inbreeding and outbreeding for conservation and management. *Molecular Ecology*, 16, 463–475.

Edwards, M. (2014). A review of management problems arising from reintroductions of large carnivores. *Journal of Young Investigators*, 27, 11–16.

Eilenberg, J., Hajek, A., & Lomer, C. (2001). Suggestions for unifying the terminology in biological control. *BioControl*, 46, 387–400.

Elvira, B., Leal, S., Doadrio, I., & Almodóvar, A. (2015). Current occurrence of the Atlantic sturgeon *Acipenser oxyrinchus* in northern Spain: A new prospect for sturgeon conservation in western Europe. *PLoS One*, 10, 1–14.

Essl, F., Dullinger, S., Rabitsch, W., Hulme, P. E., Hulme, K., Jarošík, V., ..., Pyšek, P. (2011). Socio-economic legacy yields an invasion debt. *Proceedings of the National Academy of Sciences of the United States of America*, 108, 203–207.

Evans, D. M., Che-Castaldo, J. P., Crouse, D., Davis, F. W., Epanchin-Niell, R., Flather, C. H., ..., Williams, B. K. (2016). Species recovery in the United States: Increasing the effectiveness of the endangered species act. *Issues in Ecology*, 20, 1–28.

Ferrarini, A., Selvaggi, A., Abeli, T., Alatalo, J. M., Orsenigo, S., Gentili, R., & Rossi, G. (2016). Planning for assisted colonisation of plants in a warming world. *Scientific Reports*, 6, 6–11.

Finkelfeldt, M. Z. K., Snyder, N. F., & Schmitt, N. J. (2015). California Condor (*Gymnogyps californianus*), version 2.0. In A. E. Poole (Ed.), *The birds of North America*.2, Ithaca, NY: Cornell Lab of Ornithology.

Fischer, J., & Lindenmayer, D. B. (2000). An assessment of the published results of animal relocations. *Biological Conservation*, 96, 1–11.

Fischer, M., Parkins, K., Maizels, K., Sutherland, D. R., Allan, B. M., Coulson, G., & Di Stefano, J. D. (2018). Biotelemetry matches on: A cost-effective GPS device for monitoring terrestrial wildlife. *PLoS One*, 13, 1–15.

Fowler, S. V. (2004). Biological control of an exotic scale, *Orthozia insignis* Browne (Homoptera: Ortheziidae), saves the endemic gumwood tree, *Commidendrum robustum* (Roxb.) DC. (Asteraceae) on the Island of St. Helena. *Biological Control*, 29(3), 367–374.

Fox, K., Vitt, P., Anderson, K., Fauske, G., Travers, S., Vik, D., & Harris, M. O. (2013). Pollination of a threatened orchid by an introduced hawk moth species in the tallgrass prairie of North America. *Biological Conservation*, 167, 316–324.

Frankham, R. (2015). Genetic rescue of small inbred populations: Meta-analysis reveals large and consistent benefits of gene flow. *Molecular Ecology*, 24, 2610–2618.

Frankham, R., Ballou, J. D., Ralls, K., Eldridge, M. D. B., Dudash, M. R., Fenster, C. B., ..., Sunnucks, P. (2017). Genetic management of fragmented animal and plant populations (1st ed., p. 401). Oxford, UK: Oxford University Press.

Furlan, E. M., Gruber, B., Attard, C. R. M., Wager, R. N. E., Kereczy, A., Faulks, L. K., ..., Unmack, P. J. (2020). Assessing the benefits and risks of translocations in depauperate species: A theoretical framework with an empirical validation. *Journal of Applied Ecology*, 00, 1–11.

Gallardo, B., Clavero, M., Sánchez, M. I., & Vilà, M. (2015). Global ecological impacts of invasive species in aquatic ecosystems. *Global Change Biology*, 22(1), 151–163.

George, A. L., Kuhajda, B. R., Williams, J. D., Cantrell, M. A., Rakes, P. L., & Shute, J. R. (2009). Guidelines for propagation and translocation for freshwater fish conservation. *Fisheries*, 34, 529–545.

Global Invasive Species Database. (2020a). http://www.iucngisd.org/gisd/.

Global Invasive Species Database. (2020b). *100 of the world's worst invasive alien species*. Retrieved from http://www.iucngisd.org/gisd/100_worst.php.

Goudie, R. I. (1978). Red squirrels, *Tamiuicuarius Hudsonicus*, in the Salmonier River Valley, Newfoundland. *The Canadian Field Naturalist*, 92(2), 193–194.

Greenwald, N., Suckling, K. F., Hartl, B., & Mehrhoff, L. A. (2019). Extinction and the U.S. endangered species act. *PeerJ*, 2019, 1–9.

Gummer, H. (2013). *Best practice techniques for translocations of burrow-nesting petrels and shearwaters* (p. 97). Wellington, New Zealand: The New Zealand Department of Conservation.

Habel, J. C., Gossner, M. M., Meyer, S. T., Eggermont, H., Lens, L., Dengler, J., & Weisser, W. W. (2013). Mind the gaps when using science to address conservation concerns. *Biodiversity and Conservation*, 22, 2413–2427.

Hardy, M. A., Hull, S. D., & Zuckerberg, B. (2018). Swift action increases the success of population reinforcement for a declining prairie grouse. *Ecology and Evolution*, 8, 1906–1917.

Hawaiian Invasive Species Council. (n.d.) *Mongoose (Herpestes javanicus)*. Hawaii.gov. Retrieved from https://dlnr.hawaii.gov/hisc/info/invasive-species-profiles/mongoose/.

Hays, W. S. T., & Conant, S. (2007). Biology and impacts of Pacific Island invasive species. 1. A worldwide review of effects of the small Indian mongoose, *Herpestes javanicus* (Carnivora: *Herpestidae*). *Pacific Science*, 61, 3–16.

Heffelfinger, J. R., Geist, V., & Wishart, W. (2013). The role of hunting in North American wildlife conservation. *International Journal of Environmental Studies*, 70, 399–413.

Heimpel, G. E., & Cock, M. J. W. (2018). Shifting paradigms in the history of classical biological control. *BioControl*, 63, 27–37.

Hodgson, J. C., Mott, R., Baylis, S. M., Pham, T. T., Wotherspoon, S., Kilpatrick, A. D., ..., Koh, L. P. (2018). Drones count wildlife more accurately and precisely than humans. *Methods in Ecology and Evolution*, 9, 1160–1167.
Hogg, J. T., Forbes, S. H., Steele, B. M., & Luikart, G. (2006). Genetic rescue of an insular population of large mammals. *Proceedings of the Royal Society B: Biological Sciences, 273*, 1491–1499.

Hollenbeck, C. M., & Johnston, I. A. (2018). Genomic tools and selective breeding in molluscs. *Frontiers in Genetics, 9*, 1–15.

Huff, D. D., Miller, L. M., Chizinski, C. J., & Vondracek, B. (2011). Mixed-source reintroductions lead to outbreeding depression in second-generation descendents of a native North American fish. *Molecular Ecology, 20*, 4246–4258.

Hunter, M. E., Hoban, S. M., Bruford, M. W., Segelbacher, G., & Bernatchez, L. (2018). Next-generation conservation genetics and biodiversity monitoring. *Evolutionary Applications, 11*, 1029–1034.

Ingvarsson, P. K. (2001). Restoration of genetic variation lost – The genetic rescue hypothesis. In *Trends in Ecology and Evolution, 16*, 62–63.

IUCN/SSC. (2013). *Guidelines for reintroductions and other conservation translocations. Version 1*. Gland, Switzerland: IUCN Species Survival Commission viii+57 p.

Jahner, J. P., Matocq, M. D., Malaney, J. L., Cox, M., Wolff, P., Gritts, M. A., & Parchman, T. L. (2019). The genetic legacy of 50 years of desert bighorn sheep translocations. *Evolutionary Applications, 12*, 198–213.

Johnson, R., Crafton, R. E., & Upton, H. F. (2017). *Invasive species: Major laws and the role of selected federal agencies* (p. 1–47). Washington, D.C.: Congressional Research Service. https://crsreports.congress.gov/product/details?prodcode=R43258.

Johnson, W. E., Onorato, D. P., Roelke, M. E., Land, E. D., Cunningham, M., Belden, R. C., ... O’Brien, S. J. (2010). Genetic restoration of the Florida Panther. *Science, 329*, 1641–1645.

Kelly, E., & Phillips, B. L. (2016). Targeted gene flow for conservation. *Conservation Biology, 30*, 259–267.

Koch, R. L. (2003). The multicolored Asian lady beetle, *Harmonia axyridis*: A review of its biology, uses in biological control, and off-target impacts. *Journal of Insect Science, 3*(2), 1–16. https://doi.org/10.1093/jis/3.1.32

Kock, R. A., Woodford, M. H., & Rossiter, P. B. (2010). Disease risks associated with the translocation of wildlife. *OEJ Revue Scientifique et Technique, 29*, 329–350.

Kolman, R., Kapusta, A., Duda, A., & Wiszniewski, G. (2011). Review of the current status of the Atlantic sturgeon *Acipenser oxyrinchus oxyrinchus* Mitchill 1815, in Poland: Principles, previous experience, and results. *Journal of Applied Ichthyology, 27*, 186–191.

Kowarik, I. (1995). Time lags in biological invasions with regard to the success and failure of alien species. In P. Pyšek, K. Prach, M. Rejmánek, & M. Wade (Eds.), *Plant invasions: General aspects and special problems* (pp. 15–38). Amsterdam, Netherlands: SPB Academic Publishing.

Lincoln Park Zoo. (n.d.) *Avian reintroduction and translocation database*. Retrieved from http://www.lpzoozites.org/artd/.

Ludwig, A., Arndt, U., Lippold, S., Benecke, N., Debus, L., King, T. L., & Matsumura, S. (2008). Tracing the first steps of American sturgeon pioneers in Europe. *BMC Evolutionary Biology, 8*, 1–14.

Ludwig, A., Debus, L., Lieckfeldt, D., Wirgin, I., Benecke, N., Jenneckens, I., ... Pitra, C. (2002). Fish populations: When the American sea sturgeon swam east. *Nature, 419*, 447–448.

Lusardi, R. A., Stephens, M. R., Moyle, P. B., McGuire, C. L., & Hull, J. M. (2015). Threat evolution: Negative feedbacks between management action and species recovery in threatened trout (Salmonidae). *Reviews in Fish Biology and Fisheries, 25*, 521–535.

Luther, D., & Gentry, K. (2019). Threatened vertebrate species: Associations between conservation actions, funding and population trends. *Endangered Species Research, 39*, 105–114.

Luther, D., Skelton, J., Fernandez, C., & Walters, J. (2016). Conservation action implementation, funding, and population trends of birds listed on the Endangered Species Act. *Biological Conservation, 197*, 229–234.

Lynch, L. D., & Thomas, M. B. (2000). Nontarget effects in the biocontrol of insects with insects, nematodes and microbial agents: The evidence. *Biocontrol News and Information, 21*, 117–130.

Mahoney, S. (2004). The land mammals of insular Newfoundland. *Antigonish Review, 124*, 87–94.

Marshall, T. C., & Spalton, J. A. (2000). Simultaneous inbreeding and outbreeding depression in reintroduced Arabian oryx. *Animal Conservation, 3*, 241–248.

McRoberts, J. T., Wallace, M. C., & Eaton, S. W. (2014). *Wild Turkey (Meleagris gallopavo)*, version 2.0. In A. E. Poole (Ed.), *The birds of North America*. Ithaca, NY: Cornell Lab of Ornithology.

Miller, R. R., Williams, J. D., & Williams, J. E. (1989). Extinctions of North American fishes during the past century. *Fisheries, 14*, 22–38.

Molloy, S. W., Burbidge, A. H., Comer, S., & Davis, R. A. (2020). Using climate change models to inform the recovery of the western ground parrot *Pezoporus flaviventris*. *Oryx, 54*, 52–61.

Morrison, S. A. (2014). A bird in our hand: Weighing uncertainty about the past against uncertainty about the future in Channel Islands National Park. In *The George Wright Forum, 31*(1), 77–93.

Moseley, K. E., Lollback, G. W., & Lynch, C. E. (2018). Too much of a good thing: Successful reintroduction leads to overpopulation in a threatened mammal. *Biological Conservation, 219*, 78–88.

Mozelewski, T., & Scheller, R. (2021). Forecasting for intended consequences. In *Conservation Science and Practice, 3*(4), e570. https://doi.org/10.1111/csp2.370

Müller, H., & Eriksson, O. (2013). A pragmatic and utilitarian view of species translocation as a tool in conservation biology. *Biodiversity and Conservation, 22*, 1837–1841.

Musgrave, R., Flynn-O’Brien, J., Lambert, P. A., Smith, A. A., & Marinkais, Y. D. (1998). *Federal wildlife laws handbook with related laws* (p. 679). Rockville, MD: Government Institutes.

NAISA. (2003). *National Aquatic Invasive Species Act*. 109th U.S. Congress. Retrieved from https://www.congress.gov/bill/109th-congress/house-bill/1591/text.

National Species Reintroduction Forum. (2014). *The Scottish code for conservation translocations: Best practice guidelines for conservation translocations in Scotland* (p. 1.1, 1–79). Scotland.Scottish Natural Heritage.

National Wild Turkey Foundation. (2017). *Wild Turkey population history and overview*. Retrieved from https://www.nwtf.org/resource-library/detail/population-overview.

Nogués-Bravo, D., Simberloff, D., Rahbek, C., & Sanders, N. J. (2016). Rewilding is the new Pandora’s box in conservation. *Current Biology, 26*, R87–R91.

Olsen, A., Sedransk, J., Edwards, D., Gotway, C. A., Liggett, W., Rathbun, S., ... Young, L. J. (1999). Statistical issues for monitoring ecological and natural resources in the United States. *Environmental Monitoring and Assessment, 54*, 1–45.

Olson, Z. H., Whittaker, D. G., & Rhodes, O. E. (2012). Evaluation of experimental genetic management in reintroduced bighorn sheep. *Ecology and Evolution, 2*, 429–443.
Patterson, K. R. (2015). *Ecological restoration in wilderness: What the fish to do?* (p. 56). Missoula, MT: University of Montana.

Pister, E. P. (2001). Wilderness fish stocking: History and perspective. *Ecosystems, 4*, 279–286.

Purvina, S., & Medne, R. (2018). Reintroduction of sturgeon, *Acipenser oxyrinchus*, in the Gulf of Riga, East-Central Baltic Sea. *Archives of Polish Fisheries, 26*, 39–46.

Régnier, C., Fontaine, B., & Bouchet, P. (2009). Not knowing, not recording, not listing: Numerous unnoticed mollusk extinctions. *Conservation Biology, 23*, 1214–1221.

Reichard, S. H., & White, P. (2001). Horticulture as a pathway of invasive plant introductions in the United States. *Bioscience, 51*, 103–113.

Ricciardi, A., & Simberloff, D. (2009). Assisted colonization is not a viable conservation strategy. *Trends in Ecology and Evolution, 24*, 248–253.

Saremi, N. F., Supple, M. A., Byrne, A., Cahill, J. A., Coutinho, L. L., Dalén, L., ... Shapiro, B. (2019). Puma genomes from North and South America provide insights into the genomic consequences of inbreeding. *Nature Communications, 10*, 4769. https://doi.org/10.1038/s41467-019-12741-1

Schiebelhut, L. M., Puritz, J. B., & Dawson, M. N. (2018). Decimation by sea star wasting disease and rapid genetic change in a keystone species, *Pisaster ochraceus*. *Proceedings of the National Academy of Sciences of the United States of America, 115*, 7069–7074.

Schwarzländer, M., Hinz, H. L., Winston, R. L., & Day, M. D. (2018). Biological control of weeds: An analysis of introductions, rates of establishment and estimates of success, worldwide. *BioControl, 63*, 319–331.

Seddon, P. J., & Armstrong, D. P. (2016). Reintroduction and other conservation translocations: History and future developments. In D. S. Jachowski, J. J. Millspaugh, P. L. Angermeier, & R. Slotow (Eds.), *Reintroduction of fish and wildlife populations* (pp. 7–28). Oakland, CA: University of California Press.

Seddon, P. J., Strauss, W. M., & Innes, J. (2012). Animal translocations: What are they and why do we do them? In John G. E., Doug P. A., Kevin A. P., Philip J. S. (Eds.), *Reintroduction Biology: Integrating Science and Management*, (pp. 1–32). Blackwell. https://doi.org/10.1002/9781444355833.ch1

Shine, R. (2010). The ecological impact of invasive cane toads (*Bufo marinus*) in Australia. *The Quarterly Review of Biology, 85*, 253–291.

Simberloff, D. (2012). Risks of biological control for conservation purposes. *BioControl, 57*, 263–276.

Smith, D., & Peterson, R. (2021). Intended and unintended consequences of Wolf restoration to Yellowstone and Isle Royale National Parks. *Conservation Science and Practice, 3*(4), e413. https://doi.org/10.1111/csp2.413

Soorae, P. S. (Ed.). (2008). *Global re-introduction perspectives: Re-introduction case-studies from around the globe* (pp. 1–284). Abu Dhabi, UAE: IUCN/SSC Re-introduction Specialist Group & Environment Agency.

Soorae, P. S. (Ed.). (2010). *Global re-introduction perspectives: 2010. Additional case-studies from around the globe* (pp. 1–352). Abu Dhabi, UAE: IUCN/SSC Re-introduction Specialist Group & Environment Agency.

Soorae, P. S. (Ed.). (2011). *Global re-introduction perspectives: 2011. More case studies from around the globe* (pp. 1–250). Abu Dhabi, UAE: IUCN/SSC Re-introduction Specialist Group.

Soorae, P. S. (Ed.). (2013). *Global re-introduction perspectives: 2013. Further case-studies from around the globe* (pp. 1–282). Abu Dhabi, UAE: IUCN/SSC Re-introduction Specialist Group & Environment Agency.

Soorae, P. S. (Ed.). (2016). *Global re-introduction perspectives: 2016. Case-studies from around the globe* (pp. 1–292). Abu Dhabi, UAE: IUCN/SSC Re-introduction Specialist Group & Environment Agency.

Soorae, P. S. (Ed.). (2018). *Global reintroduction perspectives: 2018. Case studies from around the globe* (pp. 1–286). Abu Dhabi, UAE: IUCN/SSC Re-introduction Specialist Group & Environment Agency.

Stanturf, J. A., Palik, B. J., & Dumroese, R. K. (2014). Contemporary forest restoration: A review emphasizing function. *Forest Ecology and Management, 331*, 292–323.

Swan, K. D., Lloyd, N. A., & Moehrensclager, A. (2018). Projecting further increases in conservation translocations: A Canadian case study. *Biological Conservation, 228*, 175–182.

Swan, K. D., McPherson, J. M., Seddon, P. J., & Moehrenclager, A. (2016). Managing marine biodiversity: The rising diversity and prevalence of marine conservation translocations. *Conservation Letters, 9*, 239–251.

Taylor, G., Canessa, S., Clarke, R. H., Ingwersen, D., Armstrong, D. P., Seddon, P. J., & Ewen, J. G. (2017). Is reintroduction biology an effective applied science? *Trends in Ecology and Evolution, 32*, 873–880.

Tejeda, A. (2020). *What you need to know about a ballot effort to bring wolves back to Colorado*. The Colorado Independent. Retrieved from https://www.coloradoindpependent.com/2020/01/06/colorado-reintroduction-gray-wolf-ballot-measure-explainer/.

Tennyson, A., & Martinson, P. (2006). *Extinct birds of New Zealand* (p. 180). Wellington, New Zealand: Te Papa Press.

Thévenin, C., Mouchet, M., Robert, A., Kerbiriou, C., & Sarrazin, F. (2018). Reintroductions of birds and mammals involve evolutionarily distinct species at the regional scale. *Proceedings of the National Academy of Sciences of the United States of America, 115*, 3404–3409.

Vail, P. V., Coulson, J. R., Kauffman, W. C., & Dix, M. E. (2001). History of biological control programs in the United States department of agriculture. *American Entomologist, 47*, 24–49.

van de Kerk, M., Onorato, D. P., Hostetler, J. A., Bolker, B. M., & Oli, M. K. (2019). Dynamics, persistence, and genetic management of the endangered Florida Panther Population. *Wildlife Monographs, 203*, 3–35.

van Driesche, R. G., Carruthers, R. I., Center, T., Hoddle, M. S., Hough-Goldstein, J., Morin, L., ... van Klinken, R. D. (2010). Classical biological control for the protection of natural ecosystems. *Biological Control, 54*, S2–S33.

Vilà, C., Sundqvist, A.-K., Flagstad, Ø., Seddon, J., Björnerfeldt, S., Kojola, I., ... Ellegren, H. (2003). Rescue of a severely bottlenecked wolf (*Canis lupus*) population by a single immigrant. *Proceedings. Biological Sciences/The Royal Society, 270*, 91–97.

Warner, R. E. (1968). The role of introduced diseases in the extinction of the endemic Hawaiian avifauna. *The Condor, 70*, 101–120.
Webster, C. R., Jenkins, M. A., & Jose, S. (2006). Woody invaders and the challenges they pose to forest ecosystems in the eastern United States. *Journal of Forestry, 104*, 366–374.

West, R. S., Tilley, L., & Moseby, K. E. (2019). A trial reintroduction of the western quoll to a fenced conservation reserve: Implications of returning native predators. *Australian Mammalogy, 42*, 257. https://doi.org/10.1071/am19041

Wild Sheep Working Group. (2015). *Records of wild sheep translocations United States and Canada 1922-present* (p. 178). USA: Western Association of Fish and Wildlife Agencies.

Williams, D. R., Balmford, A., & Wilcove, D. S. (2020). The past and future role of conservation science in saving biodiversity. *Conservation Letters, 13*(4), 1–7.

Wilson, A. C., & Price, M. R. S. (1994). Reintroduction as a reason for captive breeding. In P. J. S. Olney, G. M. Mace, & A. T. C. Feistner (Eds.), *Creative conservation* (pp. 243–264). Dordrecht: Springer.

Yan, X., Zhang, H., Li, D., Wu, D., Zhou, S., Sun, M., ... Huang, Y. (2019). Acoustic recordings provide detailed information regarding the behavior of cryptic wildlife to support conservation translocations. *Scientific Reports, 9*, 1–11.

Zimmer, H. C., Auld, T. D., Cuneo, P., Offord, C. A., & Commander, L. E. (2019). Conservation translocation – An increasingly viable option for managing threatened plant species. *Australian Journal of Botany, 67*, 501–509.

Zimmerman, S. J., Aldridge, C. L., Apa, A. D., & Oyler-McCance, S. J. (2019). Evaluation of genetic change from translocation among Gunnison Sage-Grouse (*Centrocercus minimus*) populations. *Condor, 121*, 1–14.

**SUPPORTING INFORMATION**

Additional supporting information may be found online in the Supporting Information section at the end of this article.

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