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# Decision support framework for the conservation translocation of SARA-listed freshwater fishes and mussels

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**Foreword**

This series documents the scientific basis for the evaluation of aquatic resources and ecosystems in Canada. As such, it addresses the issues of the day in the time frames required and the documents it contains are not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations.

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## ABSTRACT

For freshwater fishes and mussels listed under the *Species at Risk Act* (SARA), a federal recovery strategy or management plan identifies recovery measures for best achieving population and distribution objectives. Recovery strategies and management plans for SARA-listed freshwater fish and mussel species often identify conservation translocation, specifically supplementation or reintroduction, as a potential approach for improving survival and/or recovery. However, there has been limited progress in undertaking conservation translocations primarily due to basic information gaps on species ecology and uncertainty about how to assess the potential ecological benefits and risks to freshwater species and ecosystems. The objectives of this research document were to: 1) identify and evaluate the potential benefits and risks of conservation translocation as a tool for improving the survival, recovery, or management of SARA-listed freshwater fish and mussel species; and, 2) identify science-based considerations and methods for determining when conservation translocation would be expected to improve the survival, recovery, or management of SARA-listed freshwater fishes and mussels. Benefits and risks of conservation translocations to the focal species and broader ecosystem components are presented, including a decision support framework for evaluating the potential benefits and risks of conservation translocation for SARA-listed freshwater fishes and mussels. The decision support framework consists of five steps: 1) Identify fundamental and means objectives for conservation translocations; 2) Assess the probability of achieving the fundamental and means objectives; 3) Identify and assess the likelihood and magnitude of unintended consequences; 4) Compile and weigh the scientific evidence to inform the translocation decision; and, 5) Implement and monitor the conservation translocation. Using the decision support framework can help determine if and when the ecological benefits of conservation translocations outweigh the uncertainties and ecological risks when attempting to improve the survival and recovery of SARA-listed freshwater fishes and mussels. Future refinement of the decision support framework would benefit from several case studies in addition to those presented in the appendices.

# INTRODUCTION

For species listed under the *Species at Risk Act* (SARA), a federal recovery strategy or management plan is required that identifies species recovery targets and recovery measures for best achieving population and distribution objectives. **Recovery**, as defined within federal species recovery strategies, is "*the process by which the decline of an endangered, threatened, or extirpated species is arrested or reversed and threats are removed or reduced to improve the likelihood of the species' persistence in the wild. A species* [as defined under SARA] *will be considered recovered when its long-term persistence in the wild has been secured*" (Fisheries and Oceans Canada 2012). Two commonly identified tools for improving survival or recovery of SARA-listed freshwater fishes and mussels are **reintroduction** and **supplementation**. Species reintroduction describes the intentional movement and release of an organism to a location within its native range from which it has disappeared (IUCN/SSC 2013). Supplementation describes the intentional release of individuals of a focal species to an area presently occupied by conspecifics (Seddon et al. 2012). Together, these terms fall under the umbrella term of **conservation translocations**, which describes the intentional movement of species in an effort to improve survival or recovery (IUCN/SSC 2013), and represents a long-term, experimental restoration strategy for species at risk of extinction.

As of March 2021, 75 freshwater fish and 20 freshwater mussel species [and/or Designatable Units (DUs)] were listed under Schedule 1 of SARA as Extirpated, Endangered, Threatened, or Special Concern. Among them, 24 have reintroduction and (or) supplementation identified as potential methods for recovery in federal recovery strategies, action plans, and (or) management plans, including 13 freshwater fish Designatable Units (DUs) and 11 freshwater mussel DUs. However, reintroduction or supplementation efforts have only occurred for seven freshwater fish DUs (*n* = 7; Atlantic Whitefish *Coregonus huntsmani*, Copper Redhorse *Moxostoma hubbsi*, White Sturgeon *Acipenser transmontanus* [Kootenay River, Nechako River, and Upper Columbia River DUs], and Westslope Cutthroat Trout *Oncorhynchus clarkii lewisi* [Alberta and Pacific DUs]; Lamothe and Drake 2019), with none performed for freshwater mussels. Poor progress toward initiating reintroduction or supplementation has been the result of basic information gaps on species ecology (e.g., species distribution, abundance), as certain SARA-listed species have only recently been the focus of dedicated monitoring and research (Castañeda et al. 2021; Drake et al. 2021), and concern that such gaps may limit translocation success or cause increased harm. Adding to the lack of standard ecological information, limited progress has also been the result of uncertainty about how to assess the potential ecological benefits and risks of proposed reintroduction or supplementation efforts. In the absence of clear advice, decisions about whether conservation translocations should occur will be made on an ad-hoc, case-by-case basis. Such an approach may fail to identify situations in which conservation translocations would provide meaningful benefit to SARA-listed wildlife species, or alternatively, identify situations where a net benefit to the focal species is unlikely to be achieved. As such, two primary objectives are addressed:

1. Identify and evaluate the potential benefits and risks of conservation translocation as a tool for improving the survival, recovery, or management of SARA-listed freshwater fish and mussel species; and,
2. Identify science-based considerations and methods for determining when conservation translocation would be expected to improve the survival, recovery, or management of SARA-listed freshwater fishes and mussels.

These objectives are intended to help persons involved in species recovery planning and implementation make more robust and consistent decisions around the situations in which conservation translocations would be expected to improve the survival or recovery of SARA-listed species. Thus, information contained in this document will be useful at several stages of recovery planning, from drafting recovery strategies to identifying the information needed to implement translocations.

In the next section, terminology is presented to ensure clarity for the remainder of the document. Following this introduction, a decision support framework is presented for determining in what situations conservation translocation may be a suitable strategy for improving the survival or recovery of SARA-listed freshwater fishes and mussels. The decision support framework considers the potential ecological benefits for undertaking four different types of conservation translocation (supplementation, reintroduction, **mitigation translocations**, and **assisted colonization**) and the potential risks of unintended consequences to focal and non-focal species in source and recipient ecosystems. Only once the risks of unintended ecological consequences are considered against the probability of achieving conservation translocation objectives can a decision be made on how to proceed (informed by a parallel socioeconomic analysis and management considerations).

Following the description of the conservation translocation decision support framework, two examples are presented in the appendix to demonstrate how the support framework can be used to assess whether conservation translocation could improve survival or recovery of SARA-listed species: Appendix 1) reintroduction of Eastern Sand Darter *Ammocrypta pellucida* (Ontario population; Threatened) to Big Otter Creek, Ontario; and, Appendix 2) reintroduction of Snuffbox *Epioblasma triquetra* (Endangered) to the Thames River, Ontario.

## TERMINOLOGY

For the remainder of the document, the term '**species**' refers to '**wildlife species**' as assessed by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC 2019) and listed under SARA (2002). A wildlife species is defined as a species, subspecies, variety, or geographically or genetically distinct population of animal, plant, or other organism, other than a bacterium or virus, that is wild by nature and is native to Canada or has extended its range into Canada without human intervention and has been present in Canada for at least 50 years. The term **population** is used to describe geographically or otherwise distinct groups of the taxonomic species between which there is little demographic or genetic exchange. Finally, the term **Designatable Unit (DU)** is only used to refer to species, subspecies, varieties, or geographically or genetically distinct populations as defined and assessed by COSEWIC.

Terminology surrounding conservation translocation varies, but most often follows guidelines presented by the IUCN/SSC (2013). Broadly, the intentional movement and release of an organism where the primary objective is to improve the survival or recovery of the species describes a conservation translocation. There are several approaches for conducting conservation translocations, with their use depending on the identified means for achieving improved survival or recovery of the species (i.e., means objectives; Figure 1). Three common means for achieving improved survival or recovery via conservation translocation are to:

1. Improve population recruitment;
2. Establish a population; or,
3. Rescue individuals or populations at imminent risk of extirpation (Figure 1).

Achieving these means objectives will confer unique benefits that will differ among species. Therefore, early consideration is needed on whether the species would benefit from increased population recruitment, newly created or restored connection of populations, and (or) population rescue. The factors underlying the decision about which means to choose for improving survival or recovery are described in greater detail in the next sections.

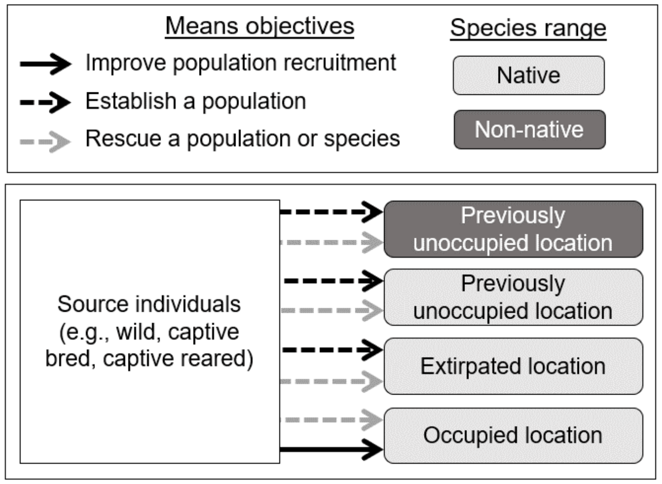


*Figure 1. Means for conducting conservation translocations depending on the objective (means objective; arrow shade, type) and scale of intended movement (species range; box shade).*

For all conservation translocation approaches, individuals must be removed from the wild and are either relocated to an alternative location in the wild (i.e., **translocation**), or collected and raised under human care. When raised under human care, **captive breeding** or **captive rearing** practices may be used (Figure 1). Captive breeding describes the act of removing individuals from the wild with the goal of breeding those individuals to support reintroduction efforts (Williams and Hoffman 2009). Captive rearing describes the raising of captured, wild individuals from eggs or young to adults with the intention of release back into the wild within a single generation (i.e., no facilitation of reproduction; George et al. 2009). When wild individuals are unavailable for translocation, captive rearing and (or) breeding could assist in improving population survival and (or) recruitment when used for supplementation and mitigation translocation, and could assist in establishing populations when used for reintroduction.

The geographic scale at which conservation translocations occur varies. Conservation translocations most often occur within the **native range** of the focal species, which includes supplementation of occupied locations, reintroduction to extirpated locations, and the establishment of populations in previously unoccupied locations (Figure 1). The term 'native range' or 'native species distribution' is used here to characterize the best available knowledge on total historical **extent of occurrence** for the focal species prior to human-caused reductions in population abundance and (or) distribution. Extent of occurrence describes the area included in a polygon without concave angles that encompasses the geographic distribution of all known populations of a wildlife species (COSEWIC 2019). **Area of occupancy** is the area within the extent of occurrence that is occupied by a taxon, excluding cases of vagrancy. The area of occupancy measure reflects the fact that the extent of occurrence may contain unsuitable or unoccupied habitats (COSEWIC 2019). Present-day distributions of SARA-listed freshwater fishes and mussels are often geographically discontinuous within the total extent of occurrence, which may or may not reflect distributions prior to human-caused reductions in population abundance and distribution (Guiaşu and Labib 2021).

The native range of a species reflects the interaction between the **ecological niche** of a species and potential dispersal constraints. The ecological niche of a species represents a complex set of ecological and evolutionary relationships that support its persistence. Efforts to quantify species distributions commonly incorporate abiotic (i.e., physiological tolerances) and biotic factors (i.e., inter- and intra-specific interactions) that may influence the persistence of the species, along with potential dispersal constraints (Holt 2003; Sexton et al. 2009). Figure 2 presents the "BAM" heuristic (Soberón and Peterson 2005; Holloway and Miller 2017), which can be used to conceptualize the determinants of species distributions. The distribution of a species ($G_0$) is represented by the interaction between biotic (B), abiotic (A), and movement (M) factors (Soberón and Peterson 2005; Figure 2a). Although $G_i$ represents habitat where the abiotic and biotic conditions may be suitable for the species, the inability to disperse to that habitat limits species occupancy (Figure 2a).

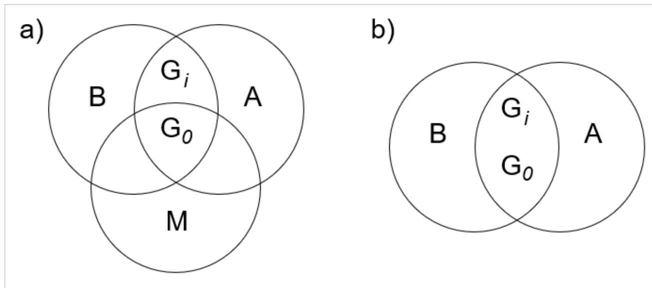


*Figure 2. The "BAM" heuristic. a) The distribution of a species ($G_0$) results from interactions between biotic (B), abiotic (A), and movement (M) factors. $G_i$ represents habitat where the abiotic and biotic conditions may be suitable for the species, but the distribution is limited by the ability to disperse. b) Removal of the dispersal constraint (M) allows the potential establishment of species in previously unoccupied, but suitable habitat ($G_i$). Recreated from Soberón and Peterson (2005) and Holloway and Miller (2017).*

The concept of the ecological niche is presented because translocations of individuals can also occur in areas with favourable abiotic and biotic characteristics outside of the species' native range (Figure 1). Assisted colonization, also known as managed relocation (Richardson et al. 2009; Lawler and Olden 2011; Olden et al. 2011; Schwartz et al. 2012; Karasov-Olson et al. 2021), is defined as the translocation of a species to favourable habitat beyond the native range to protect it from human-induced threats (Ricciardi and Simberloff 2009). The term assisted colonization broadened the original term 'assisted migration', which was specific to the movement of individuals outside the native range of the species in response to climate change (McLachlan et al. 2007; Chauvenet et al. 2013). Ultimately, translocation, reintroduction, and assisted colonization remove the dispersal constraint that may limit natural (re)colonization, potentially providing opportunities for the persistence of species (i.e., $G_i + G_0$; Figure 2b).

## DECISION SUPPORT FRAMEWORK FOR USING CONSERVATION TRANSLOCATIONS FOR SARA-LISTED FRESHWATER FISH AND MUSSEL SPECIES

Identifying the potential ecological benefits for species and ecosystems, as well the risks of negative ecological consequences, is needed when considering conservation translocation as a recovery measure. A decision support framework is presented below for evaluating whether conservation translocation would be expected to improve the survival or recovery of SARA-listed freshwater fishes and mussels (Figure 3). Each step is explored in detail to describe the process of identifying suitable candidate species, weighing the potential ecological benefits and risks, and implementing recovery actions.

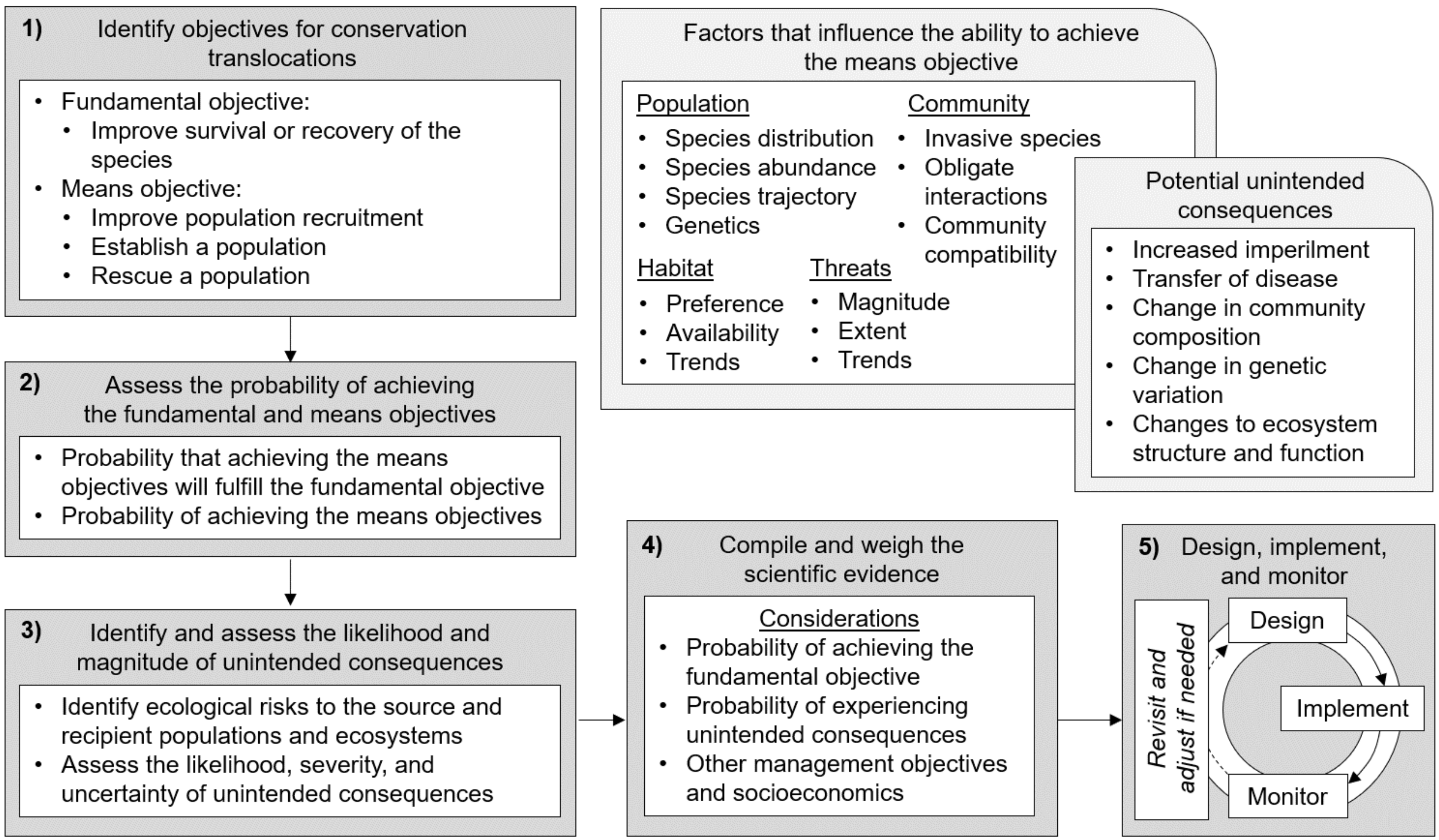


*Figure 3. Decision support framework outlining science considerations for the use of conservation translocation as a tool to improve the survival or recovery of freshwater fish or mussel species listed under the* Species at Risk Act *(SARA 2002). The framework begins at the top left.*

# 1. IDENTIFY OBJECTIVES FOR CONSERVATION TRANSLOCATIONS

Determining whether a species should be considered for conservation translocation requires understanding the underlying motivations for the demand-driven action. The development of a **problem statement, fundamental objective**, and **means objective(s)** is the first step for using the decision support framework (Figure 3). A problem statement is a concise description of the issue, setting the context for the decision-making process. This includes identifying the intended taxon for management (i.e., species, population, and (or) DU) and the temporal and spatial scale of management efforts. Fundamental objectives reflect the broadest objectives that decision makers and stakeholders value most (Robinson and Jennings 2012). For all SARA-listed freshwater fishes and mussels, the fundamental objective of recovery actions under SARA (2002) is to improve the survival or recovery of the species.

There are several mechanisms, or means, for which the fundamental objective of improved survival or recovery can be achieved using conservation translocation. Described as the means objectives, three mechanisms for improving survival or recovery of species (i.e., achieving the fundamental objective) through conservation translocation are to:

1. Improve recruitment of extant populations;
2. Establish a population; or,
3. Rescue individuals or populations at imminent risk of extirpation.

Fulfilling the means objectives will have unique benefits for SARA-listed species that are dependent on the context of the species and its imperilment. A series of high-level questions are presented in Figure 4 that can help decide which means may be relevant to the focal species. For example, supplementation would be considered when populations of the species remain extant, but abundance and (or) trajectories are below recovery targets (Figure 4). Reintroduction is considered when population extirpations have occurred, and formerly occupied habitats can support the species (Figure 4). A review of the conservation translocation literature is provided below to describe how the three primary mechanisms for conservation translocation are used to improve survival or recovery.

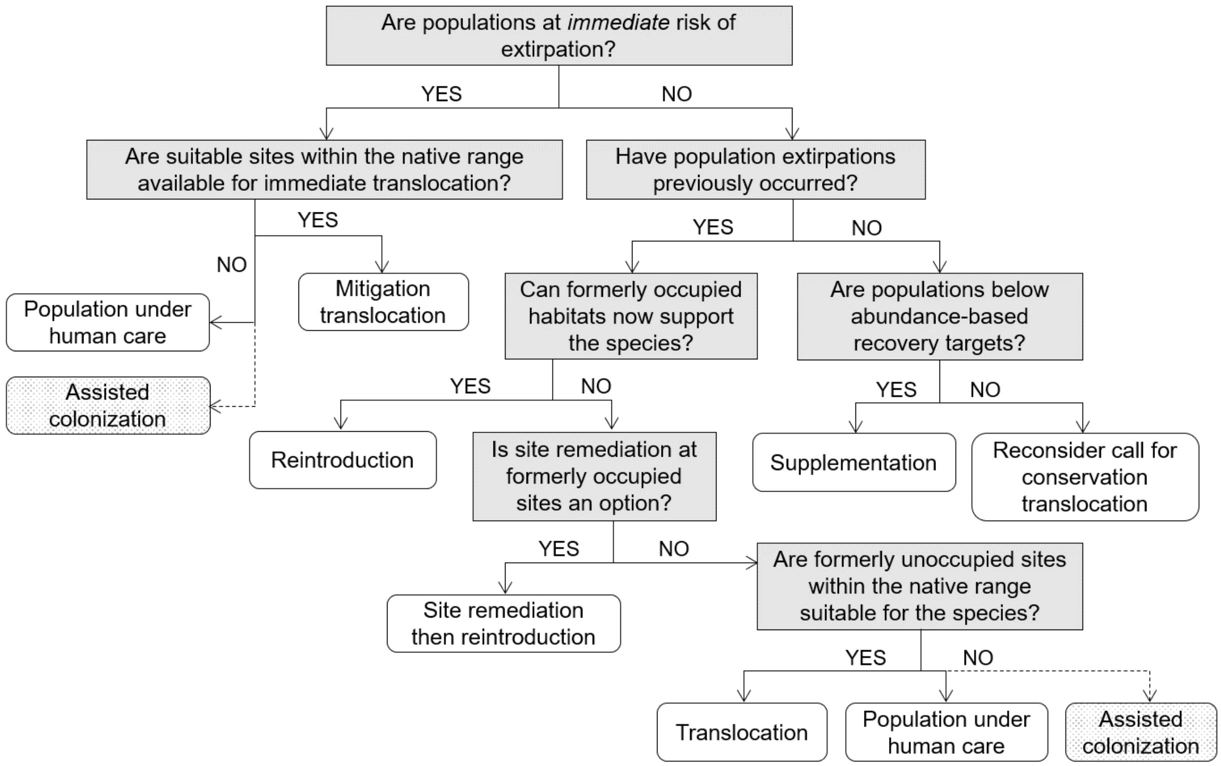


*Figure 4. High-level decision framework for initial consideration of means objectives. Management actions are identified in white boxes and questions for the user are identified in grey boxes. Assisted colonization is hashed to identify this as a last resort option owing to greater potential for ecological risks. Note that each path through the decision framework requires extensive consideration of ecological benefits and risks.*

## 1.1. Improve population recruitment

Conservation translocations are undertaken with the aim of having population-level effects (i.e., improved population abundance or distribution) that benefit the species. Increased population recruitment is one way to improve the survival or recovery of SARA-listed freshwater fishes and mussels. Population size and trends in population size over time are the best predictors of extinction risk; smaller populations and populations declining at higher rates are at greater risk of extirpation (O'Grady et al. 2004). Supplementing extant populations with individuals is one approach for increasing population size, reducing inbreeding within recipient populations, and reversing declining population trajectories (Figure 5). The act of supplementation immediately increases local population abundance (Figure 5b, c) and the opportunity to mate, which can provide a demographic boost for the focal species (Janowitz-Koch et al. 2018) and can reduce the risks of local genetic and demographic collapse that can occur when populations are small (Seddon 2010; Seddon et al. 2012). Ultimately, supplementation efforts require reproduction in the wild to achieve improved survival or recovery of the species, which can only be confirmed through genetic monitoring post-translocation (e.g., Fisheries and Oceans Canada 2010).

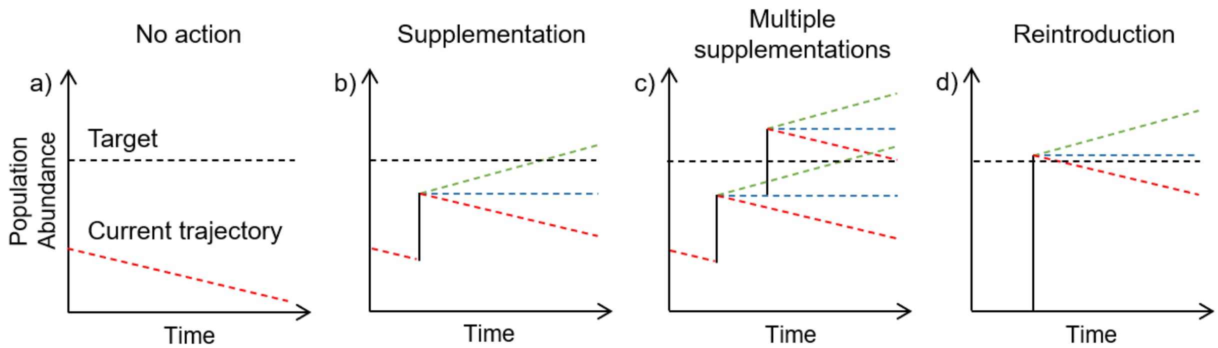


*Figure 5. Examples of time series that demonstrate how supplementation and reintroduction could be used to achieve the hypothetical population recovery target (dashed black lines). a) No conservation translocation action is undertaken, and current population trajectory is insufficient to meet the recovery target. b) Supplementation to an abundance less than the population recovery target, with three potential growth scenarios thereafter (green = growth, blue = neutral, red = negative). Population growth post-supplementation is the only scenario that reaches the population abundance target by the end of the time series. c) Multiple supplementations resulting in an abundance greater than the population-level target. d) Reintroduction to the target abundance with three population growth scenarios thereafter.*

The frequency and magnitude of introductions needed to achieve population-level objectives will be context dependent (e.g., Figure 5b, c). Although rarely successful, a single supplementation event could theoretically provide a large enough boost to population-level recruitment that future supplementations are unnecessary (i.e., population growth post-supplementation; Figure 5b). Alternatively, multiple supplementations may be needed to achieve population objectives (Figure 5c). A situation can also occur where supplementation is conducted for a declining population, which increases abundance in the short-term, but is unable to improve recruitment among recipient populations, potentially masking population decline (Post et al. 2002; Figure 5c). Supplementation is considered a relatively short-term management strategy because the potential for unintended consequences increases over time (IUCN/SSC 2013). A potential long-term reliance on supplementation suggests that other factors may need to be managed (e.g., threats).

Supplementation of freshwater fishes and mussels can be performed through two basic mechanisms: 1) the translocation of individuals from wild populations to wild populations, or 2) captive breeding or rearing of individuals and their release to wild populations (Figure 1). The decision on whether to use captive breeding or rearing versus translocating individuals from wild populations depends on characteristics of the source and recipient populations (e.g., abundance, distribution, genetics, and condition). Specifically, a decision must be made about whether suitable source populations are available that can withstand the removals needed to improve population-level recruitment at the recipient site and how recipient populations will respond to the supplementation (e.g., Fisheries and Oceans Canada 2018a; Lamothe et al. 2021). Evaluating both factors concurrently is needed to demonstrate a net improvement in survival for the species (Figure 6). In cases where abundance is low across potential source populations, the removal of individuals to improve recruitment at the release site could place increased risk of extirpation on the source population(s). Ultimately, the choice of how, when, and how frequently supplementation should occur for a population must be considered against the potential benefits and risks to the survival or recovery of the species (described in more detail in the following steps).

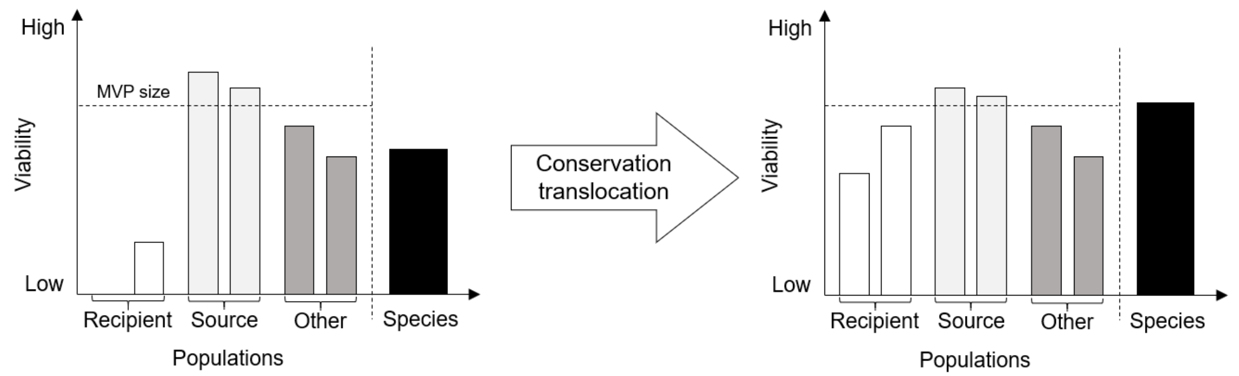


*Figure 6. Conceptual assessment of the ability of the means objective to fulfill the fundamental objective of improved survival or recovery of the species. Metrics for assessing viability are measured before and after translocations for the recipient, source, and other (e.g., reference) remaining populations. MVP = minimum viable population size. In this example, reintroduction and supplementation are performed for two recipient populations from two source populations.*

Captive breeding has been used for many decades in North America to propagate freshwater fishes and mussels, with the propagation of endangered species gaining more attention over the last few decades. For example, captive breeding techniques were developed for the Blackside Dace (*Phoxinus cumberlandensis*), Boulder Darter (*Etheostoma wapiti*), Smoky Madtom (*Noturus baileyi*), Spotfin Chub (*Cyprinella monachus*), and Yellowfin Madtom (*Noturus flavipinnis*) in the mid-1980s and early-1990s for the purposes of reintroduction (Rakes et al. 1999; Shute et al. 2005). Similarly, captive breeding and release of Oyster Mussel (*Epioblasma capsaeformis*), Combshell (*Epioblasma brevidens*), Fanshell (*Cyprogenia stegaria*), Tan Riffleshell (*Epioblasma florentina walkeri*), Purple Bean (*Villosa perpurpurea*), Snuffbox (*Epioblasma triquetra*), Dromedary (*Dromus dromas*), Birdwing Pearlymussel (*Lemiox rimosus*), and Crackling Pearlymussel (*Hemistena lata*) began in the late 1990s (Neves 2004). Whereas the early motivation for fish hatcheries was more often related to the production of recreational and commercial species (e.g., Pister 2001), captive breeding and rearing for freshwater mussels were used for species recovery after commercial harvest decimated populations (Haag 2012; Patterson et al. 2018). However, there remains a paucity of literature on the effectiveness of captive breeding for recovering imperilled freshwater mussels (Eveleens and Febria 2021), with few areas of conservation-based captive breeding rigorously evaluated for freshwater fishes (Rytwinski et al. 2021).

Supplementation of SARA-listed freshwater mussels has yet to be performed in Canada. Three SARA-listed freshwater fish species have been bred under human care and released to supplement populations in Canada: Copper Redhorse, White Sturgeon, and Westslope Cutthroat Trout (Lamothe et al. 2019). The Québec Ministère des Resources Naturelles et de la Faune (MRNF) began captive breeding efforts for the Copper Redhorse in 2004 to supplement natural recruitment, and has since released over three million larvae into the Rivière Richelieu (Fisheries and Oceans Canada 2012; Vachon et Sirois 2019; Vachon 2021). Similarly, White Sturgeon is bred under human care and released for conservation purposes to supplement non-recruiting populations (Fisheries and Oceans Canada 2014; Hildebrand et al. 2016). Westslope Cutthroat Trout has been bred under human care and released in stocked ponds for decades, including in Canada, but for the purposes of maintaining a recreational catch and harvest fishery (i.e., not a conservation translocation).

Although not listed under SARA, the Government of Québec has provided guidance on the supplementation and reintroduction of Walleye (*Sander vitreus*) for the purposes of conservation

(Ministère du Développement durable, de l'Environnement, de la Faune et des Parcs 2013), and more generally for freshwater fishes (MRNF 2008). Similarly, the Government of Alberta has provided guidance on supplementation and reintroduction approaches for Athabasca Rainbow Trout (*Oncorhynchus mykiss*; Alberta Athabasca Rainbow Trout Recovery Team 2014). Supplementation has been used in an effort to rehabilitate fishes in the Great Lakes, such as Lake Trout (*Salvelinus namaycush*; Lake Ontario; Elrod et al. 1995; Lake Erie: Cornelius et al. 1995). Despite supplementation being used for relatively few SARA-listed freshwater fishes or mussels, the supplementation efforts thus far for other freshwater species have provided an important foundation for how conservation translocation can be used to improve survival or recovery of imperilled taxa.

## 1.2. Establish a population

Establishing a population is another approach to improve the survival or recovery of SARA-listed species (Figure 1). A greater number of populations provides a greater likelihood of survival for the species against stochastic and catastrophic events (Figure 7). Moreover, the addition of a species to an ecosystem, even in low abundance (Downing et al. 2014), can have broader ecosystem benefits such as stabilizing effects on community and ecosystem attributes (Oliver et al. 2015), generating redundancy in functional traits (Micheli and Halpern 2005), and, ultimately, providing insurance against declines in the provisioning of ecosystem services caused by environmental fluctuations (i.e., insurance hypothesis of biodiversity; Yachi and Loreau 2001).

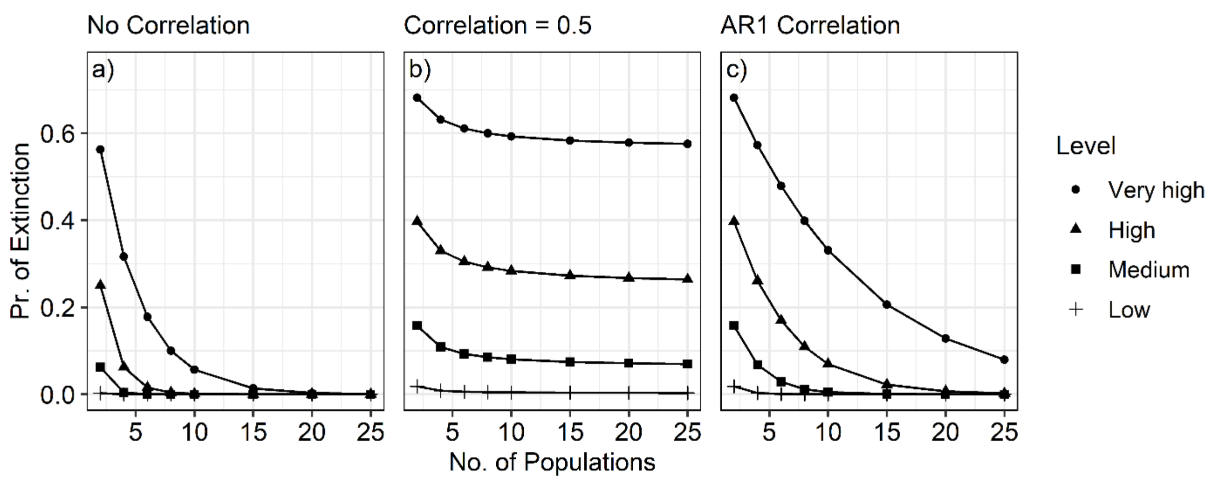


*Figure 7. Hypothetical probabilities of species extinction (y-axis). Individual plots represent different correlation structures of extirpation probabilities across populations, including a) no correlation, b) correlation = 0.5, and c) spatial correlation (first-order autoregressive model; AR1). Population-level extirpation probabilities are depicted as low ($P_{pop}$ = 0.05), medium ($P_{pop}$ = 0.25), high ($P_{pop}$ = 0.50), and very high ($P_{pop}$ = 0.75). The true shape of this relationship and correlation structure is typically unknown and will vary among species.*

Establishment of populations can occur in different ways, including the movement of individuals to formerly occupied locations within the native range of a species (i.e., situations where the species has since been extirpated), formerly unoccupied locations within the native range of a species (i.e., situations outside known area of occupancy), or areas outside of the species' native range (Figure 1). Similar to supplementation, the establishment of populations requires the removal of individuals from the wild to act as a source, and may require the use of captive breeding or rearing efforts (Figure 1). The choice of whether to remove individuals from the wild and (or) use captive breeding or rearing to facilitate translocations is evaluated by considering the potential harm caused by removals and the potential suitability of populations under human care.

Species reintroduction describes the intentional release of individuals of a focal species to a historically occupied area (i.e., locally extirpated) in an effort to re-establish a population. Species reintroduction is considered less ecologically complex, and therefore with relatively fewer ecological risks, than the establishment of populations in formerly unoccupied locations or locations outside of the native range. This is based on the assumption that the historically occupied locations provide suitable abiotic and biotic conditions for supporting the focal species (Figure 2), that conditions at the historically occupied location have been restored to a sufficient state or that significant ecological changes since extirpation have not occurred, and that there is a lower likelihood of the reintroduced species causing a reduction in co-occurring species abundance, distribution, and (or) desired ecosystem processes in formerly occupied habitat than if introduced to novel habitat outside of the species' native range.

Aquatic species distributions are often non-contiguous. As a result, translocations could be proposed for habitats located within the native range of the species, but for areas that are outside known areas of occupancy. This scenario may occur when there is a lack of historical monitoring data to confirm prior site occupancy, or if historically occupied sites are deemed less likely to support the species than relatively proximate sites with favourable habitat conditions. If the original cause of extirpation continues to limit the presence of the species at a historically occupied site, but nearby watercourses are less affected by the stressor and unable to be naturally colonized (Figure 2b), then considerations may be warranted for introductions within the native range but outside areas of occupancy.

There may also be situations when historically or presently occupied habitats located within the species' native range can no longer support the persistence of a population, or when anticipated environmental changes (e.g., climate change) to historically or presently occupied locations are predicted to make habitats within the native range unsuitable. In such cases, assisted colonization beyond the native range may provide opportunities for establishment in the absence of human-induced threats (e.g., non-native species; Fisheries and Oceans Canada 2018a). However, few examples of assisted colonization in the wild exist (e.g., Dade et al. 2014; Mitchell et al. 2016), and its use as a tool to support the persistence of species has been strongly debated (Ricciardi and Simberloff 2009; Lawler and Olden 2011; Loss et al. 2011; Gallagher et al. 2015). The primary caution regarding the use of assisted colonization as a recovery action is the potential for unexpected short- and (or) long-term consequences for species or ecosystems, including biological invasions, structural and compositional changes in community composition, and hybridization resulting in reduced genetic diversity of wild populations (Ricciardi and Simberloff 2009). Freshwater mussel researchers have cautioned against the use of assisted colonization for imperilled freshwater mussels because of the limited literature on assisted colonization for freshwater mussels compared to other taxa being considered for the approach (Strayer et al. 2019). Nevertheless, a comprehensive assessment of the ecological benefits and risks that considers the focal species and broader ecosystem components, as presented in Figure 3, is needed for proactive policy to inform future decisions on assisted colonization (Lawler and Olden 2011; Swan et al. 2018; Strayer et al. 2019; Karasov-Olson et al. 2021).

Examples of reintroduction or assisted colonization are rare for SARA-listed freshwater fishes or mussels. Westslope Cutthroat Trout alevin were translocated to unoccupied stream sections within their native range to expand their occupied habitat and to support a larger, more connected and genetically pure population. Assisted colonization has been conducted for Atlantic Whitefish, a fish species that is endemic to Nova Scotia (Bradford 2017; Fisheries and

Oceans Canada 2018a). Thought to be historically distributed within the Tusket and Annis rivers, and the Petite Rivière, the distribution of Atlantic Whitefish has been reduced to three interconnected, semi-natural lakes (Milipsigate Lake, Minamkeak Lake, and Hebb Lake; Bradford et al. 2004). In 2000, the first documented successful breeding of Atlantic Whitefish was recorded at the Mersey Biodiversity Facility in Milton, Nova Scotia (Bradford et al. 2015) with conservation-focused introductions performed thereafter. Specifically, captive-bred Atlantic Whitefish were experimentally translocated to areas within the Petite Rivière below the Hebb Lake Dam (Bradford et al. 2015) and over 12,000 individuals were released into Anderson Lake, a formerly unoccupied habitat approximately 100 km northeast of the known species distribution (i.e., assisted colonization; Bradford et al. 2015; Lamothe et al. 2019). The translocations of Atlantic Whitefish were implemented to reduce surplus of captive-bred individuals, not necessarily to achieve a self-sustaining population. Neither Atlantic Whitefish translocation appears to have resulted in recruitment (Bradford 2017).

Although no longer listed under SARA, captive breeding, reintroduction, and assisted colonization were performed for Aurora Trout (*Salvelinus fontinalis timagamiensis*), a variant of Brook Trout (*S. fontinalis*) that was extirpated from two small lakes northeast of Sudbury, Ontario as a result of lake acidification (Snucins et al. 1995; COSEWIC 2011). Captive breeding of Aurora Trout began in 1958, shortly before the extirpation of the two native populations. After three decades of maintaining a captive broodstock, reductions in smelting emissions, and significant habitat restoration efforts (i.e., whole-lake liming), Aurora Trout was reintroduced to the historically occupied Whirligig and Whitepine lakes (Snucins et al. 1995; COSEWIC 2011), along with several assisted colonizations (COSEWIC 2011).

## 1.3. Rescue individuals or populations at imminent risk of extirpation

Rescuing individuals or populations at **imminent risk of extirpation** is a clear mechanism for improving the survival or recovery of a species. Imminent extirpation or extinction, as described by COSEWIC (2019), involves a 20% or greater probability of extirpation or extinction within 20 years or 5 generations (up to a maximum of 100 years), whichever is longer. Several terms have been used to describe the intentional movement of animals as a means for rescue. Mitigation translocation is the most used term in the mollusc literature, defined as the human-mediated, intentional movement of individuals from an occupied location with the objective of reducing the inevitable effects of a development project on local biota (Germano et al. 2015). For fishes, mitigation translocations are often referred to as salvage operations (e.g., Higgins and Bradford 1996). However, the term 'salvage operation' does not differentiate between the capture and relocation of fishes prior to or following anthropogenic perturbation and is therefore not used in this document. Ultimately, mitigation translocation is performed when no other options remain for preventing the imminent loss of individuals at their natal location.

Mitigation translocations are typically small in geographic scope, where movement of individuals only occurs within the focal ecosystem (Cope and Waller 1995; Mackie et al. 2008; Bradley et al. 2020). Rather than aiming to improve the viability of recipient populations for long-term conservation benefits, mitigation translocations are typically conducted to save individuals or populations from imminent extirpation (Bradley et al. 2020). For these scenarios, the risk to the survival and recovery of freshwater fishes or mussels at the extant location is higher than the risk of moving individuals away from the site, as the development actions at the extant site are inexorable. There may be situations when immediately translocating individuals between suitable habitats in the wild within the species' native range is unachievable and individuals must be moved to locations under human care (Figure 4), where captive rearing and (or) breeding techniques could be performed. Although there may be scenarios where the rescue of species requires movement outside its native range (Figure 1). This form of rescue is considered under ‘assisted colonization’ for the purposes of this document and is typically a last-resort option (Figure 4).

Guidance documents have been developed for mitigation translocations of freshwater mussels in Canada both provincially (e.g., British Columbia – FLNRORD 2018) and federally (Mackie et al. 2008). Fisheries and Oceans Canada provided guidelines for “relocating” freshwater mussels in anticipation of development projects that may have a negative effect on mussel survival or recovery (Mackie et al. 2008). The guidance document outlines that mussel relocation must only occur within a single drainage area, preferably as close to the construction site as possible to avoid unintended consequences on focal species and broader ecosystem components or processes (Mackie et al. 2008). Moreover, guidance was provided on the application of mitigation translocations, referencing proper methods for collection, handling, and marking of mussels, timing of relocations, and monitoring post-release (Mackie et al. 2008). The guidance provided in Mackie et al. (2008) has been used for more than 20 mussel relocations in Canada (e.g., Argyle Street Bridge in Caledonia, Ontario; Natural Resource Solutions Inc. 2021).

## 2. ASSESS THE PROBABILITY OF ACHIEVING THE FUNDAMENTAL AND MEANS OBJECTIVES

After describing the context of imperilment and identifying the fundamental and means objectives for the focal species, the next steps are to: 1) assess the probability that achieving the means objective will improve survival or recovery of the species (i.e., achieve the fundamental objective); and, 2) assess the probability of achieving the means objective. Assessing the probability that achieving the means objective will fulfill the fundamental objective requires identifying how removals will change the viability of source populations, the degree to which translocated individuals will improve species viability at the recipient location, and the degree to which the establishment of a new population or improved recruitment of an existing population will improve the viability of species relative to the change in viability from removals of the source population(s; Figure 6).

Below, a summary of general approaches for predicting changes in the viability of source and recipient populations resulting from conservation translocations is provided that is relevant across species and means objectives. Following this summary, factors that may influence the ability to achieve the means objectives, and how to quantify these factors, are discussed (Figure 3). The final section provides an approach for assessing the likelihood of confounding factors that reduce the ability to achieve the means objective. The review of quantitative approaches in this section is not meant to be exhaustive, but to provide an understanding of the questions and considerations underlying these approaches and their use when assessing the potential ecological benefits and risks resulting from conservation translocations.

### 2.1. Estimate the probability that achieving the means objective improves survival or recovery of the species

Improving survival or recovery of SARA-listed species using conservation translocation can be achieved in multiple ways (i.e., three means objectives), but ultimately, each approach can be broadly linked to the need to attain long-term viability of the species in the wild (Figure 6). Achieving long-term viability requires an adequate number of populations characterized by an adequate number of individuals, where rates of recruitment and survival are greater than or equal to rates of mortality. Understanding the adequate number of populations for the species, and abundance of individuals within those populations, can help determine which conservation translocation approach(es) to use for achieving the fundamental objective. Developing this quantitative understanding is difficult for SARA-listed freshwater fishes and mussels and requires species and habitat data, which can come from laboratory or field studies, along with simulations and modelling approaches.

Simulations can demonstrate how the number of populations, and the persistence of those populations, affects overall long-term species viability (Figure 7). For example, the probability of species extinction ($P_{Sp}$) can be estimated as a function of the total number of populations ($n$), the persistence of each population ($P_{pop,i}$; extirpation probability for population $i$), and the correlation matrix between the persistence of populations ($\boldsymbol{\rho}$):

$$P_{Sp} = \prod_{i=1}^{n} P_{pop,i}^{\frac{n}{\boldsymbol{d}^{\mathsf{T}}\boldsymbol{\rho}\boldsymbol{d}}}$$

(Figure 7; van der Lee and Koops 2020). For a species with 10 populations, where each population has a 5% probability of extirpation that is independent from other populations, the probability of species extinction is calculated as the product of the 10 probabilities (i.e., 0.05^10; $9.77 \times 10^{-14}$); reduce the number of populations to two and the probability of species extinction drastically increases (Figure 7a). Incorporating correlation structure in the probability of extirpation among populations increases the probability of species extinction relative to independent populations (Figure 7b, c). Populations typically share sensitivities to particular threats (e.g., habitat stressors) or may be geographically close and, therefore, if a threat were to cause extirpation in one population, a positively correlated extirpation probability would suggest that a second population is now more likely to be extirpated than it would be based on the hypothesis of independence. Moreover, the correlation structure among populations can significantly alter inference about the relationship between the number of populations and probability of species extinction (Figure 7). Relatively small reductions in extinction probability per population are observed when populations have a strict correlation structure of 0.5 (Figure 7b), whereas the benefits of adding populations are greater following a first-order autoregressive model (AR1; Figure 7c). An assessment of current and future threats, geographic proximity, and environmental stochasticity among populations can help inform the most suitable correlation structure for the focal species.

Population viability analysis (PVA) is a tool that can be used to estimate the effects of adding or removing individuals on recipient and source populations. Broadly describing a large set of demographic models that vary in complexity, PVAs are used to understand how the structure of populations change over time. Stochastic matrix population models are the most used type of PVA, which incorporate species- or population-specific vital rates to project age- or stage-specific estimates of abundance over time. Based on repeated simulations, stochastic matrix population models generate estimates of long-term population growth rates and the probability of extinction, and when combined with an analysis of elasticity (Vélez-Espino et al. 2006), provide an assessment of the relative importance of ages or life-stages on population persistence (Fieberg and Ellner 2001). Population viability analyses are foundational components of recovery potential assessments for SARA-listed freshwater fish species, being used to inform species recovery targets, minimum viable population sizes, and generate estimates of allowable harm (Vélez-Espino and Koops 2009; van der Lee and Koops 2020; van der Lee et al. 2020), and can be further used to inform the decision of which conservation translocation approach to use for the species in question.

Population viability analysis can be used to assess the viability of source and recipient populations resulting from conservation translocations and compare across scenarios with different means objectives. A study by Fisheries and Oceans Canada (Lamothe et al. 2021) used PVA to quantitatively evaluate scenarios for re-establishing an Eastern Sand Darter population in Ontario. Each scenario was considered based on its risk of causing source population extirpation against the probability of success (Figure 8a), where success was defined as the persistence of a reintroduced population with an abundance greater than the estimated minimum viable population size over the last 15 years of the simulation. For example, considering a high rate of translocation mortality (70%), the potential for Allee effects, and a population growth rate in the recipient habitat of 2.13, it was estimated that approximately 550 individuals need to be removed and translocated pre-spawn for five years from a source population of 20,000 individuals to achieve reintroduction success with a low probability of source population extirpation (≤ 1%; Figure 8b). If population growth rate in the recipient habitat was equal to 1.56, 863 individuals would need to be removed annually and translocated for five years from a source population of nearly 50,000 individuals to achieve the thresholds of ≥ 90% probability of successful establishment and ≤ 1% probability of extirpation (Figure 8b).

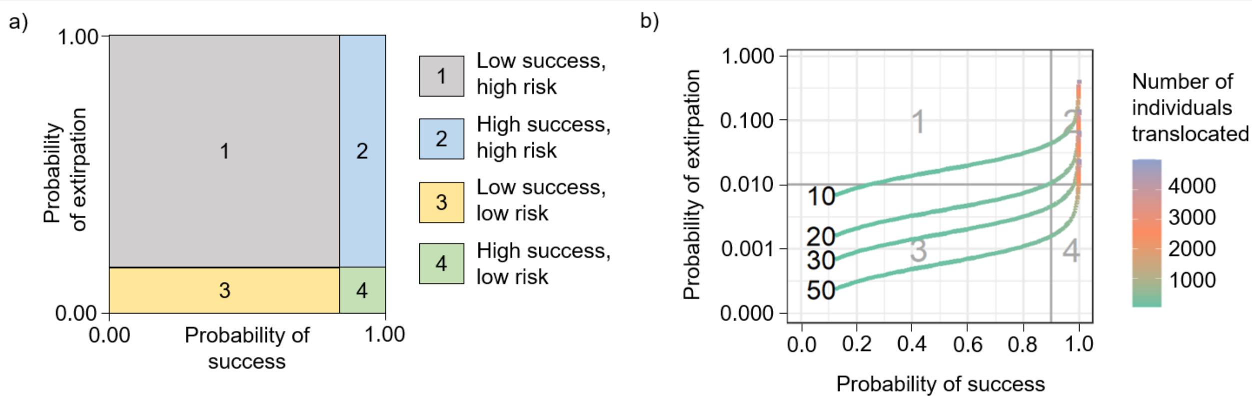


*Figure 8. a) Conceptual framework for assessing the probability of source population extirpation versus achieving the means objective (probability of success). The asymmetrical quadrants represent weighted ecological benefits and risks, where an optimal outcome is defined by a much higher probability of success than source population extirpation. b) Simulated probability of source population extirpation (log-scale) for various source population abundances (10 = 10,000, 20 = 20,000, 30 = 30,000, 50 = 50,000 individuals) versus the probability of successful translocation. Presented are the results when removing individuals for five consecutive years pre-spawn from a source population and releasing them immediately, where source and recipient populations show population growth rate equal to 2.13. Probability of success is defined as maintaining an adult population after reintroduction with a geometric mean population size greater than the minimum viable population size (95%) over the last 15 years of the simulation. The grey boxes and numbers represent the boundaries of the cost-benefit outcomes with an optimal outcome of ≤ 1% probability of extirpation for a ≥ 90% probability of success indicated by the number 4. Original figure presented in Lamothe et al. (2021).*

## 2.2. Identify factors that may influence the ability to achieve the means objective

There are many factors that can influence the ability to achieve the means objectives and, therefore, fulfill the fundamental objective of improved survival and recovery for the species. These factors can be broadly grouped into four categories: population, habitat, community, and threat considerations. Below, a summary of these factors is provided, including how each can influence the ability to achieve the means objective and quantitative approaches for assessing their influence. Following this summary, a qualitative scoring approach is presented to assist in evaluating the likelihood of these factors influencing the ability to achieve the means objective.

### 2.2.1. Population considerations

Populations can have unique responses to conservation translocations, which can therefore affect the ability to achieve the means objectives. In some instances, there will be few, or only one, options for selecting source populations, whereas in other scenarios, there may be several. When multiple populations are available, consideration is needed regarding the potentially unique responses to conservation translocations (i.e., ecological or adaptive differences) that individual populations may have.

Ecological and evolutionary theory are the foundations for conservation-based source population selection (Meffe 1995; Houde et al. 2015). Three related, high-level approaches have been suggested for selecting source populations that require knowledge of species habitat requirements: **ancestry matching**, **environmental matching**, and the use of **adaptive potential** (Figure 9; Houde et al. 2015). Ancestry matching, which is the preferred approach, describes the selection of source populations that share genetic similarity to the extant or historically present population. This assumes that shared genetic variation will confer adaptation to environmental conditions at the recipient site of translocations. Environmental matching describes the selection of a source population(s) based on the similarity of habitat conditions between recipient and source locations, following similar logic as the ancestry matching approach. Finally, the adaptive potential approach seeks to translocate individuals from multiple populations to provide the best potential to adapt to conditions at the recipient site (i.e., experimental population; Kreuger et al. 1981; Figure 9).

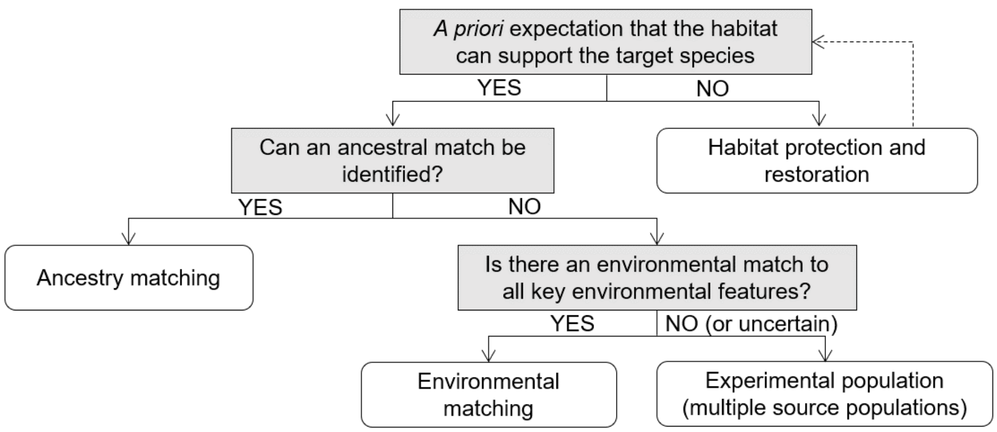


*Figure 9. A source population selection framework for conservation translocation that requires knowledge of abiotic and biotic requirements for the species being considered for translocation.*

Generally, an ancestry matching approach is considered the most likely to succeed with the least uncertainty, whereas environmental matching and the use of adaptive potential should only be considered when an ancestral match no longer exists (Figure 9). Situations may arise where a population has been extirpated for a significant amount of time, preventing opportunities to determine the ancestral relationship between the extirpated population and extant source populations. Rather than simply choosing an environmental match or resorting to the use of an experimental population, consideration of the biogeographical patterns from non-related taxa with similar distributions can provide insight on the best approach for source population selection. For example, reintroduction of the Spotfin Chub (*Erimonax monachus*) was being considered for Shoal Creek in the middle Tennessee River drainage of Alabama and Tennessee, USA in 2004 (George et al. 2009). Multiple options were available for source populations that differed in geographic proximity to the historic site. One potential source population was from the Buffalo River, a relatively proximate location within the same lowland physiographic province (Highland Rim province). Another potential source was further upriver in a different physiographic province (Emory River; Cumberland Plateau province). Due to a lack of genetic information on the extirpated Spotfin Chub population, managers assessed patterns of biogeography of other fishes (e.g., Coppercheek Darter *Etheostoma aquali,* Tennessee Darter *E. tennesseense*, Duck Darter *E. planasaxatile*) in the Tennessee River and connected tributaries. Based on genetic relationships among these and other species, it was determined that populations and species in Shoal Creek were more closely related to those in the Emory River than populations and species in the geographically proximate Buffalo River. As a result, the Emory River population was chosen as a source for Spotfin Chub reintroduction (George et al. 2009).

Species monitoring programs can help to characterize population specific life-history parameters that can be used to inform the selection of individuals for translocation. Life-history characteristics can inform decisions around the frequency of introductions and number of individuals needed to achieve a means objective. For example, efforts to reintroduce fish species with a periodic life-history strategy (i.e., late maturation, large body size, and large home range) likely require more individuals for translocation to achieve a means objective than for species with an opportunistic life-history (i.e., early maturation, small-bodied, short life-span, narrow geographic range; Winemiller and Rose 1992; George et al. 2009). Moreover, an understanding of local life-history characteristics (i.e., population growth rate, fertility) can inform the probability of source population extirpation if removals are too extreme (e.g., population growth rate is < 1 or if relatively few breeders exist), and be informative for deciding whether to translocate wild individuals or initiate captive breeding efforts. Finally, conservation translocations inherently involve the movement of genes and individuals with potentially unique adaptations (Weeks et al. 2011; Keller et al. 2012). Failing to consider genetic structure of populations could result in reductions in fitness of source populations and (or) the inability of translocated individuals to survive in the recipient habitat (e.g., lack of local adaptation). Overall, a thorough understanding of species abundance, distribution, genetic structure, and life-history strategies can reduce uncertainty and help inform conservation translocation decisions for SARA-listed species, ultimately improving the likelihood of achieving the means objective.

### 2.2.2. Habitat considerations

A sufficient quantity of productive habitat for a species is needed when considering conservation translocation efforts (Maitland and Lyle 1992; Galloway et al. 2016). Failing to translocate individuals to areas with optimal habitat reduces the probability of survival for translocated individuals and, therefore, reduces the likelihood of achieving the means objective. As defined under SARA (2002), habitat for aquatic species includes all spawning grounds, areas for nursery, rearing, food supply, migration, and any other areas on which aquatic species depend directly or indirectly to carry out their life processes, or areas where aquatic species formerly occurred and have the potential to be reintroduced. Moreover, microhabitats are an important consideration for SARA-listed freshwater fish and mussel species (Bolland et al. 2010), and for their obligate host species, which may rely on particular microhabitat features (e.g., substrate types, pool habitat).

Improvements in data availability and computing technology over the last few decades have led to the development of numerous tools for predicting species distributions based on multiple measured and projected spatial and environmental predictors (Elith and Leathwick 2009; Zurell et al. 2020). Broadly termed '**species distribution models**', these models use species occurrence data and environmental variables gathered from across a spatial landscape to reconstruct the native range or present-day extent of occurrence for a species. The first step in building a species distribution model is to conceptualize the study and gather spatially-explicit biodiversity (e.g., presence/absence) and environmental data hypothesized to be important to the species (Figure 10). Next, model algorithms are chosen (Elith and Graham 2009), fit to the data, and assessed for performance (Mouton et al. 2010). Finally, the models are used to make predictions on the distribution of species in space and time (Zurell et al. 2020; Figure 10). Generating these predictions can provide insight into potentially suitable sites for translocation, how site conditions and distributions might change over time, or, potentially, areas where the species has yet to be detected.

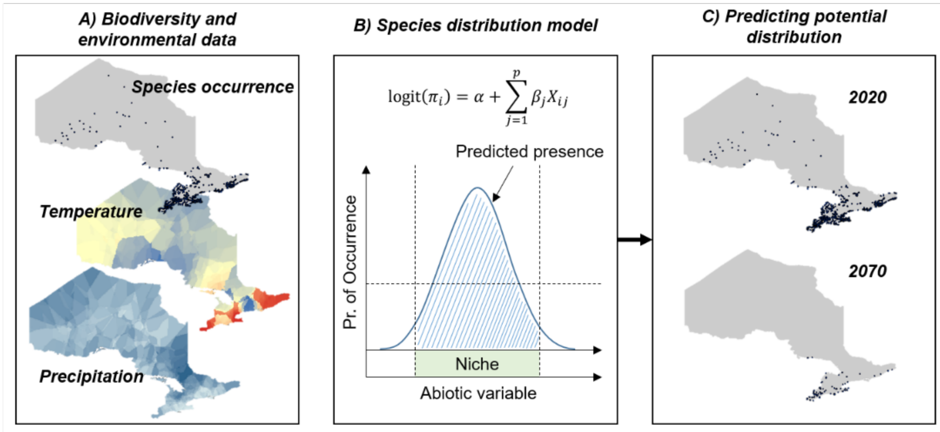


*Figure 10. Conceptual representation of the process of species distribution modelling for a hypothetical species in Ontario, Canada. A) Biodiversity data, here species occurrence data, are gathered alongside environmental covariates (temperature and precipitation) hypothesized to affect the species' distribution. B) The data are used to generate a generalized additive model, which considers the relationship between the biodiversity data and the environmental covariates. C) Model-based predictions of the present and future distribution of the species can be made and mapped. Figure modified from Zurell (2020).*

### 2.2.3. Community considerations

The biotic composition of recipient ecosystems can influence the likelihood that conservation translocations achieve their intended outcomes of improved survival or recovery (i.e., Figure 6). Understanding the significant biotic interactions for candidate species can help ensure that recipient sites meet biotic expectations (Letty et al. 2007). Interactions between species can broadly be described as negative, including parasitism, competition, predation, amensalism, or hybridization, or positive, including mutualisms or commensalisms. Each of the interaction types are described below with respect to how they can affect the probability of achieving the means objectives. A brief summary of methods to quantify biotic interactions follows.

#### *2.2.3.1. Parasitism*

Parasitism describes the situation where the focal species benefits from the presence of a co-occurring species, while harming the co-occurring species in the process. Freshwater mussels (order Unionoida) are unique in that the larvae are obligate parasites on host species (e.g., freshwater fishes; Haag and Stoeckel 2015), meaning that freshwater mussels are unable to complete their life cycle without the exploitation of a host. This is exemplified by the extirpation of Dwarf Wedgemussel (*Alasmidonta heterodon*) in New Brunswick, Canada. The extirpation of Dwarf Wedgemussel was primarily caused by the elimination of its host (American Shad *Alosa sapidissima*) after construction of the Moncton-Riverview causeway in 1968 (Fisheries and Oceans Canada 2007). Without the presence of suitable hosts, conservation translocations for freshwater mussels will fail to achieve their intended objectives.

#### *2.2.3.2. Competition and predation*

Freshwater species often share habitat and resource requirements leading to potentially negative interactions between individuals (Larkin 1956). Competitive interactions describe the situation where species vie for similar resources, which can include predation by the competitors on each other or competition for survival under environmental conditions that differentially affect the competitors (Larkin 1956).

Predation is commonly identified as a mechanism structuring freshwater communities and can have a significant effect on the outcome of conservation translocations. Freshwater mussels typically live in aggregated multispecies assemblages, which have been hypothesized to benefit individual mussels by reducing the risk of predation through dilution and (or) encounter effects (Wilson et al. 2011). Aggregations of individuals reduce the probability of predating on any one mussel (i.e., dilution effect; Wrona and Dixon 1991) and also reduce the attack efficiency of the predator post encounter as a result of increased vigilance of the group relative to that of solitary living (Wilson et al. 2011). Predators of freshwater mussels include certain fishes (e.g., Freshwater Drum *Aplodinotus grunniens*; Redhorse species *Moxostoma* spp.) and mammals (e.g., Muskrat *Ondatra zibethicus*; Raccoon *Procyon lotor*). Instances of mussel predation are typically sporadic, localized, and rarely witnessed (Haag and Warren Jr. 1998). The presence of middens along the stream bank can provide indirect evidence of mammal predation on freshwater mussels (e.g., characteristic claw marks on shells). Ultimately, an understanding of the present-day predator dynamics in the source and recipient ecosystems can provide opportunities to control, or reduce the likelihood of, predation events affecting recipient sites. For example, in-stream enclosures were built for performing reintroductions of freshwater mussels in the Pigeon River, North Carolina, USA, which allowed researchers the ability to more easily monitor translocated individuals, while limiting the effects of predation (e.g., Rooney 2010).

A literature exists on the effects of predator introductions on freshwater ecosystems, including the introduction of non-native fishes (e.g., Smallmouth Bass *Micropterus dolomieu;* Loppnow et al. 2013; salmonids; Krueger and May 1991; Eby et al. 2006). Included in these effects are increased predation on native fishes, increased competition for resources between species, environmental modifications of benthic substrates (e.g., through nest building or feeding), the introduction of parasites and disease, along with direct and indirect genetic effects (Krueger and May 1991; Jackson et al. 2001). Moreover, an abundance of predators can limit the number of small-bodied fishes, alter habitat use of co-occurring species, and influence foraging behaviour, potentially forcing the co-occurring species into less desirable locations or situations (Jackson et al. 2001). Failed efforts to reintroduce Lahontan Cutthroat Trout (*Oncorhynchus clarkii henshawi*) into Fallen Leaf Lake, California, USA were attributed to predation by non-native salmonids as the primary limiting factor, with the two observations of Lahontan Cutthroat Trout being in less-desirable nearshore, epilimnetic refugia (Al-Chokhachy et al. 2009). Similarly, the invasive Western Mosquitofish (*Gambusia affinis*) has been identified as a significant predator and competitor to the endangered Barrens Topminnow (*Fundulus julisia*) in Tennessee, USA, limiting the success of long-term reintroduction efforts (Ennen et al. 2021). Ultimately, elevated rates of predation on translocated individuals will reduce the probability of achieving the means objectives.

#### *2.2.3.3. Amensalism*

When the presence of a co-occurring species negatively affects the persistence of the focal species, but the presence of the focal species has no net effect on the co-occurring species, the interaction is considered amensalism. Amensalism can result in a unidirectional displacement of the focal species and is most commonly acknowledged in the context of species invasions (e.g., deprivation of focal species resources by invader). The presence of a species with an amensalistic relationship to the focal species of conservation concern will significantly reduce the likelihood of fulfilling the means and fundamental objectives.

#### *2.2.3.4. Inter- and intra-specific hybridization*

Hybridization describes the act of reproduction between genetically distinct individuals or species (Barton and Hewitt 1985). Hybridization reduces native genotypes, can alter genetic variation and break down gene complexes coadapted to local environments, and can ultimately lead to the loss of genetically pure native individuals (Simberloff 1996; Huxel 1999; Storfer 1999). The Westslope Cutthroat Trout (Alberta population) illustrates the challenges of introgressive hybridization for SARA-listed species, as the greatest threat to this population is hybridization with Rainbow Trout (*Oncorhynchus mykiss*) and Yellowstone Cutthroat Trout (*Oncorhynchus clarkii bouvieri*; Fisheries and Oceans Canada 2019). Like other threats, hybridization limits the likelihood of achieving the fundamental and means objectives for SARA-listed freshwater fish or mussel species when performing conservation translocations.

#### *2.2.3.5. Positive interactions*

Positive interactions in freshwater ecosystems are studied less often than negative interactions (Holomuzki et al. 2010; Silknetter et al. 2020); however, ignoring these relationships could lead to the inability to achieve the means objectives. A mutualistic interaction suggests that the presence of one species benefits the focal species, and vice versa. A commensalistic interaction indicates that the presence of a co-occurring species benefits the focal species, but the presence of the focal species has no net effect on the co-occurring species. For example, Redside Dace (*Clinostomus elongatus*; Endangered) is a nest associate, which is a common reproductive strategy in small-bodied freshwater fishes where the species spawns in active nests built by co-occurring species. Although Redside Dace capitalizes on the presence of the nest-building species, the building of nests and active guarding of eggs by co-occurring species would occur with or without the presence of Redside Dace eggs. Furthermore, nest-building species may benefit from the presence of nest associates and their eggs through dilution effects, whereby rates of predation on nest-builder eggs is reduced as a result of the increased abundance of eggs from co-occurring species (Silknetter et al. 2019).

Experimental manipulations of co-occurring cyprinid abundance (Yellowfin Shiner *Notropis lutipinnis*) with a nest associate (Bluehead Chub *Nocomis leptocephalus*) in South Carolina, USA suggests that the nest association relationship is heterogenous and context dependent; a transition between mutualism and parasitism was observed between Yellowfin Shiner and Bluehead Chub as the abundance of Yellowfin Shiner (i.e., the nest-building species) diminished with a constant number of Bluehead Chub (Silknetter et al. 2019). Whether this relationship is considered mutualistic or parasitic, the presence of co-occurring, breeding species is required for the persistence of nest associates if translocation were to occur. Without the presence of the co-occurring species, the probability that the means objective will be achieved is significantly reduced.

#### *2.2.3.6. Quantifying biotic interactions*

The probability of achieving the means objectives should be informed by the potential effects of interspecific biotic interactions, particularly when considering reintroduction, mitigation translocation, or assisted colonization. The goal of exploring co-occurrence and abundance relationships between species is to rationalize the suitability of the biotic community for the translocation of individuals. There is an increased probability of achieving the means objective at recipient sites when species composition reflects expectations from extant, thriving populations or suspected biotic composition prior to species reduction or extirpation. Alternatively, a heightened presence of negatively associated species, or the lack of obligate or beneficial interactions, can restrict the ability to achieve improvements in recruitment, establish a population, or rescue the species, therefore limiting the ability to achieve the fundamental objective.

The development of methods for characterizing biotic interactions is an evolving field, made challenging by the simultaneous interactions of abiotic and biotic factors within continuously changing environments. Data for assessing biotic interactions between freshwater taxa are most often observational, as experiments in the laboratory are difficult and direct manipulation in the field is rare (Mittelbach et al. 1995). Although lacking complexity relative to the wild, experiments have been instrumental in developing an understanding of predator-prey relationships (Mills et al. 2004), competitive interactions (Hart et al. 2018), and the relationships between parasite and host (Österling and Larsen 2013).

Rather than direct interactions between species, analyses based on observational data are typically describing species co-occurrence patterns (i.e., species associations), which may or may not be direct, and can be positive, negative, or independent. Statistical methods exist for quantifying associations using observational data, such as incorporating interactions as predictors of species occurrence in linear models (Hein et al. 2013), applying a probabilistic approach to determine significant pairwise co-occurrence patterns (Veech 2013), performing multivariate ordinations with community data for identifying positive, negative, or neutral relationships between species (Hinch et al. 1991), and building joint hierarchical multispecies models that assess the simultaneous response of co-occurring species to environmental factors (Pollock et al. 2014; Dormann et al. 2018; Rodríguez et al. 2021). The goal of using these approaches is to evaluate the suitability of the recipient biotic community for translocating the focal species, informing whether community composition will likely influence the ability to achieve the means objective.

#### 2.2.4. Threat considerations

The most frequently cited reason for unsuccessful conservation translocations is failure to reduce or eliminate the original cause of species reduction or extirpation (Armstrong and Seddon 2008; Bolland et al. 2010; Thomas et al. 2010; Cochran-Biederman et al. 2014; McMurray and Roe 2017; Bubac et al. 2019). There is a need to assess how present-day and future threats could influence the viability of source and translocated individuals, and therefore, how threats may affect the probability of achieving the means objectives for the species (Figure 6). Threats to the survival and recovery of freshwater fishes and mussels in Canada have been well-described and include habitat degradation and destruction, flow modifications, invasive species, and pollution, among others (Dextrase and Mandrak 2006; Dudgeon et al. 2006; McCune et al. 2013; Reid and Parna 2017; Reid et al. 2019). Threat summaries in species recovery strategies, recovery potential assessments, management plans, and COSEWIC reports can provide information on the relative influence of threats on populations and species.

Individuals, populations, and species can demonstrate unique responses to abiotic conditions (e.g., temperature) that can limit species abundance and distributions. Species response curves depict the relationship between population state variables (y-axis) and environmental gradients or stressors (x-axis; Figure 11). Population state variables include the probability of occurrence, often informed by observational data, or the probability of survival, often informed by experimental data. Knowledge of the shape of species response curves can help assess whether potential recipient sites of conservation translocations have sufficient conditions to support the translocation of individuals.

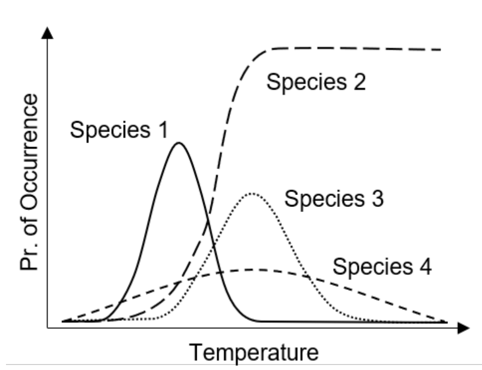


*Figure 11. Example of species response curves. Plotted is the hypothetical probability of occurrence for four species against temperature.*

Developing species response curves from field data can be challenging, but experimental approaches are promising. For example, species response curves to salinity, pH, and temperature were developed for Atlantic Whitefish and used to make recommendations on site selection for conservation translocations (Cook et al. 2010). Similarly, the effects of hypoxia on critical thermal maximum (i.e., the temperature at which a fish loses equilibrium in response to acute temperature stress) across a thermal gradient was measured for Pugnose Shiner (*Notropis anogenus*; Threatened; Potts et al. 2021). Experiments with wild Redside Dace have demonstrated population-specific thermal tolerances across the species range (Leclair et al. 2020; Turko et al. 2020), suggesting a need to understand the link between source and recipient environmental conditions in future reintroduction efforts (Turko et al. 2021). Using observational data and experimental trials, Douda (2010) demonstrated unique responses of freshwater mussel species to nitrate nitrogen (N-$NO_3^-$) concentrations in the Lužnice River, Czech Republic. Although they are informative for predicting conservation translocation success, there are only a few examples of experimentally informed species response curves for SARA-listed freshwater species.

## 2.3. Estimate the ability to achieve the means objectives

In this step, the potential influence of confounding factors on the likelihood of achieving the means objectives are assessed based on best available data and knowledge (i.e., evidence). Sources of evidence include observations from monitoring programs, field and laboratory experiments, the scientific literature, management and recovery documents, existing assessments, and expert opinion (Karasov-Olson et al. 2021).

Table 1 provides a series of considerations that can each be assigned a likelihood for achieving the means objective. Given the general data limitations for SARA-listed freshwater fishes and mussels, a qualitative approach for scoring the influence (or likelihood) of factors on achieving the means objective is presented that considers the uncertainty associated with a factor based on the available evidence (Galloway et al. 2016; Karasov-Olson et al. 2021). The likelihood of achieving the means objective can be categorized as low, medium, high, or unknown for each row in Table 1. The definitions of each level will depend on the factor being considered. For example, the likelihood of achieving the means objective when considering Row 1, "*abundance of the source population is suitable to achieve the means objective*," might be characterized as:

- **Low**: There are few reproducing adults in the source population. Removing individuals could threaten the persistence of the source population, and given the few individuals, may not be successful in achieving the means objective at the recipient location;

- **Medium**: Source population abundance is coarsely described, but suggests enough removals can be performed to achieve the means objective in the recipient habitat;
- **High**: Source population abundance is well-described with a sufficient number of breeding individuals and is capable of withstanding removals for translocations; or,
- **Unknown:** Information on source population abundance is insufficient for informing likelihood estimates.

*Table 1. Considerations for evaluating the ability to achieve the means objective using conservation translocation. For each row, a likelihood of achieving the means objective is assigned (low, medium, high, unknown) and the evidence strength (limited, medium, robust), agreement between evidence sources (low, medium, high), and overall confidence in that likelihood assignment are scored. Confidence is the outcome of the evidence strength and agreement scores (Figure 12).*

***Focal species**:*
***Problem statement**:*
***Fundamental objective**:*
***Means objective**:*

| *Category* | *Factors* | *Likelihood* | *Evidence strength* | *Agreement* | *Confidence* | *References* | *Additional considerations* |
|---|---|---|---|---|---|---|---|
| Population considerations | Abundance of the source population(s) is suitable to achieve the means objective | - | - | - | - | - | - |
| | Age-structure of the source population is suitable to achieve the means objective | - | - | - | - | - | - |
| | Genetic diversity and variation of the source population is suitable to achieve the means objective | - | - | - | - | - | - |
| | Genetic diversity and variation of the recipient population is suitable to achieve the means objective | - | - | - | - | - | - |
| | Life-history strategy of the source population is suitable to achieve the means objective | - | - | - | - | - | - |
| | Captive breeding or captive rearing techniques are available to achieve the means objective | - | - | - | - | - | - |
| Habitat | Habitat in the recipient site(s) reflect species requirements (e.g., water clarity, water velocity, depth, vegetation, substrate) | - | - | - | - | - | - |
| | A sufficient quantity of habitat exists in the recipient location to support all life-stages | - | - | - | - | - | - |
| | Sufficient connectivity exists in the recipient habitat to support all life-stages | - | - | - | - | - | - |
| Community considerations | Obligate, facultative, or parasitic species dependencies limit the ability to achieve the means objective | - | - | - | - | - | - |
| Threats | Pertinent threats limit the ability to achieve the means objective, including: | - | - | - | - | - | - |
| | Invasive species | - | - | - | - | - | - |
| | Residential and commercial development | - | - | - | - | - | - |
| | Agriculture and aquaculture | - | - | - | - | - | - |
| | Energy production and mining | - | - | - | - | - | - |
| | Biological resource use | - | - | - | - | - | - |
| | Transportation and service corridors | - | - | - | - | - | - |
| | Human intrusions and disturbance | - | - | - | - | - | - |
| | Natural systems modification | - | - | - | - | - | - |

| *Category* | *Factors* | *Likelihood* | *Evidence strength* | *Agreement* | *Confidence* | *References* | *Additional considerations* |
|---|---|---|---|---|---|---|---|
| Threats (cont.) | Pollution | - | - | - | - | - | - |
| | Geological events | - | - | - | - | - | - |
| | Parasites or disease transfer | - | - | - | - | - | - |
| | Future habitat modification or degradation | - | - | - | - | - | - |
| | Climate change | - | - | - | - | - | - |
| | Other factors | - | - | - | - | - | - |

Confidence in the anticipated effects of the factors in each row of Table 1 is based on the type, quality, and quantity of evidence sources and the agreement between sources (Figure 12). Evidence can be described as limited, medium, or robust, where sources of information are in low, medium, or high agreement (Figure 12). Confidence in the influence of a factor on achieving the means objective increases with increasing quality and quantity of evidence and agreement between sources of evidence (Figure 12).

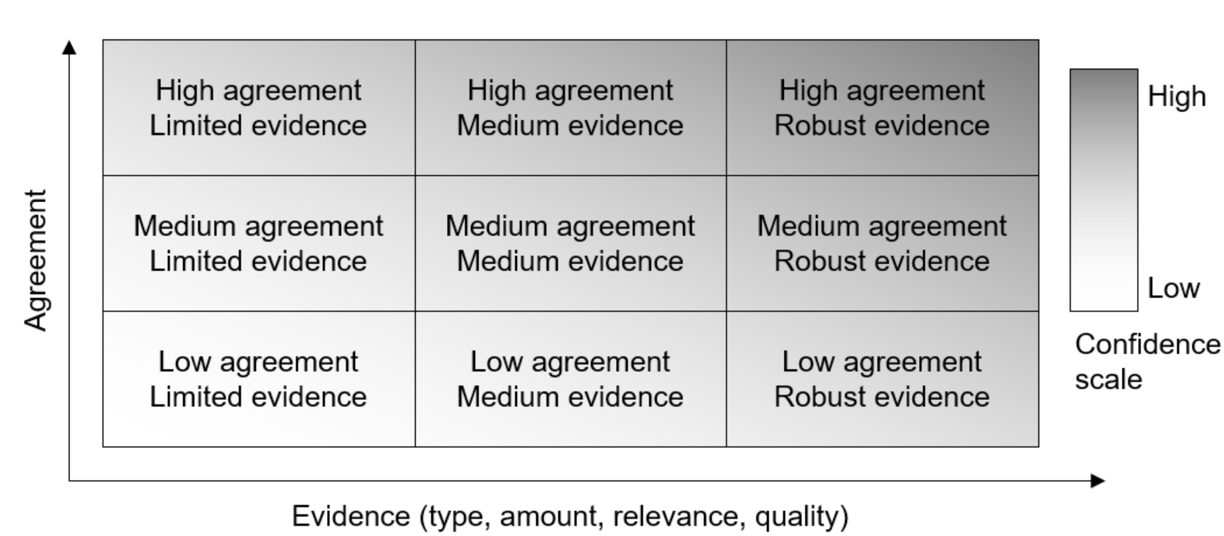


*Figure 12. Conceptual framework describing confidence in the evidence for predicting the potential effects of risk factors when performing conservation translocations. Confidence increases with the quality and quantity of evidence and the agreement of that evidence. Modified figure originally presented in IPCC (2013) and later adapted in Karasov-Olson et al. (2021).*

Continuing with the example, evidence for assessing if "*abundance of the source population is suitable to achieve the means objective*" could be described as:

- **Limited**: No primary literature or data from local monitoring of population abundance;
- **Medium**: Some data available on population abundance from local monitoring or experiments. Data could be outdated; or,
- **Robust**: Time-series data, including recent data, from monitoring efforts and several scientific articles in the primary and grey literature.

The agreement between sources of information can be then be characterized as:

- **Low**: Sources describe contradictory findings;
- **Medium**: Sources describe similar findings, but with differing magnitudes of effects; or,
- **High**: Sources are consistent with findings.

For each row in Table 1, evidence is compiled, reviewed, evaluated, and a likelihood is assigned. Ultimately, Table 1 should be used to qualify the potential improvement in survival or recovery when performing conservation translocations. This understanding of the ability to achieve the means objective will be used in Step 4 for making a decision, where the potential benefits of achieving the means and fundamental objectives are weighed against the potential ecological risks.

# 3. ASSESS THE ECOLOGICAL RISKS OF PERFORMING CONSERVATION TRANSLOCATIONS

Although conservation translocation is intended to benefit SARA-listed freshwater fishes and mussels, translocation could result in negative consequences (i.e., ecological risks) on the focal species or other ecosystem components (Bradford 2017). Below, the potential ecological risks of negative consequences when performing conservation translocations are reviewed. Following this review, a framework for scoring the relative ecological risks of the translocation is provided.

## 3.1. Identify the ecological risks of performing conservation translocations

There are ecological risks of performing conservation translocations. The scale of the potential consequences can vary from short-term, or systemically minor, to systemically transformative with consequences occurring within or beyond the site of translocation. Moreover, negative consequences of conservation translocations could be experienced by other ecosystem components (e.g., non-SARA species) even when the fundamental objective of improved survival or recovery of the SARA-listed species is achieved. The ecological risks of conservation translocations are presented below, organized as the potential reduction in source population persistence, change in source population genetic variation, change in recipient population genetic variation and persistence, change in short- and long-term community and ecosystem dynamics in the source and recipient ecosystems, and the transfer of disease to the recipient population and ecosystem.

### 3.1.1. Reduction in source population persistence

The worst possible outcome of a conservation translocation for the source population is extirpation. Therefore, understanding how the number and frequency of individuals removed from source populations will affect population persistence is a critical step prior to initiating translocation efforts. Consistent with advice regarding allowable harm of species (Fisheries and Oceans Canada 2004; Vélez-Espino and Koops 2009), removals of individuals to supplement populations should not jeopardize the survival or recovery of the SARA-listed species and, therefore, should only occur from robust (i.e., growing or large) populations. Removing too many individuals from a source population could immediately threaten long-term source population persistence. Moreover, a small population size resulting from excessive removals for translocations increases the potential for genetic effects, potentially reducing adaptive potential of wild source populations (expanded in the next section). Removing too few individuals from a source population, however, could lead to failed translocations in the recipient habitat while imposing unnecessary harm to the source population and, by extension, the species. If significant removals are expected to harm the source population, captive breeding or rearing can be considered to reduce impacts of removals on source populations.

### 3.1.2. Change in source population genetic variation

Maintaining or enhancing local genetic diversity and quality is a priority for conservation translocation efforts (Minckley 1995; Neff et al. 2011; Weeks et al. 2011; Malone et al. 2018). Sourcing individuals from the wild for conservation translocations risks the loss of source genetic variation, particularly when important breeding individuals or large numbers of individuals are removed from relatively small populations (a concern for most SARA-listed species). Rates of mating between closely related individuals can become elevated if too many individuals are removed from the wild, threatening **inbreeding depression** among the individuals remaining in the source habitat, and possibly reducing long-term adaptive potential. Inbreeding depression describes the relative reduction in fitness of offspring resulting from the mating of closely related individuals compared to those of randomly mated individuals (Hedrick and Kalinowski 2000). Inbreeding can increase the probability of potentially deleterious mutations being present together within populations, thus resulting in reduced fitness. A literature on the effects of inbreeding and how to avoid these effects has been developed (e.g., Ryman and Laikre 1991; Wang et al. 2002; Neff et al. 2011; Rollinson et al. 2014). This literature is particularly relevant when considering the use of captive breeding for sourcing conservation translocations (Fraser 2008). Overall, it is unlikely that removals of individuals from the source population will lead to inbreeding unless source population abundance or genetic variation is very low, emphasizing the need to assess population abundance and genetic diversity prior to removals.

Genetic evaluation and monitoring of populations have become increasingly common, including for SARA-listed species. Quantifying source population genetic diversity and effective population size can help inform decisions on which populations are most suitable for reintroduction and how many individuals are needed to maintain sufficient genetic variation in captive settings (e.g., Malone et al. 2018; VanTassel et al. 2021). For example, up to 50 individuals and as few as 7 females have been recommended as the minimum number needed to effectively maintain 95% of the genetic variation in endangered or critically endangered species (Weeks et al. 2011; VanTassel et al. 2021). Similarly, ratios of effective population size and adult census size have been used to approximate the minimum number of freshwater fish needed for conservation translocation over time while maintaining comparable levels of genetic diversity relative to the source population (Frankham 1995; Waples 2005; Palstra and Ruzzante 2008; Palstra and Fraser 2012; Frankham et al. 2014; Malone et al. 2018).

Genomics and transcriptomics techniques are becoming more accessible for informing source population selection (He et al. 2016). Genomics refers to the study of the entire genome, or the complete set of DNA within an organism. Transcriptomics describes the study of the RNA produced as a result of gene expression (i.e., transcription). Differential expression of genes in response to environmental stressors can be directly quantified and compared between potential source populations using custom oligonucleotide microarrays (He et al. 2015). For example, microarrays were developed to evaluate the differential expression of genes between two Atlantic Salmon (*Salmo salar*) strains translocated to Lake Ontario and predict the relative success of translocation for the two strains (He et al. 2015). Similarly, RNA sequencing was used to investigate the effects of different contaminants on gene expression in the Freshwater Pearl Mussel (*Margaritifera margaritifera*) with the objective of measuring the potential impact of illegal dumping on stress responses of the species (Bertucci et al. 2017). Access to information on the genetic structure of populations and differential gene expression resulting from exposure to environmental stressors can inform conservation translocation management efforts and, therefore, reduce the probability of negative genetic consequences.

#### 3.1.3. Change in recipient population genetic variation and persistence

Genetic concerns for recipient populations are primarily related to the issues with a small founding population when performing reintroductions or assisted colonization, and the mixing of previously disconnected gene pools when performing supplementation or if the introduced individuals disperse from the site of introduction. The **founder effect** describes the reduction in genetic variation incurred as a result of re-establishing a population with a small number of individuals unrepresentative of the species pool. This effect is commonly described for invasive species (Chen et al. 2012) and captive breeding programs (Saavedra and Guerra 1996), and can be of concern for conservation translocations (Hoftyzer et al. 2008; Jamieson 2010). Multiple small translocations of randomly selected (or stratified random selection), unrelated individuals with sufficient genetic diversity from the source to recipient location can reduce the likelihood of experiencing founder effects (Le Gouar et al. 2008; Alcaide et al. 2010).

**Outbreeding depression**, defined as the reduction in fitness of individuals caused by the crossing of individuals from two genetically distinct populations, is a potential consequence of supplementing populations. Outbreeding depression can result from the breakdown of coadapted gene complexes or from the disruption of beneficial interactions between genes and the environment (Templeton 1986; Edmands and Timmerman 2003). The negative effects of outbreeding depression can be observed as demographic and genetic swamping. **Demographic swamping** describes the situation where hybridization is widespread, but the fitness of hybrid individuals is lower than replacement rates for sustaining the population (Todesco et al. 2016). **Genetic swamping** describes the situation where population growth rates of hybrid individuals outpace the recipient population, which can lead to the loss of pure parental genomes (Todesco et al. 2016).

The probability of experiencing outbreeding depression increases with divergence of parental populations (Todesco et al. 2016) and when individuals are translocated to environments with conditions dissimilar to the source habitat. Based on species generation times and the glacial history of Canada, outbreeding depression is unlikely to occur when translocating wild freshwater fishes or mussels (in contrast to certain anadromous fishes) but is of particular concern when wild individuals mate with individuals bred under human care. **Domestication selection** is a common and particularly challenging problem when raising animals in non-native, controlled settings (Huntingford 2004; Hagen et al. 2019), with supplemented populations often demonstrating reductions in effective population size due to a variety of mechanisms (Lennox et al. 2021; Milla et al. 2021). Although genetic diversity can often be maintained for species bred under human care (e.g., Fisheries and Oceans Canada 2018b; Vachon et al. 2019; Wacker et al. 2019; VanTassel et al. 2021), phenotypic variation can lead to observed differences between wild and captive-bred individuals in reproduction and survival post-translocation (Kostow 2004). These observed differences highlight the urgency of conserving wild populations to avoid the need for captive breeding and release.

Experimental crossing of individuals from geographically or genetically distant populations (i.e., common garden experiments) can inform the probability of fitness reductions occurring as a result of mixing potentially divergent populations (Weeks et al. 2011) but may be difficult to observe in artificial environments. In situations where experiments cannot occur, inferences can be made about the potential for outbreeding depression based on information from taxonomic resolution, chromosomal differences between populations, rates of historical gene flow, and differences in environmental characteristics of occupied habitats (Frankham et al. 2011; Weeks et al. 2011). Similarly, this information can inform the potential that the focal species of conservation translocation could overcome interspecific reproductive barriers and hybridize with co-occurring species. The probability of interspecific hybridization occurring is increased when introduced to ecosystems with sister species. Analogous to the effects of crossing divergent populations, interspecific introgressive hybridization can result in changes to both pure parental genomes and a potential reduction in fitness for each species. Overall, consideration of the genetic effects of conservation translocations are required to maximize the probability of achieving the means objective and reducing the risk of negative genetic and fitness-related effects.

#### 3.1.4. Change in community and ecosystem dynamics in source ecosystems

All species play a functional role in local ecosystem conditions, and the removal of individuals could lead to an altered ecological state with new community and ecosystem dynamics. For example, the effects of predator removal from an ecosystem are well-documented, including changes in the dominant species (Stantial et al. 2021) and shifts in food webs (Sieben et al. 2011). Experimental removals of Smallmouth Bass from a moderately sized lake (271 ha) in New York, USA led to an immediate and sustained increase in native littoral fish species abundance for the duration of the six-year study (Weidel et al. 2007). Large removals of freshwater mussels could similarly cause ecosystem-level changes. For example, the removal of freshwater mussels and their replacement with spent shells led to reductions in organic matter and invertebrate biodiversity when compared to when live mussels were present in field experiments (Spooner and Vaughn 2006).

Regardless of the species, the removal of individuals from a source habitat will enable resources for other individuals or species to exploit. If too many individuals of the focal species are removed from a population, there is the risk that other species may capitalize on the open resources and become more dominant in the source population habitat, leading to potentially transformative change beyond expectations from removals alone. Quantifying species co-occurrence patterns and community dynamics, along with simulations of management actions and expert opinion, can inform the probability of significant alterations in community dynamics resulting from removals, and the potential cascading risk of altering ecosystem structures and functions in source habitats. Although the ecological impacts in source communities may be low if SARA-listed species represent a low proportion of total abundance or have a perceived small functional role, it is prudent to identify the potential likelihood of impacts from removals on source ecosystems.

#### 3.1.5. Change in community and ecosystem dynamics in the recipient ecosystems

Conservation translocation of freshwater fishes or mussels could lead to changes in the biotic composition, structure, or function of recipient ecosystems, including other non-focal SARA-listed species. Many studies have documented the effects of recreational or commercial freshwater fish introductions to formerly unoccupied areas (Gozlan et al. 2010; Sandlund et al. 2013), which tend to be top predators (e.g., Cowx and Gerdeaux 2004; Eby et al. 2006). For example, introductions of Westslope Cutthroat Trout for recreational purposes into fishless lakes of the Canadian Rocky Mountains shifted benthic invertebrate communities to be more similar to lakes with non-native salmonids (Banting et al. 2021). Originally conducted to improve food supply for recreationally important Lake Trout (Fry 1939; Matuszek et al. 1990), introductions of Cisco (*Coregonus artedii*) to Lake Opeongo, Ontario led to a transformation in the diatom and zooplankton community (St. Jacques et al. 2005) and reduced co-occurring fish species abundance (e.g., Yellow Perch *Perca flavescens*; Martin and Fry 1972).

The incidence and magnitude of indirect effects on co-occurring species and ecosystem functions have been evaluated less frequently when performing conservation-based translocations. Experimental reintroductions of Northern Crayfish (*Oronectes virilis*) to a lake recovering from acidification in the Experimental Lakes Area, Ontario (Lake 302S) led to a suppression of periphyton biomass, a reduction in large predatory odonates, and changes to the taxonomic composition of the algal community (Phillips et al. 2009). The reintroduction of American Eel (*Anguilla rostrata*) into the Susquehanna River and its tributaries at locations in Pennsylvania and Maryland, USA led to an increase in native common Eastern Elliptio (*Elliptio complanata*) due to the parasitic relationship between species (Galbraith et al. 2018).

In cases where the translocated species reaches high abundance in the recipient site, changes in ecosystem structure and function could occur, potentially well beyond the initial site of translocation. If severe, this scenario mimics the consequences of species invasions. Most SARA-listed species have not had an opportunity to spread to areas outside of their historical range due to abiotic or biotic constraints, physical limitations, or small population sizes that do not favour dispersal (Figure 2). Moreover, most freshwater SARA-listed species lack the characteristic traits of common invaders (Olden et al. 2011; Karasov-Olson et al. 2021). Specifically, most SARA-listed species have restricted distributions, where the species are often in low abundance, with a low tolerance to stressors, specialized forms of reproduction, and with relatively low rates of dispersal. Nevertheless, it is possible that the translocation of SARA-listed species may pose ecological impacts on recipient communities.

Modeling and postdiction studies of freshwater species invasions can help to understand the potential risk of severe community changes. For example, joint models of environmental invasibility with propagule pressure have been used to predict long-term distributional patterns of the zebra mussel (*Dreissena polymorpha*) post-introduction (Leung and Mandrak 2007). Similarly, species distribution models have been developed to estimate the abundance and distribution of Common Carp (*Cyprinus carpio*) in Minnesota, USA, and then used to predict future dispersal (Kulhanek et al. 2011). Experimental studies suggest that species are more likely to invade in environments with conditions that match their native habitat (Iacarella et al. 2015), which is of concern given the goal of identifying recipient locations for conservation translocation that match conditions in source locations (i.e., environmental matching), or former areas of occurrence. In the context of evaluating risk, understanding what limits dispersal for SARA-listed species in their current distribution, and whether conditions at the site of introduction will lift that constraint, is warranted. Quantitative biological risk assessment tools have been developed by Fisheries and Oceans Canada (Mandrak et al. 2012; Moore et al. 2012) to quantify the potential impact of invasive species during each stage of the invasion process (i.e., arrival, survival, establishment, and spread; Blackburn et al. 2012). Given the similarity of the invasion and conservation translocation processes, the analytical techniques used to model species distributions, invasibility, and risk can be similarly used to predict the potential effects of conservation translocations on recipient ecosystems.

The likelihood of changes to the recipient community will be, in part, influenced by the ecological naivety of the source community to the translocated species. The greatest potential for community changes will occur in sites that never supported the SARA species; potentially fewer changes may occur in sites with recent extirpations and sites where the species already exists.

### 3.1.6. Transfer of disease to recipient populations and ecosystems

Recipient locations of conservation translocations could be at risk of the accidental introduction of non-native parasites and pathogens. Given that infectious diseases are widespread in many wild populations, the Government of Canada (2017) has a National Code on Introductions and Transfers of Aquatic Organisms that provides an "*objective decision-making framework and consistent national process for assessing and managing the potential ecological, disease, and genetic risks associated with intentionally moving live aquatic organisms into, between, or within Canadian watersheds and fish rearing facilities*" and requires a risk assessment of disease transfer. Moreover, qualitative and quantitative frameworks for performing disease risk assessments are well-documented (Sainsbury and Vaughan-Higgins 2011). As a rule, risk assessments should begin early and be reviewed periodically (IUCN/SSC 2013). Given existing guidelines and assessment processes, the potential for disease transfer is not reviewed here in extensive detail.

## 3.2. Estimate the risk of performing conservation translocations

At this stage in the decision support framework, the context of imperilment for the focal species has been described, including the potential source and recipient populations and ecosystems being considered for conservation translocation. As well, the fundamental and means objectives have been identified, the anticipated improvement in survival or recovery of the species resulting from the conservation translocation has been estimated, the anticipated benefits and confounding factors of achieving the means objective have been described, and the potential risks of negative consequences have been reviewed. In the present step, the potential risks of conservation translocations to focal and non-focal species and ecosystems are evaluated and scored based on their expected probability and magnitude of effects in the source and recipient habitat. Similar to the approach for assessing the likelihood of factors influencing the ability to achieve the means objective (i.e., Table 1), a qualitative approach is recommended in data-limited situations for assessing the likelihood and magnitude of risk to source and recipient populations and ecosystems. Such an approach can by default incorporate qualitative evidence or can be fully quantitative when suitable information is available. Table 2 provides an overview of the risks that should be evaluated for a given conservation translocation based on their expected likelihood and magnitude of effects in the source and recipient habitats (*sensu* Karasov-Olson et al. 2021).

*Table 2. Ecological risk considerations for the focal taxa and other ecosystem components in source and recipient habitats of proposed conservation translocations. For each row, the likelihood and magnitude of risk (low, medium, high, unknown), evidence strength (limited, medium, robust), agreement between evidence sources (low, medium, high), and overall confidence are scored. Focal = focal taxa being considered for conservation translocation. Other = other ecosystem components.*

| *Subject* | *Location* | *Risk category* | *Risk outcome* | *Risk likelihood* | *Risk magnitude* | *Evidence strength* | *Agreement* | *Confidence* | *References* | *Additional considerations* |
|---|---|---|---|---|---|---|---|---|---|---|
| Focal | Source | Population persistence | Reduced or altered population abundance | - | - | - | - | - | - | - |
| Focal | Source | Genetic variation | Altered genetic variation | - | - | - | - | - | - | - |
| Focal | Source | Genetic variation | Inbreeding depression | - | - | - | - | - | - | - |
| Focal | Recipient | Population persistence | Individual mortality | - | - | - | - | - | - | - |
| Focal | Recipient | Genetic variation | Founder effect | - | - | - | - | - | - | - |
| Focal | Recipient | Genetic variation | Outbreeding depression | - | - | - | - | - | - | - |
| Focal | Recipient | Genetic variation | Hybridization | - | - | - | - | - | - | - |
| Other | Source | Community and ecosystem dynamics | Increased negative interactions | - | - | - | - | - | - | - |
| Other | Source | Community and ecosystem dynamics | Reduced positive interactions | - | - | - | - | - | - | - |
| Other | Source | Community and ecosystem dynamics | Reduced habitat availability | - | - | - | - | - | - | - |
| Other | Source | Community and ecosystem dynamics | Altered ecosystem processes | - | - | - | - | - | - | - |
| Other | Recipient | Community and ecosystem dynamics | Enhanced negative interactions | - | - | - | - | - | - | - |
| Other | Recipient | Community and ecosystem dynamics | Reduced positive interactions | - | - | - | - | - | - | - |
| Other | Recipient | Community and ecosystem dynamics | Transformative changes within site of introduction | - | - | - | - | - | - | - |

| *Subject* | *Location* | *Risk category* | *Risk outcome* | *Risk likelihood* | *Risk magnitude* | *Evidence strength* | *Agreement* | *Confidence* | *References* | *Additional considerations* |
|---|---|---|---|---|---|---|---|---|---|---|
| Other | Recipient | Community and ecosystem dynamics | Transformative changes beyond site of introduction | - | - | - | - | - | - | - |
| Other | Recipient | Community and ecosystem dynamics | Reduced habitat availability | - | - | - | - | - | - | - |
| All | Both | Disease transfer | Individual mortality / reductions in fitness | - | - | - | - | - | - | - |

Risk likelihood and magnitude can be categorized as low, medium, high, or unknown, with definitions based on the risk factor being considered (Table 2). For example, the likelihood associated with "*reduced population abundance*" for the source population might be characterized as:

- **Low**: Low probability of reducing source population abundance. No risk of source population extirpation;
- **Medium**: Medium probability of reducing source population abundance. Low probability of source population extirpation;
- **High**: High probability of reducing source population abundance. Moderate probability of source population extirpation; or,
- **Unknown**: No information available on the potential for reducing source population abundance.

Similarly, the magnitude associated with "*reduced population abundance*" for the source population might be characterized as:

- **Low**: Small reduction in abundance. No risk of source population extirpation;
- **Medium**: Medium reduction in abundance. Small increase in the probability of source population extirpation;
- **High**: Large reduction in abundance. Moderate increase in the probability of source population extirpation; or,
- **Unknown**: No information available on the potential magnitude of reduced source population abundance.

Continuing with the example, evidence for the likelihood and magnitude of "*reduced population abundance*" for the source population could be characterized as:

- **Limited**: No primary literature or data from local monitoring. Evidence available from related species and expert opinion;
- **Medium**: A few correlative scientific articles in the primary literature relevant to the species. Limited data available from local monitoring or experiments; or,
- **Robust**: Several scientific articles in the primary literature as well as a thorough grey literature. Data from local monitoring programs and experiments are available.

The agreement between sources of information can then be characterized as:

- **Low**: Sources describe contradictory findings;
- **Medium**: Sources describe similar findings, but with differing magnitudes of effects; or,
- **High**: Sources are consistent with findings.

Finally, confidence can be assessed for each ecological risk outcome (Figure 12). Ultimately, for each row in the risk table, evidence is compiled, reviewed, and evaluated, with risk levels assigned. However, a final step is to consider the expected improvement in survival or recovery of the species against the potential risks, other fishery and ecosystem management objectives, and a parallel socioeconomic assessment, which are described in Step 4.

## 4. COMPILE AND WEIGH SCIENTIFIC EVIDENCE TO INFORM THE CONSERVATION TRANSLOCATION DECISION

In Step 4, the compiled evidence is used to decide whether the anticipated improvement in survival or recovery of the SARA-listed freshwater fish or mussel species resulting from conservation translocation (Figure 3; Table 1) outweigh the ecological risks to the focal species and other ecosystem components (Table 2). A three-step process is recommended for making this decision that sequentially considers risks to 1) the focal species, 2) other ecosystem components, 3) and other management objectives.

First, a decision should be made on whether the benefits of conservation translocation, which are evaluated as the probability that achieving the means objective fulfills the fundamental objective, and the probability of achieving the means objective, outweigh the risks to the SARA-listed freshwater fish or mussel species (Table 2). Certain risks directly influence the ability to improve survival and recovery of the focal species, and thus can be evaluated in terms of the net expected outcome for the species. For example, the act of removing individuals for conservation translocation may pose immediate reductions in source population viability that some species will not be able to withstand and can be evaluated quantitatively when sufficient information exists (e.g., Lamothe et al. 2021). Other factors, such as genetic considerations, may be more difficult to evaluate in terms of their role in achieving a short- or long-term net benefit for the species. In other cases, anticipated changes to ecosystem structure or function may have indirect links to the focal species, and may or may not affect long-term viability (e.g., altered ecosystem processes; Table 2). Only once the anticipated benefits of conservation translocation have been recognized as outweighing the potential risks to the focal species should the risks to other ecosystem components be considered (Table 2). The completion of Tables 1 and 2 will provide important summary information to aid the evaluation of net benefit to the SARA-listed species.

Quantifying risks to other ecosystem components and processes and weighing those risks against the anticipated improvement in survival or recovery of the focal species requires a careful evaluation of other management objectives that are potentially influenced by the conservation translocation, and the perceived importance of those management objectives relative to the goal of improved survival or recovery of the SARA-listed species. For example, improved survival or recovery of the SARA-listed species may lead to reductions in formerly abundant species or the loss of a numerically rare species in the recipient habitat, or other changes in ecosystem structure or function. These potential ecosystem changes should be clearly related to the ability to achieve other management objectives. It may be the case that the improvement in survival or recovery of SARA-listed species occurs with little influence on other management objectives, resulting in relatively clear ecological outcomes and straightforward management decisions. Alternatively, achieving improved survival or recovery of SARA-listed species may result in reduced ability to achieve other management goals, implying that weighting of all pertinent management goals and potential outcomes (SARA-related and otherwise) is needed before a decision on the conservation translocation can be made. In such cases, structured decision-making approaches can help identify the perceived importance of each objective, the relevant uncertainties, and, when competing objectives exist, provide clearer opportunities for decision-making.

Ultimately, an understanding of 1) whether fulfilling the means objective will achieve the fundamental objective, 2) the probability of achieving the means objective (Table 1), and 3) the probability of negative consequences (Table 2), coupled with broader approaches for evaluating multiple management objectives (i.e., structured decision-making) can inform the decision of whether to move forward with conservation translocation as a recovery strategy for SARA-listed freshwater fishes or mussels.

# 5. IMPLEMENT AND MONITOR THE EFFECTS OF CONSERVATION TRANSLOCATIONS

If the decision is made to perform conservation translocations, protocols must be established for how they will be implemented and for the monitoring programs to document changes to the focal species, non-focal species, and other ecosystem components (Figure 3). Given the early identification of the means objectives, source population selection, and the number and frequency of translocations needed to improve survival or recovery estimated in Step 2, protocols can be confirmed as feasible and implemented (or not), while considering any logistical constraints that may need to be accounted for.

There are inevitable complications and logistical constraints when implementing field-based restoration or management actions (Fiumera et al. 2004). This is particularly true for conservation translocations, which represent long-term, experimental studies and the related commitment to monitoring. Logistical constraints can range from the ability to acquire enough individuals to implement the translocation (Fetherman et al. 2015; Malone et al. 2018) to the methods available for transporting individuals between source and recipient locations (e.g., Vachon 2021). In some instances, such as when source populations are very small, captive breeding or rearing approaches will be needed to supply an adequate number of individuals for translocation, which requires significant investment in infrastructure (Patterson et al. 2018). For example, there are only a few facilities in Canada that can support the propagation and rearing of warm-water SARA-listed fishes or mussels. Moreover, captive breeding requires significant expertise about species reproductive strategies and husbandry. Further science advice will be needed on key science considerations for the species-specific implementation and experimental design of conservation translocations (e.g., release schedule, state variables for monitoring, stopping points for translocations).

Conservation translocations of SARA-listed freshwater fish and mussel species are long-term ecosystem restoration experiments that must be carefully designed, documented, and monitored. Experimentation describes the process of using the scientific method to test hypotheses and learn from the results. Taking an adaptive approach to conservation translocation implementation can be beneficial as it requires recurring (e.g., annual) evaluation of translocation protocols and allows experimental design to be revisited, or terminated, when implementation is not producing the expected results. Identifying when to stop translocation efforts is an important decision when planning conservation translocations, as the risk of unintended consequences increases over time (IUCN/SSC 2013). Although monitoring of the effects of conservation translocations must continue over long time frames (i.e., many years to decades depending on the generation time of the species and other factors), supplementing or reintroducing wildlife species should be considered as relatively short-term actions.

For any species recovery action, monitoring of outcomes (intended and unintended) and documenting decisions related to implementation are critical. Well-designed monitoring programs for measuring the effects of conservation translocations must ensure an adequate number of sampling sites and frequency of sampling in source and recipient locations to ensure conclusions are made with sufficient statistical power. There are many attributes of conservation translocations that can be measured to define success, but all should relate back to the intended means objectives and fundamental objective of improved survival or recovery. Common metrics for evaluating the effects of conservation translocations include changes in population abundance and genetic variation, changes in size-structure or physiological metrics, or increased rates of survival, spawning, and (or) recruitment at the recipient site (Vachon 2010, 2021; Cochran-Biederman et al. 2014; Tarszisz et al. 2014; Sard et al. 2016; Rytwinski et al. 2021). Such metrics are measured at the site-level and evaluated at the population scale where benefits to the population are inferred as benefits to the species overall (Figure 6). Using the described metrics and approaches to evaluation aligns with the indicators used for COSEWIC status assessments, where recovery actions are recommended to reverse declining trajectories and increase population size and distribution. Ultimately, metrics for evaluating conservation translocation actions should consider the outcome of a conservation translocation over time for the recipient population, the source population, and the overall species (Figure 6) relative to some historical information (i.e., before-after), reference location (i.e., control-impact), or ideally both (i.e., before-after control-impact).

## CONCLUSIONS

This document provided a decision support framework for identifying and evaluating the potential ecological benefits and risks of performing conservation translocations and determining the scientific considerations and methods for determining if and when conservation translocation could improve the survival or recovery of SARA-listed freshwater fishes and mussels (Figure 3). The decision support framework followed five general steps:

1. Identify objectives for conservation translocations;
2. Assess the probability of achieving the fundamental and means objectives;
3. Identify and assess the likelihood and magnitude of negative consequences;
4. Compile and weigh scientific evidence to inform the translocation decision; and,
5. Implement and monitor the conservation translocation.

Fundamental objectives describe the most general objective for performing the management action, which for conservation translocation of SARA-listed freshwater fishes and mussels is to improve the survival or recovery of the species. Means objectives describe how the fundamental objectives will be achieved. There are three primary means objectives for the context of conservation translocations:

1. Improve recruitment of extant populations;
2. Establish a population; or,
3. Rescue individuals or populations at imminent risk of extirpation (Figure 1).

Once the fundamental and means objectives have been considered (Step 1), an assessment is needed of whether achieving the means objective at recipient habitats will improve survival or recovery, including an assessment of whether the means objective can be achieved (Step 2). Assessing whether achieving the means objective will improve survival or recovery requires identifying potential source and recipient populations and ecosystems, and using population models and simulations to make predictions on population viability after performing a conservation translocation. There are many factors that may preclude the ability to achieve the means objective that must be considered that were broadly categorized into population, habitat, community, and threat considerations. A qualitative approach was provided to score the effects of confounding factors on the likelihood of achieving the means objective (Table 1), but quantitative information should be used when available.

Following an assessment of factors that may preclude the ability to achieve the means objective, the magnitude and likelihood of risks to the source and recipient populations and ecosystems need to be identified and assessed (Step 3). Risks of negative consequences when performing conservation translocation include changes in population persistence and genetic variation, short- and long-term changes in ecosystem structure and function, along with the potential transfer of disease. Assessing the potential for these risks (Table 2) is necessary to decide whether a net improvement in survival or recovery of the species is likely to occur.

The next step (Step 4) is to take the compiled information on ecological benefits and risks and decide whether the act of conservation translocation is an ecologically sound approach for attaining species recovery. Informing this decision is an understanding of the probability of achieving the means and fundamental objectives, consideration of risks to the focal species and broader ecosystem components, and attention to potential conflicting management objectives. Completed information in Table 1 and Table 2 will help guide this decision-making process, while also identifying knowledge gaps in the information needed to make the translocation decision. If the decision to perform a conservation translocation is made, protocols for implementation and monitoring must be designed (Step 5).

The decision support framework presented here provides a structured science-based approach for determining if and when conservation translocation would be expected to improve survival or recovery of SARA-listed freshwater fishes and mussels. Like most recovery actions, conservation translocations are experiments that come with risks and uncertainties, which must be weighed against the potential benefits. Included in this consideration is the risk of inaction. In some cases, the fundamental objective for SARA-listed freshwater fishes or mussels may never be achieved without the use of conservation translocation (e.g., reintroduction). In other cases, however, alternative strategies may be justified. For example, habitat restoration or non-native species removal may be suitable recovery actions for achieving improved survival or recovery of the focal species with fewer risks than conservation translocations and a greater likelihood of success. In addition to guiding decisions on conservation translocation, Steps 1–5 of the decision support framework could be modified to evaluate the ecological benefits and risks of alternative recovery actions. Ultimately, the decision support framework presented here can help determine if and when the ecological benefits of recovery actions outweigh the uncertainties and risks of unintended consequences when aiming to improve survival and recovery of SARA-listed freshwater fishes and mussels. Future refinement of the decision support framework would benefit from case studies in addition to those that follow in the appendices.

## GLOSSARY

**Adaptive potential approach**: A source population selection approach where populations are selected for translocation based on their potential to adapt to the key environmental features of the recipient habitat.

**Ancestry matching approach**: A source population selection approach where populations are chosen for translocation based on their genetic similarity to the extant or historically present population (Houde et al. 2015).

**Area of occupancy**: the area within the **extent of occurrence** that is occupied by a taxon, excluding cases of vagrancy. The measure reflects the fact that the extent of occurrence may contain unsuitable or unoccupied habitats, and that occupied habitats are often disjunct in freshwater systems. In some cases (e.g., irreplaceable colonial nesting sites or crucial feeding sites for migratory taxa), the area of occupancy is the smallest area essential at any stage to the survival of the wildlife species/designatable unit considered (COSEWIC 2019).

**Assisted colonization**: The translocation of a species to favourable habitat beyond the native range to protect it from human-induced threats such as climate change (Ricciardi and Simberloff 2009).

**Captive breeding**: The act of intentionally propagating plants or animals in controlled environments (e.g., hatcheries, zoos) for eventual introduction into the wild.

**Captive rearing**: The act of raising captured, wild individuals from eggs or young to adults with the intention of release back into the wild (i.e., no facilitation of reproduction).

**Conservation translocation**: The intentional movement and release of an organism where the primary objective is a conservation benefit (IUCN/SSC 2013). For SARA-listed species in Canada, the conservation benefit is an improvement in survival or recovery of the focal species.

**Demographic swamping**: The situation where hybridization is widespread, but the fitness of hybrid individuals (i.e., recipient population x extant population individuals) is lower than replacement rates for sustaining the population (Todesco et al. 2016).

**Domestication selection**: Natural selection on traits that affect survival and reproduction in a human-controlled (domestic) environment (Doyle 1983).

**Ecological niche**: The defined ecological boundaries within which an organism can carry out its life-history processes. An organism's realized niche is constrained by the physical environment and interactions with other species.

**Environmental matching approach**: A source population selection approach where a population is selected for translocation based on the environmental similarity between the source and recipient locations (Houde et al. 2015).

**Extent of occurrence**: The area included in a polygon without concave angles that encompasses the geographic distribution of all known populations of a wildlife species. (COSEWIC 2019).

**Founder effect**: The resulting loss of genetic variation resulting from the establishment of a new population using a relatively small number of founding individuals.

**Fundamental objective**: The underlying motivation for why a particular management approach is being considered.

**Genetic swamping**: The situation where population growth rates of hybrid individuals outpace the recipient population, leading to the loss of pure parental genomes (Todesco et al. 2016).

**Imminent extirpation or extinction**: A 20% or greater probability of extinction or extirpation within 20 years or 5 generations (up to a maximum of 100 years), whichever is longer (COSEWIC 2019).

**Inbreeding depression**: The relative reduction in fitness of offspring resulting from the mating of closely related individuals compared to those of randomly mated individuals (Hedrick and Kalinowski 2000).

**Native range**: The known or inferred distribution generated from historical (written or verbal) records, or physical evidence of the species' occurrence. Where direct evidence is inadequate to confirm previous occupancy, the existence of suitable habitat within ecologically appropriate proximity to proven range may be taken as adequate evidence of previous occurrence (IUCN/SSC 2013).

**Means objective**: The way of achieving an end point or **fundamental objective**.

**Mitigation translocation**: Translocations initiated to reduce immediate animal deaths as a result of development activities (Germano et al. 2015).

**Outbreeding depression**: A reduction in the fitness of individuals caused by the crossing of genetically distinct populations.

**Problem statement**: A concise description of the conservation issue to be addressed.

**Recovery**: The process by which the decline of a species is arrested or reversed, and threats are removed or reduced to improve the likelihood of the species' persistence in the wild. A species is considered recovered when its long-term persistence in the wild has been secured.

**Reintroduction**: The intentional movement and release of an organism inside its native range to a location from which it has disappeared (IUCN/SSC 2013).

**Risk**: The probability and magnitude of negative or undesirable events.

**Species**: See **wildlife species**.

**Species distribution models**: Numerical tools that combine observations of species occurrence or abundance with environmental variables to reconstruct the extent of occurrence, or project changes to this extent, for a species (Elith and Leathwick 2009).

**Supplementation**: The intentional release of individuals of a focal species to an area presently occupied by conspecifics within the native range of the species (Seddon et al. 2012).

**Translocation**: The human-mediated movement of living organisms from one area, with release in another (IUCN/SSC 2013).

**Wildlife species**: A species, subspecies, variety, or geographically or genetically distinct population of animal, plant or other organism, other than a bacterium or virus, that is wild by nature and is native to Canada or has extended its range into Canada without human intervention and has been present in Canada for at least 50 years (SARA 2002).

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## REFERENCES CITED

Alberta Athabasca Rainbow Trout Recovery Team. 2014. Alberta Athabasca Rainbow Trout recovery plan, 2014-2019. Alberta Environment and Sustainable Resource Development, Alberta Species at Risk Recovery Plan No. 36. Edmonton, AB. 111 pp.

Alcaide, M., Negro, J.J., Serrano, D., Antolín, J.L., Casado, S., and Pomarol, M. 2010. Captive breeding and reintroduction of the lesser kestrel *Falco naumanni*: a genetic analysis using microsatellites. Conservation Genetics 11: 331-338.

Al-Chokhachy, R., Peacock, M., Heki, L.G., and Thiede, G. 2009. Evaluating the reintroduction potential of Lahontan Cutthroat Trout in Fallen Leaf Lake, California. North American Journal of Fisheries Management 29: 1296-1313.

Armstrong, D.P., and Seddon, P.J. 2008. Directions in reintroduction biology. Trends in Ecology & Evolution 23(1): 20-25.

Banting, A.L.K., Taylor, M.K., Vinebrooke, R.D., Carli, C.M., and Poesch, M.S. 2021. Assisted colonization of a regionally native predator impacts benthic invertebrates in fishless mountain lakes. Conservation Science and Practice 3: e344. doi: 10.1111/csp2.344.

Barton, N.H., and Hewitt, G.M. 1985. Analysis of hybrid zones. Annual Review in Ecology and Systematics 16: 113-148.

Bertucci, A., Pierron, F., Thébault, Klopp, C., Bellec, J., Gonzalez, P., and Baudrimont, M. 2017. Transcriptomic responses of the endangered freshwater mussel *Margaritifera margaritifera* to trace metal contamination in the Dronne River, France. Environmental Science and Pollution Research 24: 27145-27159.

Blackburn, T.M., Pyšek, P., Bacher, S., Carlton, J.T., Duncan, R.P., Jarošík, V., Wilson, J.R.U., and Richardson, D.M. 2011. A proposed unified framework for biological invasions. Trends in Ecology & Evolution 26: 333-339.

Bolland, J.D., Bracken, L.J., Martin, R., and Lucas, M.S. 2010. A protocol for stocking hatchery reared freshwater pearl mussel *Margaritifera margaritifera*. Aquatic Conservation: Marine and Freshwater Ecosystems 20: 695-704.

Bradford, R.G. 2017. Supplementation options to aid recovery of the Endangered Atlantic Whitefish (*Coregonus huntsmani*). Can. Manuscr. Rep. Fish. Aquat. Sci. 3124: vi +29p.

Bradford, R.G., Longard, D.A., and Longue, P. 2004. Status, trend, and recovery considerations in support of an allowable harm assessment for Atlantic Whitefish (*Coregonus hunstmani*). Can. Sci. Advis. Secret. Res. Doc. 2004/109: iv + 38p.

Bradford, R.G., Themelis, D., LeBlanc, P., Campbell, D.M., O'Neil, S.F., and Whitelaw, J. 2015. Atlantic Whitefish (*Coregonus huntsmani*) Stocking in Anderson Lake, Nova Scotia. Can. Tech. Rep. Fish. Aquat. Sci. 3142: vi + 45p.

Bradley, H.S., Tomlinson, S., Craig, M.D., Cross, A.T., and Bateman, P.W. 2020. Mitigation translocation as a management tool. Conservation Biology. doi: 10.1111/cobi.13667.

Bubac, C.M., Johnson, A.C., Fox, J.A., and 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. doi: 10.1016/j.biocon.2019.108239.

Castañeda, R.A., Ackerman, J.D., Chapman, L.J., Cooke, S.J., Cuddington, K., Dextrase, A.J., Jackson, D.A., Koops, M.A., Krkošek, M., Loftus, K.K., Mandrak, N.E., Martel, A.L., Molnár, P.K., Morris, T.J., Pitcher, T.E., Poesch, M.S., Power, M., Pratt, T.C., Reid, S.M., Rodríguez, M.A., Rosenfeld, J., Wilson, C.C., Zanatta, D.T., & Drake, D.A.R. 2021. Approaches and research needs for advancing the protection and recovery of imperilled freshwater fishes and mussels in Canada. Canadian Journal of Fisheries and Aquatic Sciences 78: 1356-1370.

Chauvenet, A.L.M., Ewen, J.G., Armstrong, D.P., Blackburn, T.M., and Pettorelli, N. 2013. Maximizing the success of assisted colonization. Animal Conservation 16: 161-169.

Chen, Q., Wang, C., Lu, G., Zhao, J., Chapman, D.C., Zsigmond, J., and Li, S. 2012. Microsatellite genetic diversity and differentiation of native and introduced grass carp populations in three continents. Genetica 140: 115-123.

Cochran-Biederman, J.L., Wyman, K.E., French, W.E., and Loppnow, G.L. 2014. Identifying correlates of success and failure of native freshwater fish reintroductions. Conservation Biology 29(1): 175-186.

Cook, A.M., Bradford, R.G., Hubley, B., and Bentzen,P. 2010. Effects of pH, Temperature and Salinity on Age 0+ Atlantic Whitefish (*Coregonus huntsmani*) with Implications for Recovery Potential. DFO Can. Sci. Advis. Sec. Res. Doc. 2010/055.

Cope, W.G., and Waller, D.L. 1995. Evaluation of freshwater mussel relocation as a conservation and management strategy. Regulated Rivers: Research & Management 11: 147-155.

Cornelius, F.C., Muth, K.M., and Kenyon, R. 1995. Lake Trout rehabilitation in Lake Erie: A case history. Journal of Great Lakes Research 21(Suppl. 1): 65-82.

COSEWIC. 2011. COSEWIC report on the eligibility for the Aurora Trout *Salvelinus fontinalis timagamiensis* in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa. x + 27 pp.

COSEWIC. 2019. COSEWIC definitions and definitions and abbreviations. Accessed: 2021-07-29.

Cowx, I.G., and Gerdeaux, D. 2004. The effects of fisheries management practises on freshwater ecosystems. Fisheries Management and Ecology 11: 145-151.

Dade, M.C., Pauli, N., and Mitchell, N.J. 2014. Mapping a new future: using spatial multiple criteria analysis to identify novel habitats for assisted colonisation of endangered species. Animal Conservation 17: 4-17.

Dextrase, A.J., and Mandrak, N.E. 2006. Impacts of alien invasive species on freshwater fauna at risk in Canada. Biological Invasions 8: 13-24.

DFO. 2018a. Strategy for the establishment of self-sustaining Atlantic Whitefish population(s) and development of a framework for the evaluation of suitable lake habitat. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2018/045

DFO. 2018b. Review of the Science Associated with the Inner Bay of Fundy Atlantic Salmon Live Gene Bank and Supplementation Programs. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2018/041.

Dormann, C.F., Bobrowski, M., Dehling, D.M., Harris, D.J., Hartig, F., Lischke, H., Moretti, M.D., Pagel, J., Pinkert, S., Schleuning, M., Schmidt, S.I., Sheppard, C.S., Steinbauer, M.J., Zeuss, D., and Kraan, C. 2018. Biotic interactions in species distribution modelling: 10 questions to guide interpretation and avoid false conclusions. Global Ecology and Biogeography 27(9): 1004-1016.

Douda, K. 2010. Effects of nitrate nitrogen pollution on Central European unionid bivalves revealed by distributional data and acute toxicity testing. Aquatic Conservation: Marine and Freshwater Ecosystems 20: 189-197.

Downing, A.L., Brown, B.L., and Leibold, M.A. 2014. Multiple diversity-stability mechanisms enhance population and community stability in aquatic food webs. Ecology 95(1): 173-184.

Doyle, R.W. 1983. An approach to the quantitative analysis of domestication selection in aquaculture. Aquaculture 33(1-4): 167-185.

Drake, D.A.R., Lamothe, K.A., Thiessen, K.E., Morris, T.J., Koops, M.A., Pratt, T.C., Reid, S.M., and Mandrak, N.E. 2021. Fifteen years of Canada’s Species at Risk Act: Evaluating research progress for aquatic species in the Great Lakes-St. Lawrence River basin. Canadian Journal of Fisheries and Aquatic Science 78(9): 1205-1218.

Dudgeon, D., Arthington, A.H., Gessner, M.O., Kawabata, Z.I., Knowler, D.J., Lévêque, C., Naiman, R.J., Prieur-Richard, A.H., Soto, D., Stiassny, M.L.J., and Sullivan, C.A. 2006. Freshwater biodiversity: importance, threats, status and conservation challenges. Biological Reviews 81(2): 163-182.

Eby, L.A., Roach, W.J., Crowder, L.B., and Stanford, J.A. 2006. Effects of stocking-up freshwater food webs. Trends in Ecology & Evolution 21(10): 576-584.

Edmands, S., and Timmerman, C.C. 2003. Modeling factors affecting the severity of outbreeding depression. Conservation Biology 17(3): 883-892.

Elith, J., and Graham, C.H. 2009. Do they? How do they? Why do they differ? On finding reasons for differing performances of species distribution models. Ecography 32(1): 66-77.

Elith, J., and Leathwick, J.R. 2009. Species distribution models: ecological explanation and prediction across space and time. Annual Reviews in Ecology, Evolution, and Systematics 40: 677-697.

Elrod, J.H., O’Gorman, R., Schneider, C.P., Eckert, T.H., Schaner, T., Bowlby J.N., and Schleen, L.P. 1995. Lake Trout rehabilitation in Lake Ontario. Journal of Great Lakes Research 21(Suppl. 1): 83-107.

Ennen, J.R., Kuhadja, B.R., Fix, S., Sweat, S.C., Zuber, B., Watts, A.V., Mattingly, H.T., and Cecala, K.K. 2021. Assessing the success of conservation efforts for a North American topminnow at risk of extinction from spatially variable mosquitofish invasions. Freshwater Biology 66(3): 458-467.

Eveleens, R.A., and Febria, C.M. 2021. A systematic review of the global freshwater mussel restoration toolbox. Aquatic Conservation: Marine and Freshwater Ecosystems. doi: 10.1002/aqc.3750.

Fetherman, E.R., Winkelman, D.L., Bailey, L.L., Schisler, G.J., and Davies, K. 2015. Brown Trout removal effects on short-term survival and movement of *Myxobolus cerebralis*-Resistant Rainbow Trout. Transactions of the American Fisheries Society 144: 610-626.

Fieberg, J., and Ellner, S.P. 2001. Stochastic matrix models for conservation and management: a comparative review of methods. Ecology Letters 4(3): 244-266.

Fisheries and Oceans Canada. 2004. Allowable Harm Assessment for Atlantic Whitefish. DFO Can. Sci. Advis. Sec. Stock Status Rep. 2004/052.

Fisheries and Oceans Canada. 2007. Recovery Strategy for the Dwarf Wedgemussel (*Alasmidonta heterodon*) in Canada. Species at Risk Act Recovery Strategy Series. Department of Fisheries and Oceans, Ottawa. vi + 9 pp.

Fisheries and Oceans Canada. 2010. Recovery strategy for the Atlantic Salmon (*Salmo salar*), inner Bay of Fundy populations [Final]. *In* Species at Risk Act Recovery Strategy Series. Ottawa: Fisheries and Oceans Canada. xiii + 58 pp. + Appendices.

Fisheries and Oceans Canada. 2012. Recovery Strategy for the Copper Redhorse (*Moxostoma hubbsi*) in Canada. Species at Risk Act Recovery Strategy Series. Fisheries and Oceans Canada, Ottawa. xi + 60 pp.

Fisheries and Oceans Canada. 2014. Recovery strategy for White Sturgeon (*Acipenser transmontanus*) in Canada [Final]. In Species at Risk Act Recovery Strategy Series. Ottawa: Fisheries and Oceans Canada. 252 pp.

Fisheries and Oceans Canada. 2019. Recovery Strategy and Action Plan for the Westslope Cutthroat Trout (*Oncorhynchus clarkii lewisi*) Alberta population (also known as Saskatchewan-Nelson River populations) in Canada. Species at Risk Act Recovery Strategy Series. Fisheries and Oceans Canada, Ottawa. vii + 61 pp + Part 2.

Fiumera, A.C., Porter, B.A., Looney, G., Asmussen, M.A., and Avise, J.C. 2004. Maximizing offspring production while maintaining genetic diversity in supplemental breeding programs of highly fecund managed species. Conservation Biology 18(1): 94-101.

FLNRORD. British Columbia Ministry of Forests, Lands, Natural Resource Operations and Rural Development. 2018. Guidance for Freshwater Mussels in the Okanagan. Thompson Okanagan Region. ii - 19 p.

Frankham, R. 1995. Effective population size/adult population size ratios in wildlife: A review. Genetics Research 66(2): 95-107.

Frankham, R., Ballou, J.D., Eldridge, M.D.B., Lacy, R.C., Ralls, K., Dudash, M.R., and Fenseter, C.B. 2011. Predicting the probability of outbreeding depression. Conservation Biology 25(3): 465-475.

Frankham, R., Bradshaw, C.J.A., and Brook, B.W. 2014. Genetics in conservation management: Revise recommendations for the 50/500 rules, Red List criteria and population viability analyses. Biological Conservation 170: 56-63.

Fraser, D.J. 2008. How well can captive breeding programs conserve biodiversity? A review of salmonids. Evolutionary Applications 1: 535-586.

Fry, F.E.J. 1939. A comparative study of lake trout fisheries in Algonquin Park, Ontario. University of Toronto Studies, Biological Series 46. (Publication of the Ontario Fisheries Research Laboratory 58.)

Galbraith, H.S., Devers, J.L., Blakeslee, C.J., Cole, J.C., St. John White, B., Minkkinen, S., and Lellis, W.A. 2018. Reestablishing a host-affiliate relationship: migratory fish reintroduction increases native mussel recruitment. Ecological Applications 28(7): 1841-1852.

Gallagher, R.V., Makinson, R.O., Hogbin, P.M., and Hancock, N. 2015. Assisted colonization as a climate change adaptation tool. Austral Ecology 40(1): 12-20.

Galloway, B.T., Muhlfeld, C.C., Guy, C.S., Downs, C.C., Fredenberg, W.A. 2016. A framework for assessing the feasibility of native fish conservation translocations: applications to threatened Bull Trout. North American Journal of Fisheries Management 36: 754-768.

George, A.L., Kuhajda, B.R., Williams, J.R., Cantrell, M.A., Rakes, P.L., and Shute, J.R. 2009. Guidelines for the propagation and translocation for freshwater fish conservation. Fisheries 34(11): 529-545.

Germano, J.M., Field, K.J., Griffiths, R.A., Clulow, S., Foster, J., Harding, G., and Swaisgood, R.R. 2015. Mitigation-driven translocations: are we moving wildlife in the right direction? Frontiers in Ecology and the Environment 13(2): 100-105.

Government of Canada. 2017. National code on introductions and transfers of aquatic organisms. Fisheries and Oceans Canada. Accessed: 2021-09-01.

Gozlan, R.E., Britton, J.R., and Copp, G.H. 2010. Current knowledge on non-native freshwater fish introductions. Journal of Fish Biology 76: 751-786.

Guiaşu, R.C., and Labib, M. 2021. The unreliable concept of native range as applied to the distribution of the rusty crayfish (*Faxonius rusticus*) in North America. Hydrobiologia 848: 1177-1205.

Haag, W.R. 2012. North American freshwater mussels: natural history, ecology, and conservation. Cambridge University Press. New York, NY.

Haag, W.R., and Stoeckel, J.A. 2015. The role of host abundance in regulating populations of freshwater mussels with parasitic larvae. Oecologia 178: 1159-1168.

Haag, W.R., and Warren Jr., M.L. 1998. Role of ecological factors and reproductive strategies in structuring freshwater mussel communities. Canadian Journal of Fisheries and Aquatic Sciences 55: 297-306.

Hagen, I.J., Jensen, A.J., Bolstad, G.H., Diserud, O.H., Hindar, K., Lo, H., and Karlsson, S. 2019. Supplementary stocking selects for domesticated genotypes. Nature Communications 10(199). doi: 10.1038/s41467-018-08021-z.

Hart, S.P., Freckleton, R.P., and Levine, J.M. 2018. How to quantify competitive ability. Journal of Ecology 106(5): 1902-1909.

He, X., Johansson, M.L., and Heath, D.D. 2016. Role of genomics and transcriptomics in selection of reintroduction source populations. Conservation Biology 30(5): 1010-1018.

He, X., Wilson, C.C., Wellband, K.W., House, A.L.S., Neff, B.D., and Heath, D.D. 2015. Transcriptional profiling of two Atlantic salmon strains: implications for reintroduction into Lake Ontario. Conservation Genetics 16: 277-287.

Hein, C.L., Öhlund, G., and Englund, G. 2013. Fish introductions reveal the temperature dependence of species interactions. Proceedings of the Royal Society B 281: 20132641.

Hedrick, P.W., and Kalinowski, S.T. 2000. Inbreeding depression in conservation biology. Annual Review of Ecology and Systematics 31: 139-162.

Higgins, P.S., and Bradford, M.J. 1996. Evaluation of a large-scale fish salvage to reduce the impacts of controlled flow reduction in a regulated river. North American Journal of Fisheries Management 16: 666-673.

Hildebrand, L.R., Schreier, A.D., Lepla, K., McAdam, S.O., McLellan, J., Parsley, M.J., Paragamian, V.L., and Young, S.P. 2016. Status of White Sturgeon (*Acipenser transmontanus* Richardson, 1863) throughout the species range, threats to survival, and prognosis for the future. Journal of Applied Ichthyology 32(Suppl. 1): 261-312.

Hinch, S.G., Collins, N.C., and Harvey, H.H. 1991. Relative abundance of littoral zone fishes: biotic interactions, abiotic factors, and postglacial colonization. Ecology 72(4): 1314-1324.

Hoftyzer, E., Ackerman, J.D., Morris, T.J., and Mackie, G.L. 2008. Genetic and environmental implications of reintroduction laboratory-reared unionid mussels to the wild. Canadian Journal of Fisheries and Aquatic Sciences 65: 1217-1229.

Holloway, P., and Miller, J.A. 2017. A quantitative synthesis of the movement concepts used within species distribution modelling. Ecological Modelling 356: 91-103.

Holomuzki, J.R., Feminella, J.W., and Power, M.E. 2010. Biotic interactions in freshwater benthic habitats. Journal of the North American Benthological Society 29(1): 220-244.

Holt, R.D. 2003. On the evolutionary ecology of species' ranges. Evolutionary Ecology Research 5: 159-178.

Houde, A.L.S., Garner, S.R., and Neff, B.D. 2015. Restoring species through reintroductions: strategies for source population selection. Restoration Ecology 23(6): 746-753.

Huntingford, F.A. 2004. Implications of domestication and rearing conditions for the behaviour of cultivated fishes. Journal of Fish Biology 65(Suppl. A): 122-142.

Huxel, G.R. 1999. Rapid displacement of native species by invasive species: effects of hybridization. Biological Conservation 89(2): 143-152.

Iacarella, J.C., Dick, J.T.A., Alexander, ME., and Ricciardi, A. 2015. Ecological impacts of invasive alien species along temperature gradients: testing the role of environmental matching. Ecological Applications 25(3): 706-716.

International Panel on Climate Change [IPCC]. 2013. Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA. 1535 pp.

IUCN/SSC. 2013. Guidelines for Reintroductions and Other Conservation Translocations. Version 1.0. Gland, Switzerland: IUCN Species Survival Commission, viiii + 57 pp.

Jackson, D.A., Peres-Neto, P.R., and Olden, J.D. 2001. What controls who is where in freshwater fish communities – the roles of biotic, abiotic, and spatial factors. Canadian Journal of Fisheries and Aquatic Sciences 58: 157-170.

Jamieson, I.G. 2010. Founder effects, inbreeding, and loss of genetic diversity in four avian reintroduction programs. Conservation Biology 25(1): 115-123.

Janowitz-Koch, I., Rabe, C., Kinzer, R., Nelson, D., Hess, M.A., and Narum, S.R. 2019. Long-term evaluation of fitness and demographic effects of a Chinook Salmon supplementation program. Evolutionary Applications 12: 456-469.

Karasov-Olson, A., Schwartz, M.W., Olden, J.D., Skikne, S., Hellmann, J.J., Allen, S., Brigham, C., Buttke, D., Lawrence, D.J., Miller-Rushing, A.J., Morisette, J.T., Schuurman, G.W., Trammell, M., and Hawkins Hoffman, C. 2021. Ecological risk assessment of managed relocation as a climate change adaptation strategy. Natural Resource Report NPS/NRSS/CCRP/NRR—2021/2241. National Park Service, Fort Collins, Colorado. doi: 10.36967/nrr-2284919.

Keller, L.F., Biebach, I., Ewing, S.R., and Hoeck, P.E.A. 2012. The genetics of reintroductions: inbreeding and genetic drift. In Reintroduction Biology: Integrating Science and Management. Edited by J.G. Owen, D.P. Armstrong, K.A. Parkerand, and P.J. Seddon. Wiley-Blackwell. Chichester. pp. 360–394.

Kostow, K.E. 2004. Differences in juvenile phenotypes and survival between hatchery stocks and a natural population provide evidence for modified selection due to captive breeding. Canadian Journal of Fisheries and Aquatic Sciences 61: 577-589.

Krueger, C.C., and May, B. 1991. Ecological and genetic effects of salmonid introductions in North America. Canadian Journal of Fisheries and Aquatic Sciences 48(Suppl. 1): 66-77.

Kulhanek, S.A., Leung, B., and Ricciardi, A. 2011. Using ecological niche models to predict the abundance and impact of invasive species: application to the common carp. Ecological Applications 21(1): 203-213.

Lamothe, K.A., and Drake, D.A.R. 2019. Moving repatriation efforts forward for imperilled Canadian freshwater fishes. Canadian Journal of Fisheries and Aquatic Sciences 76(10): 1914-1921.

Lamothe, K.A., Drake, D.A.R., Pitcher, T.E., Broome, J.E., Dextrase, A.J., Gillespie, A., Mandrak, N.E., Poesch, M.S., Reid, S.M., and Vachon, N. 2019. Reintroduction of freshwater fishes in Canada: a review of research progress for SARA-listed species. Environmental Reviews 27(4): 575-599.

Lamothe, K.A., van der Lee, A.S., Drake, D.A.R., and Koops, M.A. 2021. The translocation trade-off for eastern sand darter (*Ammocrypta pellucida*): balancing harm to source populations with the goal of re-establishment. Canadian Journal of Fisheries and Aquatic Sciences 78(9): 1321-1331.

Larkin, P.A. 1956. Interspecific competition and population control in freshwater fish. Journal of the Fisheries Research Board of Canada 13(3): 327-342.

Lawler, J.J., and Olden, J.D. 2011. Reframing the debate over assisted colonization. Frontiers in Ecology and the Environment 9(10): 569-574.

Le Gouar, P., Rigal, F., Boisselier-Dubayle, M.C., Sarrazin, F., Arthur, C., Choisy, J.P., Hatzofe, O., Henriquet, S., Lécuyer, P., Tessier, C., Susic, G., and Samadi, S. 2008. Genetic variation in a network of natural and reintroduction populations of Griffon vulture (*Gyps fulvus*) in Europe. Conservation Genetics 9: 349-359.

Leclair, A.T.A., Drake, D.A.R., Pratt, T.C., and Mandrak, N.E. 2020. Seasonal variation in thermal tolerance of redside dace *Clinostomus elongatus*. Conservation Physiology 8(1): coaa081.

Lennox, R.J., Alexandre, C.M., Almeida, P.R., Bailey, K.M., Barlaup, B.T., Bøe, K., Breukelaar, A., Erkinaro, J., Forseth, T., Gabrielsen, S.-E., Halfyard, E., Hanssen, E.M., Karlsson, S., Koch, S., Koed, A., Langåker, R.M., Lo, H., Lucas, M.C., Mahlum, S., Perrier, C., Pulg, U., Sheehan, T., Skoglund, H., Svenning, M., Thorstad, E.B., Velle, G.,.Whoriskey, F.G., and Wiik Vollset, K. 2021. The quest for successful Atlantic salmon restoration: perspectives, priorities, and maxims. ICES Journal of Marine Science. fsab201. doi: 10.1093/icesjms/fsab201.

Letty, J., Marchandeau, S., and Aubineau, J. 2007. Problems encountered by individuals in animal translocations: Lessons from field studies. Ecoscience 14(4): 420-431.

Leung, B., and Mandrak, N.E. 2007. The risk of establishment of aquatic invasive species: joining invasibility and propagule pressure. Proceedings of the Royal Society B 274(1625): 2603-2609.

Loppnow, G.L., Vascotto, K., and Venturelli, P.A. 2013. Invasive smallmouth bass (*Micropterus dolomieu*): history, impacts, and control. Management of Biological Invasions 4(3): 191-206.

Loss, S.R., Terwilliger, L.A., and Peterson, A.C. 2011. Assisted colonization: Integrating conservation strategies in the face of climate change. Biological Conservation 144(1): 92-100.

Mackie, G., Morris, T.J., and Ming, D. 2008. Protocol for the detection and relocation of freshwater mussel species at risk in Ontario-Great Lakes Area (OGLA). Can. Manuscr. Rep. Fish. Aquat. Sci. 2790: vi +50 p.

Maitland, P.S., and Lyle, A.A. 1992. Conservation of freshwater fish in the British Isles: proposals for management. Aquatic Conservation: Marine and Freshwater Ecosystems 2: 165-183.

Malone, E.W., Perkin, J.S., Leckie, B.M., Kulp, M.A., Hurt, C.R., and Walker, D.M. 2018. Which species, how many, and from where: Integrating habitat suitability, population genomics, and abundance estimates into species reintroduction planning. Global Change Biology 24: 3729-3748.

Mandrak, N.E., Cudmore, B., and Chapman, P.M. 2012. National Detailed-Level Risk Assessment Guidelines: Assessing the Biological Risk of Aquatic Invasive Species in Canada. DFO Can. Sci. Advis. Sec. Res. Doc. 2011/092. vi + 17 p.

Martin, N.V., and Fry, F.E.J. 1972. Lake Opeongo: effects of exploitation and introductions on the salmonid community. Journal of the Fisheries Research Board of Canada 29: 795-805.

Matuszek, J.E., Shuter, B.J., and Casselman, J.M. 1990. Changes in Lake Trout growth and abundance after introduction of Cisco into Lake Opeongo, Ontario. Transactions of the American Fisheries Society 119: 718-729.

McCune, J.L., Harrower, W.L., Avery-Gomm, S., Brogan, J.M., Csergő, A.M., Davidson, L.N.K., Garani, A., Halpin, L.R., Lipsen, L.P.J., Lee, C., Nelson, J.C., Prugh, L.R., Stinson, C.M., Whitney, C.K., and Whitton, J. 2013. Threats to Canadian species at risk: An analysis of finalized recovery strategies. Biological Conservation 166: 254-265.

McLachlan, J.S., Hellmann, J.J., and Schwartz, M.W. 2007. A framework for debate of assisted migration in an era of climate change. Conservation Biology 21:297-302.

McMurray, S.E., and Roe, K.J. 2017. Perspectives on the controlled propagation, augmentation, and reintroduction of freshwater mussels (Mollusca: Bivalvia: Unionoida). Freshwater Mollusk Biology and Conservation 20: 1-12.

Meffe, G.K. 1995. Genetic and ecological guidelines for species reintroduction programs: application to Great Lakes fishes. Journal of Great Lakes Research 21(Suppl. 1): 3-9.

Micheli, F., and Halpern, B.S. 2005. Low functional redundancy in coastal marine assemblages. Ecology Letters 8(4): 391-400.

Milla, S., Pasquet, A., El Mohajer, L., and Fontaine, P. 2021. How domestication alters fish phenotypes. Reviews in Aquaculture 13: 388-405.

Mills, M.D., Rader, R.B., and Belk, M.C. 2004. Complex interactions between native and invasive fish: the simultaneous effects of multiple negative interactions. Oecologia 141: 713-721.

Minckley, W.L. 1995. Translocation as a tool for conserving imperiled fishes: experiences in western United States. Biological Conservation 72: 297-309.

Ministère du Développement durable, de l'Environnement, de la Faune et des Parcs. 2013. Outil d'aide à l'ensemencement des plans d'eau – Doré jaune (*Sander vitreus*). Direction générale de l'expertise sur la faune et ses habitats, Direction de la faune aquatique, Québec 12 pages.

Ministère des Ressources Naturelles et de la Faune [MRNF]. 2008. Lignes directrices sur les ensemencements de poissons. Secteur Faune Québec, Direction de l'expertise sur la faune et ses habitats. Québec. 41 p.

Mitchell, N., Rodriguez, N., Kuchling, G., Arnall, S., and Kearney, M.R. 2016. Reptile embryos and climate change: modelling limits of viability to inform translocation decisions. Biological Conservation 204: 134-147.

Mittelbach, G.G., Turner, A.M., Hall, D.J., Rettig, J.E., and Osenberg, C.W. 1995. Perturbation and resilience: a long-term whole-lake study of predator extinction and reintroduction. Ecology 76(8): 2347-2360.

Moore, J.E., Koops, M.A., and Cudmore, B. 2006. Quantitative Biological Risk Assessment Tool, v2. Fisheries and Oceans Canada, Burlington, ON.

Mouton, A.M., De Baets, B., and Goethals, P.L.M. 2010. Ecological relevance of performance criteria for species distribution models. Ecological Modelling 221(16): 1995-2002.

Natural Resource Solutions Inc. 2021. Argyle Street Bridge, Grand River, Caledonia: Mussel Species at Risk Summary Report. Permitting Report MTO Contract 3019-C-0668.

Neff, B.D., Garner, S.R., and Pitcher, T.E. 2011. Conservation and enhancement of wild fish populations: preserving genetic quality versus genetic diversity. Canadian Journal of Fisheries and Aquatic Sciences 68: 1139-1154.

Neves, R. 2004. Propagation of endangered freshwater mussels in North America. Journal of Conchology Special Publication 3: 69-80.

O'Grady, J.J., Reed, D.H., Brook, B.W., and Frankham, R. 2004. What are the best correlates of predicted extinction risk? Biological Conservation 118(4): 513-520.

Olden, J.D., Kennard, M.J., Lawler, J.J., and Poff, N.L. 2011. Challenges and opportunities in implementing managed relocation for conservation of freshwater species. Conservation Biology 25(1): 40-47.

Oliver, T.H., Heard, M.S., Isaac, N.J.B., Roy, D.B., Procter, D., Eigenbrod, F., Freckleton, R., Hector, A., Orme, C.D.L., Petchey, O.L., Proença, V., Raffaelli, D., Suttle, K.B., Mace, G.M., Martín-López, B., Woodcock, B.A., and Bullock, J.M. 2015. Biodiversity and resilience of ecosystem functions. Trends in Ecology & Evolution 30(11): 673-684.

Österling, M.E., and Larsen, B.M. 2013. Impact of origin and condition of host fish (*Salmo trutta*) on parasitic larvae of *Margaritifera margaritifera*. Aquatic Conservation: Marine and Freshwater Ecosystems 23(4): 564-570.

Palstra, F.P., and Fraser, D.J. 2012. Effective/census population size ratio estimation: A compendium and appraisal. Ecology and Evolution, 2(9): 2357–2365.

Palstra, F.P., and Ruzzante, D.E. 2008. Genetic estimates of contemporary effective population size: What can they tell us about the importance of genetic stochasticity for wild population persistence? Molecular Ecology 17(15): 3428-3447.

Patterson, M.A., Mair, R.A., Eckert, N.L., Gatenby, C.M., Brady, T., Jones, J.W., Simmons, B.R., and Devers, J.L. 2018. Freshwater Mussel Propagation for Restoration. Cambridge University Press. New York, New York.

Phillips, I.D., Vinebrooke, R.D., and Turner, M.A. 2009. Experimental reintroduction of the crayfish species *Oronectes virilis* into formerly acidified Lake 302S (Experimental Lakes Area, Canada). Canadian Journal of Fisheries and Aquatic Sciences 66: 1982-1902.

Pister, E.P. 2001. Wilderness fish stocking: history and perspective. Ecosystems 4: 279-286.

Pollock, L.J., Tingley, R., Morris, W.K., Golding, N., O'Hara, R.B., Parris, K.M., Vesk, P.A., and McCarthy, M.M. 2014. Understanding co-occurrence by modelling species simultaneously with a Joint Species Distribution Model (JSDM). Methods in Ecology and Evolution 5: 397-406.

Post, J.R., Sullivan, M., Coz, S., Lester, N.P., Walters, C.J., Parkinson, E.A., Paul, A.J., Jackson, L., and Shuter, B.J. 2002. Canada's recreational fisheries: the invisible collapse? Fisheries 27(1): 6-17.

Potts, L.B., Mandrak, N.E., and Chapman, L.J. 2021. Coping with climate change: Phenotypic plasticity in an imperilled freshwater fish in response to elevated water temperature. Aquatic Conservation: Marine and Freshwater Ecosystems. doi: 10.1002/aqc.3620.

Rakes, P.L., Shute, J.R., and Shute, P.W. 1999. Reproductive behavior, captive breeding, and restoration ecology of endangered fishes. Environmental Biology of Fishes 55: 31-42.

Reid, A.J., Carlson, A.K., Creed, I.F., Eliason, E.J., Gell, P.A., Johnson, P.T.J., Kidd, K.A., MacCormack, T.J., Olden, J.D., Ormerod, S.J., Smol, J.P., Taylor, W.W., Tockner, K., Vermaire, J.C., Dudgeon, D., and Cooke, S.J. 2019. Emerging threats and persistent conservation challenges for freshwater biodiversity. Biological Reviews 94(3): 849-873.

Reid, S.M., and Parna, S. 2017. Urbanization, long-term stream flow variability, and Redside Dace status in Greater Toronto Area streams. Can. Manuscr. Rep. Fish. Aquat. Sci. 3118: iv + 20 p.

Ricciardi, A., and Simberloff, D. 2009. Assisted colonization is not a viable conservation strategy. Trends in Ecology & Evolution 24(5): 248-253.

Richardson, D.M., Hellmann, J.J., McLachlan, J.S., Sax, D.F., Schwartz, M.W., Gonzalez, P., Brennan, E.J., Camacho, A., Root, T.L., Sala, O.E., Schneider, S.H., Ashe, D.M., Rappaport Clark, J., Early, R., Etterson, J.R., Fielder, E.D., Gill, J.L., Minteer, B.A., Polasky, S., Safford, H.D., Thompson, A.R., and Vellend, M. 2009. Multidimensional evaluation of managed relocation. PNAS 106(24): 9721-9724.

Robinson, K.F., and Jennings, C.A. 2012. Maximizing Age-0 spot export from a South Carolina estuary: an evaluation of coastal impoundment management alternatives via structured decision making. Marine and Coastal Fisheries: Dynamics, Management, and Ecosystem Science 4: 156-172.

Rodríguez, M.A., Marselli, G., and Mandrak, N.E. 2021. Responses of vulnerable fishes to environmental stressors in the Canadian Great Lakes basin. Canadian Journal of Fisheries and Aquatic Sciences 78: 1278-1292.

Rollinson, N., Keith, D.M., Houde, A.L.S., Debes, P.V., McBride, M.C., and Hutchings, J.A. 2014. Risk assessment of inbreeding and outbreeding depression in a captive-breeding program. Conservation Biology 28(2): 529-540.

Rooney, C.E. 2010. In-situ feasibility study of freshwater mussel reintroduction: survival and growth of the Wavy-rayed Lampmussel (*Lampsilis fasciola*) in the Pigeon River, NC. M.Sc. Thesis. Western Carolina University, North Carolina, USA. 48 pp.

Ryman, N., and Laikre, L. 1991. Efforts of supportive breeding on the genetically effective population size. Conservation Biology 5(3): 325-329.

Rytwinski, T., Kelly, L.A., Donaldson, L.A., Taylor, J.J., Smith, A., Drake, D.A.R., Martel, A., Geist, J., Morris, T.J., George, A.L., Dextrase, A.J., Bennett, J.R., and Cooke, S.J. 2021. What evidence exists for evaluating the effectiveness of conservation-oriented captive breeding and release programs for imperilled freshwater mussels and fishes. Canadian Journal of Fisheries and Aquatic Sciences 78(9): 1332-1346.

Saavedra, C., and Guerra, A. 1996. Allozyme heterozygosity, founder effect and fitness traits in a cultivated population of the European oyster, *Ostrea edulis*. Aquaculture 139: 203-224.

Sainsbury, A.W., and Vaughan-Higgins, R.J. 2011. Analyzing disease risks associated with translocations. Conservation Biology 26(3): 442-452.

Sard, N.M., Johnson, M.A., Jacobson, D.P., Hogansen, M.J., O'Malley, K.G., and Banks, M.A. 2016. Genetic monitoring guides adaptive management of a migratory fish reintroduction program. Animal Conservation 19(6): 570-577.

Schwartz, M.W., Hellman, J.J., McLachlan, J.M., Sax, D.F., Borevitz, J.O., Brennan, J., Camacho, A.E., Ceballos, G., Clark, J.R., Doremus, H., Early, R., Etterson, J.R., Fielder, D., Gill, J.L., Gonzalez, P., Green, N., Hannah, L., Jamieson, D.W., Javeline, D., Minteer, B.A., Odenbaugh, J., Polasky, S., Richardson, D.M., Root, T.L., Safford, H.D., Sala, O., Schneider, S.H., Thompson, A.R., Williams, J.W., Vellend, M.W., Vitt, P., and Zellmer, S. 2012. Managed relocation: integrating the scientific, regulatory, and ethical challenges. BioScience 62(8): 732-743.

Seddon, P.J. 2010. From reintroduction to assisted colonization: moving along the conservation translocation spectrum. Restoration Ecology 18(6): 796-802.

Seddon, P.J., Strauss, W.M., and Innes, J. 2012. Animal translocations: what are they and why do we do them? In: Reintroduction Biology: Integrating Sciences and Management. First edition. J.G. Ewen, D.P. Armstrong, K.A. Parker, and P.J. Seddon (eds). Blackwell Publishing Ltd. pg. 1-32.

Sexton, J.P., McIntyre, P.J., Angert, A.L., and Rice, K.J. 2009. Evolution and ecology of species range limits. Annual Review of Ecology, Evolution, and Systematics 40: 415-436.

Shute, J.R., Rakes, P.L., and Shute, P.W. 2005. Reintroduction of four imperiled fishes in Abrams Creek, Tennessee. Southeastern Naturalist 4(1): 93-110.

Sieben, K., Rippen, A.D., and Klemens Eriksson, B. 2011. Cascading effects from predator removal depend on resource availability in a benthic food web. Marine Biology 158: 391-400.

Silknetter, S., Creed, R.P., Brown, B.L., Frimpong, E.A., Skelton, J., and Peoples, B.K. 2020. Positive biotic interactions in freshwaters: A review and research directive. Freshwater Biology 65: 811-832.

Silknetter, S., Kanno, Y., Kanapeckas Métris, K.L., Cushman, E., Darden, T.L., and Peoples, B.K. 2019. Mutualism of parasitism: Partner abundance affects host fitness in a fish reproductive interaction. Freshwater Biology 64: 175-182.

Simberloff, D. 1995. Hybridization between native and introduced wildlife species: importance for conservation. Wildlife Biology 2(3): 143-150.

Snucins, E.J., Gunn, J.M., and Keller, W. 1995. Restoration of the Aurora Trout to its acid-damaged native habitat. Conservation Biology 9(5): 1307-1311.

Soberón, J., and Peterson, A.T. 2005. Interpretation of models of fundamental ecological niches and species' distributional areas. Biodiversity Informatics 2: 1-10.

Species at Risk Act [SARA]. 2002. Species at Risk Act S.C. 2002, c. 29. Accessed: 10/22/2021.

Spooner, D.E., and Vaughn, C.C. 2006. Context-dependent effects of freshwater mussels on stream benthic communities. Freshwater Biology 51: 1016-1024.

St. Jacques, J.-M., Douglas, M.S.V., Price, N., Drakulic, N., and Gubala, C.P. 2005. The effect of fish introductions on the diatom and cladoceran communities of Lake Opeongo, Ontario, Canada. Hydrobiologia 549: 99-113.

Stantial, M.L., Cohen, J.B., Darrah, A.J., Farrell, S.L., and Maslo, B. 2021. The effect of top predator removal on the distribution of a mesocarnivore and nest survival of an endangered shorebird. Avian Conservation & Ecology 16(1): 8. doi: 10.5751/ACE-01806-160108.

Storfer, A. 1999. Gene flow and endangered species translocations: a topic revisited. Biological Conservation 87(2): 173-180.

Strayer, D.L., Geist, J., Haag, W.R., Jackson, J.K., and Newbold, J.D. 2019. Essay: Making the most of recent advances in freshwater mussel propagation and restoration. Conservation Science and Practice 1:e53. doi: 10.1111/csp2.53

Sundland, O.T., Hesthagen, T., and Brabrand, Å. 2013. Coregonid introductions in Norway: well-intended and successful, but destructive. Advances in Limnology 64: 345-362.

Swan, K.D., Lloyd, N.A., and Moehrenschlager, A. 2018. Projecting further increases in conservation translocations: A Canadian case study. Biological Conservation 228: 175-182.

Tarszisz, E., Dickman, C.R., and Munn, A.J. 2014. Physiology in conservation translocations. Conservation Physiology 2: cou054.

Templeton, A.R. 1986. Coadaptation and outbreeding depression. Pages 105–166 in M. Soulé, editor. Conservation biology: the science of scarcity and diversity. Sinauer Associates, Sunderland, Massachusetts.

Thomas, G.R., Taylor, J., and Garcia de Leaniz, C. 2010. Captive breeding of the endangered freshwater pearl mussel *Margaritifera margaritifera*. Endangered Species Research 12:1-9.

Todesco, M., Pascual, M.A., Owens, G.L., Ostevik, K.L., Moyers, B.T., Hübner, S., Heredia, S.M., Hahn, M.A., Caseys, C., Bock, D.G., and Rieseberg, L.H. 2016. Hybridization and extinction. Evolutionary Applications 9(7): 892-908.

Turko, A.J., Leclair, A.T.A., Mandrak, N.E., Drake, D.A.R., Scott, G.R., and Pitcher, T.E. 2021. Choosing source populations for conservation reintroductions: lessons from variation in thermal tolerance among populations of the imperilled redside dace. Canadian Journal of Fisheries and Aquatic Sciences 78(9): 1347-1355.

Turko, A.J., Nolan, C.B., Balshine, S., Scott, G.R., and Pitcher, T.E. 2020. Thermal tolerance depends on season, age and body condition in imperilled redside dace *Clinostomus elongatus*. Conservation Physiology 8(1): coaa062.

Vachon, N. 2010. Reproduction artificielle, ensemencements et suivi du recrutement du chevalier cuivré en 2009, ministère des Ressources naturelles et de la Faune, Unité de gestion des ressources naturelles et de la faune de Montréal-Montérégie, Longueuil, Rapp. tech.16-44, vii + 28 p. + 5 annexes.

Vachon, N. 2021. Reproduction artificielle, ensemencements et suivi de la population du chevalier cuivré (*Moxostoma hubbsi*) en 2018. Direction de la gestion de la faune de l'Estrie, de Montréal, de la Montérégie et de Laval, Secteur des opérations régionales, ministère des Forêts, de la Faune et des Parcs. Rapport technique 16-57, viii + 43 p.

Vachon, N., et Sirois, C. 2019. Reproduction artificielle, ensemencements et suivi de la population du chevalier cuivré (*Moxostoma hubbsi*) en 2014, ministère des Forêts, de la Faune et des Parcs, Direction de la gestion de la faune de l'Estrie, de Montréal, de la Montérégie et de Laval, Rapport technique 16-54, 18 p.

Vachon, N., Velásquez-Medina, S., et Grondin, P. 2019. Motilité des spermatozoïdes du chevalier cuivré dans les différents traitements de cryopréservation en 2013. Ministère des Forêts, de la Faune et des Parcs. Direction de la gestion de la faune de l'Estrie, de Montréal, de la Montérégie et de Laval, Secteur de la faune, Rapport technique 16-47, 24 p.

van der Lee, A.S., and Koops, M.A. 2020. Recovery Potential Modelling of Westslope Cutthroat Trout (*Oncorhynchus clarkii lewisi*) in Canada (Saskatchewan-Nelson River populations). DFO Can. Sci. Advis. Sec. Res. Doc. 2020/046. v + 26 p.

van der Lee, A.S., Poesch, M.S., Drake, D.A.R., and Koops, M.A. 2020. Recovery Potential Modelling of Redside Dace (*Clinostomus elongatus*) in Canada. DFO Can. Sci. Advis. Sec. Res. Doc. 2019/034. v + 40 p.

VanTassel, N.M., Morris, T.J., Wilson, C.G., and Zanatta, D.T. 2021. Genetic diversity maintained in comparison of captive-propagated and wild populations of *Lampsilis fasciola* and *Ptychobranchus fasciolaris* (Bivalvia: Unionidae). Canadian Journal of Fisheries and Aquatic Sciences 78(9): 1312-1320.

Vaughn, C.C., Nichols, S.J., and Spooner, D.E. 2008. Community and foodweb ecology of freshwater mussels. Freshwater Science 27(2): 409-423.

Veech, J.A. 2013. A probabilistic model for analysing species co-occurrence. Global Ecology and Biogeography 22: 252-260.

Vélez-Espino, L.A., and Koops, M.A. 2009. Recovery potential assessment for Lake Sturgeon in Canadian Designatable Units. North American Journal of Fisheries Management 29: 1065-1090.

Vélez-Espino, L.A., Fox, M.H., and McLaughlin, R.L. 2006. Characterization of elasticity patterns of North American freshwater fishes. Canadian Journal of Fisheries and Aquatic Sciences 63: 2050-2066.

Wacker, S., Mejdell Larsen, B., Jakobsen, P., and Karlsson, S. 2019. Multiple paternity promotes genetic diversity in captive breeding of a freshwater mussel. Global Ecology and Conservation 17: e00564. doi: 10.1016/j.gecco.2019.e00564.

Wang, S., Hard, J.J., and Utter, F. 2002. Salmonid inbreeding: a review. Reviews in Fish Biology and Fisheries 11: 30-319.

Waples, R.S. 2005. Genetic estimates of contemporary effective population size: To what time periods do the estimates apply? Molecular Ecology 14(11): 3335-3352.

Weeks, A.R., Sgro, C.M., Young, A.G., Frankham, R., Mitchell, N.J., Miller, K.A., Byrne, M., Coates, D.J., Eldridge, M.D.B., Sunnucks, P., Breed, M.F., James, E.A., and Hoffman, A.A. 2011. Assessing the benefits and risks of translocations in changing environments: a genetic perspective. Evolutionary Applications 4: 709-725.

Weidel, B.C., Josephson, D.C., and Kraft, C.E. 2007. Littoral fish community response to smallmouth bass removal from an Adirondack lake. Transactions of the American Fisheries Society 136: 778-789.

Williams, S.E., and Hoffman, E.A. 2009. Minimizing genetic adaptation in captive breeding programs: A review. Biological Conservation 142(11): 2388-2400.

Wilson, C.D., Arnott, G., and Elwood, R.W. 2012. Freshwater pearl mussels show plasticity of responses to different predation risks but also show consistent individual differences in responsiveness. Behavioural Processes 89: 299-303.

Winemiller, K.O., and Rose, K.A. 1992. Patterns of life-history diversification in North American fishes: implications for population regulation. Canadian Journal of Fisheries and Aquatic Sciences 49(10): 2196-2218.

Wrona, F.J., and Dixon, R.W.J. 1991. Group size and predation risk: a field analysis of encounter and dilution effects. The American Naturalist 137(2): 186-201.

Yachi, S., and Loreau, M. 2001. Biodiversity and ecosystem productivity in a fluctuating environment: The insurance hypothesis. Proceedings of the National Academy of Sciences USA 96: 1463-1468.

Zurell, D. 2020. Introduction to species distribution modelling (SDM) in R. Personal website. Accessed: 2021-06-09.

Zurell, D., Franklin, J., König, C., Bouchet, P.J., Dormann, C.F., Elith, J., Fandos, G., Feng, X., Guillera-Arroita, G., Guisan, A., Lahoz-Monfort, J.J., Leitão, P.J., Park, D.S., Peterson, A.T., Rapacciuolo, G., Schmatz, D.R., Schröder, B., Serra-Diaz, J.M., Thuiller, W., Yates, K.L., Zimmermann, N.E., and Merow, C. 2020. A standard protocol for reporting species distribution models. Ecography 43(9): 1261-1277.

## APPENDIX A: REINTRODUCTION OF EASTERN SAND DARTER (*AMMOCRYPTA PELLUCIDA*; ONTARIO POPULATIONS) TO BIG OTTER CREEK, ONTARIO, CANADA USING THE CONSERVATION TRANSLOCATION FRAMEWORK

### INTRODUCTION

Eastern Sand Darter (*Ammocrypta pellucida*) is a small (< 8 cm) freshwater fish species distributed across central North America, including Ontario and Québec. Ontario and Québec populations are both listed as Threatened as separate Designatable Units (DUs) under the *Species at Risk Act* (Fisheries and Oceans Canada 2012, 2014). Population extirpations have occurred across its Canadian range, including in the Ausable River, Catfish Creek, and Big Otter Creek in Ontario (Fisheries and Oceans Canada 2012), and the Châteauguay, Yamaska, and Saint-François rivers in Québec (Fisheries and Oceans Canada 2014). Reintroduction has been identified as a potential recovery strategy for the Ontario DU but has not yet been initiated.

Below, an example of how to use the conservation translocation decision support framework is presented for Eastern Sand Darter in Big Otter Creek, Ontario. Specifically, worked examples of Steps 1–3 are provided to illustrate the use of available information for informing decision-making (Figure A1). Although not meant to be used directly for management of Eastern Sand Darter in Big Otter Creek, this example lays the groundwork for future species-specific assessments.

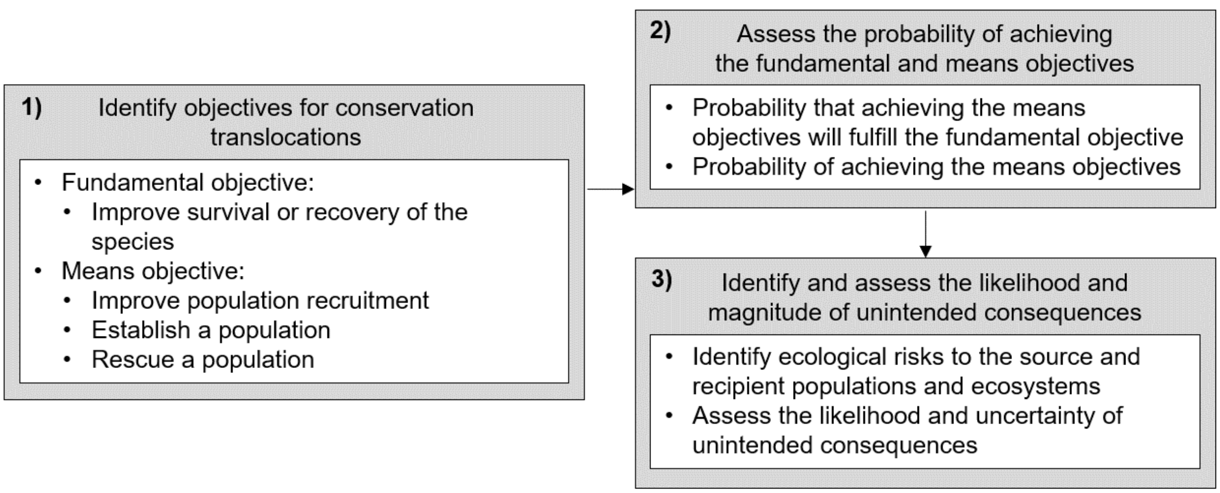


*Figure A1. The first three steps in the conservation translocation decision support framework being considered in the example appendices.*

### STEP 1: IDENTIFY OBJECTIVES FOR CONSERVATION TRANSLOCATIONS

The first step in considering conservation translocations is to identify the fundamental and means objectives. Here, the fundamental objective is to improve the survival or recovery of Eastern Sand Darter (Ontario population). The means objective for achieving the fundamental objective is to re-establish a population in Big Otter Creek, Ontario (i.e., reintroduction). Reintroduction is warranted over supplementation as long-term sampling has failed to detect Eastern Sand Darter in the formerly occupied Big Otter Creek (e.g., Barnucz et al. 2020).

## STEP 2: ASSESS THE PROBABILITY OF ACHIEVING THE FUNDAMENTAL AND MEANS OBJECTIVES

The next step in the conservation translocation decision support framework is to assess the probability that achieving the means objective will improve survival or recovery of the species, and to assess the probability of achieving the means objective. Included is the need to gather information on factors that will influence the ability to achieve the means and fundamental objectives and to score the likelihood and magnitude of their influence. Here, efforts to estimate the probability of the means objective achieving the fundamental objective for Eastern Sand Darter are described, along with potential factors that may influence the ability to achieve the means objective. However, scoring of the influence of individual factors on the ability to achieve the means objective has not been performed given that the goal of this appendix is to act as a functional guide for using the decision support framework, and not for the use of context-specific management actions.

### Probability of the means objective achieving the fundamental objective

Basic simulations suggest that increasing the number of populations for a species decreases the probability of extinction, thus increasing long-term species viability (Figure 7). Given low impact on existing populations when sourcing a reintroduction, the establishment of a self-sustaining population in Big Otter Creek would increase the long-term persistence of the Ontario DU. Population models using measured vital rates have been developed for Eastern Sand Darter to evaluate the potential for species recovery (Finch et al. 2011, 2018; Fisheries and Oceans Canada 2011), including to assess the potential for re-establishing a population by species reintroduction (Lamothe et al. 2021). Lamothe et al. (2021) used population models to understand:

1. How the establishment probability of reintroduced populations is affected by life-history characteristics and the number of individuals released;
2. The consequence of removing individuals from source populations at different frequencies and magnitudes given different life-history characteristics; and,
3. The optimal trade-off between removals from source populations and the probability of successful re-establishment.

A variety of scenarios were modelled in Lamothe et al. (2021) due to the uncertainty regarding Eastern Sand Darter life-history, population characteristics, and the response of individuals to translocation. Uncertainty was incorporated into the models for population growth rates, Allee effects, and mortality during the act of transportation, along with the potential effects of environmental and demographic stochasticity. Moreover, models were run for different source population sizes and various management strategies (i.e., frequency and magnitude of removals). Each scenario was considered for its probability of achieving successful re-establishment against the probability of causing source population extirpation, where success was defined as the persistence of a translocated population with an abundance greater than the estimated minimum viable population size over the last 15 years of the simulation. For example, considering a high rate of translocation mortality (70%) and Allee effects, approximately 105 individuals need to be translocated annually from a source population of at least 10,759 individuals for a decade to achieve translocation success if population growth rate of the introduced individuals = 2.69, where the probability of success is ≥ 90% and risk of source population extirpation is ≤ 1% (Figure A2). However, if population growth rate = 1.56, a source population of approximately 46,817 individuals would be needed with more than double the number of individuals (*n* = 235) removed annually for a decade to achieve a ≥ 90% probability of successful establishment with a low probability of extirpation (i.e., ≤ 1%; Figure A2; Lamothe et al. 2021). Overall, the results of Lamothe et al. (2021) suggest that the risk of population extirpation is relatively low unless very large numbers of individuals are removed from a relatively small source population.

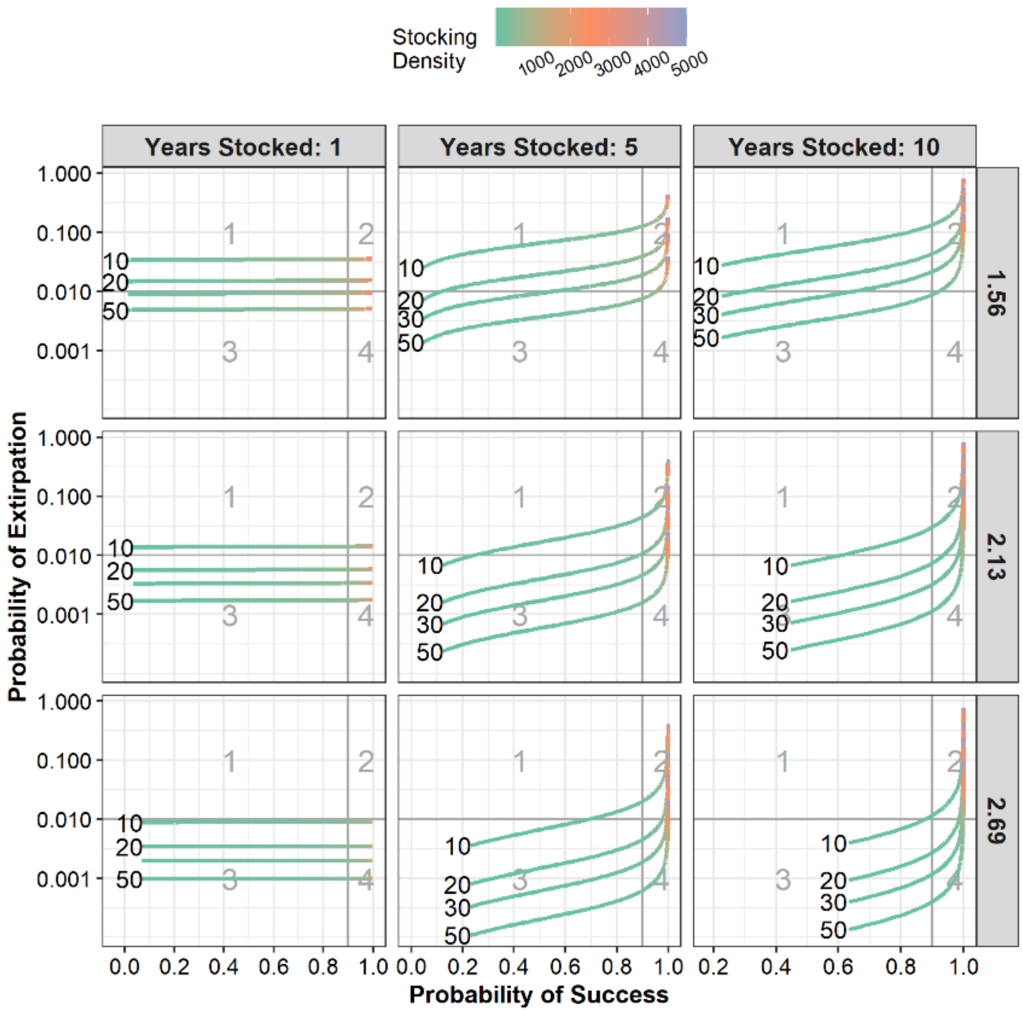


*Figure A2. Simulated probability of source population extirpation (log-scale) for various carrying capacities (10 = 10,000, 20 = 20,000, 30 = 30,000, 50 = 50,000 individuals) versus the probability of successful translocation. Presented are the results when removing individuals for one, five, or 10 consecutive years pre-spawn from a source population and releasing them immediately, whereby populations show differences in population growth rate (λ = 1.56, 2.13, 2.69). Probability of success is defined as maintaining an adult population after reintroduction with a geometric mean population size greater than the minimum viable population size (95%) over the last 15 years of the simulation. The grey boxes and numbers represent the boundaries of the cost-benefit outcomes with an optimal outcome of ≤ 1% probability of extirpation for a ≥ 90% probability of success indicated by the number 4. Original figure presented Lamothe et al. (2021).*

## Population considerations

The distribution of the Eastern Sand Darter Ontario DU is well-described (Table A1; Figure A3), with eight populations considered extant and three known extirpated populations. Populations in Ontario are geographically disconnected from each other, and from populations in Québec or the United States. Natural dispersal to recolonize historically occupied sites is unlikely given the physical limitations of the species and geographic distance between sites. The abundance of Ontario Eastern Sand Darter is generally unknown across river systems, however, the Grand and Thames rivers are considered to contain the largest and most stable populations (Table A1). The number of populations and abundance of individuals across southern Ontario suggests that the species is not at imminent risk of extinction.

*Table A1. Relative abundance index, population trajectory, and population status of each Eastern Sand Darter population in Ontario, sorted by drainage. Numbers inside parentheses indicate certainty rankings, listed as: 1 = quantitative analysis, 2 = catch per unit effort or standardized sampling, or 3 = expert opinion. Originally modified from Bouvier and Mandrak (2010).*

| Population | Relative abundance index | Population trajectory | Population status |
|---|---|---|---|
| **Lake Huron** | | | |
| *Ausable River* | Extirpated (2) | Not applicable | Extirpated |
| **Lake St. Clair** | | | |
| *Lake St. Clair* | Low (2) | Declining (3) | Poor |
| *Thames River* | High (1) | Stable (1) | Good |
| *Sydenham River* | Low (2) | Unknown (3) | Poor |
| **Lake Erie** | | | |
| *Pelee Island* | Unknown (3) | Unknown (3) | Unknown |
| *Rondeau Bay* | Unknown (3) | Unknown (3) | Unknown |
| *Long Point Bay* | Low (2) | Declining (2) | Poor |
| *Catfish Creek* | Extirpated (3) | Not applicable | Extirpated |
| *Big Otter Creek* | Extirpated (3) | Not applicable | Extirpated |
| *Big Creek* | Low (3) | Unknown (3) | Poor |
| *Grand River* | High (2) | Stable (2) | Good |

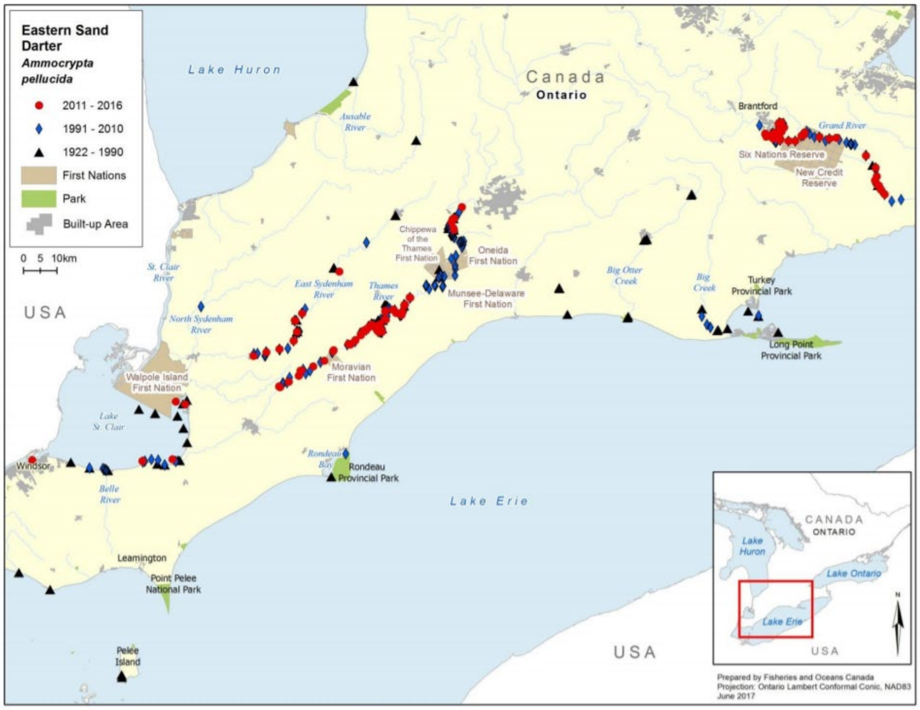


*Figure A3. Historical distribution and recent detections of Eastern Sand Darter in southwestern Ontario. Originally published in Fisheries and Oceans Canada (2018). Not pictured is West Lake.*

The genetic structure of Eastern Sand Darter has been resolved and generally reflects the northward spread of the species after the Wisconsinan glacial period (Ginson et al. 2015). Based on this information, and given the location of Big Otter Creek, individuals from the Lake Erie drainage within the Ontario DU should be considered as a source for reintroduction. Within the Lake Erie drainage, only the Grand River is considered to support a relatively strong population (Table A1), and is therefore the most likely candidate for sourcing individuals for conservation translocations. However, the abundance of the Eastern Sand Darter population in the Grand River is unknown.

## Habitat considerations

Habitat for aquatic species includes all spawning grounds, areas for nursery, rearing, food supply, migration, and any other areas on which aquatic species depend directly or indirectly to carry out their life processes, or areas where aquatic species formerly occurred and have the potential to be reintroduced. Much is known about habitat requirements of Eastern Sand Darter (Scott and Crossman 1973; Trautman 1981; Daniels 1993; COSEWIC 2009; Dextrase 2013; Dextrase et al. 2014). It is a warm-water, non-migratory species that requires relatively silt-free, sand and fine gravel substrate in relatively clear waters. The species is typically found in sandy depositional bends of rivers where moderate current helps to maintain relatively clean substrates (Daniels 1993; Trautman 1981). Spawning takes place in the sand and fine gravel substrates during late spring to summer, when water temperature ranges between 14.4°C and 25.5°C (Williams 1975; Johnston 1989; Simon and Wallus 2006; Fisheries and Oceans Canada 2014). The species is insectivorous, with a diet composed primarily of larval midges (Chironomidae) and blackflies (Simuliidae; Scott and Crossman 1973; COSEWIC 2009; Fisheries and Oceans Canada 2014). A more complete summary of the habitat requirements for Eastern Sand Darter in Ontario is provided in COSEWIC (2009) and Fisheries and Oceans Canada (2012, 2014).

Silt-free, sand and fine gravel substrate bars in Big Otter Creek have been restored over time after changes were made to historical agricultural practices. As noted above, present-day sampling in Big Otter Creek suggests that sites composed primarily of sand substrates are available, but turbidity may remain a concern (Table A1). Research activities are underway to better understand the similarity in microhabitat characteristics (i.e., substrate) between Big Otter Creek and potential source locations (e.g., Grand River).

### Community considerations

Eastern Sand Darter is small-bodied benthic species. It is assumed that Eastern Sand Darter is vulnerable to predation by non gape-limited fishes. Competition between other benthic species for resources is also likely. A positive association has been demonstrated between Silver Shiner and Eastern Sand Darter, which is likely a function of habitat suitability rather than biotic interactions (Lamothe et al. 2019a, b). Eastern Sand Darter has no known obligate biotic interactions. Many invasive species have been introduced to Ontario since the extirpation of Eastern Sand Darter from Big Otter Creek. For example, Round Goby (*Neogobius melanostomus*) has been documented as far north as Tillsonburg, Ontario within Big Otter Creek (Barnucz et al. 2020). An analysis of the community composition at Big Otter Creek relative to other populations has not occurred but can be performed with available data.

### Threat considerations

The threats to the Eastern Sand Darter DU have been described (Table A2; Fisheries and Oceans Canada 2014), with threats to Eastern Sand Darter critical habitat described in Fisheries and Oceans Canada (2018). Identified threats to Eastern Sand Darter populations include turbidity and sediment loading, contaminants and toxic substances, nutrient loading, barriers to movement, altered flow regimes, shoreline modifications, exotic species, and disease. The magnitude of the effects varies by location (Table A2).

*Table A2. Summary of threats to Eastern Sand Darter populations in Ontario. Threat Status for all Eastern Sand Darter populations in Ontario resulting from an analysis of both the Threat Likelihood and Threat Impact. The number in brackets refers to the level of certainty assigned to each Threat Status, which reflects the lowest level of certainty associated with either initial parameter (Threat Likelihood or Threat Impact). Certainty has been classified as: 1 = causative studies; 2 = correlative studies; 3 = expert opinion. Gray cells indicate that the threat is not applicable to the population due to the nature of the aquatic system where the population is located. Originally adapted from Bouvier and Mandrak (2010).*

| | **Lake Huron** | **Lake St. Clair** | **Lake St. Clair** | **Lake St. Clair** | **Lake Erie** | **Lake Erie** | **Lake Erie** | **Lake Erie** | **Lake Erie** | **Lake Erie** | **Lake Erie** |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Threats** | Ausable River | Lake St. Clair | Thames River | Sydenham River | Pelee Island | Rondeau Bay | Long Point Bay | Catfish Creek | Big Otter Creek | Big Creek | Grand River |
| **Turbidity and sediment loading** | High (3) | High (3) | High (3) | High (3) | Medium (3) | High (3) | High (3) | High (3) | High (3) | High (3) | High (3) |
| **Contaminants and toxic substances** | High (3) | High (3) | High (3) | High (3) | - | High (3) | - | - | - | - | Medium (3) |
| **Nutrient loading** | Medium (3) | Medium (3) | Medium (3) | Medium (3) | Low (3) | Medium (3) | Medium (3) | Medium (3) | Medium (3) | Medium (3) | Medium (3) |
| **Barriers to movement** | - | - | - | High (3) | - | - | - | - | Medium (3) | Low (3) | Medium (3) |
| **Altered flow regime** | High (3) | - | High (3) | High (3) | - | - | - | High (3) | Medium (3) | High (3) | High (3) |
| **Shoreline modification** | Medium (3) | Medium (3) | Medium (3) | Medium (3) | Medium (3) | High (3) | Low (3) | Medium (3) | TBD | TBD | High (3) |
| **Exotic species and disease** | High (3) | High (3) | High (3) | High (3) | High (3) | High (3) | High (3) | High (3) | High (3) | High (3) | High (3) |
| **Incidental harvest** | Low (3) | Low (3) | Low (3) | Low (3) | Low (3) | Low (3) | Low (3) | Low (3) | Low (3) | Low (3) | Low (3) |

Turbidity and sediment loading, and exotic species are the most concerning threats to Eastern Sand Darter across its range, including in Big Otter Creek. Big Otter Creek is within what was historically known as the Tobacco Belt in southwestern Ontario. Due to historical agricultural practices for tobacco, freshwater ecosystems were left impacted by sediment loads, excessive turbidity, and changes in water chemistry. Changes in agricultural practices have improved water quality conditions in these areas relative to the periods when the species was extirpated. Present-day sampling in Big Otter Creek suggests historically occupied sites are now composed primarily of sand substrates, however, turbidity remains relatively high compared to the Ausable River (Barnucz et al. 2020), another formerly occupied river system (Table A1).

Fish sampling of historically occupied sites in Big Otter Creek was performed in 2018 (Barnucz et al. 2020). Round Goby, an invasive species, was among the most abundant and most frequently encountered invasive species in the system (Barnucz et al. 2020). Recent sampling in Big Otter Creek suggests Round Goby occupies habitats as far north as Tillsonburg (i.e., the complete historic range of Eastern Sand Darter in this system; Barnucz et al. 2020). Given the perceived threat of Round Goby to Eastern Sand Darter (Fisheries and Oceans Canada 2011; Raab et al. 2018), the abundance of Round Goby in Big Otter Creek could reduce the ability to achieve the means objective of re-establishing a population. In the last 10 years, Round Goby and Eastern Sand Darter have begun to co-occur in locations considered to be in “Good” status (i.e., Grand and Thames rivers; Table A2), however, long-term outcomes of this co-occurrence are still uncertain. More information on threats that could potentially affect the ability to achieve the means objective can be found in Fisheries and Oceans Canada (2014) and Fisheries and Oceans Canada (2018).

### Estimate the expected benefits of performing conservation translocation

Given the factors described above, and the modeling performed to estimate the probability of achieving a successful reintroduction, the potential influence of population, habitat, community, and threat considerations on achieving the means objective must be scored. Given the need for input from additional Eastern Sand Darter experts, Table A3 has not been filled.

*Table A3. Considerations for evaluating the ability to re-establish an extirpated population of Eastern Sand Darter in Big Otter Creek, Ontario.*

***Focal species***: *Eastern Sand Darter (Ammocrypta pellucida) Ontario population - THREATENED*
***Problem statement***: *Eastern Sand Darter – Ontario DU is listed as Threatened under SARA. Population extirpations have occurred across its Canadian range, including in Big Otter Creek, Ontario. Reintroduction has been identified as a potential recovery strategy but has not yet occurred. The Grand River population is considered to be the best source for performing reintroductions to Big Otter Creek.*
***Fundamental objective***: *Improve survival or recovery of Eastern Sand Darter in Canada.*
***Means objective***: *Re-establish a population of Eastern Sand Darter in Big Otter Creek, Ontario.*

| *Category* | *Factors* | *Likelihood* | *Evidence strength* | *Agreement* | *Confidence* | *References* | *Additional considerations* |
|---|---|---|---|---|---|---|---|
| Population considerations | Abundance of the source population(s) is suitable to achieve the means objective | - | - | - | - | - | - |
| | Age-structure of the source population is suitable to achieve the means objective | - | - | - | - | - | - |
| | Genetic diversity and variation of the source population is suitable to achieve the means objective | - | - | - | - | - | - |
| | Genetic diversity and variation of the recipient population is suitable to achieve the means objective | - | - | - | - | - | - |
| | Life-history strategy of the source population is suitable to achieve the means objective | - | - | - | - | - | - |
| | Captive breeding or captive rearing techniques are available to achieve the means objective | - | - | - | - | - | - |
| Habitat | Habitat in the recipient site(s) reflect species requirements (e.g., water clarity, water velocity, depth, vegetation, substrate) | - | - | - | - | - | - |
| | A sufficient quantity of habitat exists in the recipient location to support all life-stages | - | - | - | - | - | - |
| | Sufficient connectivity exists in the recipient habitat to support all life-stages | - | - | - | - | - | - |
| Community considerations | Obligate, facultative, or parasitic species dependencies limit the ability to achieve the means objective | - | - | - | - | - | - |
| Threats | Pertinent threats limit the ability to achieve the means objective, including: | - | - | - | - | - | - |
| | Invasive species | - | - | - | - | - | - |
| | Residential and commercial development | - | - | - | - | - | - |
| | Agriculture and aquaculture | - | - | - | - | - | - |
| | Energy production and mining | - | - | - | - | - | - |
| | Biological resource use | - | - | - | - | - | - |
| | Transportation and service corridors | - | - | - | - | - | - |
| | Human intrusions and disturbance | - | - | - | - | - | - |
| | Natural systems modification | - | - | - | - | - | - |
| | Pollution | - | - | - | - | - | - |
| | Geological events | - | - | - | - | - | - |

| *Category* | *Factors* | *Likelihood* | *Evidence strength* | *Agreement* | *Confidence* | *References* | *Additional considerations* |
|---|---|---|---|---|---|---|---|
| Threats (cont.) | Parasites or disease transfer | - | - | - | - | - | - |
| | Future habitat modification or degradation | - | - | - | - | - | - |
| | Climate change | - | - | - | - | - | - |
| | Other factors | - | - | - | - | - | - |

## STEP 3: IDENTIFY AND ASSESS THE LIKELIHOOD AND MAGNITUDE OF UNINTENDED CONSEQUENCES

The next step in the decision support framework is to identify and assess the likelihood and magnitude of negative consequences of reintroduction on the source and recipient populations and ecosystems. For the context of Eastern Sand Darter reintroduction to Big Otter Creek, this includes changes in source population persistence and genetic variation, changes in short- and long-term community and ecosystem dynamics in the source and recipient ecosystems, and the transfer of disease to the recipient ecosystem.

As described above, population models have been developed to estimate the probability of reducing source population persistence as a function of direct removals (Lamothe et al. 2021). The results suggested that there was a low probability of source population extirpation resulting from the frequency and magnitude of removals needed to re-establish a population. However, given the uncertainty about the abundance and trajectory of Eastern Sand Darter in the Grand River (Table A1), the likelihood and magnitude of risk cannot be definitively stated. Research on the genetic structure of Eastern Sand Darter populations suggests low within-river genetic variation (Ginson et al. 2015), which may suggest a relatively low risk of altering source population genetic structure when performing removals. There is no evidence of hybridization between Eastern Sand Darter and co-occurring species in Canada. An assessment has yet to be performed on the risk of disease transfer when performing conservation translocations for Eastern Sand Darter.

### Estimate the risk of performing conservation translocations

The next step is to score the likelihood and magnitude of potential risks on the ability to achieve the means and fundamental objectives. Similar to Table A3, Table A4 has not been filled given the scope of this document and need for input from species experts.

*Table A4. Risk considerations for Eastern Sand Darter populations and other ecosystem components in source and recipient habitats of proposed conservation translocations.*

| *Subject* | *Location* | *Risk category* | *Risk outcome* | *Risk likelihood* | *Risk magnitude* | *Evidence strength* | *Agreement* | *Confidence* | *References* | *Additional considerations* |
|---|---|---|---|---|---|---|---|---|---|---|
| Focal | Source | Population persistence | Reduced or altered population abundance | - | - | - | - | - | - | - |
| Focal | Source | Genetic variation | Altered genetic variation | - | - | - | - | - | - | - |
| Focal | Source | Genetic variation | Inbreeding depression | - | - | - | - | - | - | - |
| Focal | Recipient | Population persistence | Individual mortality | - | - | - | - | - | - | - |
| Focal | Recipient | Genetic variation | Founder effect | - | - | - | - | - | - | - |
| Focal | Recipient | Genetic variation | Outbreeding depression | - | - | - | - | - | - | - |
| Focal | Recipient | Genetic variation | Hybridization | - | - | - | - | - | - | - |
| Other | Source | Community and ecosystem dynamics | Increased negative interactions | - | - | - | - | - | - | - |
| Other | Source | Community and ecosystem dynamics | Reduced positive interactions | - | - | - | - | - | - | - |
| Other | Source | Community and ecosystem dynamics | Reduced habitat availability | - | - | - | - | - | - | - |
| Other | Source | Community and ecosystem dynamics | Altered ecosystem processes | - | - | - | - | - | - | - |
| Other | Recipient | Community and ecosystem dynamics | Enhanced negative interactions | - | - | - | - | - | - | - |
| Other | Recipient | Community and ecosystem dynamics | Reduced positive interactions | - | - | - | - | - | - | - |
| Other | Recipient | Community and ecosystem dynamics | Transformative changes within site of introduction | - | - | - | - | - | - | - |
| Other | Recipient | Community and ecosystem dynamics | Transformative changes beyond | - | - | - | - | - | - | - |

| *Subject* | *Location* | *Risk category* | *Risk outcome* | *Risk likelihood* | *Risk magnitude* | *Evidence strength* | *Agreement* | *Confidence* | *References* | *Additional considerations* |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | site of introduction | | | | | | | |
| Other | Recipient | Community and ecosystem dynamics | Reduced habitat availability | - | - | - | - | - | - | - |
| All | Both | Disease transfer | Individual mortality / reductions in fitness | - | - | - | - | - | - | - |

## REFERENCES


Barnucz, J., Reid, S.M., and Drake, D.A.R. 2020. Targeted surveys for Eastern Sand Darter in the upper Ausable River and Big Otter Creek, 2018. Can. Data Rep. Fish. Aquat. Sci. 1312: vi + 26 p.

Bouvier, L.D., and N.E. Mandrak. 2010. Information in support of a recovery potential assessment of Eastern Sand Darter (*Ammocrypta pellucida*) in Ontario. DFO Can. Sci. Advis. Sec. Res. Doc. 2010/093. vi + 43pp.

COSEWIC. 2009. COSEWIC assessment and status report on the Eastern Sand Darter *Ammocrypta pellucida*, Ontario populations and Quebec populations, in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa, ON. vii + 49 pp.

Daniels, R.A. 1993. Habitat of the Eastern Sand Darter, *Ammocrypta pellucida*. Journal of Freshwater Ecology 8(4): 287-295.

Dextrase, A.J. 2013. Modelling occupancy and abundance of Eastern Sand Darter (*Ammocrypta pellucida*) while accounting for imperfect detection. PhD dissertation, Trent University, Peterborough, Ontario.

Dextrase, A.J., Mandrak, N.E., and Schaeffer, J.A. 2014. Modelling occupancy of an imperilled stream fish at multiple scales while accounting for imperfect detection: implications for conservation. Freshwater Biology 59: 1799-1815.

Finch, M., Vélez-Espino, L.A., Doka, S.E., Power, M. and Koops, M.A. 2011. Recovery potential modelling of Eastern Sand Darter (*Ammocrypta pellucida*) in Canada. DFO Can. Sci. Advis. Sec. Res. Doc. 2011/020. vi + 34 p.

Finch, M., Koops, M.A., Doka, S.E., and Power, M. 2018. Population viability and perturbation analyses to support recovery of imperilled Eastern Sand Darter (*Ammocrypta pellucida*). Ecology of Freshwater Fish 27(1): 378-388.

Fisheries and Oceans Canada. 2011. Recovery potential assessment of Eastern Sand Darter (*Ammocrypta pellucida*) in Canada. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2011/020.

Fisheries and Oceans Canada. 2012. Recovery strategy for the Eastern Sand Darter (*Ammocrypta pellucida*) in Canada: Ontario Populations. Species at Risk Act Recovery Strategy Series, Fisheries and Oceans Canada, Ottawa. vii + 56 pp.

Fisheries and Oceans Canada. 2014. Recovery strategy for the Eastern Sand Darter (*Ammocrypta pellucida*) in Canada: Quebec Populations. Species at Risk Act Recovery Strategy Series, Fisheries and Oceans Canada, Ottawa. vii + 47 pp.

Fisheries and Oceans Canada. 2018. Report on the Progress of Recovery Strategy Implementation for the Eastern Sand Darter (*Ammocrypta pellucida*) in Canada (Ontario Populations) for the Period 2012 – 2017. *Species at Risk Act* Recovery Strategy Report Series. Fisheries and Oceans Canada, Ottawa. v + 33 p.

Ginson, R., Walter, R.P., Mandrak, N.E., Beneteau, C.L., and Heath, D.D. 2015. Hierarchal analysis of genetic structure in the habitat-specialist Eastern Sand Darter (*Ammocrypta pellucida*). Ecology and Evolution 5: 695-708.

Johnston, C.E. 1989. Spawning in the Eastern Sand Darter, *Ammocrypta pellucida* (Pisces: Percidae), with comments on the phylogeny of *Ammocrypta* and related taxa. Transactions of the Illinois Academy of Sciences 82(3-4): 163-168.

Lamothe, K.A., Dextrase, A.J., and Drake, D.A.R. 2019a. Aggregation of two imperfectly detected imperilled freshwater fishes: Understanding community structure and co-occurrence for multispecies conservation. Endangered Species Research 40: 123-132.

Lamothe, K.A., Dextrase, A.J., and Drake, D.A.R. 2019b. Characterizing species co-occurrence patterns of imperfectly detected stream fishes to inform species reintroduction efforts. Conservation Biology 33(6): 1392-1403.

Lamothe, K.A., van der Lee, A.S., Drake, D.A.R., and Koops, M.A. 2021. The translocation trade-off for eastern sand darter (*Ammocrypta pellucida*): balancing harm to source populations with the goal of re-establishment. Canadian Journal of Fisheries and Aquatic Sciences. doi: 10.1139/cjfas-2020-0288.

Raab, D., Mandrak, N.E., and Ricciardi, A. 2018. Low-head dams facilitate Round Goby *Neogobius melanostomus* invasion. Biological Invasions 20: 757-776.

Scott, W.B., and Crossman, E.J. 1973. Freshwater Fishes of Canada. Fisheries Research Board of Canada. Bulletin 184. Ottawa, Canada.

Simon, T.P., and Wallus, R. 2006. Reproductive biology and early life history of fishes in the Ohio River: Percidae – perch, pikeperch and darters, Volume 4. CRC Press, Boca Raton, Florida, USA.

Trautman, M.B. 1981. The Fishes of Ohio with Illustrated Keys: Revised Edition. Ohio State University Press. Ohio, United States.

Williams, J.D. 1975. Systematics of the percid fishes of the subgenus *Ammocrypta*, genus *Ammocrypta*, with descriptions of two new species. Bulletin of the Alabama Museum of Natural History 1: 1-56.

## APPENDIX B: REINTRODUCTION OF SNUFFBOX (*EPIOBLASMA TRIQUETRA*) TO THE THAMES RIVER, ONTARIO, CANADA USING THE CONSERVATION TRANSLOCATION FRAMEWORK

## INTRODUCTION

Snuffbox (*Epioblasma triquetra*; Rafinesque 1820) is a relatively small, sexually dimorphic freshwater mussel listed as Endangered under SARA (COSEWIC 2011; Fisheries and Oceans Canada 2019). As of 2011, there were 31 historical records of Snuffbox from southern Ontario, including from Lake Erie, Lake St. Clair, and the Ausable, Sydenham, Thames, Grand, and Niagara rivers; however, Snuffbox can only be presently found in a 93 km stretch of the East Sydenham River and at five sites within a 91 km stretch of the Ausable River (Figure A4; Fisheries and Oceans Canada 2019). Conservation translocation has been identified as a potential recovery strategy for the species but has not yet occurred.

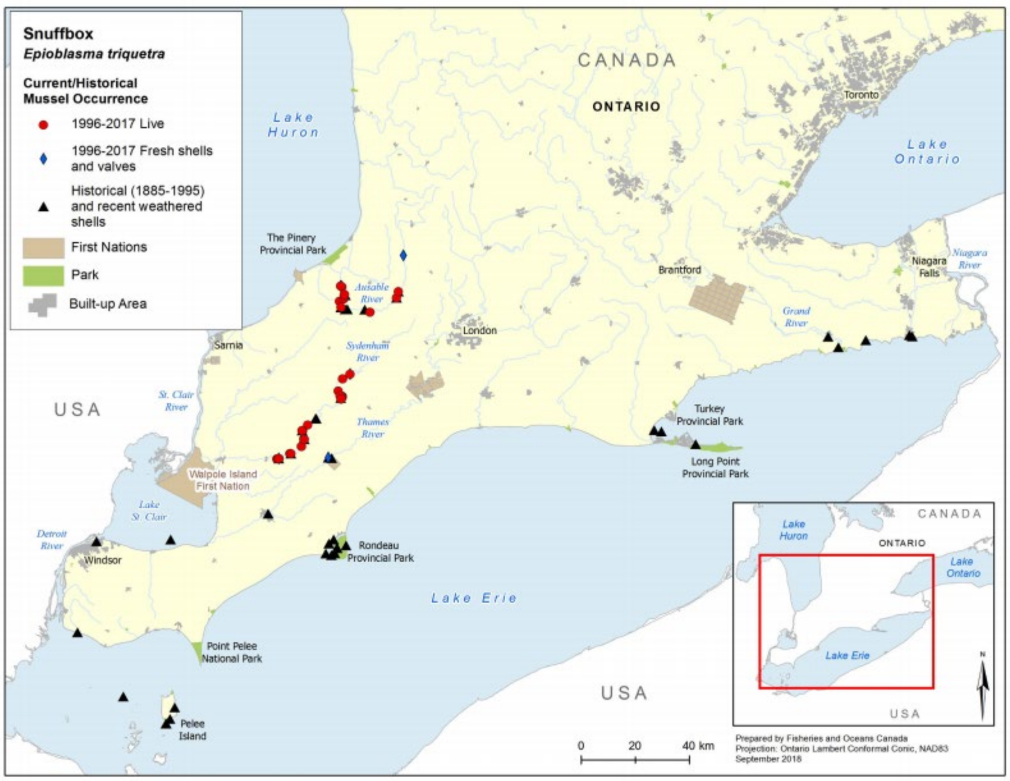


*Figure A4. Historical distribution and recent detections of Snuffbox in southwestern Ontario. Originally published in Fisheries and Oceans Canada (2019).*

Below, an example of how to use the conservation translocation decision support framework is presented for Snuffbox in the Thames River, Ontario. Specifically, Steps 1-3 are navigated to understand the level of available information for informing decision-making (Figure A1). Although not meant to be used directly for management of Snuffbox, this demonstration provides a foundation for future assessments.

## STEP 1: IDENTIFY OBJECTIVES FOR CONSERVATION TRANSLOCATIONS

The first step in considering conservation translocations is to identify the fundamental and means objectives. Here, the fundamental objective is to improve the survival or recovery of

Snuffbox in Canada. The approach to achieving the fundamental objective is reintroduction to the Thames River, Ontario, between Chatham and Moraviantown.

## STEP 2: ASSESS THE PROBABILITY OF ACHIEVING THE FUNDAMENTAL AND MEANS OBJECTIVES

The next step in the conservation translocation decision support framework is to assess the probability that achieving the means objective will improve survival or recovery of the species, and to assess the probability of achieving the means objective. Included in this is the need to gather information on factors that will influence the ability to achieve the means and fundamental objectives and to score the likelihood and magnitude of their influence. Here, efforts to estimate the probability of the means objective achieving the fundamental objective for Snuffbox are described, along with potential factors that may influence the ability to achieve the means objective. However, scoring of the influence of individual factors on the ability to achieve the means objective is not performed. The appendix is intended to act as a functional guide for using the decision support framework and would require additional input from species experts for the completion of scoring.

### Probability of the means objective achieving the fundamental objective

Modeling and simulation work to identify how species reintroduction could benefit the survival or recovery of Snuffbox in Canada has yet to be performed. Theory and basic simulations ignoring life-history and species information suggests that the probability of extirpation of a species is intrinsically related to the number of extant populations (i.e., Figure 7), of which, Snuffbox has two. Recovering populations of Snuffbox can therefore have immediate benefits for the species.

Young and Koops (2011) developed a general matrix population model for unionid mussels to assess the sensitivity of population growth to changes in a range of life-history parameters. Their results suggested that the population growth rate of freshwater mussels with relatively low fecundity and late age-of-maturity was most sensitive to adult survival, and somewhat sensitive to juvenile survival. Snuffbox age-of-maturity is generally unknown but estimated at 5-10 years (Dennis 1987; Yeager and Saylor 1995; Fisheries and Oceans Canada 2019), with survival rates and fecundity unknown (Butler 2007). Unlike the models presented in Young and Koops (2011), Snuffbox may demonstrate a male-biased sex ratio in the wild (Butler 2007), which could heighten the risk of local extirpation if removals of source individuals are too large.

### Population considerations

The distribution of Snuffbox has been well-described (Butler 2007; Fisheries and Oceans Canada 2019; Figure A4; Table A5). Fewer than 50 self-sustaining populations of Snuffbox are estimated to be remaining in North America, with only two known populations remaining in Canada (Fisheries and Oceans Canada 2019; Table A5). The two populations of Snuffbox in Ontario are geographically disconnected from each other, with the natural recovery of extirpated sites unlikely given the relatively low dispersal of Logperch (*Percina caprodes*), the obligate host species (McNichols 2007; Schwalb et al. 2011). The genetic structure of Snuffbox populations has been well-described (Zanatta and Murphy 2008; Zanatta and Wilson 2011; Galbraith et al. 2015; Beaver et al. 2019). Across the species range, evidence of population differentiation matches expectations based on post-glacial colonization routes (Beaver et al. 2019) and populations show greater among-population differentiation than within-population (Galbraith et al. 2015).

*Table A5. Relative abundance index, population trajectory, and population status of each Snuffbox population in Ontario. Certainty rankings listed as: 2 = catch per unit effort or standardized sampling, or 3 = expert opinion.*

| Population | Relative abundance index | Population trajectory | Population status |
|---|---|---|---|
| **Lake Huron** | | | |
| *Ausable River* | Low (2) | Stable (2) | Poor |
| **Lake St. Clair** | | | |
| *Thames River* | Extirpated (2) | Not applicable | Extirpated |
| *East Sydenham River* | Medium (2) | Stable (2) | Fair |
| **Lake Erie** | | | |
| *Grand River* | Extirpated (2) | Not applicable | Extirpated |
| *Niagara River* | Extirpated (3) | Not applicable | Extirpated |

Snuffbox tends to occur in extremely low densities within diverse freshwater mussel aggregations. Previous surveys (Baitz et al. 2008) from the Ausable River suggest that Snuffbox is present at a density of approximately 0.09 individuals/$m^2$ (range: 0.01 – 0.25). Given the low suspected abundance, removing individuals from the wild to directly act as a source stock would likely pose heightened risk of source population decline. Snuffbox has been propagated under human care and released for the purposes of conservation in the United States (Barnhart 2002).

## Habitat considerations

Snuffbox occupies riffle-run habitat within small- to medium-sized rivers and streams where substrate sizes range from sand to boulders (Clarke 1981; Buchanan 1990; Metcalfe-Smith et al. 2003, 2007; Baitz et al. 2008; Fisheries and Oceans Canada 2019). In Ontario, the species has been detected in areas of the Ausable and East Sydenham rivers where depths ranged from 12-21 cm with a site velocity of 0.03-0.38 m/s. Most often, the species is buried in the substrate. Logperch, the host species of Snuffbox, can occupy a wide variety of habitats, including lakes and slow- or fast-flowing streams (Scott and Crossman 1973; Robinson and Buchanan 1988; Holm et al. 2009). Logperch prefers rocky substrates but can be found in areas with a variety of substrates and vegetation (Scott and Crossman 1973; Holm et al. 2009).

## Community considerations

SARA-listed mussels tend to occur in diverse mussel and benthic macroinvertebrate communities (Eveleens 2021). The Thames River is composed of a relatively diverse freshwater mussel assemblage, with at least 28 co-occurring species (Table A6).

*Table A6. Freshwater mussel species previously captured in the Thames River (2004-2018).*

| *Genus* | *Species* | *Common Name* |
|---|---|---|
| *Alasmidonta* | *marginata* | Elktoe |
| *Alasmidonta* | *viridis* | Slippershell |
| *Amblema* | *plicata* | Threeridge |
| *Anodontoides* | *ferussacianus* | Cylindrical Papershell |
| *Cambarunio* | *iris* | Rainbow |
| *Cyclonaias* | *pustulosa* | Pimpleback |
| *Cyclonaias* | *tuberculata* | Purple Wartyback |
| *Eurynia* | *dilatata* | Spike |
| *Fusconaia* | *flava* | Wabash Pigtoe |
| *Lampsilis* | *cardium* | Plain Pocketbook |
| *Lampsilis* | *fasciola* | Wavyrayed Lampmussel |
| *Lampsilis* | *siliquoidea* | Fatmucket |
| *Lasmigona* | *complanata* | White Heelsplitter |
| *Lasmigona* | *compressa* | Creek Heelsplitter |
| *Lasmigona* | *costata* | Flutedshell |
| *Ligumia* | *recta* | Black Sandshell |
| *Obliquaria* | *reflexa* | Threehorn Wartyback |
| *Ortmanniana* | *ligamentina* | Mucket |
| *Paetulunio* | *fabalis* | Rayed Bean |
| *Pleurobema* | *sintoxia* | Round Pigtoe |
| *Potamilus* | *alatus* | Pink Heelsplitter |
| *Potamilus* | *fragilis* | Fragile Papershell |
| *Pyganodon* | *grandis* | Giant Floater |
| *Quadrula* | *quadrula* | Mapleleaf |
| *Strophitus* | *undulatus* | Creeper |
| *Truncilla* | *donaciformis* | Fawnsfoot |
| *Truncilla* | *truncata* | Deertoe |
| *Utterbackia* | *imbecillis* | Paper Pondshell |

Logperch, the host species for Snuffbox, is often characterized as a species of relatively low abundance, which may be due to its ability to avoid a seine by occupying waters deeper than 1-m (Scott and Crossman 1973). Nevertheless, Logperch has been sampled from the Thames River in relatively high abundance (Lamothe et al. 2020). There is no evidence of other biotic associations for Logperch besides the parasitic relationship with Snuffbox. Given the size of Logperch, it may be predated on by fishes with sufficient gape size.

## Threat considerations

The Thames River is considered to be moderately impacted by several threats. The likelihood and magnitude of impacts from contaminants and toxic substances, invasive species, and habitat removal and alteration in the Thames River are considered to be relatively high, with impacts from nutrient loading and sediment loading being considered medium (Table A7). More broadly, threats to Snuffbox across the species range have been reviewed (Butler 2007).

*Table A7. Summary of threats to Snuffbox populations in Ontario. Threat Status for all populations resulting from an analysis of both the Threat Likelihood and Threat Impact originally performed for Threehorn Wartyback (Obliquaria reflexa; Sydenham River, Thames River, Grand River; Fisheries and Oceans Canada 2014) and Rainbow (Villosa iris; Ausable River; Fisheries and Oceans Canada 2018), considered suitable surrogates for threat impacts. The number in parentheses refers to the level of certainty assigned to each Threat Status, which reflects the lowest level of certainty associated with either initial parameter (Threat Likelihood or Threat Impact). Certainty has been classified as 2 = correlative studies, or 3 = expert opinion.*

| *Threats* | *Sydenham River* | *Thames River* | *Grand River* | *Ausable River* |
|---|---|---|---|---|
| **Contaminants and toxic substances** | High (3) | High (3) | High (3) | High (3) |
| **Nutrient loading** | Medium (3) | Medium (3) | High (3) | High (3) |
| **Turbidity** | Medium (3) | Unknown (3) | Unknown (3) | High (3) |
| **Sediment Loading** | Medium (3) | Medium (3) | Medium (3) | High (3) |
| **Invasive species** | Low (2) | High (2) | High (2) | Medium (2) |
| **Habitat removal and alteration** | High (3) | High (3) | High (3) | Medium (3) |
| **Altered flow regimes** | Low (3) | Low (3) | Medium (3) | Medium (3) |
| **Host fish (Invasive species)** | Unknown (3) | Unknown (3) | Unknown (3) | Medium (3) |

Threats to the host species must also be considered during the conservation translocation of freshwater mussels. Logperch is considered to be a relatively hardy species (Robinson and Buchanan 1988; Argent and Kimmel 2011). Threats to the persistence of Logperch include siltation and turbidity, and invasive species (Robinson and Buchanan 1988; Holm et al. 2009). Round Goby occupies a similar ecological niche to Logperch, with a common habitat preference (Raab et al. 2018) and similar diet across life-stages (Burkett and Jude 2015). Experimental trials have demonstrated the superior competitive ability of Round Goby to Logperch (Balshine et al. 2005; Leino and Mensinger 2017); however, a significant reduction in Logperch populations has not been observed in areas invaded by Round Goby, which may relate to the better ability of Logperch to occupy deeper, soft-bottomed habitats (Burkett and Jude 2015; Leino and Mensinger 2017), the reproductive strategy of burying eggs into the substrate enabling continued recruitment (Page 1983), or a lack of sampling effort and research.

## Estimate the expected benefits of performing conservation translocation

Based on the factors described above, the expected influence of factors on the ability to achieve a successful reintroduction must be scored. Table A8 has not been filled given the need for additional input from species experts.

*Table A8. Factors that may influence the ability to re-establish an extirpated population of Snuffbox.*

***Focal species**: Snuffbox (Epioblasma triquetra) - ENDANGERED*
***Problem statement**: Snuffbox is listed as Endangered under SARA. Reintroduction has been identified as a potential recovery strategy for the species in the Thames River, Ontario, but efforts have yet to occur. A source population for performing conservation translocations has not been determined.*
***Fundamental objective**: Improve the survival or recovery of Snuffbox in Canada.*
***Means objective**: Re-establish a population of Snuffbox in the Thames River, Ontario, between Chatham and Moraviantown.*

| *Category* | *Factors* | *Likelihood* | *Evidence strength* | *Agreement* | *Confidence* | *References* | *Additional considerations* |
|---|---|---|---|---|---|---|---|
| Population considerations | Abundance of the source population(s) is suitable to achieve the means objective | - | - | - | - | - | - |
| | Age-structure of the source population is suitable to achieve the means objective | - | - | - | - | - | - |
| | Genetic diversity and variation of the source population is suitable to achieve the means objective | - | - | - | - | - | - |
| | Genetic diversity and variation of the recipient population is suitable to achieve the means objective | - | - | - | - | - | - |
| | Life-history strategy of the source population is suitable to achieve the means objective | - | - | - | - | - | - |
| | Captive breeding or captive rearing techniques are available to achieve the means objective | - | - | - | - | - | - |
| Habitat | Habitat in the recipient site(s) reflect species requirements (e.g., water clarity, water velocity, depth, vegetation, substrate) | - | - | - | - | - | - |
| | A sufficient quantity of habitat exists in the recipient location to support all life-stages | - | - | - | - | - | - |
| | Sufficient connectivity exists in the recipient habitat to support all life-stages | - | - | - | - | - | - |
| Community considerations | Obligate, facultative, or parasitic species dependencies limit the ability to achieve the means objective | - | - | - | - | - | - |
| Threats | Pertinent threats limit the ability to achieve the means objective, including: | - | - | - | - | - | - |
| | Invasive species | - | - | - | - | - | - |
| | Residential and commercial development | - | - | - | - | - | - |
| | Agriculture and aquaculture | - | - | - | - | - | - |
| | Energy production and mining | - | - | - | - | - | - |
| | Biological resource use | - | - | - | - | - | - |
| | Transportation and service corridors | - | - | - | - | - | - |
| | Human intrusions and disturbance | - | - | - | - | - | - |
| | Natural systems modification | - | - | - | - | - | - |
| | Pollution | - | - | - | - | - | - |
| | Geological events | - | - | - | - | - | - |

| *Category* | *Factors* | *Likelihood* | *Evidence strength* | *Agreement* | *Confidence* | *References* | *Additional considerations* |
|---|---|---|---|---|---|---|---|
| Threats (cont.) | Parasites or disease transfer | - | - | - | - | - | - |
| | Future habitat modification or degradation | - | - | - | - | - | - |
| | Climate change | - | - | - | - | - | - |
| | Other factors | - | - | - | - | - | - |

## STEP 3: IDENTIFY AND ASSESS THE LIKELIHOOD AND MAGNITUDE OF UNINTENDED CONSEQUENCES

The next step in the conservation translocation decision support framework is to identify and assess the likelihood and magnitude of unintended consequences of reintroduction on the source population and on both the source and recipient ecosystems. For the context of Snuffbox reintroduction, this includes changes in source population persistence and genetic variation, change in short- and long-term community and ecosystem dynamics in the source and recipient ecosystems, and the transfer of disease to the recipient ecosystem.

The removal of individuals from a population to source a translocation can have immediate negative impacts, particularly when populations are small or when there are few breeding individuals. Based on the best available knowledge, the abundance of the two remaining populations in Canada may limit the number of available individuals for translocations. Captive rearing may be an option as Snuffbox has been experimentally reared in Canada (Wilson et al. 2021) and successfully propagated and released in the United States (Barnhart 2002). If captive rearing is considered as an approach, research on other SARA-listed unionids suggests less than 15 gravid females are required to retain natural levels of genetic variation (VanTassel et al. 2021). The reintroduction of Snuffbox is unlikely to have measurable negative effects on co-occurring unionids in the recipient habitat. A risk assessment for the potential transfer of disease has yet to be performed for Snuffbox translocations.

### Estimate the risk of performing conservation translocations

The next step is to score the likelihood and magnitude of potential risks on the ability to achieve the means and fundamental objectives. Similar to Table A8, Table A9 has not been filled as additional input from species experts is required.

*Table A9. Risk considerations for Snuffbox populations and other ecosystem components in source and recipient habitats of proposed conservation translocations.*

| *Subject* | *Location* | *Risk category* | *Risk outcome* | *Risk likelihood* | *Risk magnitude* | *Evidence strength* | *Agreement* | *Confidence* | *References* | *Additional considerations* |
|---|---|---|---|---|---|---|---|---|---|---|
| Focal | Source | Population persistence | Reduced or altered population abundance | - | - | - | - | - | - | - |
| Focal | Source | Genetic variation | Altered genetic variation | - | - | - | - | - | - | - |
| Focal | Source | Genetic variation | Inbreeding depression | - | - | - | - | - | - | - |
| Focal | Recipient | Population persistence | Individual mortality | - | - | - | - | - | - | - |
| Focal | Recipient | Genetic variation | Founder effect | - | - | - | - | - | - | - |
| Focal | Recipient | Genetic variation | Outbreeding depression | - | - | - | - | - | - | - |
| Focal | Recipient | Genetic variation | Hybridization | - | - | - | - | - | - | - |
| Other | Source | Community and ecosystem dynamics | Increased negative interactions | - | - | - | - | - | - | - |
| Other | Source | Community and ecosystem dynamics | Reduced positive interactions | - | - | - | - | - | - | - |
| Other | Source | Community and ecosystem dynamics | Reduced habitat availability | - | - | - | - | - | - | - |
| Other | Source | Community and ecosystem dynamics | Altered ecosystem processes | - | - | - | - | - | - | - |
| Other | Recipient | Community and ecosystem dynamics | Enhanced negative interactions | - | - | - | - | - | - | - |
| Other | Recipient | Community and ecosystem dynamics | Reduced positive interactions | - | - | - | - | - | - | - |
| Other | Recipient | Community and ecosystem dynamics | Transformative changes within site of introduction | - | - | - | - | - | - | - |
| Other | Recipient | Community and ecosystem dynamics | Transformative changes beyond | - | - | - | - | - | - | - |

| *Subject* | *Location* | *Risk category* | *Risk outcome* | *Risk likelihood* | *Risk magnitude* | *Evidence strength* | *Agreement* | *Confidence* | *References* | *Additional considerations* |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | site of introduction | | | | | | | |
| Other | Recipient | Community and ecosystem dynamics | Reduced habitat availability | - | - | - | - | - | - | - |
| All | Both | Disease transfer | Individual mortality / reductions in fitness | - | - | - | - | - | - | - |

## REFERENCES


Argent, D.G., and Kimmel, W.G. 2011. Influence of navigational lock and dam structures on adjacent fish communities in a major river system. River Research and Applications 27: 1325-1333.

Baitz, A., M. Veliz, H. Brock, and Staton, S. 2008. Monitoring program to track the recovery of endangered freshwater mussels in the Ausable River, Ontario [DRAFT]. Prepared for the Ausable River Recovery Team, the Interdepartmental Recovery Fund and Fisheries and Oceans Canada. 26 p.

Balshine, S., Verma, A., Chant, V., and Theysmeyer, T. 2005. Competitive interactions between Round Gobies and Logperch. Journal of Great Lakes Research 31: 68-77.

Barnhart, M.C. 2002. Propagation and culture of mussel species of concern annual report for 2002. Springfield: Southwest Missouri State University. Prepared for the Missouri Department of Conservation and U.S. Fish and Wildlife Service. iii + 37 pp.

Beaver, C.E., Woolnough, D.A., and Zanatta, D.T. 2019. Assessment of genetic diversity among populations of *Epioblasma triquetra* in the Laurentian Great Lakes drainage. Freshwater Science 38(3): 527-542.

Buchanan, A.C. 1980. Mussels (naiades) of the Meramec River basin, Missouri. Aquatic Series No. 17, Missouri Department of Conservation, Jefferson City, MO: 68 p.

Burkett, E.M., and Jude, D.J. 2015. Long-term impacts of invasive round goby *Neogobius malanostomus* on fish community diversity and diets in the St. Clair River, Michigan. Journal of Great Lakes Research 41(3): 862-872.

Butler, R.S. 2007. Status assessment report for the snuffbox, *Epioblasma triquetra*, a freshwater mussel occurring in the Mississippi River and Great Lakes Basins. Report for the Ohio River Valley Ecosystem Team Mollusk Subgroup. Asheville, North Carolina, USA. 211 pp.

Clarke, A.H. 1981. The Freshwater Molluscs of Canada. National Museums of Canada, Ottawa. 446 p.

COSEWIC. 2011. COSEWIC assessment and status report on the Snuffbox *Epioblasma triquetra* in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa. xi + 50 pp.

Dennis, S.D. 1987. An unexpected decline in populations of the freshwater mussel, *Dysnomia* (= *Epioblasma*) *capsaeformis*, in the Clinch River of Virginia and Tennessee. Virginia Journal of Science 38: 281-288.

DFO. 2014. Recovery potential assessment of Threehorn Wartyback (*Obliquaria reflexa*) in Canada. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2014/014.

Eveleens, R.A. 2021. Does it take a community to save a species? Examining the use of community interactions to restore unionid species at risk. M.Sc. Thesis. University of Windsor. Windsor, Ontario, Canada. 173 pp.

Fisheries and Oceans Canada. 2018. Recovery strategy and action plan for the Rainbow (*Villosa iris*) in Canada [Proposed]. In *Species at Risk Act* Recovery Strategy Series. Fisheries and Oceans Canada, Ottawa. v + 63 pp.

Fisheries and Oceans Canada. 2019. Recovery Strategy for Northern Riffleshell, Snuffbox, Round Pigtoe, Salamander Mussel, and Rayed Bean in Canada. In *Species at Risk Act* Recovery Strategy Series. Ottawa: Fisheries and Oceans Canada. ix + 96 p.

Galbraith, H.S., Zanatta, D.T., and Wilson, C.C. 2015. Comparative analysis of riverscape genetic structure in rare, threatened and common freshwater mussels. Conservation Genetics 16: 845-857.

Holm, E., Mandrak, N.E., and Burridge, M.E. 2009. The ROM Field Guide to Freshwater Fishes of Ontario. Friesens Printers, Altona, Manitoba, Canada.

Lamothe, K.A., Ziegler, J.P., Gáspárdy, R., Barnucz, J., and Drake, D.A.R. 2020. Abiotic and biotic associations between the round goby *Neogobius melanostomus* and tubenose goby *Proterorhinus marmoratus* with the endangered northern madtom *Noturus stigmosus* in Canada. Aquatic Conservation: Marine and Freshwater Ecosystems 30: 691-700.

Leino, J.R., and Mensinger, A.F. 2017. Interspecific competition between the round goby, *Neogobius melanostomus*, and the logperch, *Percina caprodes*, in the Duluth-Superior Harbour. Ecology of Freshwater Fish 26: 34-41.

McNichols, K.A. 2007. Host fish and population dynamics of Species at Risk freshwater mussels in Ontario. M.Sc. Thesis. The University of Guelph, Guelph, Ontario, Canada. 167 pp.

Metcalfe-Smith, J.L., Di Maio, J., Staton, S.K., and de Solla, S.R. 2003. Status of the freshwater mussel communities of the Sydenham River, Ontario, Canada. The American Midland Naturalist 150(1): 37-50.

Metcalfe-Smith, J.L, McGoldrick, D.J., Zanatta, D.T., and Grapentine, L.C. 2007. Development of a monitoring program for tracking the recovery of endangered freshwater mussels in the Sydenham River, Ontario. Prepared for the Sydenham River Recovery Team, the Interdepartmental Recovery Fund and Fisheries and Oceans Canada. 61 p.

Page, L. 1983. Handbook of Darters. T.F.H. Publications, Inc. United States. 271 p.

Raab, D., Mandrak, N.E., and Ricciardi, A. 2018. Low-head dams facilitate Round Goby *Neogobius melanostomus* invasion. Biological Invasions 20: 757-776.

Robinson, H.W., and Buchanan, T.M. 1988. Fishes of Arkansas. The University of Arkansas Press. Fayetteville, Arkansas, United States

Schwalb, A.N., Poos, M.S., and Ackerman, J.D. 2011. Movement of logperch—the obligate host fish for endangered snuffbox mussels: implications for mussel dispersal. Aquatic Sciences 73: 223-231.

Scott, W.B., and Crossman, E.J. 1973. Freshwater Fishes of Canada. Fisheries Research Board of Canada. Bulletin 184. Ottawa, Canada.

VanTassel, N.M., Morris, T.J., Wilson, C.G., and Zanatta, D.T. 2021. Genetic diversity maintained in comparison of captive-propagated and wild populations of *Lampsilis fasciola* and *Ptychobranchus fasciolaris* (Bivalvia: Unionidae). Canadian Journal of Fisheries and Aquatic Sciences 78(9).

Wilson, C., McNichols-O'Rourke, K., Pierman, J., and Johnson, P. 2021. Pond rearing of *Lampsilis fasciola*, *Ptychobranchus fasciolaris* and *Epioblasma triquetra* to investigate feasibility of broodstock development in Morris, T.J., McNichols-O'Rourke, K.A., Goguen, M.N., and Reid, S.M. (Editors). In review. Proceedings of the 2021 Canadian Freshwater Mollusc Research Meeting: December 7-8, 2021, Burlington, Ontario. Can. Tech. Rep. Fish. Aquat. Sci. XXXX: vi + 37 p.

Yeager, B.L., and Saylor, C.F. 1995. Fish hosts for four species of freshwater mussels (Pelecypoda: Unionidae) in the Upper Tennessee River Drainage. American Midland Naturalist 133: 1-6.

Young, J.A.M., and Koops, M.A. 2011. Recovery potential modelling of Eastern Pondmussel (*Ligumia nasuta*), Fawnsfoot (*Truncilla donaciformis*), Mapleleaf (*Quadrula quadrula*), and Rainbow (*Villosa iris*) in Canada. DFO Can. Sci. Advis. Sec. Res. Doc. 2010/119. iv + 10 p.

Zanatta, D.T., and Murphy, R.W. 2008. The phylogeographic and management implications of genetic population structure in the imperiled snuffbox mussel, *Epioblasma triquetra* (Bivalvia: Unionidae). Biological Journal of the Linnean Society 93: 371-384.

Zanatta, D.T., and Wilson, C.C. 2011. Testing congruency of geographic and genetic population structure for a freshwater mussel (Bivalvia: Unionoida) and its host fish. Biological Journal of the Linnean Society 102: 669-685.