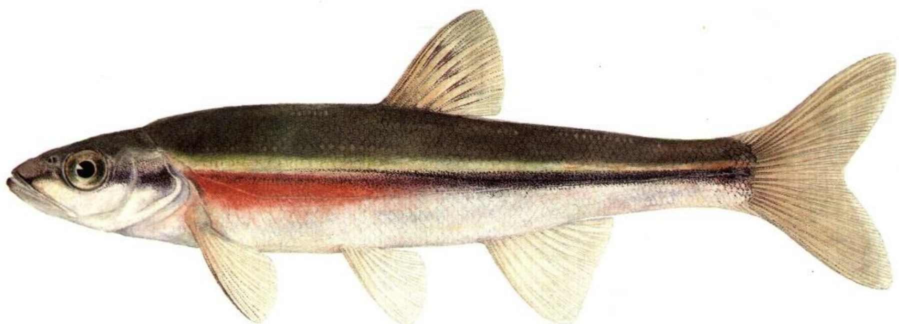
Fisheries and Oceans Canada / Pêches et Océans Canada
Ecosystems and Oceans Science / Sciences des écosystèmes et des océans

Central and Arctic Region

Canadian Science Advisory Secretariat
Science Advisory Report 2023/021

SCIENTIFIC ADVICE ON THE DESIGN OF A COMPREHENSIVE LONG-TERM MONITORING PROGRAM FOR REDSIDE DACE (*CLINOSTOMUS ELONGATUS*) TO INFORM RECOVERY AND MANAGEMENT DECISIONS



*Redside Dace (*Clinostomus elongatus*). Illustration by © Ellen Edmonson, NYSDEC*

Context:

*The Ontario Recovery Strategy for Redside Dace (*Clinostomus elongatus*) identified monitoring actions that, if implemented, would help to evaluate the persistence of Redside Dace and its habitat (Redside Dace Recovery Team 2010). Similarly, the federal Recovery Strategy and Recovery Potential Assessment for Redside Dace (DFO 2019) indicated that long-term monitoring to inform abundance and distribution, the status of habitat, and potential threats would benefit species recovery. Species and habitat information obtained from monitoring could provide baseline information about the occurrence or abundance of Redside Dace at multiple spatial scales, including in areas subject to development impacts or recovery actions, thereby informing decision-making by DFO's Species at Risk and Fish and Fish Habitat Protection programs. Although Redside Dace is detected by several agencies during the course of fish community monitoring and other targeted sampling, comprehensive efforts to conduct standardized, range-wide monitoring have not occurred. One factor that has prevented implementation of range-wide monitoring efforts is uncertainty about the most appropriate monitoring program objective(s) and related aspects of program design. Science advice about monitoring program design is needed to inform the implementation of Redside Dace monitoring efforts in Canada, thereby contributing necessary actions outlined in provincial and federal recovery strategies.*

*This Science Advisory Report is from the February 4th, 2020 regional peer review meeting on Scientific Advice on the Design of a Comprehensive Long-term Monitoring Program for Redside Dace (*Clinostomus elongatus*) to Inform Recover and Management Decisions. Additional publications from this meeting will be posted on the [Fisheries and Oceans Canada (DFO) Science Advisory Schedule]() as they become available.*

SUMMARY

- Redside Dace, a species listed as Endangered under the *Species at Risk Act*, has experienced severe declines throughout its Canadian range over the past 30 years. Federal and provincial recovery strategies indicate that the development of a long-term monitoring program to inform recovery and management decisions is a high priority recovery action.
- Distribution- and abundance-based indicators should be chosen to allow Redside Dace to be assessed relative to management objectives. Failure to clearly specify monitoring program objectives can lead to poor study design and an inability to understand the conservation status of the species or the influence of threats and recovery measures. The ability to detect changes through time is contingent on the application of a standardized monitoring approach.
- Monitoring design can include several spatial scales (site-level, sub-watershed, population, Canadian range). The choice of sampling scale is dependent on management objectives.
- Measuring the distribution and abundance of Redside Dace can be biased by imperfect detection, which is the failure to detect the species despite its occurrence. Field sampling design based on repeated surveys, and related modelling approaches, exist to account for imperfect detection. Addressing imperfect detection will improve upon previous guidance for monitoring Redside Dace.
- The ability to detect changes in distribution (occupancy) or trends through time is contingent on sampling efficiency, the occurrence of the species, the number of sampling sites, and the frequency of sampling. Many sites are required to detect small changes in occupancy; whereas, fewer sites are needed to detect large changes. Improved sampling efficiency will reduce effort requirements. Greater confidence in monitoring results will require increased sampling effort.
- Several gears exist to detect Redside Dace. Improvements to sampling design advice will require further evaluation of detection probability and harm imposed by each gear.

INTRODUCTION

Redside Dace (*Clinostomus elongatus*) is a colourful minnow found in slow-flowing riffle-pool sections of small creeks and streams. It is often found near overhanging riparian vegetation and shows a preference for middle water column positions in the deepest parts of pools (McKee and Parker 1982, Novinger and Coon 2000). There are 17 locations in Canada where Redside Dace is extant and nine locations where Redside Dace is considered extirpated (COSEWIC 2007). 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.

As a result of recent declines in abundance and distribution of Redside Dace and ongoing threats, the species has been assessed as Endangered in the province of Ontario and in Canada (COSEWIC 2017), and was listed as Endangered under Schedule 1 of the *Species at Risk Act* (DFO 2019). The status of each extant population has been assessed by DFO (2019) from an analysis of relative abundance and population trajectories 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). Declines in population abundance and distribution have been attributed to agricultural practices and urban development activities; 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 development and implementation of a long-term monitoring program to characterize spatial and temporal changes in the abundance and distribution of the species and its habitats (e.g., Wilson and Dextrase 2008). Long-term monitoring programs can inform management activities such as the review of development and in-stream work proposals, population status assessments, and the planning of restoration projects. To date, monitoring of Redside Dace has been completed through targeted sampling efforts and watershed-level fish community sampling by several agencies in Ontario; a database of distribution records has been compiled to ensure that this information is available for appropriate planning authorities.

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 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 two-tiered monitoring program, if implemented, 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, statistical evaluation of the proposed two-tiered framework to monitor the abundance and distribution of Redside Dace has not occurred, and needs to be addressed prior to widespread implementation.

An increasing number of sources identify the need to consider imperfect detection when monitoring rare species (Dextrase et al. 2014, MacKenzie et al. 2018, Lamothe et al. 2019a). Imperfect detection describes the situation when a species is undetected during sampling despite being present (MacKenzie et al. 2002, 2018) and is a common issue when trying to estimate species distribution and abundance, particularly for imperilled species. Imperilled species often have restricted distributions across the landscape or exist at very low population densities. As a result, these species can easily be missed when sampling. To account for imperfect detection when sampling the landscape, a repeat-survey design is frequently recommended to allow the calculation of site-specific species detection probabilities (p - probability that a species is detected at a site in an individual survey given that it is present) to inform estimates of species occupancy (percent of sites occupied by a species while accounting for imperfect detection) using multinomial likelihood occupancy models (Mackenzie et al. 2002, 2018). Furthermore, extensions of occupancy models (i.e., *N*-mixture models) can be used to estimate species abundance across sites using spatially and temporally replicated count data (Royle and Nichols 2003, Royle 2004).

This document reports the conclusions and advice from the Canadian Science Advisory Secretariat (CSAS) peer-review meeting, held in Burlington, Ontario on February 4th, 2020. It summarizes the research by Lamothe et al. (2023), which identifies potential objectives and assessment variables for a Redside Dace monitoring program designed to inform species occupancy and (or) abundance, with a focus on the use of repeat-surveys. Considerations around the allocation of effort for a Redside Dace monitoring program were 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.

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 key objectives and the assessment variables used to inform the objectives. 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 (Nichols and Williams 2006, Sauer and Knutson 2008). 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. Therefore, the potential objectives of a Redside Dace monitoring program 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 1). 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.

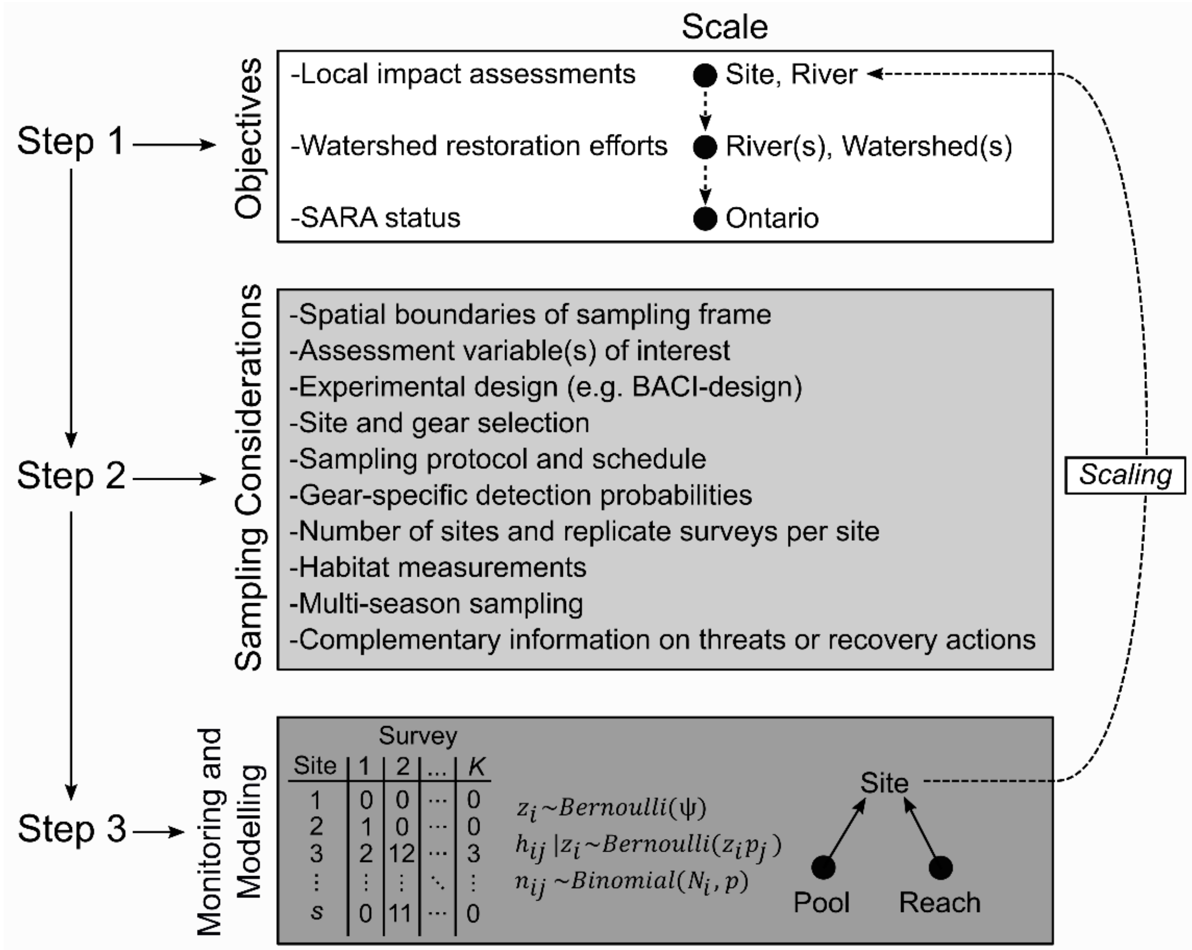


Figure 1. Initial considerations when developing a species monitoring program. Step 1 includes identifying the particular objective, including the scale at which the objective is to be addressed. Considerations for sampling are identified in Step 2 after the initial objective and scale are described.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

The use of occupancy as an assessment variable has a variety of benefits (MacKenzie et al. 2002, 2018) that have been demonstrated across taxa (e.g., Chen et al. 2013, Miller and Grant 2015), including imperilled freshwater fishes experiencing range reductions (Dextrase et al. 2014, Lamothe et al. 2020). Using a repeat-survey design, the probability of detecting Redside Dace can be calculated and modelled as a function of site and survey specific habitat measurements. An occupancy-based approach for monitoring Redside Dace: (i) directly supports the draft federal recovery strategy to monitor presence and absence at large spatial scales; (ii) will require less sampling effort relative to abundance-based monitoring; (iii) can be used to inform trends in Redside Dace distribution over time by directly linking to AO (area of occupancy) or EO (extent of occurrence); and (iv) 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 estimates of the minimum area for a viable population (MAPV) generated from population matrix models (e.g., van der Lee et al. 2020), allowing the probability of one or multiple populations being above or below MAPV as an outcome of program design. However, it should be noted that no single assessment variable is faultless.

Many studies have used estimates of occupancy 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), 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, estimates of occupancy remain a promising, and cost effective (Joseph et al. 2006) approach for evaluating the distribution and conservation status of Redside Dace.

Although the primary focus of the research was to evaluate sampling effort required to measure changes in occupancy over time, thereby informing extensive sampling, a worked example of estimating local species abundance using *N*-mixture models was provided to inform intensive sampling, given the importance of abundance as an assessment variable (COSEWIC 2017). One benefit of using *N*-mixture models, which require a similar repeat-survey design, is the ability to generate site-specific abundance estimates and extrapolate those estimates to the population or national scale, depending on the sampling design (i.e., sample site selection, number of sample sites; Figure 1). Furthermore, extrapolated estimates can then be 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, *N*-mixture models can perform relatively poorly compared to occupancy models (Ward et al. 2017), and as a result, presence-absence 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).

ANALYSIS

Power Analysis

Prospective power analyses can help to ensure that monitoring program designs are likely to detect changes in assessment variables (i.e., occupancy probability) through time. The approach 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 distribution and abundance of Ontario fishes at risk. Power analysis was used to illustrate how different sampling strategies influence the ability of monitoring program designs to identify changes in Redside Dace distribution.

A maximum-likelihood approach for assessing the power to detect differences in occupancy (ψ) between two points in space or time that assumes a standard repeat sampling design with K repeat surveys (e.g., seine hauls) at S survey sites was developed by Guillera-Arroita 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 the chosen significance level (α), statistical power ($1 - \beta$), 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 1), 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.

Table 1. 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; [3] = Reid et al. 2009; [4] = Poesch et al. 2012; [5] = Poos and Jackson 2012; [6] = Castañeda et al. 2020

The power to detect proportional reductions (30% - solid lines; 50% - dotted lines) in occupancy is a function of S given that ψ_1= 0.4, 0.5, 0.6, or 0.7, p = 0.4, 0.5, 0.6, 0.7, 0.8, 0.9, or 1.0, K = 3, 4, 5, 6, 7, 8, 9, or 10, and α = 0.05 is plotted in Figure 2. As ψ_1, p, K, and proportional reductions in occupancy are reduced, S increases nonlinearly (Figure 2). 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 2). 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 2). 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. Results indicate that the number of sites S needed to detect proportional reductions in Redside Dace ψ is reduced if α is increased from 0.05 to 0.20 (Figure 3). In such a case, the researcher accepts a 20% probability of concluding that a proportional reduction in occupancy has occurred when, in fact, no reduction has occurred. For example, to detect a proportional reduction of 30% in ψ 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 $\alpha = 0.05$. 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%.

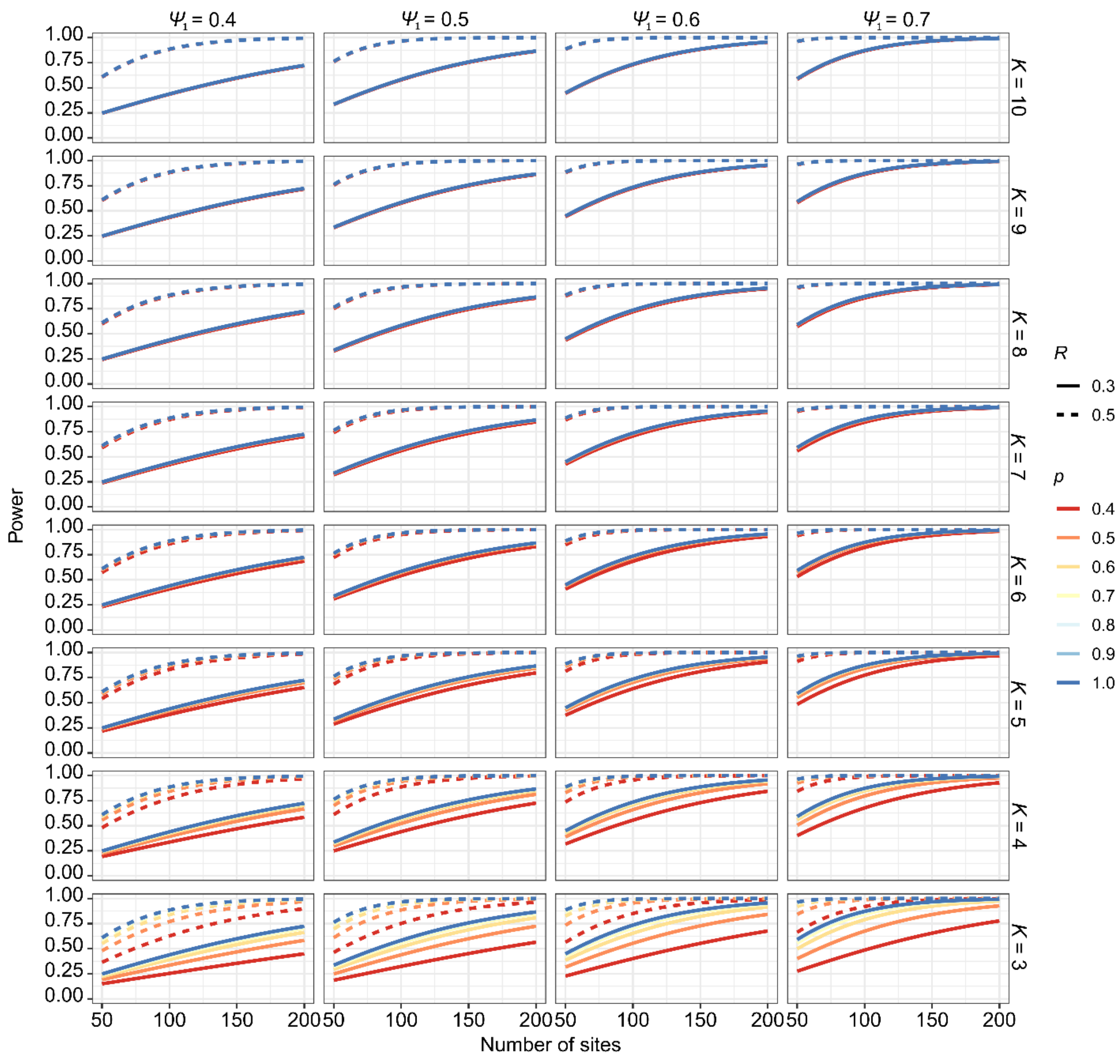


Figure 2. Power to detect proportional reductions (R) in Redside Dace occupancy probabilities of 0.5 (dotted lines) and 0.3 (solid lines) across various detection probability (p) thresholds (0.4–1.0; colors), where initial occupancy probabilities (ψ_1) range between 0.4 and 0.7, the number of surveys per site (K) ranges from 3–10, and α = 0.05.

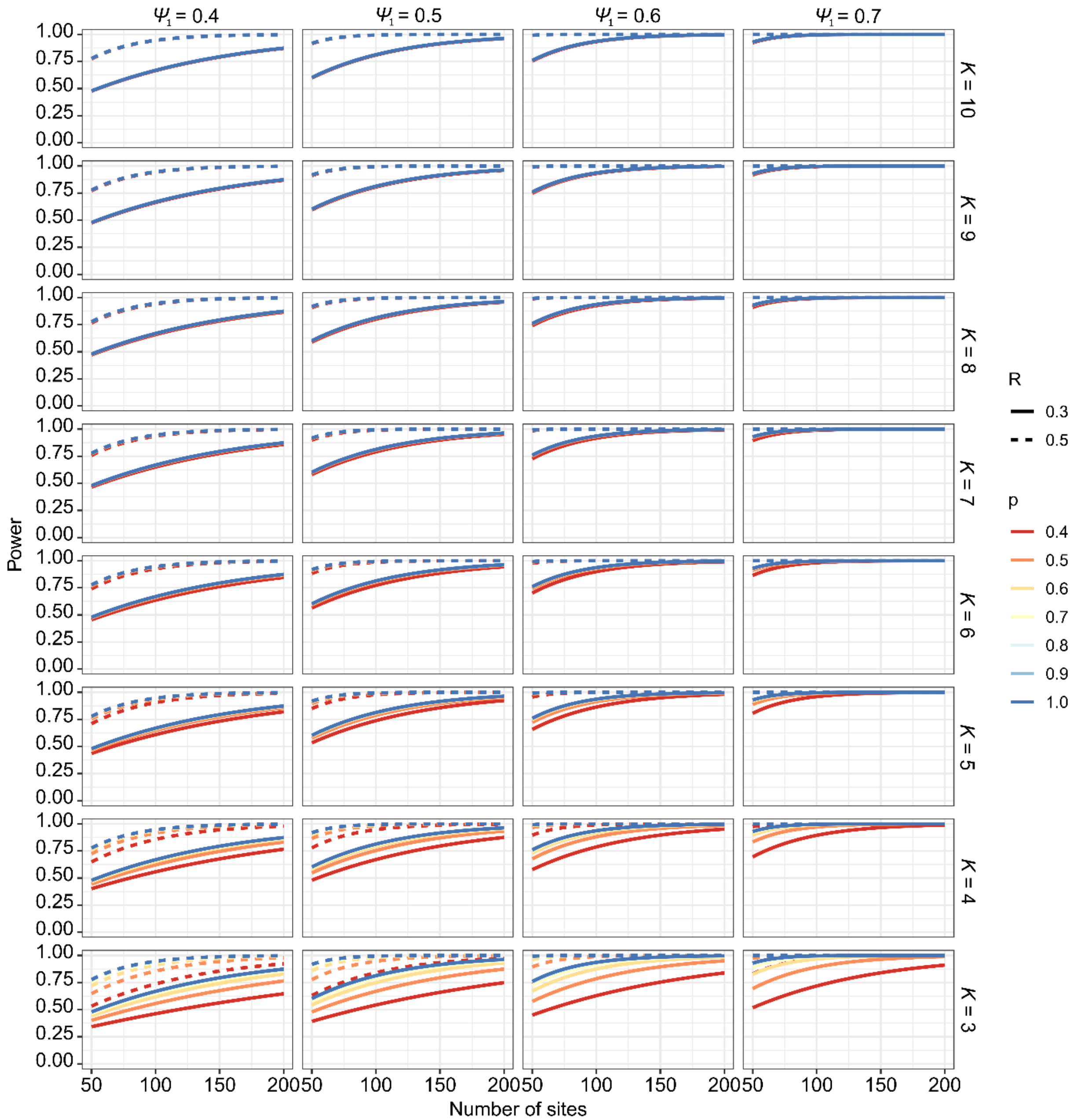


Figure 3. Power to detect proportional reductions (R) in Redside Dace occupancy probabilities of 0.5 (dotted lines) and 0.3 (solid lines) across various detection probability (p) thresholds (0.4–1.0; colors), where initial occupancy probabilities (ψ_1) range between 0.4 and 0.7 and the number of surveys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

Based on the results presented in Figure 4, small reductions in ψ require large numbers of sampling sites, particularly when p and ψ are reduced; 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 1), gear choice is a less important factor than S and the desired proportional reduction in ψ.

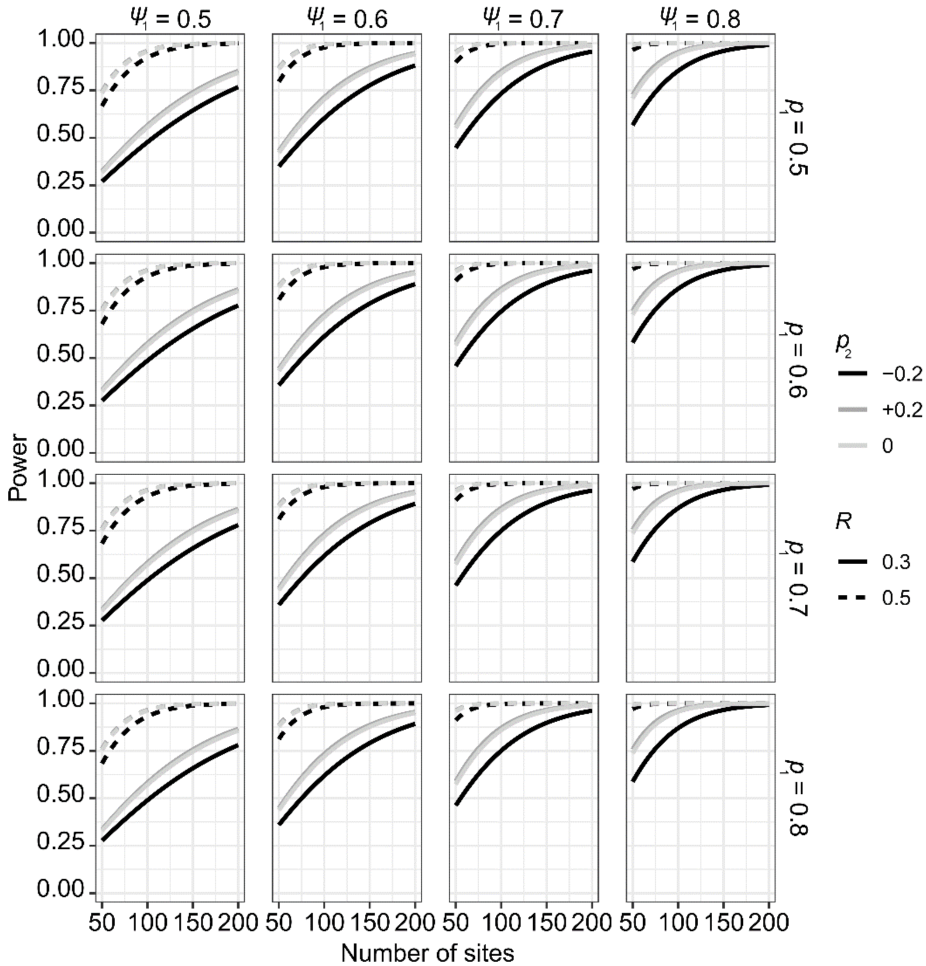


Figure 4. Power to detect proportional reductions (R) in Redside Dace occupancy probabilities of 0.5 (dotted lines) and 0.3 (solid lines) across various initial occupancy (ψ_1; columns) and detection (p_1; rows) probability thresholds (0.5–0.8), where K = 5, α = 0.05, and detection probabilities are constant (black), increase by 0.2 (dark grey), or decrease by 0.2 (light grey) over time (p_2).

The minimum number of repeated surveys required to reliably detect the species can be back-calculated based on the probability that the site is occupied even though the species was not detected and relies on site-specific estimates of occupancy and detection (Wintle et al. 2012). 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 5). As ψ increases, more repeat samples are needed to be sure that Redside Dace is absent (Figure 5). Note, however, that Redside Dace absence during sampling does not imply that Redside Dace does not use the habitat; but rather, that Redside Dace was not present at time of sampling. Multi-season sampling would be needed to confirm lack of use.

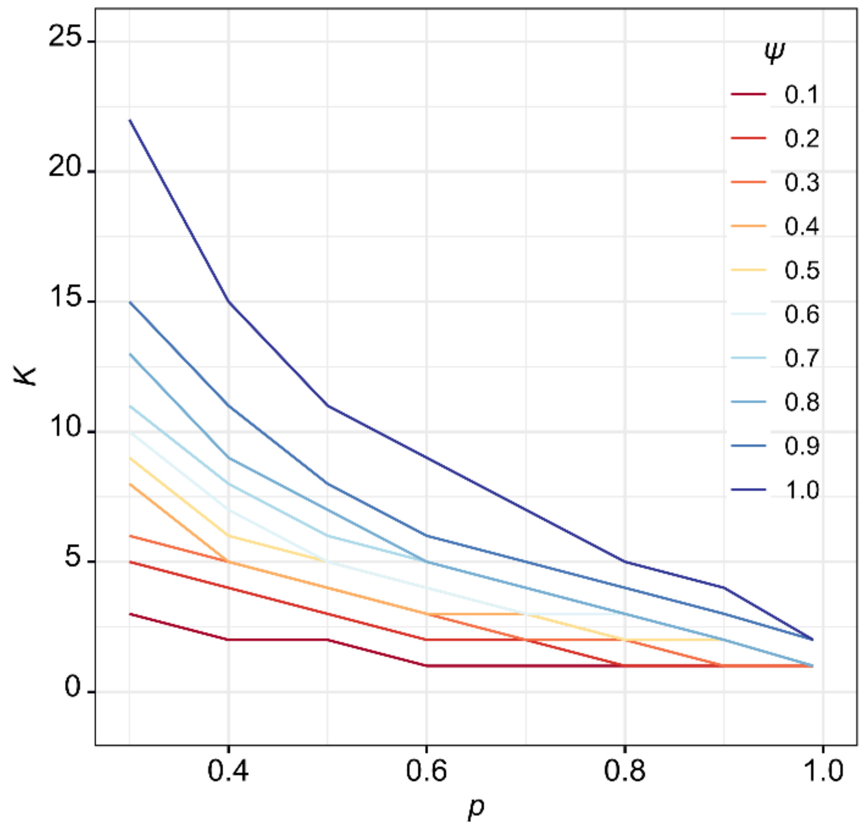


Figure 5. Minimum number of repeated surveys (K') required to detect Redside Dace at α = 0.05 based on the probability that the site is occupied even though the species was not detected and relies on site-specific estimates of occupancy (ψ'; colours) across differing detection probabilities (p' = 0.3–0.7).

Considerations around the allocation of effort for a Redside Dace monitoring program

Based on the current knowledge of Redside Dace in Canada, both the allocation of effort and sampling unit were considered to inform a statistically rigorous, occupancy based monitoring program; specifically: (i) how site definition (i.e., pool, reach, or multi-unit) 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.

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 sampling effort in the dominant microhabitat for Redside Dace, potentially improving the probability of Redside Dace occupancy and detection, and restricting the amount of habitat (m^2) needed to sample. However, defining the pool as the unit of sampling neglects other microhabitats where Redside Dace may occur (e.g., within runs or riffles). Pool-focused sampling for Redside Dace in southern 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).

In 2019, Fisheries and Oceans Canada conducted pool-specific sampling in the main stem of Gully Creek, a tributary of Lake Huron, to confirm the presence of a previously documented Redside Dace population and evaluate Redside Dace habitat (Gáspárdy and Drake 2021). The sampling frame consisted of a single Aquatic Ecosystem Classification (AEC) stream segment (Melles et al. 2013), which contained historical records of Redside Dace. Sampling was performed non-randomly, whereby the sampling crew aimed to increase the geographic area sampled within the segment while targeting pool habitats containing relatively little woody debris (Gáspárdy and Drake 2021). In total, fishes were sampled from 16 pools (S = 16), along with measurements of pool length (m), width (m), and depth (m).

Based on the 2019 data, preliminary estimates of detection and occupancy probability in the main stem of Gully Creek were calculated. Of the 16 sampled pools, Redside Dace was captured in 11, indicating a naïve occupancy probability that ignores imperfect detection 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, based on a relatively small sample size and assuming a completely random sample (which it was not), approximately 73% of pool habitat in the AEC segment of Gully Creek is occupied by Redside Dace.

Based on a constant detection probability of p = 0.606, ψ_1 = 0.732, K = 3, S = 16, and α = 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 effort) would be required to demonstrate 30% reductions in ψ. 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, this sampling design provides reasonable power to identify a 50% and (or) 70% reduction in ψ, estimated at 0.82 and 1.00, respectively, indicating that large (> 50%) changes in occupancy can be identified with sufficient statistical power with a relatively small number of sites.

It is reasonable to hypothesize that pool size (e.g., pool length, width, area, or volume) 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 were incorporated into the single-species occupancy models, however model selection using Akaike's Information Criteria (AIC) indicated that the best model for the data was an intercept-only model (i.e., no habitat covariates).

Using the collected habitat data, an estimate of the total number of pools available for Redside Dace in the Gully Creek AEC segment was made, and subsequently, an estimate of the number of pools occupied by Redside Dace. However, these data consist of few sample sites and were not collected to reflect the entirety of Gully Creek and therefore do not reflect the entirety of the system. The results of the extrapolation 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 Gully Creek AEC segment.

The mean pool width from S = 16 pools was 4.58 m ± 0.08 SE, mean pool length was 15.61 m ± 0.44 SE, and the mean sampled reach length (S = 4) was 33.54 m ± 2.76 SE. The main stem of Gully Creek is approximately 5 km long. If approximately 47% of the main stem of Gully Creek is composed of pool habitat (15.61 m / 33.54 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 m / 15.61 m = 150.51). Based on the mean width of sampled pools, 151 pools would equal approximately 10,763 m^2 of pool habitat (2350 m * 4.58 m = 10,763 m^2). Alternatively, if the subset of sampled pools during 2019 sampling were considered to be overly large, and estimates on mean pool length minus 1 SD were used (i.e., $\bar{x}$ – 1 SD = 15.61 – 7.00 = 8.61), then there would be approximately 273 total pools (2350 m / 8.61 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). Based on estimated total pool area in the main stem of Gully Creek, the total area of habitat is less than MAPV published in the Recovery Potential Assessment of Redside Dace in Canada (van der Lee et al. 2020). This comparison to MAPV is not rigorous and is shown only as an approach for extrapolation if future field data is collected in a manner consistent with this goal in mind.

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). As well, reach-specific sampling aligns with OSAP (Ontario Stream Assessment Protocol; Stanfield 2017), 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 monitoring, sampling stream reaches will require more time at each sampling unit, therefore reducing the total number of sites 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. At each site, standard OSAP was followed, where fishes were sampled in an upstream manner along 40+ m reaches using a backpack electrofisher. Mean electrofishing effort was 7.7 $s \cdot m^{-2}$. Three (n = 35) or four (n = 5) repeated passes were performed at each site. Redside Dace was captured at 20 sites, 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 mean ψ was estimated at 0.503 ± 0.08 SE. Based on a constant p = 0.822, ψ_1 = 0.503, K = 3, S = 40, and α = 0.05, the power to identify a 30% reduction in ψ with these data is 0.28. 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 to estimate species detection probability and species occupancy is whether Redside Dace sites will be targeted or sampled randomly. 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 extreme cases, the area of stream habitat occupied by Redside Dace sub-populations is so small that it prevents the use of random site selection. Targeting sites to sample Redside Dace limits the ability to extrapolate to other populations and, instead, concentrates on 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.

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 is required for logistical reasons (e.g., crew size, distance between populations, resources) to monitor Canadian populations of Redside Dace. As well, stratification of Redside Dace populations over time: (i) 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); (ii) enables researchers the ability to extrapolate research findings to representative populations given the proper sample site selection; and, (iii) enables investigations regarding how two broad, regional-scale threats (e.g., agricultural practices, urbanization) may affect the persistence of Redside Dace populations differentially in Canada.

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 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 1), 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, or sites that are the subject of 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 because it prevents meaningful inference between the two time periods.

Extensions to the Single-Season Single-Species Occupancy Models

Although occupancy modelling is a powerful approach to evaluate distribution-based monitoring objectives such as AO, EO, and MAPV, estimating species abundance (N) and characterizing changes in N over time can directly inform the likelihood that Redside Dace populations remain above MVP. Abundance-based approaches also satisfy intensive sampling, as outlined in the provincial (RDRT 2010) and draft federal (Amy Boyko, DFO Species at Risk Program, pers. comm.) recovery strategies. Several approaches have been described to estimate N of fishes including mark-recapture methods and depletion methods (Carle and Strub 1978, Pollock et al.

1990, Ogle 2016). Unfortunately, conventional approaches for estimating species abundance, including mark-recapture and depletion methods, have yielded poor results for small-bodied fishes at risk in Ontario owing to low depletion and re-capture rates. 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), data that could be recorded if a repeat-survey design was used to sample Redside Dace. Compared to conventional approaches for estimating abundance, *N*-mixture models may be particularly well-suited for estimating Redside Dace abundance given the: (i) implicit incorporation of detection probability differences across sites within the model; (ii) the reduced effort requirements compared to traditional approaches, such as mark-recapture; and, (iii) the reduced risk of failure (and physical harm) given that marking and recapturing individuals is not required.

Pool-Specific Abundance in Gully Creek

Using Royle's (2004) *N*-mixture model, the number of Redside Dace in the sampled Gully Creek AEC segment can be estimated while incorporating p. Similar to single-season occupancy models, the effects of pool length, width, area, volume, and the additive effects of length and width were considered on p and λ. A negative-binomial distribution for the latent abundance distribution (i.e., θ) was chosen after initial tests of model fit and 75 individuals were used as the upper index of integration.

The best *N*-mixture model included pool width as a p covariate and an intercept model for θ, providing an average of 9.30 ± 3.69 SE Redside Dace per site. Therefore, 1,404 individuals were estimated to occupy the main stem of Gully Creek if it contains 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 Redside Dace abundance in the main stem would be estimated at approximately 2,539 individuals (273 * 9.30). The population abundance value could then 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 mainstem that diverges in the headwaters, where Redside Dace individuals have been captured (ABCA 2010).

Sources of Uncertainty

Occupancy monitoring has recently been used as a surrogate for abundance monitoring in ecology (Steenweg et al. 2018). Much of the literature, however, has reported positive abundance–occupancy relationships but mainly for terrestrial species. Often, this relationship is nonlinear and differs across spatial scales (Gaston et al. 2000, He and Gaston 2000, Steenweg et al. 2018). Earlier studies had shown no correlation between abundance and occupancy for freshwater fishes (Gaston and Lawton 1990, Pyron 1999). More recently, it has been shown that occupancy in freshwater stream fishes is positively related to abundance (Faulks et al. 2015, Miranda and Killgore 2019). However, for low-occupancy species such as Redside Dace, the factors that affect population size may be site-specific and could vary from one location to another (Miranda and Killgore 2019).

Much of the uncertainty in this research pertains to the non-random manner in which field data were collected. For example, estimates of detection probabilities in this study likely portray the best-case scenarios for Redside Dace as samples were not collected randomly in the field. Future declines in species abundance or sampling for the species within marginal habitat will result in lower values of detection probability relative to those presented in Table 1, re-emphasizing the importance of accounting for imperfect detection in program design. Furthermore, an estimate of the number of pools in the sampled Gully Creek AEC segment was used to extrapolate the number of pools in the main stem of Gully Creek. These data were not collected to reflect the entirety of Gully Creek and therefore should only be interpreted as an approach for extrapolation of the Gully Creek AEC segment.

CONCLUSION

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, and across strata to inform the conservation status of the species. Using Gully Creek as a case study, results confirm that imperfect detection is an important consideration when estimating the occupancy and abundance of Redside Dace and, therefore, will be equally important when evaluating future changes in distribution and abundance over time.

Although this study does not describe the considerations for implementing a broad-scale Redside Dace monitoring program (e.g., allocating sampling resources given logistical 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) 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. An understanding of the direct physical harm to individual Redside Dace (and associated population-level harm) caused by different scientific sampling techniques has yet to be fully quantified, but is an additional consideration prior to implementation.

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

LIST OF MEETING PARTICIPANTS

Name	Organization/Affiliation
Adam van der Lee	DFO – Science, Ontario and Prairie Region
Andrew Drake	DFO – Science, Ontario and Prairie Region
Dave Andrews (Chair/Rapporteur)	DFO – Science, Ontario and Prairie Region
Karl Lamothe	DFO – Science, Ontario and Prairie Region
Marten Koops	DFO – Science, Ontario and Prairie Region
Rowshyra Castañeda	DFO – Science, Ontario and Prairie Region
Tessa Brinklow	DFO – Science, Ontario and Prairie Region
Rick Kiriluk	DFO – Fisheries Protection Program, Ontario and Prairie Region
Amy Boyko	DFO – Species at Risk Management, Ontario and Prairie Region
Andrea Dunn	Halton Conservation Authority
Chris Wilson	Ministry of Natural Resources and Forestry
Cindy Chu	Ministry of Natural Resources and Forestry
Nick Jones	Ministry of Natural Resources and Forestry
Scott Reid	Ministry of Natural Resources and Forestry
Alan Dextrase	No affiliation
Mark Heaton	No affiliation
Dave Lawrie	Toronto and Region Conservation Authority
Don Jackson	University of Toronto

SOURCES OF INFORMATION

This Science Advisory Report is from the February 4th, 2020 regional peer review meeting on Scientific Advice on the Design of a Comprehensive Long-term Monitoring Program for Redside Dace (*Clinostomus Elongatus*) to Inform Recovery Management Decisions. Additional publications from this meeting will be posted on the Fisheries and Oceans Canada (DFO) Science Advisory Schedule as they become available.

Ausable Bayfield Conservation Authority (ABCA). 2010. Habitat assessment for Redside Dace in Gully Creek. Ausable Bayfield Conservation Authority. Exeter, ON. 15 p.

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.

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.

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.

Dextrase, A.J., Mandrak, N.E., and Schaefer, J.A. 2014. Modelling occupancy of an imperilled stream fish at multiple scales while accounting for imperfect detection: implications for conservation. Freshwater Biol. 59(9): 1799–1815.

DFO. 2019. Recovery Potential Assessment of Redside Dace (*Clinostomus elongatus*) in Canada. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2019/012.

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.

Faulks, L., Svanbäck, R., Ragnarsson-Stabo, H., Eklöv, P., and Östman, Ö. 2015. Intraspecific niche variation drives abundance–occupancy relationships in freshwater fish communities. Am. Nat. 186(2): 272–283.

Gáspárdy, R., and Drake, D.A.R. 2021. Targeted surveys for Redside Dace (*Clinostomus elongatus*) in Gully Creek, Ontario, 2019. Can. Dat. Rep. Fish. Aquat. Sci. 1315: vii + 48 p.

Gaston, K.J., and Lawton, J.H. 1990. The population ecology of rare species. J. Fish Biol. 37: 97–104.

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.

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.

He, F., and Gaston, K.J. 2000. Occupancy-abundance relationships and sampling scales. Ecography 23(4): 503–511

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.

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.

Lamothe, K.A., Reid, S.M., and Drake, D.A.R. 2023. Considerations around effort and power for an occupancy-based Redside Dace (*Clinostomus elongatus*) monitoring program across spatial and temporal scales. DFO Can. Sci. Advis. Sec. Res. Doc. 2023/034. iv + 37 p.

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.

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.

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.

Miranda, L.E., and Killgore, K.J. 2019. Abundance–occupancy patterns in a riverine fish assemblage. Freshw. Biol. 64(12): 2221–2233.

Nichols, J.D., and Williams, B.K. 2006. Monitoring for conservation. Trends Ecol. Evol. 21(12): 668–673.

Novinger, D.C., and Coon, T.H. 2000. Behaviour and physiology of the redside 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.

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, redside dace (*Clinostomus elongatus*), in Canada. Aquat. Conserv. Mar. Freshwater Ecosyst. 22(1): 47–57.

Pyron, M. 1999. Relationships between geographical range size, body size, local abundance, and habitat breadth in North American suckers and sunfishes. J. Biogeogr. 26(3): 549–558.

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.

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.

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.

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.

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. Ecol. 99(1): 172–183.

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.

THIS REPORT IS AVAILABLE FROM THE:

Center for Science Advice (CSA)
Central and Arctic Region
Fisheries and Oceans Canada
501 University Crescent, Winnipeg, Manitoba, R3T 2N6

E-Mail: csas-sccs@dfo-mpo.gc.ca
Internet address: www.dfo-mpo.gc.ca/csas-sccs/

ISSN 1919-5087
ISBN 978-0-660-48531-7 N° cat. Fs70-6/2023-021E-PDF

Correct Citation for this Publication:

DFO. 2023. Scientific advice on the design of a comprehensive long-term monitoring program for Redside Dace (*Clinostomus elongatus*) to inform recovery and management decisions. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2023/021.

Aussi disponible en français :

*MPO. 2023. Avis Scientifique sur la conception d’un programme exhaustif á long terme de surveillance du méné long (*Clinostomus elongatus*) afin d’éclairer les décisions en matière de rétablissement et de gestion. Secr. can. des avis sci. du MPO. Avis sci. 2023/021.*