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Considerations around effort and power for an occupancy-based Redside Dace (*Clinostomus elongatus*) monitoring program across spatial and temporal scales

Karl A. Lamothe¹, Scott M. Reid², and D. Andrew R. Drake¹

¹Great Lakes Laboratory for Fisheries and Aquatic Sciences
Fisheries and Oceans Canada
867 Lakeshore Rd.
Burlington, ON L7S 1A1

²Aquatic Research and Monitoring Section
Ontario Ministry of Natural Resources and Forestry
2140 East Bank Dr.
Peterborough, ON K9J 7B8

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

Redside Dace (*Clinostomus elongatus*) is a small minnow with a limited distribution in southern Ontario and is listed as Endangered under the *Species at Risk Act*. The draft federal recovery strategy for Redside Dace outlines the need to develop a monitoring program to quantify the abundance and distribution of the species in Canada. A repeat-survey design, whereby surveys are replicated spatially and temporally, could be used to inform models for assessing the abundance and distribution of Redside Dace and subsequent management decisions that may impact the species. An introduction to Redside Dace monitoring is presented, including a summary of previous and current monitoring efforts. Potential objectives of a Redside Dace monitoring program are described as well as an introduction to the use of a repeat-survey design for estimating species detection and occupancy. Considerations around the allocation of effort for a Redside Dace monitoring program are presented, including: (i) the effect of scale on site definition and subsequent implications for total monitoring effort and statistical power; (ii) targeted versus random sampling; and, (iii) how stratified random sampling across different gradients (e.g., space, time, threats) can inform the conservation status of the species. Occupancy is emphasized over an abundance-based approach as it provides relatively greater power per sampling effort for generating conclusions. Nevertheless, sampling Redside Dace using repeated surveys within and between sampling seasons can be used to inform abundance estimates. Overall, stratification of sampling effort for Redside Dace spatially (i.e., across populations) and over time will allow statistically powerful conclusions about changes in occupancy and (or) abundance at the local, population, and Canadian scale. Implementation of such a monitoring approach would inform management activities such as the review of development and instream work proposals, population status assessments, and future restoration projects.

INTRODUCTION

Redside Dace (*Clinostomus elongatus*) is a colourful minnow found in slow-flowing riffle-pool¹ sections of small creeks and streams, often near overhanging riparian vegetation. The species shows a preference for middle water-column positions in the deepest parts of pools (McKee and Parker 1982, Novinger and Coon 2000) and relies on nests of Creek Chub (*Semotilus atromaculatus*) and (or) Common Shiner (*Luxilus cornutus*) for reproduction (Koster 1939, Pitcher et al. 2009). The **distribution** of Redside Dace is intermittent across North America, including throughout the upper Mississippi River drainage, Great Lakes basin, Ohio River, and upper Susquehanna River (Page and Burr 1991). There are 17 locations where Redside Dace is extant in Canada and nine locations where Redside Dace is considered extirpated (Table 1, Figure 1; COSEWIC 2017). Extant Redside Dace **populations** are primarily located in tributaries of Lake Ontario in the **Greater Toronto and Hamilton Area (GTA)**, but are also found on St. Joseph's Island (Lake Huron) and in tributaries of lakes Erie and Huron (Figure 1).

Table 1. Population Status for Redside Dace in Ontario resulting from an analysis of relative abundance and population trajectories where certainty was assigned and reflective of the lowest level of certainty associated with either initial parameter; 1 = quantitative analysis; 2 = CPUE or standardized sampling; 3 = expert opinion. From Lebrun et al. 2019. Dominant threat has been identified based on expert opinion of the authors.

<i>Population</i>	<i>Status</i>	<i>Certainty</i>	<i>Dominant threat</i>	<i>Abundance estimate</i>
Two Tree River	Fair	2	Agriculture	Unknown
Saugeen River	Poor	2	Agriculture	Unknown
Gully Creek	Poor	2	Agriculture	741
South Gully Creek	Unknown	3	Agriculture	Unknown
Irvine Creek	Poor	3	Agriculture	Unknown
Spencer Creek	Poor	2	Urbanization	Unknown
Bronte Creek	Poor	3	Urbanization	Unknown
Fourteen Mile Creek	Fair	2	Urbanization	Unknown
Sixteen Mile Creek	Poor	2	Urbanization	Unknown
Credit River	Poor	2	Urbanization	Unknown
Humber River	Fair	2	Urbanization	38,582
Don River	Poor	2	Urbanization	1,607
Rouge River	Poor	2	Urbanization	9,180
Duffins Creek	Poor	2	Urbanization	2,398
Carruther's Creek	Fair	2	Urbanization	Unknown
Lynde Creek	Poor	2	Urbanization	Unknown
Holland River	Poor	2	Urbanization	Unknown
Niagara area stream	Extirpated	3	Unknown	0
Wedgewood Creek	Extirpated	3	Urbanization	0
Morrison Creek	Extirpated	2	Urbanization	0
Clarkson Creek	Extirpated	3	Urbanization	0
Etobicoke Creek	Extirpated	3	Urbanization	0
Mimico Creek	Extirpated	3	Urbanization	0
Highland Creek	Extirpated	3	Urbanization	0
Petticoat Creek	Extirpated	3	Urbanization	0
Pringle Creek	Extirpated	3	Urbanization	0

¹Terms defined in the glossary are shown in bold face the first time they appear in the body of the text.

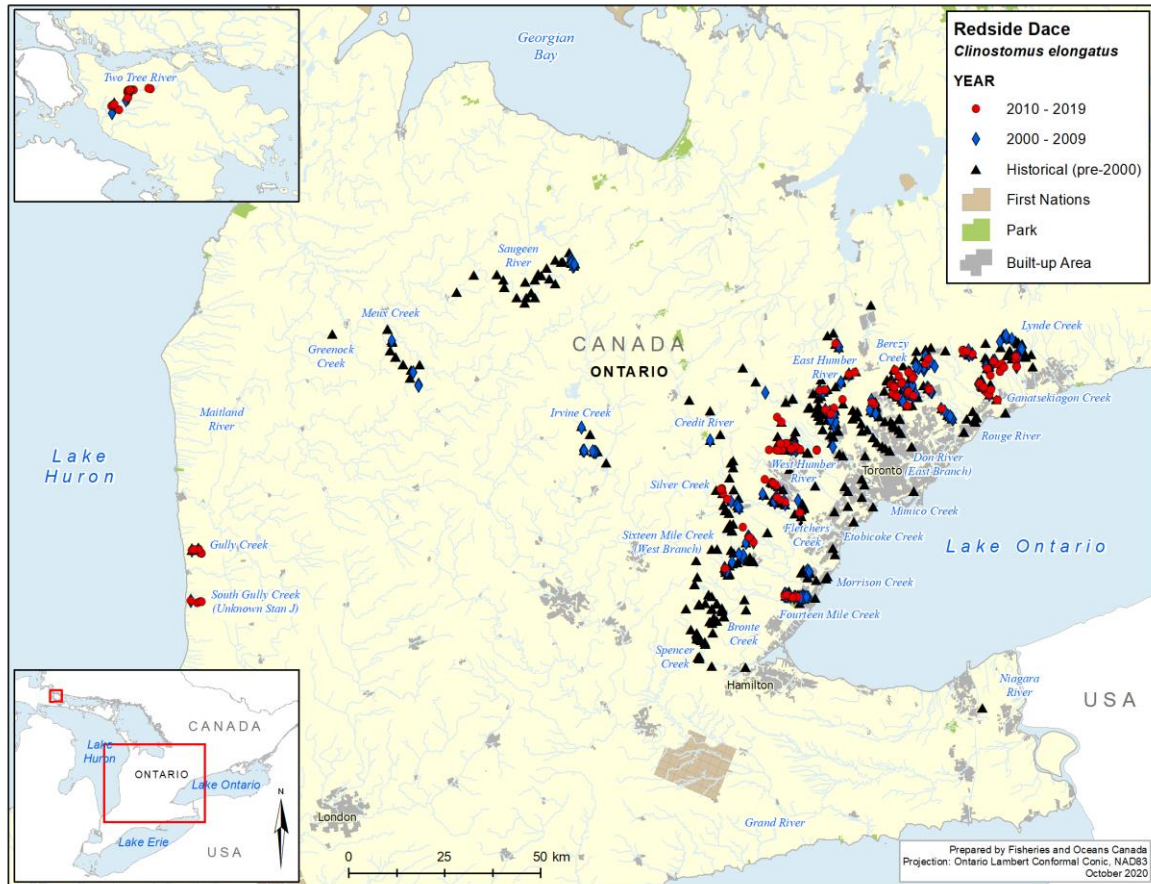


Figure 1. Distribution of Redside Dace in Canada based on samples from pre-2000 (black triangles), between 2000–2009 (blue diamonds), and between 2010–2019 (red circles).

Due to recent declines in the **abundance** and distribution of Redside Dace and ongoing threats to species persistence, Redside Dace has been assessed as Endangered in Ontario and is listed as Endangered under Schedule 1 of the *Species at Risk Act* (RDRT 2010, COSEWIC 2017, DFO 2019). The status of each extant population has been assessed based on relative abundance and population trajectory when data were available; four populations were considered to be in fair condition, whereas 12 populations were considered to be in poor condition, with one population of unknown status (South Gully Creek, Table 1; DFO 2019). Declines in the abundance and distribution of Redside Dace have been attributed to agricultural practices and urban development; the removal of riparian vegetation, channelization, pollution, siltation, and altered stream hydrology are considered detrimental to the persistence of Redside Dace (McKee and Parker 1982, Reid and Parna 2017).

A key action to support the recovery of Redside Dace is the implementation of a monitoring program to characterize spatial and temporal changes in the abundance and distribution of the species and its habitat (e.g., Portt et al. 2008, Wilson and Dextrase 2008, RDRT 2010, Mandrak and Bouvier 2014). To date, monitoring of Redside Dace has been completed through a variety of targeted sampling efforts and watershed-level fish community surveys using different gears (e.g., eDNA, electrofishers, seines, underwater cameras). Documentation of surveys directed specifically toward observing and capturing Redside Dace in Canada do not exist prior to 1979. Most of the early Redside Dace records were the result of general fish community survey work (COSEWIC 2007). Conversely, targeted surveys for Redside Dace have been conducted over

the last 20 years by Fisheries and Oceans Canada (DFO), Ontario Ministry of Natural Resources and Forestry (OMNRF), Ontario Conservation Authorities, Ontario Streams, and the Royal Ontario Museum to confirm the distribution of Redside Dace and investigate abundance at some sites. As well, watershed-level surveys have been performed (Andersen 2002, Ontario Streams 2005, Mandrak et al. 2010, Reid et al. 2017) and Redside Dace observations have been contributed by consultants and university researchers; for example, Poos et al. (2012) estimated the abundance of several Redside Dace populations using removal sampling methods (Table 1).

Scientific guidance has been provided about monitoring Redside Dace and other species at risk in Ontario (Portt et al. 2008, Wilson and Dextrase 2008, Mandrak and Bouvier 2014), focusing on appropriate sampling gear, site boundary definitions, sampling intensity, number of sites to sample, how often and when sampling should occur (i.e., timing window to avoid spawning period), and fish handling procedures. Guidance was largely developed from the Ontario Stream Assessment Protocol (OSAP; Stanfield 2017), past studies undertaken in the Laurentian Great Lakes basin to inform general fish community surveys, and expert opinion related to sampling Redside Dace and Ontario stream fishes. In particular, the *draft Sampling protocols for Redside Dace* (Wilson and Dextrase 2008) presented standardized approaches to determine the occurrence of Redside Dace at a site (referred to as extensive sampling) and monitor population trends through time at specific index sites (intensive sampling).

The provincial (RDRT 2010) and draft federal (Amy Boyko, DFO Species at Risk Program, pers. comm.) recovery strategies for Redside Dace stipulate the development of a two-tiered monitoring program based on guidance from Wilson and Dextrase (2008) that includes both intensive and extensive sampling. Intensive sampling is intended to assess the abundance or density of Redside Dace at pre-determined index sites, with the frequency of sampling dependent on current population status. Information from intensive sampling would be used to understand the abundance or trajectory of Redside Dace at a subset of sites, as well as detailed species-habitat associations. Extensive sampling would involve the collection of occurrence data over a much larger spatial scale, which would allow changes in the distribution of Redside Dace to be evaluated within and among watersheds. Together, the implementation of a two-tiered monitoring program could provide quantitative information about the conservation status of each population, while also informing the planning of restoration projects and decisions around permitting for development and instream work proposals.

However, the following observations drawn from conversations with Conservation Authority biologists characterize how the draft protocol has been applied over the past decade:

1. population monitoring at index sites using removal-sampling methods has rarely been done;
2. the number of sites regularly sampled within individual watersheds is variable across the range of Redside Dace;
3. regular Redside Dace monitoring is not the focus of most stream sampling. Monitoring for Redside Dace is indirectly accomplished through other efforts to monitor watershed health and inform land-use planning; and,
4. gear permitted for sampling Redside Dace has changed over the years. Concerns were expressed over how to interpret trends in distribution and population status over time using differing gears.

Therefore, the intent of the *draft Sampling protocols for Redside Dace* to move forward from *ad hoc*, incidental, and non-standardized sampling does not appear to be fully achieved. Moreover, a statistical evaluation of the proposed approaches to monitor the abundance and distribution of

Redside Dace in Canada has not occurred, which needs to be addressed prior to the widespread implementation of the proposed two-tiered framework.

This research document supports the development and implementation of a two-tiered monitoring program for Redside Dace by: 1) providing advice related to the sampling effort needed to evaluate changes in the occupancy of Redside Dace with sufficient statistical power, thereby informing aspects of extensive sampling design; and, 2) describing the application of a modelling technique to estimate abundance at selected sites, thereby informing intensive sampling. Design advice was developed by considering the influence of **imperfect detection** on Redside Dace monitoring efforts, sample unit boundaries, the role of habitat and threat data collection, and the utility of stratifying effort in space and time. The effectiveness of a repeat-survey design relative to other monitoring designs (e.g., single-pass approaches) is not addressed. Rather, considerations around the allocation of effort for a Redside Dace monitoring program following a repeat-survey design are focused on:

1. the effect of scale on site definition and subsequent implications for total monitoring effort and statistical power;
2. targeted versus random sampling; and,
3. how stratified random sampling across different gradients (e.g., space, time, threats) can inform the conservation status of the species.

The examples presented were developed using occupancy and *N*-mixture models to demonstrate the utility of initiating a repeat-survey protocol for short- and (or) long-term monitoring of Redside Dace in Canada. Potential objectives and assessment variables for a Redside Dace monitoring program designed to inform species conservation are discussed in the following section.

POTENTIAL OBJECTIVES AND ASSESSMENT VARIABLES FOR A REDSIDE DACE MONITORING PROGRAM IN CANADA

The first step for developing a species monitoring program is to identify the objectives and assessment variables used to inform the objectives (Figure 2). For example, one of the objectives of OSAP is to collect data on the entire species assemblage in a way that allows for meaningful comparisons across wadeable streams (Stanfield 2017). This objective informs the definition of the sample unit and the data collection approach; specifically, fishes are collected at stream segments of at least 40 m that include at least one riffle-pool sequence, beginning and ending at **crossover points** (Stanfield 2017). This precise description of the sampling unit ensures that riffle, run, and pool-dwelling species have the opportunity to be detected, and therefore, aims to characterize the composition of the entire stream fish assemblage at that site. Failing to identify prescriptive questions and (or) objectives during the development phase of a monitoring program often leads to flawed experimental designs and poor ability to make meaningful conclusions about the question of interest (Yoccoz et al. 2001, Nichols and Williams 2006, Bailey et al. 2007, Sauer and Knutson 2008).

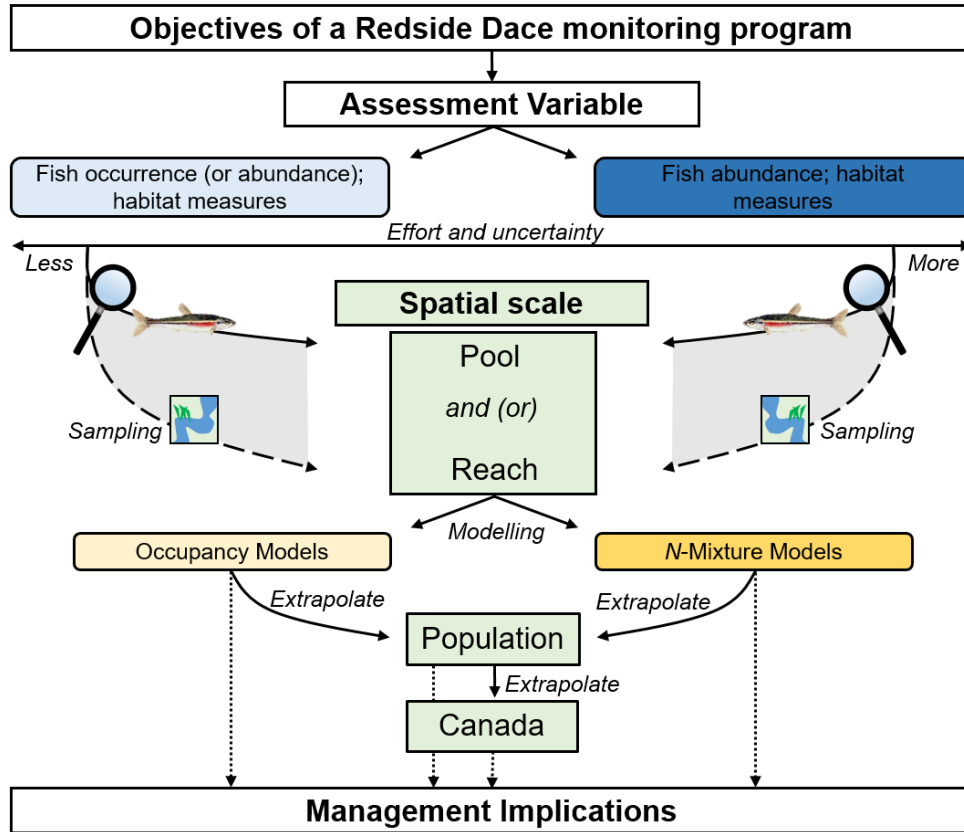


Figure 2. Flow diagram outlining the steps in developing a Redside Dace monitoring program. Planning begins with defining the specific objectives of a monitoring program, identifying assessment variables based on the stated objectives and an understanding of the relative effort and uncertainty for measurement in the field, followed by the spatial scale of sampling, approach for modelling, need for extrapolation, and ending with management implications. Modeling approaches (occupancy and N-mixture models) can inform management objectives leading to implications at multiple spatial scales.

The intent of this document is not to define a single objective for a Redside Dace monitoring program, but rather, to identify potential objectives related to extensive (i.e., distribution-based) and intensive (i.e., abundance-based) sampling as outlined in past provincial guidance (Wilson and Dextrase 2008) and the provincial (RDRT 2010) and draft federal recovery strategies (Amy Boyko, DFO Species at Risk Program, pers. comm.). Therefore, the potential objectives of a Redside Dace monitoring program addressed in this document include quantifying:

1. changes in species occupancy, thereby identifying expansions or contractions of the distribution of Redside Dace through time; and (or),
2. changes in population abundance of Redside Dace through time.

These objectives can be informed at three or more spatial scales, including the local (i.e., site), population (i.e., river or watershed), and (or) national scale (i.e., Ontario), with inference about the assessment variable (e.g., occupancy, abundance) contingent on the chosen scale (Figure 3). For example, sampling to inform the pattern of occupied sites for a single population would provide an index of the distribution of Redside Dace in that watershed; changes in occupied sites through time would indicate an underlying change in species distribution, whether due to threats, limiting factors, recovery actions, and (or) natural environmental variability. Extending sampling to multiple populations would allow similar inference within and among the populations

selected. The manner in which the **sampling frame** is identified and sites selected has implications for the interpretation of the assessment variable (e.g., changes in occupancy). For example, identifying a sampling frame that includes sites near and beyond the expected range boundary could be combined with an occupancy approach to inform changes in the range boundary through time. Alternatively, a sampling frame concentrated within the range, whereby range edge and core sites are included, would provide information about changes in the pattern of occupied sites within the range. This distinction is similar to the use of the **extent of occurrence (EO)** and the **area of occupancy (AO)** by COSEWIC (2017) as indicators of distribution; the general intent of EO is to evaluate the range boundary, whereas the intent of AO is to evaluate the amount of habitat occupied by a species. Ideally, a monitoring program that uses occupancy as an assessment variable would allow inference of EO and AO.

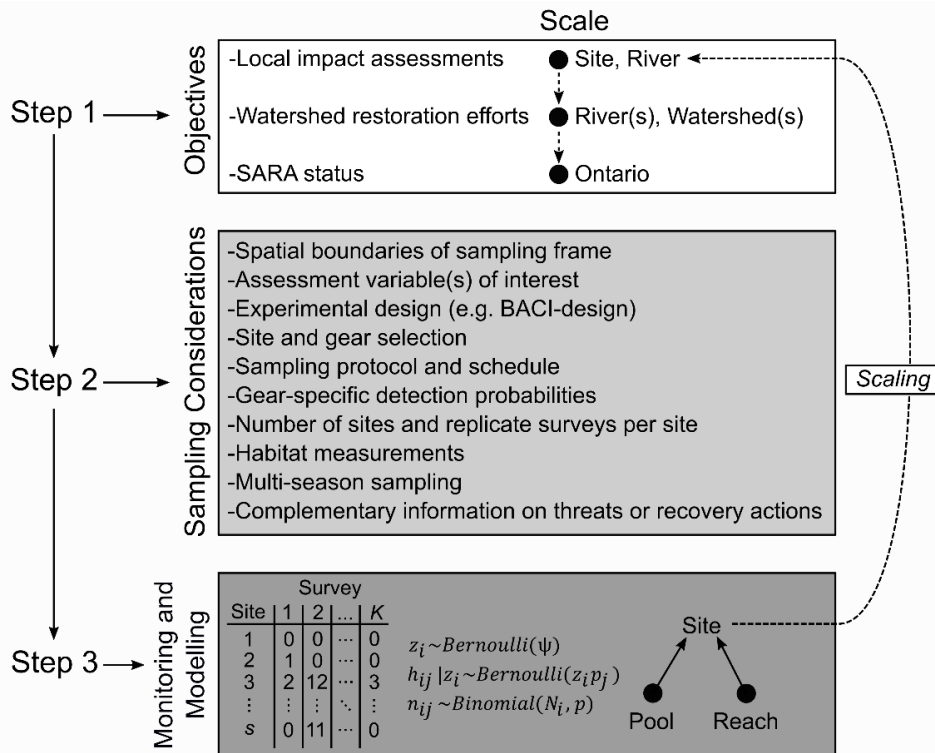


Figure 3. Initial considerations when developing a species monitoring program. Step 1 includes identifying the particular objective, including the scale at which the objective is to be addressed. Considerations for sampling are identified in Step 2 after the initial objective and scale are described. Step 3 includes modelling of collected data and scaling of the results to address the objectives identified in Step 1.

The suite of potential monitoring endpoints for animal populations has previously been described (MacKenzie et al. 2018). Although it would be ideal to directly measure changes in the component processes (i.e., vital rates) that influence species abundance or meta-population dynamics, the sampling effort to evaluate these endpoints with sufficient statistical power is large (Gryska et al. 1997, Quist et al. 2006) and usually beyond the effort capabilities of even the most well-funded programs (MacKenzie et al. 2018). This limitation has spurred the development and use of occupancy modelling approaches and inference, with justification that changes in the pattern of occupied sites reflects underlying changes in population abundance or component processes. The benefits of using an occupancy-based approach have been demonstrated across taxa (e.g., Bayne et al. 2008, Chen et al. 2013, Miller and Grant 2015), including imperilled freshwater fishes experiencing range reductions (Dextrase et al. 2014a, Lamothe et al. 2019a,b, 2020), and aligns well with the need to assess changes in species

distribution. In this document, occupancy methods and extensions for abundance-based estimation are proposed to ensure that monitoring changes in the abundance and distribution of Redside Dace is not biased by imperfect detection, especially given the potential for imperfect detection to change through time as a function of declining density or other factors.

Changes in the pattern of occupied sites across the sampled landscape can indicate changes in local- or broad-scale habitat quality, or other changes in vital rates resulting from threats (e.g., predation, reproduction failure due to habitat disturbance; MacKenzie et al. 2018). Site selection is critical when inferring the causes of change in species occupancy, and without careful consideration to the objective and scale of the sampling effort (Figures 2, 3), can result in a poor ability to generalize across the landscape. For example, intensive localized sampling among several pools may be useful when trying to assess the effects of disturbance or in-stream restoration projects, but would not be ideally suited for estimating the effect of multiple stressors across Redside Dace populations. Instead, estimating occupancy at the population scale would need to incorporate a larger sampling frame and a greater number of measurements of potential threats and (or) restoration measures alongside fish detections. Finally, occupancy probability at the national scale could be used to estimate ultimate gains or losses in Redside Dace distribution in Canada over time.

An occupancy-based approach for monitoring Redside Dace:

1. directly supports the draft federal recovery strategy to monitor presence and absence at large spatial scales;
2. would require less sampling effort relative to abundance-based monitoring;
3. can be used to inform trends in Redside Dace distribution over time by directly linking to AO or EO; and,
4. pending suitable site selection, can be used to estimate the total area occupied by the species (whether single or multiple populations).

Total area occupied can be compared to management goals or thresholds, such as the **minimum area for population viability (MAPV)** (e.g., van der Lee et al. 2020), allowing the probability of one or multiple populations being above or below particular management thresholds to be assessed as an outcome of the program design. Although occupancy models and its extensions are not designed to develop management thresholds (e.g., MAPV), they can be used to inform whether management thresholds have been met.

Many studies have used local occupancy estimates as surrogates for changes in species abundance, because intuitively, species will likely go undetected more frequently when at low abundance compared to when at high abundance. Although the relationship between occupancy and abundance is typically positive (Hartley 1998), it is often nonlinear and differs across spatial scales (Gaston et al. 2000, He and Gaston 2000, Steenweg et al. 2018), with complex, system-specific mechanisms forming these relationships (Borregaard and Rahbek 2010). As a result, assuming a 1:1 relationship between occupancy and abundance can lead to flawed inference about population dynamics and habitat relationships (Tyre et al. 2003), particularly for species that have detection probabilities < 1.0 (i.e., most imperilled species, including Redside Dace; Table 2), with implications for the identification and management of critical habitat. While focusing on occupancy will not directly provide information on changes in abundance or the component processes of abundance (i.e., vital rates) that influence local population dynamics, presence-absence estimates of occupancy remain a promising, and cost-effective (Joseph et al. 2006) approach for evaluating the distribution and conservation status of Redside Dace.

Table 2. Previous estimates of occupancy (ψ) and detection probabilities (p) for Redside Dace in Canada using a variety of gear types, where sampling occurred during differing seasons and at differing locations with differing number of surveys (K). p^* = probability of capture. Min 3 = a minimum of three hauls.

Gear type	Param.	Est.	SE	K	Season	Location
eDNA ¹	ψ	0.55	0.10	3	Spring	Multiple locations
eDNA ¹	ψ	0.59	0.10	4	Spring	Multiple locations
eDNA ¹	ψ	0.61	0.10	5	Spring	Multiple locations
eDNA ¹	ψ	0.47	0.10	3	Fall	Multiple locations
eDNA ¹	ψ	0.47	0.10	4	Fall	Multiple locations
eDNA ¹	ψ	0.52	0.09	5	Fall	Multiple locations
eDNA ¹	p	0.82	0.04	3	Spring	Multiple locations
eDNA ¹	p	0.79	0.04	4	Spring	Multiple locations
eDNA ¹	p	0.73	0.04	5	Spring	Multiple locations
eDNA ¹	p	0.73	0.06	3	Fall	Multiple locations
eDNA ¹	p	0.73	0.05	4	Fall	Multiple locations
eDNA ¹	p	0.64	0.04	5	Fall	Multiple locations
Seine ²	ψ	0.732	0.14	3	Summer	Gully Creek
Seine ²	p	0.606	0.18	3	Summer	Gully Creek
Electrofishing ³	ψ	0.509	0.08	3	Summer	Multiple locations
Electrofishing ³	p	0.738	0.06	3	Summer	Multiple locations
Seine ⁴	p^*	0.584	NA	Min 3	Summer	Gull Creek
Seine ⁴	p^*	0.612	NA	Min 3	Summer	Humber River
Seine ⁴	p^*	0.785	NA	Min 3	Summer	Don River
Seine ⁴	p^*	0.751	NA	Min 3	Summer	Rouge River – Leslie
Seine ⁴	p^*	0.718	NA	Min 3	Summer	Rouge River – Berczy
Seine ⁴	p^*	0.608	NA	Min 3	Summer	Duffins Creek
Electrofishing ³	p^*	0.62	NA	Min 3	Summer	Multiple locations
Seine ⁵	p^*	0.71	NA	Min 3	Summer	Rouge River – Leslie
Seine ⁵	p^*	0.656	NA	Min 3	Summer	Rouge River – Berczy
Electrofishing ⁶	p	0.45	0.02	1	Summer	Multiple locations
Seine ⁶	p	0.68	0.03	1	Summer	Multiple locations
Camera traps ⁶	p	0.74	0.03	4	Summer	Multiple locations

¹ = Serrao et al. 2018; ² = this document DFO data, Gáspárdy and Drake 2021; ³ = Reid et al. 2009; ⁴ = Poesch et al. 2012; ⁵ = Poos and Jackson 2012; ⁶ = Castañeda et al. 2020

The bulk of this research document evaluates the sampling effort required to measure changes in occupancy over time, thereby informing the extensive monitoring endpoint outlined in the provincial (RDRT 2010) and draft federal (Amy Boyko, DFO Species at Risk Program, pers. comm.) recovery strategy. However, given the non-linear, scale-dependent relationship between occupancy and abundance, substantial declines in abundance may be required before a change in occupancy can be detected. As such, a worked example of estimating local species abundance using N -mixture models is also provided to inform the intensive, abundance-based monitoring endpoint (e.g., Wilson and Dextrase 2008, RDRT 2010, COSEWIC 2017). A benefit of using N -mixture models, which require a similar repeat-survey design, is the ability to generate site-specific estimates of abundance and the ability to scale those estimates to the local, population, or national scale, depending on the sampling design (i.e., sample site selection, number of sample sites; Figure 3). Estimates of site-specific abundance generated by N -mixture models can be extrapolated and compared to **minimum viable population (MVP)**

estimates of Redside Dace (van der Lee et al. 2020), allowing the probability of being above or below MVP as an outcome of monitoring program design. However, it should be noted that given the same level of effort, abundance-based measures, including N -mixture models, can perform relatively poorly compared to occupancy models (Ward et al. 2017). As a result, occupancy approaches can maximize statistical power relative to count-based (i.e., abundance) methods for species with low detection rates and (or) low abundance (Pollock 2006).

OCCUPANCY AS AN ASSESSMENT VARIABLE

Occupancy models are increasingly being used to characterize the distribution of imperilled fishes, with examples from north-central Arkansas (Magoulick and Lynch 2015), Virginia (Moore et al. 2017), Arizona (Stewart et al. 2017), Florida (Dorazio et al. 2005), North Carolina (Midway et al. 2014), the Missouri River (Schloesser et al. 2012), and southern Ontario (Dextrase et al. 2014a). Single-species occupancy models have revealed differences in detection probabilities across species (Schloesser et al. 2012, Haynes et al. 2013), differences in gear-specific detection probabilities (Klein et al. 2015, Smith et al. 2015, Moore et al. 2017), and regional differences in the probability of detecting a species (Peoples and Frimpong 2011). Multi-species occupancy models, which are extensions of single-species models, have been used to estimate the probability of detection and occupancy across freshwater communities (Midway et al. 2014, Smith et al. 2015, Gibson-Reinemer et al. 2016, Potoka et al. 2016, Stewart and Long 2016) and to estimate absolute species richness and diversity of sites (Holtrop et al. 2010, Benoit et al. 2018). As occupancy models become more common in the scientific literature, improvements in computation and analyses have been made, improving the utility of these models for species conservation.

The basic single-**season** single-species occupancy model provides an intuitive starting point for describing the approach (Mackenzie et al. 2002). A repeat-survey design is used to account for differences in site-specific detection probability (p_i) when estimating species occupancy (ψ). Repeat surveys refers to sampling site i several times, consecutively, whereby the assumption of **closure** can be met (Mackenzie et al. 2002). To implement this approach, spatial and temporal replication is required, but the amount of effort across sites or time is flexible. The sampling design for performing repeat surveys is diverse (Reich 2020), but for fishes, repeat-surveys are most often completed as:

1. single fish collections from a sample unit done over multiple survey dates in a season;
2. single fish collections from multiple locations in a sample unit on the same survey date; or
3. repeated fish collections from one location in a sample unit on the same survey date.

The biological and sampling processes for detecting species can be described mathematically (Mackenzie et al. 2002), where the true presence or absence of a species at site i , z_i is a Bernoulli process with expected value, ψ :

$$z_i \sim \text{Bernoulli}(\psi). \quad \text{Equation 1.}$$

Observed occurrence (h) at site i and survey j is, therefore, also a Bernoulli process conditional on true occurrence and the probability of detecting the species during the particular survey (p_{ij}):

$$h_{ij} | z_i \sim \text{Bernoulli}(z_i p_{ij}). \quad \text{Equation 2.}$$

Using this model, the observed or complete data likelihood can be developed for parameter estimation (MacKenzie et al. 2002, 2018).

Consider the example where a researcher is interested in quantifying the distribution of a species across a geographic area. To start, sites are randomly selected to survey for the

presence or absence of that species. If the species was collected at a site, catch data would be coded as a 1, or a 0 if not observed (i.e., detection versus non-detection). If five repeat samples were performed at site i and resulted in the detection history (H) of $H = 1\ 0\ 1\ 0\ 1$, where the species was detected in the first, third, and fifth surveys, but was not detected in the second or fourth surveys, a probability statement describing the data could be written as:

$$Pr(10101) = \psi_i p_{i,1} (1 - p_{i,2}) p_{i,3} (1 - p_{i,4}) p_{i,5} \quad \text{Equation 3.}$$

where the probability of observing that specific detection history (i.e., 10101) is a function of the site-specific occupancy probability (ψ_i) multiplied by the survey-specific detection probabilities for the first ($p_{i,1}$), third ($p_{i,3}$), and fifth surveys ($p_{i,5}$), and one minus the survey-specific detection probabilities for the second ($1 - p_{i,2}$) and fourth surveys ($1 - p_{i,4}$) when the species was missed. Assuming that sites are independent, the product of terms (one for each site) creates the model likelihood for the observed set of data, which can then be maximized to obtain maximum-likelihood estimates of ψ_i and p_{ij} (MacKenzie et al. 2002, 2018). ψ_i can be modelled as a function of site-specific habitat variables and p can be modelled as a function of site- and survey-specific variables, allowing the ability to test hypotheses about changes in ψ and p over time as it relates to habitat features with perceived importance (MacKenzie et al. 2002, 2018, Noon et al. 2012, Dextrase et al. 2014a).

There are six assumptions of the static single-species occupancy model:

1. the occupancy of the sample units does not change during the period of surveying;
2. the probability of occupancy is equal across sample units or accounted for using relevant covariates;
3. the probability of detecting the species in a survey, given presence, is equal across all units or accounted for using relevant covariates;
4. the detection of the species in each survey of a unit is independent of detections during other surveys of the unit;
5. the detection histories observed at each location are independent; and,
6. there is no misidentification of species resulting in false detections (MacKenzie et al. 2002, 2018).

POWER ANALYSIS

Prospective power analyses can help to ensure that monitoring program designs are likely to detect changes in species assessment variables (i.e., occupancy probability). Power analysis has been used to compare the effectiveness of different sampling gear (e.g., bag or beach seines; Reid and Dextrase 2017), sampling strategies (e.g., the use of block nets to enclose sample units; Reid and Hogg 2014), and levels of sampling effort (e.g., time spent electrofishing; Reid and Haxton 2017) to detect changes in the abundance and distribution of Ontario fishes. In this research document, power analysis is used to illustrate how different sampling strategies influence the ability of monitoring program designs to identify changes in Redside Dace occupancy probability.

Due to the probabilistic nature of statistical tests, there is always a chance of identifying an effect (e.g., change in occupancy) when there is no such effect (Type I error; false positive) or not identifying an effect when the effect exists (Type II error; false negative), abbreviated as α and β , respectively. Statistical power is defined as the probability of identifying the effect of a treatment at a chosen significance level (α), given that the effect exists (i.e., $1 - \beta$). Conventionally, α is set to 0.05 in fisheries research, indicating that the researcher accepts a

5% risk of concluding that a difference exists when there is no actual difference. Less frequently acknowledged in fisheries science is β (Peterman 1990), or the probability of committing a Type II error. This is problematic because false negatives, whereby monitoring programs fail to identify a true reduction in occupancy, would have significant implications for the management of imperilled species.

By convention, a power of 0.80 (i.e., $\beta = 0.20$) is considered by statisticians to be sufficient for most studies (Peterman 1990, Cohen 2001, Guillera-Aroita and Lahoz-Monfort 2012), indicating an 80% probability of identifying an effect given that the effect actually exists, or alternatively, a 20% chance of not identifying an effect when the effect actually exists. This would be interpreted as an 80% probability of identifying a proportional reduction (or gain) in species occupancy given that the population has actually experienced the proportional change. Similar to chosen α levels, it is the researcher's decision to establish dependable criteria for β , particularly when considering the ability to detect changes in species persistence. For example, it may be reasonable to reduce β to 0.05 (i.e., power = 0.95) and (or) increase α to improve power, indicating that the researcher is equally, or more concerned with committing a Type II error than a Type I error.

A maximum-likelihood approach based on the two-tailed z -test was developed for assessing the power to detect differences in occupancy between two points in space or time that assumes a standard repeat-survey design with K surveys (e.g., seine hauls) at S sites (Guillera-Aroita and Lahoz-Monfort 2012). This approach takes advantage of the fact that the occupancy probability estimator is normally distributed and unbiased. Differences in occupancy probability are considered significant when:

$$\frac{|\hat{\psi}_1 - \hat{\psi}_2|}{\sqrt{\hat{\sigma}_1^2 + \hat{\sigma}_2^2}} > Z_{\alpha/2}, \quad \text{Equation 4.}$$

where $\hat{\psi}_i$ are maximum-likelihood occupancy estimates and their estimated standard errors ($\hat{\sigma}_i^2$; Guillera-Aroita and Lahoz-Monfort 2012). Using this approach and assuming that K and S remain constant from time 1 to time 2, the number of survey sites S needed to achieve a given power can be derived as a function of α , β , p_1 , p_2 , and ψ_1 . Detection and occupancy probability estimates for Redside Dace in Canada have only recently been calculated and reflect relatively small sampling effort (Table 2), but provide the basis for generating simulations to characterize the effort required to make statistically rigorous conclusions about changes in Redside Dace occupancy over time.

The power to detect proportional reductions (30% - solid lines; 50% - dotted lines) in ψ is a function of S given that $\psi_1 = 0.4, 0.5, 0.6, \text{ or } 0.7$, $p = 0.4, 0.5, 0.6, 0.7, 0.8, 0.9, \text{ or } 1.0$, $K = 3, 4, 5, 6, 7, 8, 9, \text{ or } 10$, and $\alpha = 0.05$ is plotted in Figure 4. As ψ_1 , p , K , and proportional reductions in occupancy are reduced, S increases nonlinearly (Figure 4). For example, to detect a proportional reduction in ψ of 30% (solid lines) with a statistical power of 0.80, approximately 142 sites need to be sampled given that p and $\psi_1 = 0.6$ and $K = 3$; however, if $\psi_1 = 0.5$ and $p = 0.6$, S increases to approximately 200 (Figure 4). If 50 sites are sampled using the traditional $K = 3$ repeat survey approach, and $p = 0.6$, $\psi_1 = 0.5$, and $\alpha = 0.05$, the power to detect a 30% reduction in occupancy probability is less than 0.30 (Figure 4). This would indicate that there is less than a 30% probability of identifying a 30% reduction in occupancy probability (i.e., avoiding a Type II error), while maintaining a 95% probability of avoiding a Type I error.

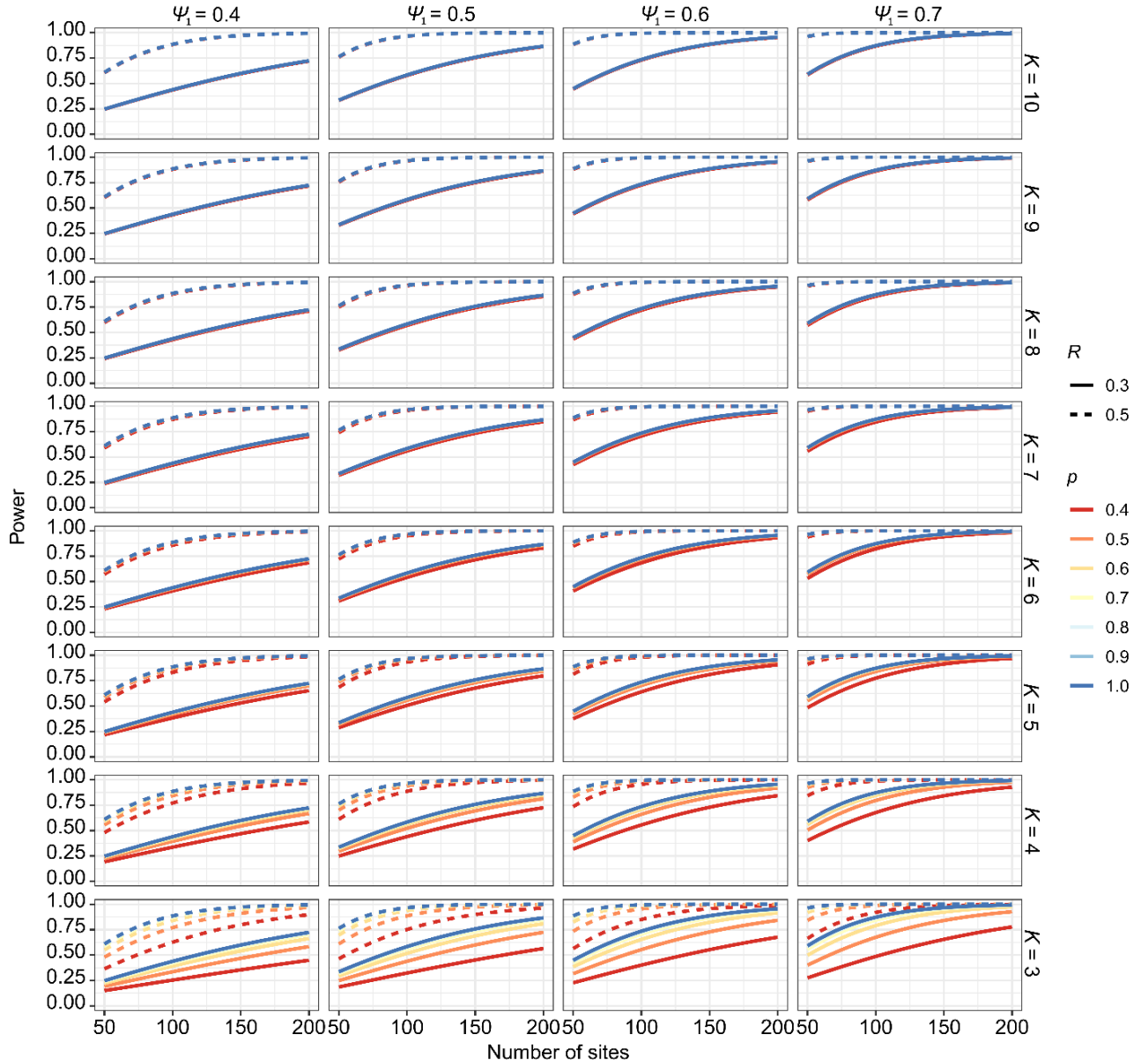


Figure 4. Power to detect proportional reductions (R) in Redside Dace occupancy probabilities of 0.5 (dotted lines) and 0.3 (solid lines) across various detection probability (p) thresholds (0.4–1.0; colors), where initial occupancy probabilities (ψ_1) range between 0.4 and 0.7, the number of surveys per site (K) ranges from 3–10, and $\alpha = 0.05$.

The number of sites S needed to detect proportional reductions in ψ is reduced if α is increased from 0.05 to 0.20 (Figure 5). In such a case, the researcher accepts a 20% probability of concluding that a proportional reduction in occupancy has occurred when, in fact, no such reduction has actually occurred. For example, to detect a proportional reduction of 30% in ψ (solid lines) with a statistical power of 0.80, approximately 82 sites need to be sampled given that p and $\psi_1 = 0.6$ and $K = 3$. Compare this to the 142 sites needed to sample when $\alpha = 0.05$ (Figure 4). Improving power to 0.95 while retaining $\alpha = 0.05$ would require 234 samples at time periods 1 and 2 to detect a proportional reduction of 30% (Figure 4).

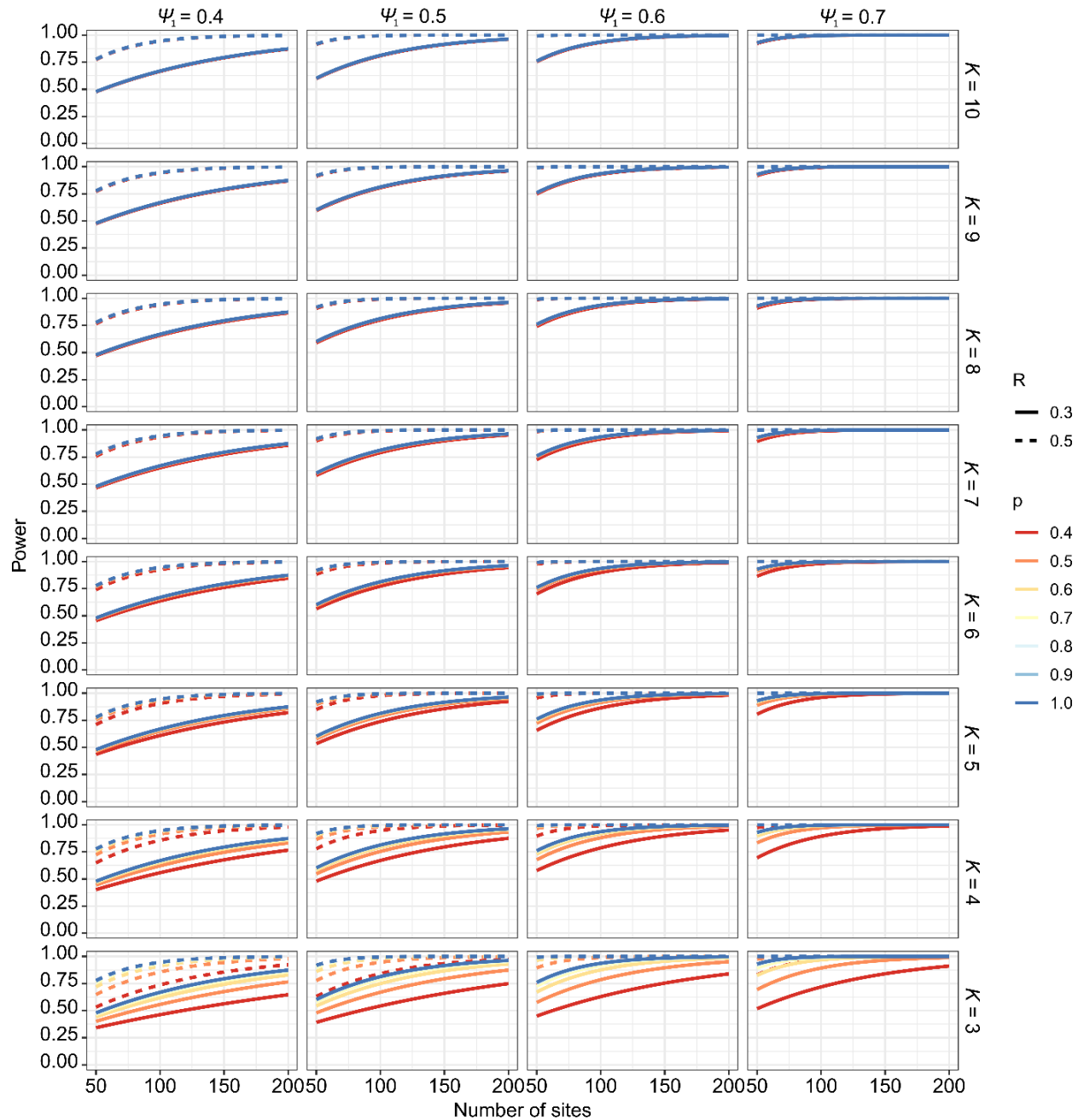


Figure 5. Power to detect proportional reductions (R) in Redside Dace occupancy probabilities of 0.5 (dotted lines) and 0.3 (solid lines) across various detection probability (p) thresholds (0.4–1.0; colors), where initial occupancy probabilities (ψ_1) range between 0.4 and 0.7 and the number of surveys per site (K) ranges from 3–10, and $\alpha = 0.20$.

It is reasonable to expect that p may change over time, which can affect the ability of the researcher to detect proportional changes in ψ (Figure 6). Reductions in p over time have greater impacts on S than improvements in p , comparatively. Moreover, as ψ_1 and p increase, the effects of changes in p over time on S to maintain sufficient power are reduced (Figure 6).

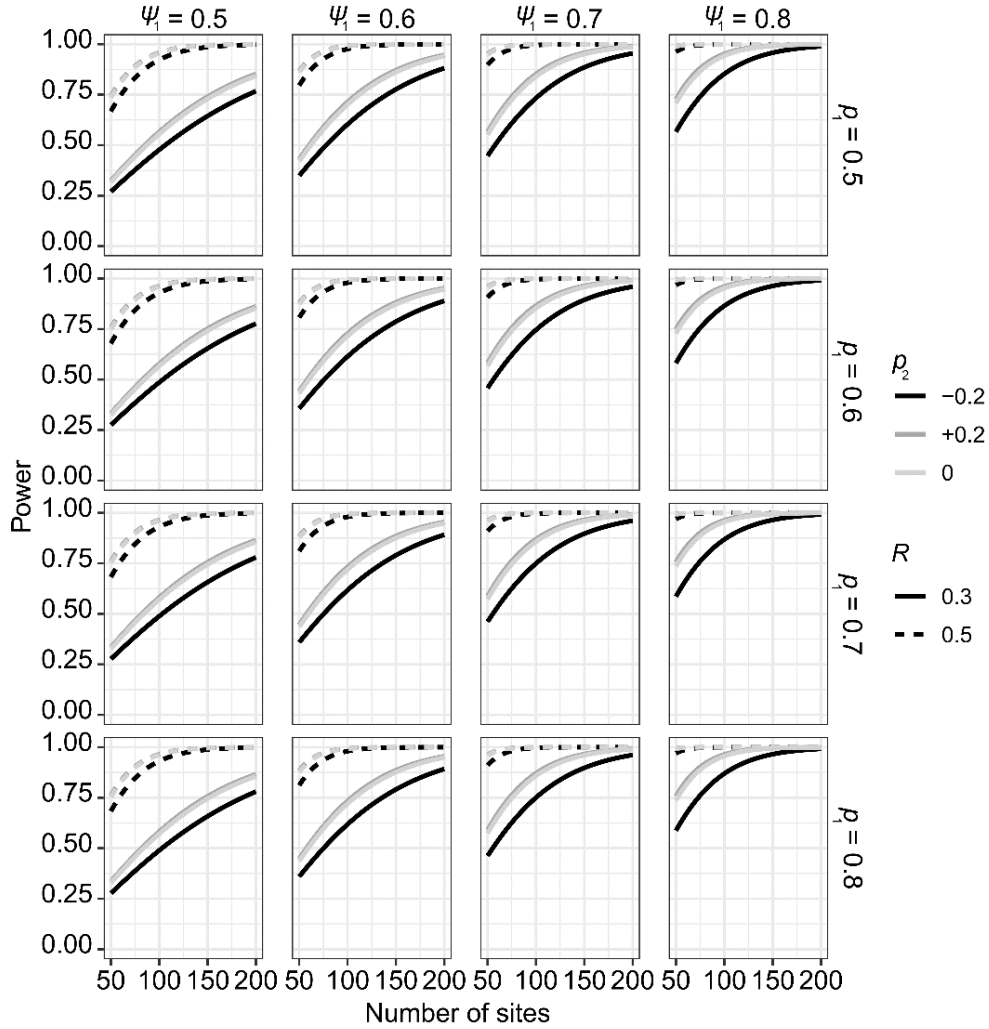


Figure 6. Power to detect proportional reductions (R) in Redside Dace occupancy probabilities of 0.5 (dotted lines) and 0.3 (solid lines) across various initial occupancy (ψ_1 ; columns) and detection (p_1 ; rows) probability thresholds (0.5-0.8), where $K = 5$, $\alpha = 0.05$, and detection probabilities are constant (black), increase by 0.2 (dark grey), or decrease by 0.2 (light grey) over time (p_2).

Overall, small reductions in ψ require large numbers of sampling sites, particularly when p is reduced (Figures 4, 6). However, these simulations also demonstrate that a relatively low level of effort is required to maintain reasonable statistical power if the goal of monitoring is to detect large changes in ψ (e.g., > 50%), such as would be expected with drastic population declines and range reductions. Furthermore, based on the overall consistency of previous estimates of p for Redside Dace across gear types (Table 2), gear choice is a less important factor than S and the desired proportional reduction in ψ .

Given an initial understanding of detection and occupancy probability at an individual site, the minimum number of repeat surveys required to reliably detect the species (K') with some level of confidence (e.g., $\alpha = 0.05$) can be back-calculated for future monitoring based on the site-specific estimates of occupancy (ψ') and detection (p' ; Wintle et al. 2012):

$$K' = \frac{\ln\left(\frac{\alpha}{1-\alpha}\right) - \ln\left(\frac{\psi'}{1-\psi'}\right)}{\ln(1-p')}.$$

Equation 5.

Given previous estimates of p between 0.6 and 0.7, between 1 and 9 repeat surveys are needed to be 95% confident that Redside Dace is absent at a site, depending on ψ' (Figure 7). As ψ' increases, more repeat samples are needed to be sure that Redside Dace is absent (Figure 7). Note, however, that Redside Dace absence during sampling does not imply that Redside Dace does not use the habitat; but instead, that Redside Dace was not present at time of sampling. Multi-season sampling would be needed to confirm lack of use.

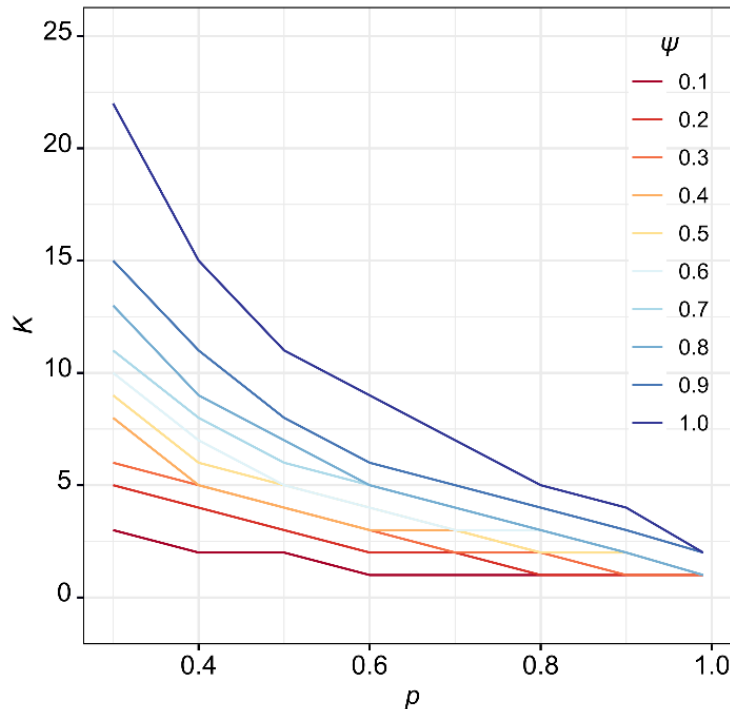


Figure 7. Minimum number of repeated surveys (K) required to detect Redside Dace at $\alpha = 0.05$ based on the probability that the site is occupied even though the species was not detected and relies on site-specific estimates of occupancy (ψ ; colours) across differing detection probabilities ($p = 0.3-1.0$).

CONSIDERATIONS AROUND THE ALLOCATION OF EFFORT FOR A REDSIDE DACE MONITORING PROGRAM

Based on the current knowledge of Redside Dace in Canada, considerations are presented in the following sections to inform a statistically rigorous, occupancy-based monitoring program. Topics include: (i) how site definition (i.e., pool, reach) can change effort requirements for ensuring statistical power when assessing changes in Redside Dace distribution; (ii) considerations around targeted versus random sampling of Redside Dace; and, (iii) how stratifying sampling effort in space and time can be used to evaluate changes in local and total species occupancy.

DEFINING THE SAMPLING UNIT FOR REDSIDE DACE

Defining the scale of the sampling unit is an important decision when designing a monitoring program and should reflect its objectives (Figures 2, 3). The size and structure of streams where Redside Dace occurs vary, which leads to challenges when developing a standardized sampling program across populations. For example, Redside Dace occurs in small creeks draining into Lake Huron (Figure 1) that can have narrow (1–2 m wide) stretches that contain several pools with large woody debris, concentrating individuals in a small area (Figure 8). As a result,

detection probabilities may differ between sampling sites and across habitats. In the following subsections, scenarios are presented where the sampling unit is defined as either a pool or stream reach (as defined by OSAP).



Figure 8. Photograph of Gully Creek sample site (2019) demonstrating block netting and habitat characteristics affecting sampling efforts. Photo credit: Robin Gáspárdy – Fisheries and Oceans Canada.

Pool-Specific Sampling

Following the spawning period, Redside Dace primarily occupies relatively deep pools, which constitutes the smallest biologically relevant sampling unit. Defining the pool as the unit of sampling concentrates efforts toward the dominant microhabitat for Redside Dace, potentially improving the probability of Redside Dace occupancy and detection, and restricting the area of habitat needed to sample. However, defining the pool as the sampling unit omits other microhabitats where Redside Dace may occur (e.g., within runs or riffles). Nonetheless, pool-focused sampling for Redside Dace in Ontario streams has been successfully used to estimate local and regional population densities (Poos et al. 2012) and to study dispersal patterns and metapopulation dynamics (Poos and Jackson 2012, Drake and Poesch 2020).

Pool habitat is described as an area of streams that is relatively deep and slow moving. Jowett (1993) determined from visual assessments and subsequent quantitative measurements that pools, runs, and riffle habitats were best differentiated by velocity:depth ratios and Froude numbers, whereby pool habitat velocity:depth ratio was less than 1.24 and Froude numbers were less than 0.18. The Froude number (Fr) is a dimensionless metric defined as:

$$Fr = \frac{V_m}{\sqrt{gY}}, \quad \text{Equation 6.}$$

where V_m is average water column velocity ($\text{m}\cdot\text{s}^{-1}$), g is acceleration due to gravity ($9.81 \text{ m}\cdot\text{s}^{-2}$), and Y is average water depth (m).

In 2019, DFO conducted pool-specific sampling in the main stem of Gully Creek, a tributary of Lake Huron (Figures 1, 8), to confirm the presence of a previously documented Redside Dace population (ABCA 2010) and characterize Redside Dace habitat (Gáspárdy and Drake 2021).

The sampling frame consisted of one Aquatic Ecosystem Classification (AEC) stream segment (R12.2051; Melles et al. 2013) that contained historical records of Redside Dace. Sampling was performed non-randomly, as the sampling crew aimed to increase the geographic area sampled while targeting sites with pool habitats containing relatively little woody debris (Gáspárdy and Drake 2021). Furthermore, permission from landowners to access all potential sampling sites was not obtained. In total, fishes were sampled from 16 pools ($S = 16$), along with measurements of pool length (m), width (m), and depth (m). Pools were defined based on water velocity, with **hydraulic head** used as a water velocity surrogate at the start and end of each pool (hydraulic head for pool = 0–3 mm).

Pools were enclosed with block-nets and fishes were sampled with a bag seine using three repeat surveys ($K = 3$). After each survey (i.e., seine haul), captured fishes were immediately placed in individual holding bins until all surveys were completed (i.e., removal sampling). Once the three seine hauls were completed, fishes were identified, counted, and subsequently released. Habitat measurements following a modified version of OSAP were recorded at four of the 16 sampled sites in areas upstream and downstream from the sampled pool (i.e., riffle upstream of pool to riffle downstream of pool). In total, these data required six days of field work (June 12, June 25–26, and July 8–10) by five experienced field crew members with additional time spent travelling and preparing gear.

Based on the 2019 data, preliminary estimates of detection and occupancy probability in the main stem of Gully Creek were calculated using the RPresence package (MacKenzie and Hines 2018) in the R statistics software (R Core Team 2019). Of the 16 sampled pools, Redside Dace was captured in 11, indicating a **naïve occupancy probability** of 0.688 (i.e., $11/16 = 0.688$). Based on an intercept-only model (i.e., no included covariates), p for Redside Dace was estimated to be 0.606 ± 0.18 SE. As a result, ψ in the main stem of Gully Creek was estimated to be 0.732 ± 0.14 SE. This suggests that, approximately 73% of pool habitat in the sampled AEC segment of Gully Creek was occupied by Redside Dace. However, given the small sample size and non-random sampling design, the occupancy estimate is interpreted as biased.

Based on a constant detection probability of $p = 0.606$, $\psi_1 = 0.732$, $K = 3$, $S = 16$, and $\alpha = 0.05$, the power to identify a 30% reduction in ψ would be 0.28. This indicates a 28% chance of identifying a 30% reduction in ψ that actually exists, or alternatively, a 72% chance of not identifying a 30% reduction in ψ . These results indicate that more sites (and therefore more sampling effort) would be required to document 30% reductions in ψ with high probability. Improving p to 1.00 and increasing K to 10 repeat surveys in time-step two would only improve the power to identify a 30% reduction in ψ to 0.35. If a $K = 3$ haul approach was retained, approximately 65 sites would be needed during initial sampling and thereafter to have attained a power of 0.80. Alternatively, if K was increased to 5, 42 pools would need to be sampled initially and thereafter to attain a reasonable statistical power (i.e., 0.80) for identifying a 30% reduction in ψ . However, the present sampling design provides reasonable power to identify a 50% and (or) 70% reduction in ψ , estimated at 0.82 and 1.00, respectively.

It is reasonable to hypothesize that pool size (e.g., pool depth, length, width) may affect estimates of p and (or) ψ . Larger pools are more difficult to sample due to depth limitations and in-stream obstructions, possibly affecting p . Alternatively, larger pools may be more attractive to Redside Dace and, therefore, a positive association between ψ and pool size might be expected. Pool-specific covariates can be incorporated into the single-species occupancy models to potentially improve the accuracy and precision of estimates, with the best model for the data selected using **Akaike's Information Criteria (AIC)**.

Model selection using AIC was performed on models that incorporated mean pool depth (m), length (m), width (m), or length + width as ψ and p covariates (all converted to z -scores). Two

pretending variables were removed from the analysis *post hoc* prior to conducting additional analysis (pool length: detection and occupancy; pool depth: detection). The best model for the data based on AIC was an intercept model (i.e., no habitat covariates), with the second best model including a negative effect of pool width on p (Figure 9) and an intercept model for ψ (i.e., no habitat covariates; Table 3). Compared to the intercept model, including pool width reduced mean p for Redside Dace to 0.548 ± 0.20 SE and raised the subsequent estimate of ψ to 0.797 ± 0.16 SE. There was similar support for three other models (i.e., $\Delta AIC < 2$) that included pool width or depth as p or ψ covariates (Table 3). Based on Pearson's χ^2 and parametric bootstrapping ($n = 1,000$) on the most global model, a goodness-of-fit test (MacKenzie and Bailey 2004) indicated high, and statistically significant ($\chi^2 = 21.07$, $p = 0.022$) levels of overdispersion (i.e., variance $>$ mean; $\hat{c} = 2.13$), suggesting that the standard errors of the model estimates were inflated.

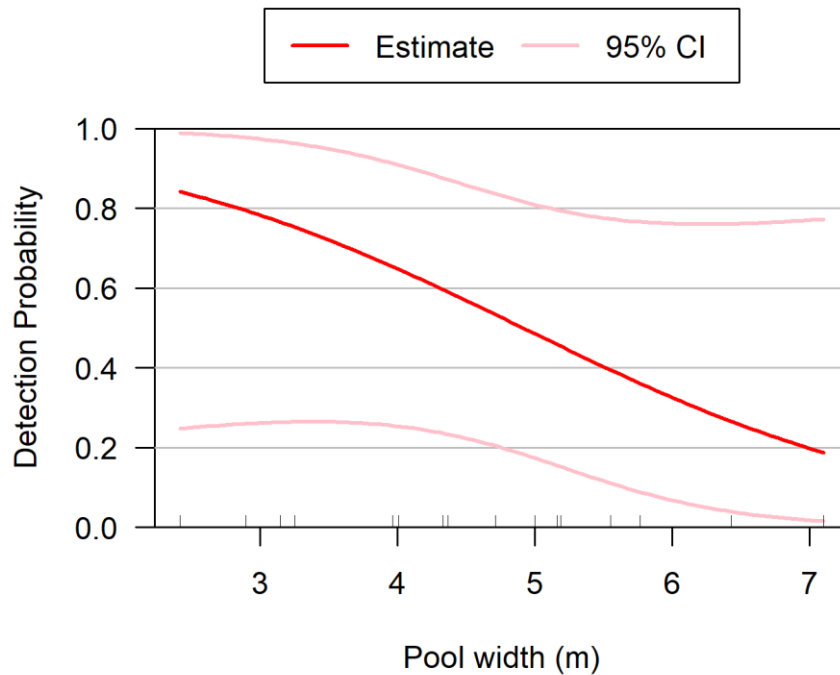


Figure 9. Probability of detecting Redside Dace as a function of pool width (m) in Gully Creek using a $K = 3$ repeat seine haul approach with block nets during the Summer of 2019. Tick marks along the x-axis indicate empirical measurements from the field.

Table 3. Summary of Gully Creek Redside Dace single-season occupancy models selected using AIC. LL = log-likelihood. npar = number of parameters in the model.

Model	AIC	ΔAIC	Akaike weight	$-2*LL$	npar
$p(\cdot), \psi(\cdot)$	42.82	-	0.25	38.82	2
$p(\text{width}), \psi(\cdot)$	43.05	0.23	0.23	37.05	3
$p(\cdot), \psi(\text{depth})$	43.89	1.07	0.15	37.89	3
$p(\cdot), \psi(\text{width})$	43.96	1.14	0.14	37.96	3
$p(\text{width}), \psi(\text{depth})$	44.05	1.23	0.14	36.05	4
$p(\text{width}), \psi(\text{width})$	44.87	2.05	0.09	36.87	4

Knowledge of the total number of pools available for Redside Dace in Gully Creek can allow for an estimate of the number of pools occupied by Redside Dace. However, the data collected from Gully Creek in 2019 do not reflect the entirety of the system given the non-random site selection and relatively few sampled sites. Furthermore, extrapolations using the poorly-fit single-species models can lead to erroneous results. Therefore, results of the extrapolation presented below cannot be related to absolute measures of AO, EO, and (or) MAPV, and should only be interpreted as an approach for extrapolation of the single Gully Creek AEC segment (R12.2051).

The mean dimensions of $S = 16$ pools were $4.58 \text{ m} \pm 0.08 \text{ SE}$ wide by $15.61 \text{ m} \pm 0.44 \text{ SE}$ long, with the mean reach length ($S = 4$) of $33.54 \text{ m} \pm 2.76 \text{ SE}$. The main stem of Gully Creek is approximately 5 km long. If 47% of the main stem of Gully Creek is composed of pool habitat ($15.61 \text{ m} / 33.54 \text{ m} = 0.47$), then 2,350 m of the creek would be composed of pools, and based on mean pool size, would result in approximately 151 total pools ($2350 \text{ m} / 15.61 \text{ m} = 150.51$). Based on the mean width of sampled pools, 151 pools would equal approximately 10,763 m² of pool habitat ($2350 \text{ m} * 4.58 \text{ m} = 10,763 \text{ m}^2$). Given that relatively few sites were sampled to estimate average pool size, it could be argued that the subset of pools sampled does not accurately reflect the true average. For example, if the subset of sampled pools in 2009 were overly large relative to the true average, and instead use the estimate of mean pool length minus 1 SD (i.e., $\bar{x} - 1 \text{ SD} = 15.61 - 7.00 = 8.61$), then there would be approximately 273 total pools ($2350 \text{ m} / 8.61 \text{ m} = 272.94$) in the main stem of Gully Creek. Based on the intercept-only occupancy model, Redside Dace would be expected to occupy approximately 110 pools if a total of 151 exist ($151 * 0.73 = 110.23$) or 199 pools if 273 exist ($273 * 0.73 = 199.29$).

Modelling has been performed to estimate MVP and MAPV of Redside Dace to inform the Recovery Potential Assessment in Canada (van der Lee et al. 2020). Assuming a catastrophe rate of 0.15 per generation using differing catastrophe scenarios, a quasi-extinction threshold of 50 adults, and three approaches for calculating the required area-per-individual, MVP for a Redside Dace population in Canada ranges from 18,000 to 75,000 individuals, and MAPV ranges from 17,000 to 463,000 m² of suitable habitat (van der Lee et al. 2020). Based on estimates of total pool area in the main stem of Gully Creek, the total area of habitat is less than MAPV.

Reach-Specific Sampling

In some situations, it may be advantageous to incorporate stream reaches as the sampling unit, particularly if Redside Dace is suspected of occupying multiple habitat types (i.e., run, pool, riffle) or if habitat is relatively continuous and difficult to differentiate. As well, reach-specific sampling aligns with OSAP, which is used to monitor southern Ontario stream fish communities and has provided a substantial amount of information on Redside Dace occurrence. Defining the reach as the sampling unit allows several habitat types to be sampled, which is advantageous when runs or riffles are occupied by transient individuals (Drake and Poesch 2020) or due to lack of access to high quality pool habitat. Compared to pool-specific sampling, sampling stream reaches will require more time at each sampling unit, reducing the total number of units that can be sampled in a year.

The availability of reach-specific, repeat-survey data for Redside Dace is generally lacking. However, in 2005 and 2006, the OMNRF sampled 7 streams occupied by Redside Dace using repeat-pass electrofishing surveys (14 Mile Creek: $S = 4$ sites; 16 Mile Creek: $S = 1$; Credit River: $S = 2$; Duffins Creek: $S = 2$; Humber River: $S = 6$; Lynde Creek: $S = 10$; Rouge River: $S = 15$; Reid et al. 2009). At each site, standard OSAP was implemented to capture fishes, where fishes were sampled in an upstream manner along 40+ m reaches using a backpack electrofisher. Mean electrofishing effort was $7.70 \pm 0.68 \text{ SE s} \cdot \text{m}^2$. Three ($n = 35$) or four ($n = 5$)

repeated passes were performed at each site. After each pass, captured fishes were placed in individual holding bins until all passes were completed. After the final pass, fishes were identified to species, counted, and subsequently released.

Estimates of p and ψ were calculated using the 2005 and 2006 OMNRF data. It should be noted that calculating such probabilities violates several assumptions of occupancy modelling, including closure and unexplained heterogeneity in p and ψ among locations, which can severely affect estimate accuracy. Therefore, at most, the results should be interpreted as the probability of the site being used rather than the probability of occupancy. Nevertheless, to the knowledge of the authors, these are the only data available at the reach scale for developing such estimates.

Among the 40 sites sampled in 2005 and 2006, Redside Dace was captured in 20, indicating a naïve occupancy probability of 0.50 ($20 / 40 = 0.50$). Based on the sequence of detections, p for Redside Dace was 0.822 ± 0.09 SE and ψ was estimated at 0.503 ± 0.08 SE. If $p = 0.822$, $\psi_1 = 0.503$, $K = 3$, and $S = 40$ are held constant, and $\alpha = 0.05$, the power to identify a 30% reduction in ψ with these data is 0.28. Unlike the previous example from Gully Creek, the present sampling design does not allow for reasonable power to identify a 50% reduction in ψ (i.e., $1 - \beta = 0.67$), but would allow conclusions regarding a 70% reduction in ψ ($1 - \beta = 0.95$). To retain a $K = 3$ repeat pass approach, approximately 54 sites would have needed to be sampled initially and thereafter to have attained a power of 0.80 to identify a 50% occupancy decline.

TARGETED VERSUS RANDOM SAMPLING

An important consideration for using repeat-surveys is whether sites will be targeted or randomly sampled. Random sampling describes the process of selecting sites to survey where each site has an equal probability of being chosen from the sampling frame. Stratified random sampling describes the division of a study area and (or) study system (i.e., Redside Dace populations) into equal or representative groupings of various factors and subsequently selecting an equal number of sites to survey across strata where each site within the stratum has an equal probability of being chosen.

Targeted sampling describes the non-random selection of survey sites to confirm the presence or abundance of Redside Dace at particular sites, and is usually done to follow up on previous detections of a species. In some cases, the area of stream habitat occupied by Redside Dace is so small that it prevents the use of random site selection. Targeting these sites to sample Redside Dace, however, limits the ability to extrapolate to other populations and, instead, limits the focus to quantifying changes in Redside Dace occupancy or abundance at the chosen sites (e.g., pools, reaches). Furthermore, targeted sampling breaks assumptions of occupancy modelling, namely that the probability of occupancy is equal across all sampling units and that the probability of detecting Redside Dace in a survey, given its presence, is equal across all sampling units (MacKenzie et al. 2018). As a result, developing occupancy models with non-random sampling can lead to inaccurate estimates of detection and occupancy. Nevertheless, the ability of researchers to target sites depends on the degree of certainty in local Redside Dace population distribution and the objective of the monitoring program; greater certainty in local site distribution allows for more targeted efforts and can inflate detection probabilities, whereas uncertainty in distribution forces a randomized design.

The process of site selection (i.e., random versus targeted) should reflect the scale of the question and objective, and the available resources. When understanding the presence of Redside Dace is required to inform impact assessments of projects with small spatial footprints, targeted sampling is appropriate. However, to obtain unbiased, comparable, and interpretable

results, a stratified random sampling design is recommended for monitoring changes among Redside Dace populations.

STRATIFYING SAMPLING EFFORT IN SPACE AND TIME

Stratifying sampling effort across populations and over time can be beneficial for initiating monitoring efforts because it:

1. provides the ability to quantify changes in Redside Dace occupancy at differing spatial scales with time-sensitive objectives (i.e., populations at greatest risk of extirpation);
2. enables researchers the ability to extrapolate research findings to representative populations given the proper sample site selection;
3. enables investigations regarding how two broad, regional-scale threats (e.g., agricultural practices, urbanization; Table 1) may affect the persistence of Redside Dace populations differentially in Canada; and,
4. may be beneficial for logistical reasons, such as to optimize sampling given field crew constraints.

There are many ways that sampling effort can be allocated temporally, such as the ‘rotating panel design’ (MacKenzie 2005, Bailey et al. 2007). Temporal allocation will be based on the need to identify changes in assessment variables in relation to species ecology and other endpoints (e.g., COSEWIC-derived assessment timelines). In some cases, temporal replication may be based entirely on policy considerations. For example, habitat protection for Redside Dace in Ontario under the *Endangered Species Act* is dependent on the habitat having been used by the species within the past 20 years, and therefore, it may be important to revisit areas where Redside Dace was previously observed within the 20-year window. Alternatively, there are no explicit requirements for reconfirming the occurrence of SARA-listed species in areas identified as critical habitat and, therefore, it may be less of a priority to revisit such areas to ensure habitat protection. Nonetheless, replicating sampling efforts allows researchers the ability to quantify changes in occupancy and abundance that may occur as a function of recovery measures or threats (Bailey et al. 2007), and also provides a mechanism to fulfill other assessment procedures (e.g., COSEWIC 2017). Furthermore, temporal replication within a year can help understand local movement patterns of Redside Dace or within-year fluctuations of occupancy and (or) abundance (e.g., Drake and Poesch 2020).

If quantifying changes in occupancy probability over time is the primary objective, a randomized approach is recommended, where sites are selected randomly at each time step at the scale of interest. That is, coarse habitat identification and random site selection should occur for time step 1 and time step 2 and used to evaluate changes in detection and (or) occupancy probability over those two time periods, where at each time step the number of sampling sites is held relatively constant. The number of pools or reaches to sample when monitoring Redside Dace populations should be chosen based on local detection and occupancy probability estimates (e.g., Table 2), pool availability within and beyond the suspected range of the species, and the accepted level of power to detect changes over time. With this approach, occupancy estimates could form the monitoring endpoint, and depending on how sites were chosen, could also be used to inform aspects of EO and (or) AO.

If, alternatively, a targeted design was chosen to evaluate changes at particular sites (e.g., historical sites, sites experiencing development pressure, sites with restoration activities), then the targeted design should be continued in the second time step and used to evaluate changes in occupancy and (or) abundance for those particular sites. Changing the design of sampling

efforts from time step 1 to time step 2 should be avoided whenever possible because it prevents meaningful inference between the two time periods.

Based on the geographic extent of Redside Dace populations across Ontario (Figure 1), the different trajectories of populations over time (Table 1), and dominant threats, one goal of a population monitoring program for Redside Dace may be to make conclusions about the status of populations experiencing threats from urban versus agricultural sources. Although these threats may jointly influence populations, each of the 17 populations supporting the species can be classified based on the dominant threat (Table 1). Populations in the GTA are predominantly threatened by urbanization, whereas populations outside of this region (i.e., Saugeen River, Two Tree River, Gully Creek, South Gully Creek, and Irvine Creek) are largely impacted by agriculture (Table 1). However, some populations within the GTA experience the cumulative effects of both threats.

The spatial distribution of Redside Dace populations in urbanized river reaches is more certain than populations in agricultural areas, comparatively. This prior knowledge will improve initial estimates of Redside Dace occupancy and detection probability at sample sites, reducing the number of sites needed to sample (Figures 4, 5). Nevertheless, the choice must be made about whether sampling particular pools or reaches is targeted or randomized. Targeting known sites limits the ability to extrapolate to other populations (e.g., other urban-influenced populations); however, it may not be important to extrapolate depending on the management objective and temporal stratification of effort. If changes in occupancy of agricultural or urban populations of Redside Dace over time is of interest, a randomized approach to sampling pools should be performed where potential sampling habitat (i.e., pools or reaches) is identified and sampling sites are randomly selected at the scale of interest to monitor.

EXTENSIONS TO THE SINGLE-SEASON SINGLE-SPECIES OCCUPANCY MODELS

Although occupancy modelling is a powerful approach to evaluate distribution-based monitoring objectives ('extensive' monitoring as outlined in recovery strategies; RDRT 2010), estimating species abundance (N) and characterizing changes in N over time can directly inform the likelihood that Redside Dace populations remain above MVP, thereby informing intensive monitoring goals. Several approaches have been described to estimate N including mark-recapture and depletion methods (Carle and Strub 1978, Pollock et al. 1990, Ogle 2016). Unfortunately, these approaches have often yielded poor results for small-bodied fishes at risk in Ontario owing to low depletion and re-capture rates (but see Poos et al. 2012).

Alternatively, extensions to the single-season single-species occupancy models, known as N -mixture models, have been developed to estimate N across sites using spatially and temporally replicated count data (Royle and Nichols 2003, Royle 2004, Royle and Dorazio 2008), and could be used if a repeat-survey design was prescribed for monitoring Redside Dace. Compared to conventional approaches for estimating abundance, N -mixture models may be well-suited for estimating Redside Dace abundance given the: (i) implicit incorporation of detection probability differences across sites within the model; (ii) smaller effort requirements compared to traditional approaches; and, (iii) the ability to estimate abundance without marking individuals.

Local species abundance (N) at site i can be viewed as an independent random latent variable arising as part of a distribution (e.g., Poisson) with density $f(N; \lambda)$ (Royle 2004). Rather than directly integrating the conditional likelihood of N and p as a beta prior (Olkin et al. 1981, Carroll and Lombard 1985), the likelihood for the catch data (C) can be integrated over the prior distribution.

$$N_i \sim \text{Poisson}(\lambda_i)$$

Equation 7.

$$C_{ij}|N_i \sim \text{Binomial}(N_i, p_{ij})$$

Equation 8.

Royle (2004) recommended the Poisson density for f , where:

$$f(N; \lambda) = \frac{e^{-\lambda} \lambda^N}{N!}.$$

Equation 9.

Similar to single-season single-species occupancy models, N and p can be modelled as functions of habitat covariates, allowing the ability to test and make inferences on changes in N and p over time as it relates to site-specific habitat features with some perceived importance. It is critical to note that p is interpreted differently between N -mixture models and occupancy models. For N -mixture models, p is the probability of detecting an individual of a species at a site given that an individual is present, whereas from an occupancy standpoint, p is interpreted as the probability of detecting the species given that the species is present. The probability of detection in occupancy models (P_{Occ}) and N -mixture models (P_{NMix}) are related, whereby:

$$P_{Occ} = 1 - (1 - P_{NMix})^N.$$

Equation 10.

There are several assumptions of N -mixture models that if violated can bias results (Barker et al. 2018, Duarte et al. 2018, Knape et al. 2018, Link et al. 2018). First, the binomial count assumption that underlies N -mixture models requires closure across sites and no double counting of individuals within a survey. These conditions can be met given proper site selection and block netting. If, however, the assumption of closure may be violated, it is recommended that the results of N -mixture models be interpreted as indices of relative abundance as opposed to absolute abundance (Barker et al. 2018). Generalizations of the Royle (2004) model have been developed to allow an open population (Dail and Madsen 2011) or when lacking a robust sampling design (Dail and Madsen 2012), but their use is outside the scope of this report. A thorough review of extensions and alternatives to the Royle (2004) model for estimating species abundance is provided in Dénes et al. (2015).

Pool-Specific Abundance in Gully Creek

The number of Redside Dace in the sampled stretch of Gully Creek can be estimated using N -mixture models to incorporate p . Similar to single-season occupancy models, the effects of pool depth, length, width, and the additive effects of length and width were considered on p and λ , and pretending variables were removed from the analysis *post hoc* prior to conducting additional analyses. A negative-binomial distribution for the latent abundance distribution was chosen after initial tests of model fit, and 75 individuals were used as the upper index of integration. Models were built using the ‘unmarked’ package (Fiske and Chandler 2011) with the ‘gmultmix’ function in the R statistical software (R Core Team 2019).

Two models were retained that did not contain pretending variables. Of the two, the best N -mixture model included pool width as a p covariate and an intercept model for N (Table 4), providing an average of 9.30 ± 3.69 SE Redside Dace per site. Mean p was estimated to be 0.698 ± 0.06 SE. Contrary to the relationship between ψ and pool width (Figure 9), pool width showed a positive effect on the probability of detecting an individual Redside Dace, p (Figure 10). However, this result demonstrates the poor model fit (residual versus predicted plot; Figure 11), as the probability of detecting an individual Redside Dace should not be greater than the probability of detecting the species (i.e., 0.606). Nevertheless, the best model for the data chosen using AIC suggested 149 (141–174 95% CI) individuals across the 16 sites, with 5 of the sites likely to not support Redside Dace (Sites 1, 2, 6, 10, and 11; Figure 12); compare this to the 135 individuals captured across 11 sites (i.e., naïve abundance).

Table 4. Summary of Gully Creek Redside Dace N-mixture models selected using AIC. LL = log-likelihood. npar = number of parameters in the model.

Model	AIC	ΔAIC	Akaike weight	$-2*LL$	npar
$p(\text{width}), \theta(\cdot)$	-270.87	-	1.00	-139.43	4
$p(\cdot), \theta(\cdot)$	-257.42	13.45	< 0.01	-131.71	3

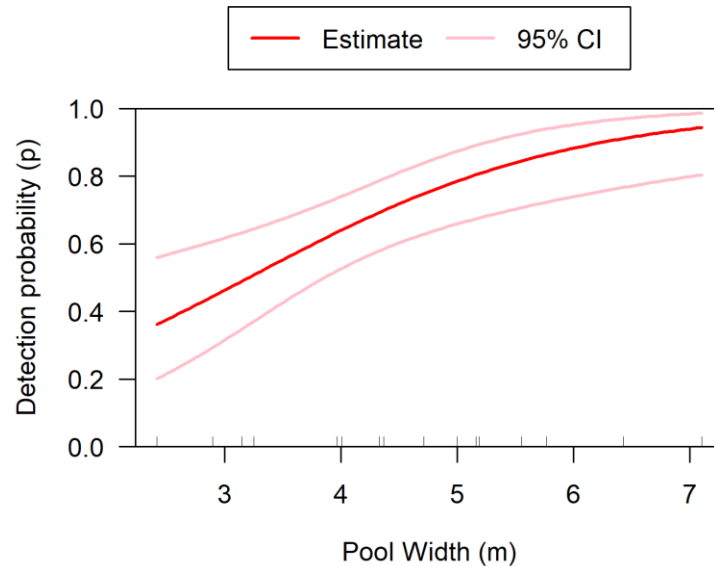


Figure 10. Probability of detecting an individual Redside Dace using the best N-mixture model as a function of pool width (m) in Gully Creek using a K = 3 repeat seine haul approach with block nets during the Summer of 2019. Tick marks along the x-axis indicate empirical measurements from the field.

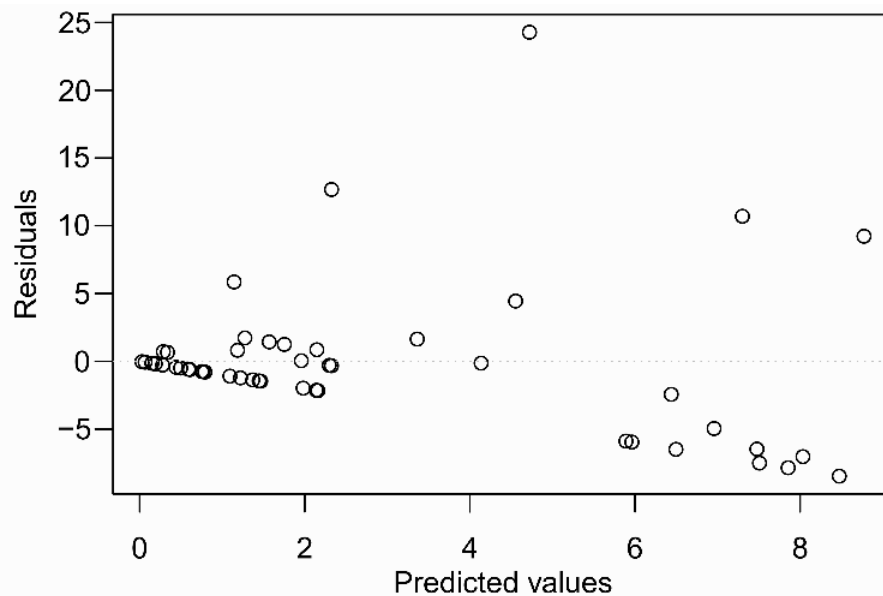


Figure 11. Residuals of the best N-mixture model versus model predictions, indicating a poor model fit. Residual plots of well-fit models would be distributed symmetrically around 0, whereas this plot shows increased variance with greater predicted values.

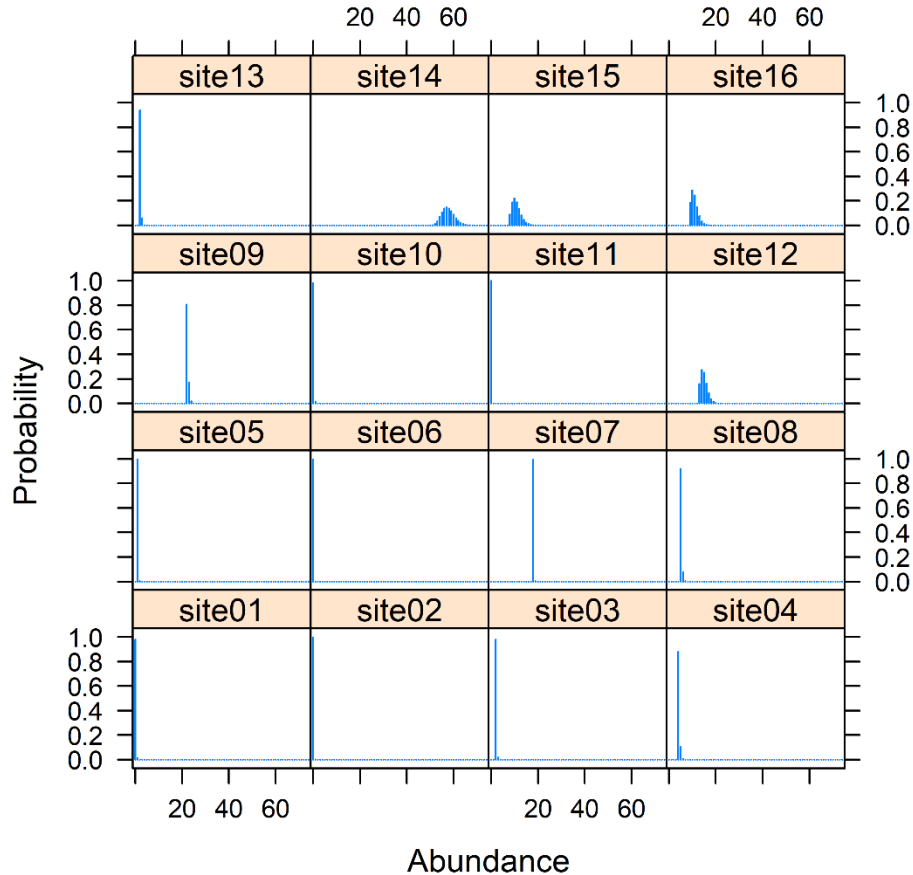


Figure 12. Probability distributions of Redside Dace abundance for 16 sites in Gully Creek using a $K = 3$ repeat seine haul approach with block nets during the Summer of 2019.

Based on the best model, approximately 1,404 individuals are estimated to occupy the main stem of Gully Creek if there are 151 pools (151 pools total * 9.30). Alternatively, if the subset of sampled pools were overly large and the population abundance estimates were based on mean pool length minus 1 SD then the Redside Dace population would be estimated at approximately 2,539 individuals (273 * 9.30). The population abundance value can be compared with the estimated MVP for Redside Dace of 18,000 to 75,000 individuals (van der Lee et al. 2020).

These calculations provide a worked example of approaches to estimate total abundance and should not be interpreted as a direct assessment of the Gully Creek population, in that Gully Creek consists of an additional ~10 km of creek habitat beyond the main stem that diverges in the headwaters where Redside Dace individuals have been captured (ABCA 2010).

Extrapolating to the entire system based on the mean pool size in the main stem, approximately 452 pools may exist throughout Gully Creek ($15,000 \text{ m} * 0.47 = 7,050 \text{ m} / 15.61 \text{ m} = 451.63$ pools). Based on this pool estimate, the entire Gully Creek population would consist of approximately 4,204 individuals ($452 * 9.30$). However, these data were not collected to represent the entirety of Gully Creek and, therefore, this population estimate likely does not reflect the true abundance and should be interpreted with extreme caution. N -mixture models were not developed for Redside Dace in other Ontario streams at the reach scale due to the few sampled sites per river system and lack of descriptive covariates.

DISCUSSION

Assessing the abundance and distribution of SARA-listed species through time is a vital component of the federal recovery planning process. However, to date, a relatively low amount of research effort has involved evaluating the ecological rationale for various monitoring endpoints, or assessing the suitability of sampling designs to evaluate those endpoints, particularly for freshwater fishes. The benefit of using a repeat-survey design for monitoring Redside Dace is that it can be modified to answer particular management and research questions at a variety of scales (i.e., local, population, and national), using different gears or a combination of gears (e.g., Lamothe and Drake 2020), and across strata to inform the conservation status of the species (Figure 2), altogether while adhering to the two-tiered monitoring framework outlined in provincial (RDRT 2010) and draft federal Redside Dace recovery strategies (Amy Boyko, DFO Species at Risk Program, pers. comm.). Using Gully Creek as a case study, results confirm that imperfect detection is an important consideration when characterizing the occupancy and abundance of Redside Dace and, therefore, will be equally important when informing future changes in distribution and abundance over time.

Compared to other SARA-listed freshwater fishes in southern Ontario (e.g., Northern Madtom, Silver Shiner, Pugnose Minnow, Eastern Sand Darter; Dextrase et al. 2014a,b, Lamothe and Drake 2020, Lamothe et al. 2019a,b, 2020), detection probability of Redside Dace in Gully Creek was relatively high, and is generally high regardless of the system or gear employed ($p = 0.45-0.74$ for backpack electrofishing; $p = 0.58-0.72$ for seining; $p = 0.74$ for underwater cameras; $p = 0.64-0.82$ for eDNA; Table 2). However, it should be acknowledged that these estimates of detection probability do not reflect a random sample and likely portray the best-case scenarios for Redside Dace. Future declines in species abundance or concentrated sampling in marginal habitat will result in lower values of detection probability relative to those presented in Table 2, further emphasizing the importance of accounting for imperfect detection in program design.

Efforts to maximize detection are among the most pressing concern for designing a statistically rigorous monitoring program. As demonstrated through power analysis, many sites need to be sampled to obtain a suitable level of power to identify proportional changes in occupancy over space or time if detection and occupancy probabilities are low, coinciding with previous studies (e.g., Guillera-Aroita et al. 2010). This is especially true when detecting small changes in occupancy probability between sample periods (< 30%); however, fewer sites are needed to detect large changes (> 50% of occupancy probability) with strong statistical power, particularly if α is reduced. Therefore, relatively fewer sites are needed to document drastic changes in species occupancy, such as for populations that are declining rapidly, assuming that detection and occupancy probability are relatively constant across populations.

The results of the case studies demonstrated repeat-survey design and application of occupancy models, and N -mixture models, and were grounded on relatively few sampled sites and models with substantial unexplained heterogeneity. Therefore, the worked examples should not be used for extrapolation across the species range. Nevertheless, the results provide a strong reference point for developing an occupancy-based Redside Dace monitoring program in Canada, and based on the power analysis, would provide the necessary empirical data for documenting a 50% or 70% reduction in Redside Dace occupancy probability in the main stem of Gully Creek. Furthermore, demonstrating the absence of Redside Dace at a site in Gully Creek, which can inform site-specific restoration or development decisions, requires a relatively low level of effort (i.e., fewer than 10 repeated seine hauls at a site) because of the moderately high estimates of occupancy and detection. However, the ability to understand why proportional reductions (or gains) in occupancy or detection have occurred would not be feasible given the data collected, and therefore, would require a greater level of effort toward collecting variables

potentially important for such changes. This is a critical issue in the development of imperilled species monitoring programs, as determining the reasons for increases or decreases in the species or population of interest are often as important as indicating that such increases or decreases have occurred.

Although this study does not describe the logistical considerations for implementing a Redside Dace monitoring program (e.g., allocating sampling resources given various crew or sampling constraints), the data needs and several considerations around sampling to develop an occupancy-based monitoring program have been presented (i.e., spatially and temporally replicated surveys with corresponding habitat measurements). Prior to implementation, initiating a Redside Dace monitoring program will require clear identification of the monitoring program objectives to ensure sufficient statistical power of the sampling design (Guillera-Arroita et al. 2010; Figure 2) and to incorporate costs of data collection that considers practical limitations such as the number of sites that can be sampled within a season. Advice about monitoring program design presented here is based on the primary objective of monitoring changes in occupancy (distribution), which can be extended to other distribution-based assessment variables such as AO, EO, and MAPV. Other endpoints, such as changes in species abundance, can be incorporated to determine the probability of a population being above MVP or some other relevant threshold. At smaller spatial scales (i.e., pools or reaches), both distribution and abundance-based approaches can be used to evaluate the effect of recovery measures or development activities.

The choice of assessment endpoint should be determined based on the objective of the program and available resources (Figure 2). If quantifying changes in the abundance of a particular Redside Dace population is the primary objective ('intensive monitoring', as per Wilson and Dextrase 2008), then it is important that fish counts (i.e., abundance) are taken at each site and, furthermore, that habitat variables are measured to reduce the effects of site-specific heterogeneity in detection, occupancy, or abundance dynamics. If the ultimate goal is to use sampling efforts to extrapolate to other populations (Figure 3), then it is important that sampling sites are chosen in a way that supports that objective. Determining the number of sites, and therefore the necessary sampling resources, will depend on the objective and the available habitat within the sampling frame.

There are several outstanding questions and uncertainties around the implementation of an occupancy-based approach to monitor Redside Dace that, if answered, would help improve future monitoring efforts. First, the relationship between species occupancy and abundance is a long-standing research topic in ecology that has been shown to differ by species and spatial scale (Gaston et al. 2000, He and Gaston 2000, Borregaard and Rahbek 2010, Steenweg et al. 2018). Although declines in species occupancy are intrinsically related to declines in species abundance, the exact shape of the underlying relationship for Redside Dace is unknown, including the extent to which the relationship differs across sites and among populations. This is important for Redside Dace if occupancy is to be assumed as a surrogate for abundance, as the species benefits from schooling behaviour (COSEWIC 2017), potentially to reduce predation risk (Poos and Jackson 2012). Given the schooling behaviour, one could expect that the relationship between occupancy and abundance is nonlinear for Redside Dace, whereby abundance increases at a faster rate than the probability of occupancy. Conversely, as the species becomes less abundant and fewer individuals occupy individual pools, detection probability would likely be reduced.

Secondly, characterizing inter-annual variability of Redside Dace habitat conditions and pool and (or) reach occupancy estimates would improve interpretations of temporal occupancy trends. For example, failing to quantify natural variability in occupancy estimates can lead to false conclusions about changes (or lack thereof) in species occupancy (MacKenzie 2005).

Furthermore, multi-season and multi-year habitat sampling in Redside Dace habitats would provide greater insight into how changes in habitat variables correlate with changes in occupancy.

Finally, an understanding of the direct physical harm to individuals (and associated population-level harm) caused by different scientific sampling techniques has yet to be fully quantified. In particular, understanding the relationship between the number of surveys, number of sample sites, size of sample sites (i.e., pools versus reaches), and different sampling gear (i.e., backpack electrofisher using different settings versus seine hauls) with Redside Dace mortality would improve efforts to minimize harm while ensuring that statistically meaningful conclusions can be made. Acquiring this knowledge will require: (i) ongoing tracking of sampling-related mortalities from stream fish community monitoring efforts; (ii) field and laboratory experiments to compare different sampling methods and identify influential covariates (e.g., stream water temperature); and, (iii) modelling-based simulations to quantify the effect of sampling designs on population growth rates.

Overall, this research document provides advice for the design of a two-tiered Redside Dace monitoring program that would provide baseline and ongoing information on range-wide occupancy while allowing the opportunity to track the trajectory of select sites or populations. Furthermore, this study reconfirms the importance of using a repeat-survey design to account for species detection probability when estimating species abundance or distribution (MacKenzie et al. 2002, 2018, Dextrase et al. 2014a,b, Lamothe and Drake 2020, Lamothe et al. 2019a,b, 2020), particularly given the low abundance and patchy distribution of Redside Dace. Failing to implement a monitoring program that considers imperfect detection may ultimately lead to erroneous conclusions about the conservation status of Redside Dace populations in Canada.

GLOSSARY

Abundance – the number of individuals at a particular sampling unit (e.g., site).

AIC (Akaike's Information Criterion) - a measure of the relative quality of a statistical model for a given set of data that represents the amount of information lost for any particular model. Lower AIC values within the same candidate set of models represent better models. $AIC = -2 \times [\log \text{likelihood}] + 2K$, where K is the number of model parameters.

AIC weight – level of support for each model in the candidate set. For a given model, the AIC weight is between 0 and 1 and the sum of AIC weights for all models in the candidate set equals 1.

Area of Occupancy (AO) – area within the extent of occurrence that is occupied by the wildlife species, excluding cases of vagrancy (COSEWIC 2017).

Closure – the assumption of occupancy models whereby the sample site is closed to immigration or emigration of species during the sampling effort.

Critical Habitat (CH) – habitat that is necessary for the survival or recovery of Extirpated, Endangered, or Threatened species, and that is identified as CH in a recovery strategy or action plan.

Crossover Point – the location where the thalweg (main concentration of flow, normally the deepest part of the channel) is in the centre of the channel during bankfull discharge.

Detection probability – the probability that at least one individual of a target species is detected in a single survey of a site given that the site is occupied by the species.

Distribution – spatial area over which the species is expected to occur, including the area of occupancy and extent of occurrence.

Extent of Occurrence (EO) - area included in a polygon without concave angles that encompasses the geographic distribution of all known populations of a wildlife species (COSEWIC 2017).

Greater Toronto and Hamilton Area (GTA) – The most populated metropolitan area in Canada. Hamilton, Ontario is west of the Greater Toronto Area, which includes the city of Toronto and six census divisions (Durham Region, Halton Region, Peel Region, York Region, Dufferin County, and Simcoe County).

Hydraulic Head – a surrogate measurement of velocity, measured as the difference in height of water between the front and back of a vertically held ruler that is placed at right angles to the flow of water.

Imperfect Detection – the condition when detection probability is less than 1. When imperfect detection exists, sampling efforts will not detect a species at some sites where it is actually present.

Meandering Stream – a stream channel form characterized as having low to moderate slope (typically < 2%), unconfined in the valley enabling lateral movement of the channel through erodible material.

Minimum Area for Population Viability (MAPV) – the total minimum habitat area required for a viable population assuming independent stage-specific habitat use.

Minimum Viable Population (MVP) – the absolute minimum population size of age 1+ individuals that has a certain probability of remaining extant over some period of time despite the continuous effects of stochasticity and catastrophic events.

Naïve occupancy – the proportion of sites sampled where the species was detected.

Occurrence – the observation of species presence at a site.

Occupancy probability (ψ) – probability that a randomly selected site or sampling unit in an area of interest is occupied by a species.

Overdispersion (\hat{c}) – occurs when the observed variance in the data is larger than the predicted variance. It is necessary to account for overdispersion in the data (i.e., $\hat{c} > 1$) when calculating AIC values (Quasi-AIC) and unconditional variances of model-averaged parameter estimates.

Pool – areas of streams that are relatively deep and have low water velocity.

Population (demographic) – a group of individuals of the same species that live together in an area of sufficient size to permit normal dispersal and/or migration behavior and in which numerical changes are largely determined by birth and death processes (i.e., emigration and immigration rates are negligible; Berryman 2002).

Population (occupancy) – proportion of the landscape where a species' occupancy state is being studied.

Power – the probability of detecting the effect of a treatment at a chosen significance level given that the effect exists.

Pretending variable: a variable that has no effect on the deviance ($-2 \times \log$ -likelihood) and results in an AIC value approximately 2 AIC units higher than the previous model. Pretending variables have no effect on the predictive value of the model (as indicated by the lack of change in deviance) and simply increase the AIC value by 2 units by the addition of an additional variable to the AIC parameter penalty term.

Reach – a stream segment of at least 40 m that includes at least one riffle-pool sequence, beginning and ending at crossover points.

Repeat surveys – refers to individual surveys that are conducted more than once at an individual site within one sample season.

Sampling frame – a list of potential sites or the geographic region from which a sample can be taken. Sampling frames can be informed by numerous ecological criteria (e.g., pools of a given size or those containing particular habitat attributes).

Season (occupancy) – a relatively short timeframe whereby all sites are closed to changes in occupancy so that sites are either always occupied or always absent.

Survey – an individual sampling event at a site. For the repeat surveys required in occupancy modelling, there are multiple surveys at each site.

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REFERENCES CITED

- Andersen, J.J. 2002. Status of Redside Dace, *Clinostomus elongatus*, in the Lynde and Pringle Creek watersheds of Lake Ontario. *Can.-Field Nat.* 116(1): 76–80.
- Ausable Bayfield Conservation Authority (ABCA). 2010. Habitat assessment for Redside Dace in Gully Creek. Ausable Bayfield Conservation Authority. Exeter, ON. 15 p.
- Bailey, L.L., Hines, J.E., Nichols, J.D., and MacKenzie, D.I. 2007. Sampling design trade-offs in occupancy studies with imperfect detection: examples and software. *Ecol. Appl.* 17(1): 281–290.
- Barker, R.J., Schofield, M.R., Link, W.A., and Sauer, J.R. 2018. On the reliability of *N*-mixture models for count data. *Biometrics* 74(1): 369–377.
- Bayne, E.M., Boutin, S., and Moses, R.A. 2008. Ecological factors influencing the spatial pattern of Canada lynx relative to its southern range edge in Alberta, Canada. *Can. J. Zool.* 86(10): 1189–1197.
- Benoit, D., Jackson, D.A., and Ridgway, M.S. 2018. Assessing the impacts of imperfect detection of estimates of diversity and community structure through multispecies occupancy modelling. *Ecol. Evol.* 8(9): 4676–4684.
- Berryman, A.A. 2002. Population: a central concept for ecology? *Oikos* 97(3): 439–442.
- Borregaard, M.K., and Rahbek, C. 2010. Causality of the relationship between geographic distribution and species abundance. *Quart. Rev. Biol.* 85(1): 3–25.
- Carle, F., and Strub, M. 1978. A new method for estimating population size from removal data. *Biometrics* 34(4): 621–630.
- Carroll, R.J., and Lombard, F. 1985. A note on *N* estimators for the binomial distribution. *J. Am. Stat. Assoc.* 80(390): 423–426.
- Castañeda, R.A., Weyl, O.L.F., and Mandrak, N.E. 2020. Using occupancy models to assess the effectiveness of underwater cameras to detect rare stream fishes. *Aquat. Conserv. Mar. Freshwater Ecosyst.* 30(3): 565–576.
- Chen, G., Kéry, M., Plattner, M., Ma, K., and Gardner, B. 2013. Imperfect detection is the rule rather than the exception in plant distribution studies. *J. Ecology* 101(1): 183–191.
- Cohen, B.H. 2001. *Explaining Psychological Statistics* (2nd ed.). John Wiley & Sons Inc. New York, NY. 792 p.
- COSEWIC (Committee on the Status of Endangered Wildlife in Canada). 2007. [COSEWIC assessment and update status report on the Redside Dace *Clinostomus elongatus* in Canada](#). Committee on the Status of Endangered Wildlife in Canada. Ottawa, ON. vii + 50 p.
- COSEWIC. 2017. [COSEWIC assessment and status report on the Redside Dace *Clinostomus elongatus* in Canada](#). Committee on the Status of Endangered Wildlife in Canada. Ottawa, ON. xii + 63 p.

-
- Dail, D., and Madsen, L. 2011. Models for estimating abundance from repeated counts of an open metapopulation. *Biometrics* 67(2): 577–587.
- Dail, D., and Madsen, L. 2012. Estimating open population site occupancy from presence-absence data lacking the robust design. *Biometrics* 69(1): 146–156.
- Dénes, F.V., Silveira, L.F., and Beissinger, S.R. 2015. Estimating abundance of unmarked animal populations: accounting for imperfect detection and other sources of zero inflation. *Meth. Ecol. Evol.* 6(5): 543–556.
- Dextrase, A.J., Mandrak, N.E., and Schaefer, J.A. 2014a. Modelling occupancy of an imperilled stream fish at multiple scales while accounting for imperfect detection: implications for conservation. *Freshwater Biol.* 59(9): 1799–1815.
- Dextrase, A.J., Mandrak, N.E., Barnucz, J., Bouvier, L.D., Gaspard, R., and Reid, S.M. 2014b. [Sampling Effort Required to Detect Fishes at Risk in Ontario](#). *Can. Manuscr. Rep. Fish. Aquat. Sci.* 3024: v + 50 p.
- Dorazio, R.M., Jelks, H.L., and Jordan, F. 2005. Improving removal-based estimates of abundance by sampling a population of spatially distinct subpopulations. *Biometrics* 61(4): 1093–1101.
- Drake, D.A.R., and Poesch, M.S. 2020. [Seasonal movement of Redside Dace \(*Clinostomus elongatus*\) in Relation to Abiotic and Biotic Factors](#). *DFO Can. Sci. Advis. Sec. Res. Doc.* 2019/077. vi + 31 p.
- Duarte, A., Adams, M.J., and Peterson, J.T. 2018. Fitting *N*-mixture models to count data with unmodeled heterogeneity: Bias, diagnostics, and alternative approaches. *Ecol. Model.* 374(24): 51–59.
- Fiske, I., and Chandler, R. 2011. unmarked: An R Package for Fitting Hierarchical Models of Wildlife Occurrence and Abundance. *J. Stat. Soft.* 43(10): 1–23.
- Gáspárdy, R.C., and Drake, D.A.R. 2021. [Targeted Sampling for Redside Dace \(*Clinostomus elongatus*\) in Gully Creek, Ontario, 2019](#). *Can. Data Rep. Fish. Aquat. Sci.* 1315: vii + 48 p.
- Gaston, K.J., Blackburn, T.M., Greenwood, J.J.D., Gregory, R.D., Quinn, R.M., and Lawton, J.H. 2000. Abundance-occupancy relationships. *J. Appl. Ecol.* 37(Suppl. 1): 39–59.
- Gibson-Reinemer, D.K., Stewart, D.R., Fritts, M.W., DeBoer, J.A., and Casper, A.F. 2016. Estimating the effects of environmental variables and gear type on the detection and occupancy of large-river fishes in a standardized sampling program using multiseason Bayesian mixture models. *N. Am. J. Fish. Manage.* 36(6): 1445–1456.
- Gryska, A.D., Hubert, W.A., and Gerow, K.G. 1997. Use of power analysis in developing monitoring protocols for the Endangered Kendal Warm Springs Dace. *N. Am. J. Fish. Manage.* 17(4): 1005–1009.
- Guillera-Arroita, G., and Lahoz-Monfort, J.J. 2012. Designing studies to detect differences in species occupancy: power analysis under imperfect detection. *Meth. Ecol. Evol.* 3(5): 860–869.
- Guillera-Arroita, G., Ridout, M.S., and Morgan, B.J.T. 2010. Design of occupancy studies with imperfect detection. *Meth. Ecol. Evol.* 1(2): 131–139.
- Hartley, S. 1998. A positive relationship between local abundance and regional occupancy is almost inevitable (but not all positive relationships are the same). *J. Anim. Ecol.* 67(6): 992–994.

-
- Haynes, T.B., Rosenberger, A.E., Lindberg, M.S., Whitman, M., and Schmutz, J.A. 2013. Method- and species-specific detection probabilities of fish occupancy in Arctic lakes: implications for design and management. *Can. J. Fish. Aquat. Sci.* 70(7): 1055–1062.
- He, F., and Gaston, K.J. 2000. Occupancy-abundance relationships and sampling scales. *Ecography* 23(4): 503–511.
- Holtrop, A.M., Cao, Y., and Dolan, C.R. 2010. Estimating sampling effort required for characterizing species richness and site-to-site similarity in fish assemblage surveys of Wadeable Illinois streams. *Trans. Am. Fish. Soc.* 139(5): 1421–1435.
- Joseph, L.N., Field, S.A., Wilcox, C., and Possingham, H.P. 2006. Presence-absence versus abundance data for monitoring threatened species. *Conserv. Biol.* 20(6): 1679–1687.
- Jowett, I.G. 1993. A method for objectively identifying pool, run, and riffle habitats from physical measurements. *N. Zeal. J. Mar. Freshwater Res.* 27(2): 241–248.
- Klein, Z.B., Quist, M.C., Rhea, D.T., and Senecal, A.C. 2015. Sampling techniques for burbot in a western non-wadeable river. *Fish. Manage. Ecol.* 22(3): 213–223.
- Knape, J., Arlt, D., Barraquand, F., Berg, Å., Chevalier, M., Pärt, T., Ruete, A., and Žmihorski, M. 2018. Sensitivity of binomial N-mixture models to overdispersion: The importance of assessing model fit. *Meth. Ecol. Evol.* 9(10): 2102–2114.
- Koster, W.J. 1939. Some phases of the life history and relationships of the cyprinid, *Clinostomus elongatus* (Kirtland). *Copeia* 1939: 201–208.
- Lamothe, K.A., and Drake, D.A.R. 2020. Habitat associations of the Threatened pugnose minnow (*Opsopoeodus emiliae*) at the northern edge of the species range. *Ecol. Freshwater Fish.* 29(2): 289–298.
- 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. *Endang. Spec. Res.* 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 for informing species reintroduction efforts. *Conserv. Biol.* 33(6): 1392–1403.
- Lamothe, K.A., Ziegler, J.P., Gáspárdy, R., Barnucz, J., and Drake, D.A.R. 2020. Ecological associations between the invasive round goby *Neogobius melanostomus* and endangered northern madtom *Noturus stigmosus*. *Aquat. Conserv. Mar. Freshwater Ecosyst.* 30(4): 691–700.
- Lebrun, D.E., Bouvier, L.D., Choy, M., Andrews, D.W., and Drake, D.A.R. 2019. [Information in support of a Recovery Potential Assessment of Redside Dace \(*Clinostomus elongatus*\) in Canada](#). DFO Can. Sci. Advis. Sec. Res. Doc. 2019/033. v + 49 p.
- Link, W.A., Schofield, M.R., Barker, R.J., and Sauer, J.R. 2018. On the robustness of N-mixture models. *Ecology* 99(7): 1547–1551.
- MacKenzie, D.I. 2005. What are the issues with “presence/absence” data for wildlife managers? *J. Wildl. Manage.* 69(3): 849–860.
- MacKenzie, D.I., and Bailey, L.L. 2004. Assessing the fit of site-occupancy models. *J. Agri. Biol. Environ. Stat.* 9: 300–318.
- MacKenzie, D.I., and Hines, J. 2018. RPresence: R interface for program PRESENCE. R package version 2.12.22.

-
- Mackenzie, D.I., Nichols, J.D., Lachman, G.B., Droege, S., Royle, J.A., and Langtimm, C.A. 2002. Estimating site occupancy rates when detection probabilities are less than one. *Ecology* 83(8): 2248–2255.
- Mackenzie, D.I., Nichols, J.D., Royle, J.A., Pollock, K.H., Bailey, L.L., and Hines, J.E. 2018. *Occupancy estimation and modeling: inferring patterns and dynamics of species occurrence*. 2nd ed. Academic Press. Cambridge, MA. 648 p.
- Magoulick, D.D., and Lynch, D.T. 2015. Occupancy and abundance modeling of the endangered yellowcheek darter in Arkansas. *Copeia* 103(2): 433–439.
- Mandrak, N.E., and Bouvier, L.D. 2014. [Standardized data collection methods in support of a classification protocol for the designation of watercourses as municipal drains](#). DFO Can. Sci. Advis. Sec. Res. Doc. 2013/077. v + 27 p.
- Mandrak, N.E., Barnucz, J., and Marson, D. 2010. [Targeted sampling of fish species at risk in the Grand River watershed, 2003](#). Can. Manuscr. Rep. Fish. Aquat. Sci. 2922: v + 29 p.
- McKee, P.M., and Parker, B.J. 1982. The distribution, biology, and status of the fishes *Campostoma anomalum*, *Clinostomus elongatus*, *Notropis photogenis* (Cyprinidae), and *Fundulus notatus* (Cyprinodontidae) in Canada. *Can. J. Zool.* 60(6): 1347–1358.
- Melles, S., Jones, N., and Schmidt, B. 2013. Aquatic Research Series 2013-05: Aquatic ecosystem classification for Ontario: a technical proposal. Ontario Ministry of Natural Resources. 52 p.
- Midway, S.R., Wagner, T., and Tracy, B.H. 2014. A hierarchical community occurrence model for North Carolina stream fish. *Trans. Am. Fish. Soc.* 143(5): 1348–1357.
- Miller, D.A., and Grant, E.H.C. 2015. Estimating occupancy dynamics for large-scale monitoring networks: amphibian breeding occupancy across protected areas in the northeast United States. *Ecol. Evol.* 5(21): 4735–4746.
- Moore, M.J., Orth, D.J., and Frimpong, E.A. 2017. Occupancy and detection of clinch dace using two gear types. *J. Fish Wild. Manage.* 8(2): 530–543.
- Nichols, J.D., and Williams, B.K. 2006. Monitoring for conservation. *Trends Ecol. Evol.* 21(12): 668–673.
- Noon, B.R., Bailey, L.L., Sisk, T.D., and McKelvey, K.S. 2012. Efficient species-level monitoring at the landscape scale. *Conserv. Biol.* 26(3): 432–441.
- Novinger, D.C., and Coon, T.H. 2000. Behaviour and physiology of the redbside dace, *Clinostomus elongatus*, a threatened species in Michigan. *Environ. Biol. Fishes* 57: 315–326.
- Ogle, D.H. 2016. *Introductory fisheries analyses with R*. CRC Press: Taylor & Francis Group. New York, NY. 317 p.
- Olkin, I., Petkau, A.J., and Zidek, J.V. 1981. A comparison of N estimators for the binomial distribution. *J. Am. Stat. Assoc.* 76(375): 637–642.
- Ontario Streams. 2005. Saugeen River watershed Redside Dace monitoring project 2004. Report prepared for Ontario Ministry of Natural Resources. 20 p. + 2 Appendices.
- Page, L.M., and Burr, B.M. 1991. *A Field Guide to Freshwater Fishes of North America North of Mexico*. The Peterson Field Guide Series, Houghton Mifflin Company, Boston, MA. 432 p.

-
- Peoples, B.K., and Frimpong, E.A. 2011. Among-pass, interregional, and single- versus multiple-season comparisons of detection probabilities of stream fishes. *Trans. Am. Fish. Soc.* 140(1): 67–83.
- Peterman, R.M. 1990. Statistical power can improve fisheries research and management. *Can. J. Fish. Aquat. Sci.* 47(1): 2–15.
- Pitcher, T.E., Beausoleil, J.J., Abbott, J.A., and Vandereerden, J.L. 2009. Sperm design and function in the reside dace *Clinostomus elongatus*. *J. Fish Biol.* 75(4): 924–931.
- Pollock, J.F. 2006. Detecting population declines over large areas with presence-absence, time-to-encounter, and count survey methods. *Conserv. Biol.* 20(3): 882–892.
- Pollock, K.H., Nichols, J.D., Brownie, C., and Hines, J.E. 1990. Statistical inference for capture-recapture experiments. *Wildl. Mono.* 107(1): 3–97.
- Poos, M.S., and Jackson, D.A. 2012. Impact of species-specific dispersal and regional stochasticity on estimates population viability on stream metapopulations. *Landscape Ecol.* 27: 405–416.
- Poos, M., Lawrie, D., Tu, C., Jackson, D.A., and Mandrak, N.E. 2012. Estimating local and regional population sizes for an endangered minnow, reddsides dace (*Clinostomus elongatus*), in Canada. *Aquat. Conserv. Mar. Freshwater Ecosyst.* 22(1): 47–57.
- Portt, C.B., Coker, G.A., Mandrak, N.E., and Ming, D.L. 2008. [Protocol for the detection of fish Species At Risk in Ontario Great Lakes Area \(OGLA\)](#). DFO Can. Sci. Advis. Sec. Res. Doc. 2008/026. v + 31 p.
- Potoka, K.M., Shea, C.P., and Bettoli, P.W. 2016. Multispecies occupancy modeling as a tool for evaluating the status and distribution of darters in the Elk River, Tennessee. *Trans. Am. Fish. Soc.* 145(5): 1110–1121.
- Quist, M.C., Gerow, K.G., Bower, M.R., and Hubert, W.A. 2006. Random versus fixed-site sampling when monitoring relative abundance of fishes in headwater streams of the Upper Colorado River basin. *N. Am. J. Fish. Manage.* 26(4): 1011–1019.
- R Core Team. 2019. R: A language and environment for statistical computing. Vienna, Austria: R Foundation for Statistical Computing.
- Redside Dace Recovery Team (RDRT). 2010. Recovery Strategy for Redside Dace (*Clinostomus elongatus*) in Ontario. Ontario Recovery Strategy Series. Prepared for the Ontario Ministry of Natural Resources, Peterborough, ON. vi + 29 p.
- Reich, H.T. 2020. Optimal sampling design and the accuracy of occupancy models. *Biometrics* 76(3): 1017–1027.
- Reid, S.M., and Dextrase, A.J. 2017. Monitoring lake populations of Eastern Sand Darter (*Ammocrypta pellucida*): a comparison of two seines. *J. Freshwater Ecol.* 32(1): 499–511.
- Reid, S.M., and Haxton, T. 2017. Backpack electrofishing effort and imperfect detection: Influence on riverine fish inventories and monitoring. *J. Appl. Ichthyol.* 33(6): 1083–1094.
- Reid, S.M., and Hogg, S. 2014. An evaluation of multiple-pass seining to monitor Blackstripe Topminnow populations in the Sydenham River (Ontario, Canada). *J. Appl. Ichthyol.* 30(5): 962–969.
- 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.
-

-
- Reid, S.M., Yunker, G., and Jones, N.E. 2009. Evaluation of single-pass backpack electric fishing for stream fish community monitoring. *Fish. Manage. Ecol.* 16(1): 1–9.
- Reid, S.M., Kopf, V., and Boothroyd, M. 2017. [An environmental DNA-based survey for Redside Dace \(*Clinostomus elongatus*\) in Greater Toronto Area watersheds](#). *Can. Manuscr. Rep. Fish. Aquat. Sci.* 3120: iv + 10 p.
- Royle, J.A. 2004. N-mixture models for estimating population size from spatially replicated counts. *Biometrics* 60(1): 108–115.
- Royle, J.A., and Dorazio, R.M. 2008. *Hierarchical Modeling and Inference in Ecology: The Analysis of Data from Populations, Metapopulations, and Communities*. Academic Press, London, UK. 464 p.
- Royle, J.A., and Nichols, J.D. 2003. Estimating abundance from repeated presence-absence data or point counts. *Ecology* 84(3): 777–790.
- Sauer, J.R., and Knutson, M.G. 2008. Objectives and metrics for wildlife monitoring. *J. Wildl. Manage.* 72(8): 1663–1664.
- Schloesser, J.T., Paukert, C.P., Doyle, W.J., Hill, T.D., Steffensen, K.D., and Travnichek, V.H. 2012. Heterogeneous detection probabilities for imperiled Missouri River fishes: implications for large-river monitoring programs. *Endang. Spec. Res.* 16: 211–224.
- Serrao, N.R., Reid, S.M., and Wilson, C.C. 2018. Establishing detection thresholds for environmental DNA using receiver operator characteristic (ROC) curves. *Conserv. Genet. Res.* 10: 555–562.
- Smith, C.D., Quist, M.C., and Hardy, R.S. 2015. Detection probabilities of electrofishing, hoop nets, and benthic trawls for fishes in two western North American rivers. *J. Fish Wildl. Manage.* 6(2): 371–391.
- Stanfield, L. 2017. *Ontario Stream Assessment Protocol. Version 10*. Ontario Ministry of Natural Resources. 550 p.
- Steenweg, R., Hebblewhite, M., Whittington, J., Lukacs, P., and McKelvey, K. 2018. Sampling scales define occupancy and underlying occupancy-abundance relationships in animals. *Ecology* 99(1): 172–183.
- Stewart, D.R., and Long, J.M. 2016. Using hierarchical Bayesian multispecies mixture models to estimate tandem hoop-net-based habitat associations and detection probabilities of fishes in reservoirs. *Trans. Am. Fish. Soc.* 145(3): 450–461.
- Stewart, D.R., Butler, M.J., Harris, G., Johnson, L.A., and Radke, W.R. 2017. Estimating abundance of endangered fish by eliminating bias from non-constant detectability. *Endang. Spec. Res.* 32: 187–201.
- Tyre, A.J., Tenhumberg, B., Field, S.A., Niejalke, D., Parris, K., and Possingham, H.P. 2003. Improving precision and reducing bias in biological surveys: estimating false-negative error rates. *Ecol. Appl.* 13(6): 1790–1801.
- 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.
- Ward, R.J., Griffiths, R.A., Wilkinson, J.W., and Cornish, N. 2017. Optimising monitoring efforts for secretive snakes: a comparison of occupancy and N-mixture models for assessment of population status. *Scient. Rep.* 7: 18074.

Wilson, C., and Dextrase, A.J. 2008. Draft Sampling protocols for Redside Dace. Ontario Ministry of Natural Resources. 4 p.

Wintle, B.A., Walshe, T.V., Parris, K.M., and McCarthy, M.A. 2012. Designing occupancy surveys and interpreting non-detection when observations are imperfect. *Divers. Distrib.* 18(4): 417–424.

Yoccoz, N.G., Nichols, J.D., and Boulinier, T. 2001. Monitoring of biological diversity space and time. *Trends Ecol. Evol.* 16(8): 446–453.