STATUS OF NORTHERN PEARL DACE AND CHROSOMID

DACE IN PRAIRIE STREAMS OF MONTANA

by

Allison Louise Stringer

A thesis submitted in partial fulfillment
of the requirements for the degree

of

Master of Science

in

Fish and Wildlife Management

MONTANA STATE UNIVERSITY
Bozeman, Montana

May 2018

ACKNOWLEDGMENTS

TABLE OF CONTENTS

iv

TABLE OF CONTENTS CONTINUED

v

LIST OF TABLES

LIST OF TABLES CONTINUED

LIST OF FIGURES

LIST OF FIGURES CONTINUED

LIST OF FIGURES CONTINUED

LIST OF FIGURES CONTINUED

ABSTRACT

Non-native Northern Pike *Esox lucius* are predators that negatively affect native fish assemblages, possibly including those in Montana prairie streams, where their effects had not been investigated heretofore. I compared fish assemblages of prairie streams with and without Northern Pike and other non-native predators, with a focus on three species of concern that are probably particularly susceptible to predation (Northern Pearl Dace *Margariscus nachtriebi* (hereafter pearl dace), Northern Redbelly Dace *Chrosomus eos*, and Northern Redbelly Dace × Finescale Dace hybrids *C. eos* × *C. neogaeus* [hereafter hybrid dace]). I documented fish assemblages at 140 sites across the historical distribution of Northern Redbelly Dace and hybrid dace (hereafter collectively referred to as chrosomid dace), including 88 sites in the historical distribution of pearl dace. I estimated percent declines in distribution by comparing the number of currently occupied historical streams with the total number of historical streams and then determined if co-occurrence of pearl dace or chrosomid dace with non-native predators was different than predicted by chance. I augmented my dataset with fish collections from 5 additional sources and evaluated whether sites with and without Northern Pike differed in native species richness (with a Poisson regression) or assemblage composition (with a discriminant function analysis). Pearl dace distribution declined 63.3 to 83.3%, and chrosomid dace distribution declined 32.0% to 67.2%, depending on how declines were calculated. Pearl dace almost never co-occurred with Northern Pike or non-native trout and chrosomid dace rarely co-occurred with them. Native minnow species richness was 52% lower at sites with Northern Pike than at sites without Northern Pike. Predation probably caused the observed changes. Pearl dace are at extreme risk and chrosomid dace are at moderate risk of extirpation from Montana, and non-native predators appear to be the biggest threat to their continued persistence. Exclusion of Northern Pike from drainages where they have not yet invaded will afford fisheries managers the best chance of conserving native minnows in Montana prairie streams.

STATUS AND DISTRIBUTION OF NORTHERN PEARL DACE, NORTHERN

REDBELLY DACE, AND NORTHERN REDBELLY × FINESCALE DACE

HYBRIDS IN MONTANA

Introduction

Freshwater biodiversity is declining globally (Allan and Flecker 1993; Sala et al.

2000; Dudgeon et al. 2006; Vorosmarty et al. 2010). Habitat degradation (e.g., pollution,

fragmentation, flow modification, channelization), non-native species invasions, and

climate change threaten lotic biodiversity (Allan and Flecker 1993; Bertrand et al. 2009;

Ricciardi and MacIsaac 2011), and anthropogenic pressures on water resources are

growing exponentially with human population growth (Sabater et al. 2013). These

stressors threaten many North American fishes (Jelks et al. 2008), including numerous

Great Plains prairie fishes (Fausch and Bestgen 1997; Oakes et al. 2005).

The Great Plains are highly altered by agriculture and urbanization (Samson and

Knopf 1994), resulting in the fragmentation, pollution, and hydrological modification of

many prairie streams (Dodds et al 2004). Prairie fish assemblages are negatively affected

by these anthropogenic changes (Cross et al. 1985; Berkman and Rabeni 1987; Winston

et al. 1991; Oakes et al. 2005; Hoagstrom et al. 2011), despite being adapted to the

naturally harsh conditions of prairie streams (i.e., highly variable flow, temperature, and

water quality; Fausch and Bestgen 1997; Dodds et al. 2004). However, the status of, and

threats to, most non-game prairie fishes are understudied because management agencies

and researchers in the western Great Plains states (e.g., Montana) have traditionally focused on sport fishes (Quist et al. 2005).

Northern Pearl Dace *Margariscus nachtriebi* (hereafter pearl dace), Northern Redbelly Dace *Chrosomus eos*, and Northern Redbelly Dace × Finescale Dace hybrids *C. eos* × *C. neogaeus* (hereafter hybrid dace) are understudied nongame taxa that are potentially at risk of extirpation from Montana. Pearl dace are listed as imperiled or vulnerable in 8 of the 21 states and provinces where they occur, and Northern Redbelly Dace and hybrid dace (hereafter collectively referred to as chrosomid dace) are listed as imperiled or vulnerable in 11 of the 24 states and provinces where they occur (NatureServe 2011). In Montana, pearl dace and chrosomid dace are listed as species of concern, and pearl dace and hybrid dace are listed by the Bureau of Land Management as sensitive species. The current statuses of pearl dace and chrosomid dace are poorly documented in Montana (Brown 1971; Bramblett and Zale 2004; Bramblett 2010), but are better known elsewhere in the Great Plains (Bestgen 1989; Cunningham 1995, 2006; Stasiak 2006; Felts and Bertrand 2014).

Hybrid dace must co-occur with Northern Redbelly Dace or Finescale Dace because they reproduce clonally by gynogenesis (Goddard et al. 1998; Angers and Schlosser 2007; Mee et al. 2013). Hybrid dace do not originate from recent hybridization events, but rather from multiple hybridization events between female Finescale Dace and male Northern Redbelly Dace that took place in the Mississippi glacial refuge during the Pleistocene epoch (< 50,000 year ago; Angers and Schlosser 2007; Mee and Taylor 2012). Hybrid dace are female and produce diploid ova, which develop after stimulation

by Northern Redbelly Dace or Finescale Dace sperm. The sperm is usually discarded after development begins (Goddard et al. 1998), but the haploid sperm is sometimes incorporated into the ovum, resulting in a triploid or diploid-triploid mosaic offspring. Hybrid dace co-occur with only Northern Redbelly Dace in Montana because no confirmed collections of Finescale Dace have been made in the state (Brown 1971; Holton and Johnson 2003; Bramblett 2010; Mee and Taylor 2012).

The southwestern peripheries of the continental ranges of pearl dace and chrosomid dace occur in Montana. Pearl dace and chrosomid dace have similar continental distributions, occurring across Canada from eastern British Columbia to Nova Scotia and across the northern United States from Montana to Maine (Lee et al. 1980; NatureServe 2011). These continental distributions probably resulted from pearl dace and chrosomid dace colonizing the Hudson Bay basin from the Missouri basin via glacial meltwaters during late Pleistocene or early Holocene time (Hoagstrom and Berry 2006). Relict populations of pearl dace also occur in Wyoming, South Dakota, and Nebraska, and relict populations of chrosomid dace occur in Colorado, South Dakota, and Nebraska (Lee et al. 1980).

Pearl dace and chrosomid dace are rare in Great Plains streams (Cunningham 2006; Stasiak 2006; Bramblett 2010) because they prefer cool water temperatures, clear water, and abundant macrophytes (Bestgen 1989; Cunningham 1995, 2006; Stasiak 2006; Felts and Bertrand 2014). These conditions exist only in cool, groundwater influenced prairie streams reaches, which are relatively rare on the Great Plains. Such narrow habitat

requirements probably limit the distributions of pearl dace and chrosomid dace in Montana.

Pearl dace and chrosomid dace appear to be declining in Montana. Pearl dace were captured at only 7 of 1,673 sites during the Montana Prairie Stream Fish Survey conducted in 1999-2007 (Bramblett 2010) and appeared to have declined relative to historical collection records; pearl dace were not captured in 1999-2007 in any of 8 creeks in which they were collected historically. Chrosomid dace were also rare or absent in 1999-2007 from large areas where others had collected them historically (Brown 1971; Holton and Johnson 2003). Few or no pearl dace and chrosomid dace were collected in prairie streams since 2007 (MFISH 2016). However, the historical distributions of pearl dace and chrosomid dace in Montana have never been fully defined, inhibiting managers from comparing their historical and current distributions and determining statuses.

A probable contributing mechanism for the putative range contractions of pearl dace and chrosomid dace in Montana is the expansion of non-native Northern Pike *Esox lucius*. Northern Pike are not native to Montana outside of the Saskatchewan River basin (Brown 1971; Holton and Johnson 2003), but were widely introduced to provide recreational fishing opportunities (MFISH 2016). Northern Pike probably dispersed to prairie streams from stocked ponds or reservoirs, or from large rivers connected to stocked reservoirs such as the Missouri, Yellowstone, and Milk Rivers, where they are now widespread (MFISH 2016). Northern Pike were associated with low abundances or extirpations of native fishes in lentic (He and Kitchell 1990; Kitchell et al. 1994; Patankar et al. 2006; Byström et al. 2007; Haught and von Hippel 2011) and lotic systems (Rincon

et al. 1990; Sepulveda et al. 2015), including in two prairie drainages (Labbe and Fausch 2000; Spurgeon et al. 2014). Northern Pike, pearl dace, and chrosomid dace have similar habitat preferences (Scott and Crossman 1973), which probably enhance the likelihood of interactions between Northern Pike and these dace where barriers do not separate them. In fact, Northern Pike were associated with reductions of pearl dace and chrosomid dace in lakes in eastern North America (Findlay et al. 2000; Trumpickas et al. 2011; Nicholson et al. 2015). However, the extent to which the expansion of Northern Pike or other influential non-native predators may have reduced the current distribution of pearl dace and chrosomid dace in Montana has not been evaluated.

Given the uncertainties about pearl dace and chrosomid dace in Montana, my goal was to evaluate the status of, and potential threats posed by non-native predators to, these taxa in Montana to enhance their conservation. My objectives were to (1) estimate the probable historical distributions of pearl dace and chrosomid dace, (2) establish the current distributions of pearl dace and chrosomid dace, and (3) determine the influence of non-native predators on these distributions. The probable (inferred) historical distributions (Objective 1) had to be established before targeting sampling efforts addressing Objectives 2 and 3 could occur.

Inferred Historical Distributions

Data Sources

I assembled all known data and publications to infer the probable historical distributions of pearl dace and chrosomid dace in Montana and the Canadian portions of

the St. Mary, Milk, and Missouri River drainages (Figures 1 and 2). Data sources were

published literature, Montana State University (MSU) Vertebrate Museum records,

records from other fish museums (accessed through the Fishnet2 Portal,

www.fishnet2.net, 11/29/2017), gray literature, and data collected by natural resource

agencies and their collaborators (Tables 1 and 2). Not all sources were considered equally

reliable. I ranked collections in descending order of reliability as follows: (1)

Vouchered—collection records with voucher specimens; (2) Published—collection

records documented in published peer-review literature; and (3) Unpublished—collection

records referenced in unpublished reports or the Montana Fisheries Information System

(MFISH) database.

The MFISH was a publicly-available database that contained information about

fish collections, distributions, and stocking in Montana. Data in MFISH were collected

by Montana Fish, Wildlife and Parks, federal agencies, universities, non-profits, and

other organizations possessing a Montana scientific collectors permit. Most MFISH data

were based on fish collections; however, some were based on professional judgement. No

MFISH records based on professional judgement alone were used to inform the inferred

historical distributions of pearl dace and chrosomid dace in Montana.

Pearl Dace

I inferred historical presence or absence of pearl dace at the HUC 10 (10-digit

hydrologic unit code; USGS 2015) level. All HUC 10s with a vouchered or published

collection were included in the distribution. I examined HUC 10s that contained an

unpublished collection record, occurred in the next larger hydrologic unit (HUC 8, 8-digit

hydrologic unit code; USGS 2015) with a vouchered or published collection, or occurred in a HUC 8 directly upstream or downstream of a HUC 8 with a vouchered or published collection for likelihood of historical occurrence. I inferred that pearl dace probably occurred in streams connected to a waterbody with a historical population. Therefore, the HUC 10s containing a waterbody that was a direct tributary to, or direct recipient of, a stream with a historical collection record were included in the historical distribution.

I used recent fish collection data (post-1999) and headwater sources to infer habitat suitability for pearl dace in those HUC 10s included in published distribution maps (Holton and Johnson 2003; NatureServe 2011; Gidmark and Simons 2014; Montana Field Guide 2016) but not directly connected to streams with historical collections. These HUC 10s were typically under-sampled; however, limited collections were made in some of them recently. I included those HUC 10s that had recent collections of native fish associated with pearl dace (i.e., Iowa Darter, Northern Redbelly Dace, Brassy Minnow, Brook Stickleback; Stagliano 2005), which suggested that habitat for pearl dace probably existed in that HUC 10. The recent collections of native fish associated with pearl dace probably did not contain pearl dace because of their limited geographical scope, or because pearl dace had already been extirpated from that HUC 10 before the collection was taken. For the Beaver HUC 8, which contains Beaver Creek, a north-flowing tributary to the Milk River, the HUC 10s that had headwaters in the Little Rocky Mountains were included in the historical distribution but those HUC 10s that had headwaters in badlands were not included because pearl dace are usually found in cool,

8

clear, groundwater influenced stream reaches (Scott and Crossman 1973; Lee et al. 1980; Tallman and Gee 1982; Cunningham 1995, 2006; Stasiak 2006).

Drainage Delineation. I combined some HUC 8s to reflect meaningful drainage boundaries to facilitate display and elucidation of the inferred historical distribution of pearl dace in Montana (Figure 1, Table 1). All four HUC 8s in the Marias River basin were combined into one drainage. The Milk River and most of its tributaries were split into three main drainages: the upper Milk (headwaters to Fresno Reservoir; two HUC 8s), middle Milk (Fresno Reservoir to Beaver Creek confluence; five HUC 8s) and the lower Milk drainages (Beaver Creek confluence to Missouri confluence; three HUC 8s). The Lodge, Battle, and Frenchman HUC 8s were not combined with the middle Milk River drainage to isolate the locations of historical pearl dace collections in Canada. The Beaver HUC 8 was not combined with the lower Milk River drainage because only part of the HUC 8 was included in the inferred distribution. Two HUC 8s were combined in both the Poplar River drainage and the lower Missouri River section. The Big Muddy and St. Mary HUC 8s were large enough to clearly display the historical collections.

Inferred Historical Distribution. Pearl dace probably occurred in portions of the Saskatchewan, Milk, and Missouri River basins in Montana (Figure 1, Table 1). The inferred historical distribution included all portions of the drainages where pearl dace probably occurred, but the actual historical distribution was probably more patchy and limited to those portions of the basins with suitable habitat. Two published records of pearl dace in the lower Yellowstone River (Gould and Brown 1968; Brown 1971; Holton

and Johnson 2003) were misidentified Creek Chubs based on reexamination of voucher specimens (R. Bramblett, Montana State University, personal communication). Subsequent surveys have failed to find pearl dace in the Yellowstone River basin in Montana (Elser et al. 1980; Appendix Table TTT).

The St. Mary River drainage in the Saskatchewan River basin was included in the inferred historical distribution of pearl dace (Figure 1, Table 1). Some of the earliest records of pearl dace in Montana were from Glacier National Park in the St. Mary River drainage. Pearl dace were also collected in the Alberta portion of the drainage and more recently at the St. Mary Diversion and on the Blackfeet Reservation.

The Marias River drainage was included in the inferred historical distribution of pearl dace (Figure 1, Table 1). Pearl dace were collected in tributaries to the upper Marias River. Although no collections of pearl dace were recorded in tributaries to the lower Marias River, I inferred that pearl dace may have inhabited portions of the lower drainage historically. This inference is consistent with post-glacial dispersal of pearl dace from proglacial lakes (Hoagstrom and Berry 2006). Moreover, recent collections in the lower Marias River basin included native fish typically associated with pearl dace (Bramblett 2010; MFISH 2016).

Most of the Milk River drainage was included in the inferred historical distribution of pearl dace (Figure 1, Table 1). Pearl dace were collected in the headwaters of the Milk River on the Blackfeet Reservation, in tributaries to the middle and lower river, and in the main stem. Pearl dace were also collected in several Milk River tributaries in Canada, including Lodge Creek, Battle Creek, and the Frenchman River.

One unpublished record exists of pearl dace in Larb Creek (MFISH 2016), but the reliability of this record is low given that the headwaters of Larb Creek originate in badlands, and its turbidity is higher than ideal for pearl dace (Cunningham and Hickey 1996; Cunningham 2006; Stringer unpublished data). No other collections in or near Larb Creek included pearl dace. The Larb HUC 10 and other nearby drainages south of the lower Milk were therefore excluded from the inferred historical distribution.

South-flowing tributaries to the Missouri River upstream of its confluence with the Milk River were not included in the inferred historical distribution, despite a single record of pearl dace in Carpenter Creek (Figure 1, Table 1). The record is from a collection card in the MSU Vertebrate Museum; however, no corresponding voucher specimen exists, and other collections from Carpenter Creek did not include pearl dace (Bramblett 2010; Appendix Table TTT). Moreover, no records of pearl dace in adjacent drainages exist; a pearl dace population in Carpenter Creek would have been disjunct. Therefore, I excluded the Carpenter Creek HUC 10 and other nearby drainages from the inferred historical distribution.

South-flowing tributaries to the Missouri River downstream of its confluence with the Milk River were included in the inferred historical distribution of pearl dace (Figure 1, Table 1; Gould and Brown 1968). Pearl dace were collected in direct tributaries to the lower Missouri River section, tributaries to the Poplar River, tributaries to Big Muddy Creek, and in the main stem. I have high confidence that pearl dace occurred in these drainages historically because voucher specimens were associated with many of these collections.

Chrosomid Dace

Too few historical collections of hybrid dace existed in Montana to reliably infer the historical distribution of hybrid dace. However, the historical distribution of hybrid dace is constrained within the distribution of Northern Redbelly Dace because hybrid dace must co-occur with Northern Redbelly Dace for reproduction (Goddard et al. 1989). Therefore, I used historical collection records of Northern Redbelly Dace to infer the historical distributions of both chrosomid dace in Montana.

I inferred historical presence or absence of chrosomid dace at the HUC 8 level because they were historically more widespread than pearl dace and their potential distribution contained many HUC 10s with no historical collections. The same logic that was used to infer which drainages were included in the historical distribution of pearl dace was used to infer the historical distribution of chrosomid dace, but at the HUC 8 scale.

Drainage Delineation. I combined some HUC 8s to reflect meaningful drainage boundaries to facilitate display and elucidation of the inferred historical distribution of chrosomid dace in Montana (Figure 2, Table 2). The St. Mary, Marias, upper Milk, Frenchman, and Poplar drainages were made up of the same HUC 8s as in Figure 1. The middle Milk drainage was expanded to include the Lodge and Battle Creek HUC 8s (seven total HUC 8s), and the lower Milk drainage to include the Beaver HUC 8 (four total HUC 8s). The lower Missouri section was expanded to include the Redwater HUC 8 (three total HUC 8s), which was not included in the inferred historical distribution of pearl dace. Eleven additional drainages were inferred in the historical distribution of

chrosomid dace that were not inferred in the historical distribution of pearl dace. These included the Teton, Sun, Madison, Gallatin, Judith, upper Musselshell, middle Musselshell, and lower Yellowstone HUC 8s. The upper Missouri section was created out of four HUC 8s, the middle Missouri section out of three HUC 8s, and the lower Musselshell drainage out of three HUC 8s.

Inferred Historical Distribution. Chrosomid dace were historically widespread in Montana (Figure 2, Table 2). They probably occurred in the St. Mary, Missouri, lower Yellowstone, and Little Missouri River basins in Montana. Hybrid dace were probably similarly widespread throughout the state but may not have occurred in every drainage where Northern Redbelly Dace occurred. Although the inferred historical distribution included all portions of the drainages where chrosomid dace probably occurred, the actual historic distribution was probably more patchy and limited to those portions of the basins with suitable habitat

The St. Mary River drainage in the Saskatchewan River basin was included in the inferred historical distribution of chrosomid dace (Figure 2, Table 2). The oldest and most reliable collections from the drainage were taken in Canada. Chrosomid dace were collected from the main stem of the St. Mary River and from its tributaries. Chrosomid dace were also collected in the Montana portion of the St. Mary River basin on the Blackfeet Reservation in the 1990s and 2010s.

Most of the Missouri River drainage in Montana to the North Dakota border was included in the inferred historical distribution of chrosomid dace (Figure 2, Table 2). Chrosomid dace were collected in the upper, middle, and lower Missouri River sections.

Chrosomid dace were also collected in many tributary drainages to the Missouri, including the Sun, Teton, Marias, Judith, Poplar, and Big Muddy drainages. Some range maps did not show chrosomid dace occurring in the Madison or Gallatin River drainages (Brown 1971; Lee et al. 1980; Morey 2004), but chrosomid dace were collected in five locations in the Madison and Gallatin drainages in the 1980s and 1990s (Table 2). Other collections from these drainages did not include chrosomid dace. However, recent informal collections included chrosomid dace in Rocky Creek (R. Bramblett, Montana State University, personal communication) and its tributary, Kelly Creek (N. Clancy, Montana State University, personal communication), in the East Gallatin River drainage. Past sampling in these drainages probably failed to detect or report chrosomid dace because most sampling in the area was focused on trout. Targeted sampling of tributaries to the Gallatin, Madison, and Jefferson River drainages would clarify chrosomid dace distribution in this area.

The entire Milk River drainage in Montana was included in the inferred historical distribution of chrosomid dace (Figure 2, Table 2). Chrosomid dace were collected in tributaries in the upper, middle, and lower Milk sections, as well as in the main stem. Chrosomid dace were collected in the Canadian portion of the Frenchman River drainage, suggesting that chrosomid dace could have been present in the Montana portion of the Frenchman River drainage historically.

The entire Musselshell River drainage was included in the inferred historical distribution of chrosomid dace (Figure 2, Table 2). Chrosomid dace were collected in the

14

upper, middle, and lower Musselshell sections. Chrosomid dace were collected in the upper and middle main stem of the Musselshell River.

The lower Yellowstone River drainage was included in the inferred historical distribution of chrosomid dace in Montana (Figure 2, Table 2). Chrosomid dace were collected in multiple tributaries to the lower Yellowstone River. All chrosomid dace from the Yellowstone River drainage were collected from tributaries downstream of Terry, Montana.

The entire Little Missouri River drainage in Montana was included in the inferred historical distribution of chrosomid dace (Figure 2, Table 2) on the basis of only one collection of chrosomid dace taken from Little Beaver Creek. The full extent of the chrosomid dace distribution in the upper Little Missouri River and its tributaries is unknown because few historical samples were taken from the area. However, based on drainage basin topology, I inferred that chrosomid dace may have been present throughout the entire upper Little Missouri River basin in Montana historically.

Zoogeographical Context

The inferred historical distributions of pearl dace and chrosomid dace in Montana are consistent with the hypothesis that pearl dace and chrosomid dace dispersed from proglacial lakes after the Laurentide ice sheet receded (Davis 2004; Hoagstrom and Berry 2006). Chrosomid dace may have been better dispersers than pearl dace following deglaciation, because chrosomid dace were historically more widespread than pearl dace. Alternatively, climatic fluctuations in the mid to late Holocene Epoch may have resulted in extirpation of pearl dace from some drainages (i.e., Upper Missouri, Musselshell,

Lower Yellowstone, Little Missouri; Hoagstrom and Berry 2006) prior to modern fish collections.

Methods

Site Selection

I used the inferred historical distributions to define the sampling frames to determine the current distributions of pearl dace and chrosomid dace in Montana. I prioritized site selection based on the likelihood of pearl dace and chrosomid dace occurrence. I prioritized sites in the inferred historical distribution of pearl dace above those in the inferred historical distribution of chrosomid dace because pearl dace appeared to be more imperiled than chrosomid dace (and therefore of greater interest to conservation biologists), and because chrosomid dace historically occurred throughout the range of pearl dace. I prioritized sites as follows: (1) sites on historical pearl dace streams, (2) sites on non-historical pearl dace streams with potential pearl dace habitat within the inferred historical distribution of pearl dace, (3) sites on historical chrosomid dace streams, and (4) sites on non-historical chrosomid dace streams with potential chrosomid dace habitat within the inferred historical distribution of chrosomid dace but outside of the inferred historical distribution of pearl dace.

I placed priority-1 and 3 sites as close to historical collection coordinates as possible considering landowner permission and stream access. If permission was denied, I solicited permission from nearby landowners. If a historical stream flowed through more than one HUC 10, I attempted to sample a site within each HUC 10. I selected

additional priority-1 sites near the end of the 2016 field season to better define the current

distribution of pearl dace; I sampled one or two additional sites on all the streams where I

collected pearl dace except on the Blackfeet Reservation. Sites were selected based on

landowner permission and the presence of springs and water.

I selected priority-2 and 4 sites on non-historical streams based on proximity to

historical streams and likelihood of habitat suitability for pearl dace and chrosomid dace.

I selected sites on streams that were direct tributaries to, or direct recipients of,

historically occupied streams. I also selected sites on streams in the inferred historical

distributions that were not well-sampled by others (Bramblett 2010; MFISH 2016). I

screened streams for the presence of springs (USGS 2015) and water by examining

satellite imagery in ArcMap 10.2.2 (ESRI 2014). If springs were present or water was

visible, and the site was not on public land, I solicited permission to sample the site.

Sampling location along a stream was randomized if possible (i.e., if permission was

obtained from multiple landowners or multiple reaches on public land were accessible

and had water).

I sampled 41 priority-1 sites, 47 priority-2 sites, 39 priority-3 sites, and 13

priority-4 sites during 2015 and 2016 (Appendix Table TTT). I sampled a total of 88 sites

on 57 streams and 2 reservoirs in the inferred historical distribution of pearl dace and 140

sites on 103 streams and 2 reservoirs in the inferred historical distribution of chrosomid

dace. I sampled 23 streams with historical pearl dace collections and 73 streams with

historical chrosomid dace collections. The Blackfeet Environmental Office sampled one

additional stream with historical pearl dace collections and one additional stream with historical chrosomid dace collections.

<u>Sampling</u>

   <u>Fish Collection</u>. I collected fish with four different gear types (Appendix Table TTT): seines, dipnets, backpack electrofishers, and minnow traps. I selected the most suitable gear type for each site based on stream conditions (e.g., water depth, vegetation density, water conductivity, current velocity, woody debris presence). Wadeable sites with predominantly small substrates, scarce woody debris, and high conductivity (> 5000 μS) were sampled with seines, except for any narrow or shallow sections (i.e., < 0.25 m in depth and < 1 m in width; Labbe and Fausch 2000), which were sampled with dipnets. Sites that were too narrow to seine and that had conductivities too high to electrofish (> 5000 μS; Reynolds and Kolz 2012) were sampled with dipnets alone. Wadeable sites that were unsuitable for seining (e.g., abundant large rocks, woody debris, or swift currents; Patton et al. 2000) and had suitable conductivities (10-5000 μS; Reynolds and Kolz 2012) were sampled with backpack electrofishers. Minnow traps were used at sites that were too deep to safely sample with seines, dipnets, or backpack electrofishers.

   I sampled 57 sites with bagless seines (6.1 m or 4.6 m long × 1.8 m tall with 0.64-cm bar mesh) and 5 sites with a bag seine (6.1 m × 1.8 m with 0.64-cm bar mesh and a 1.8 m × 1.8 m × 1.8 m bag) as the primary sampling gear. I used dipnets in small, shallow habitats at 11 of the 57 seine sites and alone at 4 sites that were too narrow to seine.

   I sampled 59 sites with backpack electrofishers. I used a Smith-Root LR-24 backpack electrofisher, except when sampling with the Blackfeet Environmental Office

18

when a Smith-Root model 12-B, a Smith-Root LR-20B, or a combination of the three backpack electrofishers was used. Electrofishing was conducted in accordance with the Montana Fish, Wildlife and Parks electrofishing policy and guidelines.

I sampled four sites with minnow traps (Promar [Gardena, California] model TR-501 polyethylene collapsible traps). I baited 30 traps with dry dog food (Goertz and Phoenix 2015) and deployed them at 10-m intervals for 8-15 hours. At one trapped site, a backwater was shallow enough to sample with a bag seine (6.1 m × 1.8 m with 0.64-cm bar mesh) allowing for collection of fish larger than the minnow traps could capture.

I sampled a 300-m stream reach when possible. Either seining a 300-m reach or electrofishing a 200-m reach was sufficient to capture all fish species in Wyoming prairie streams (Patton et al. 2000). However, land access sometimes limited the length of stream I sampled. Dry sites were photographed (8 sites in the inferred historical distribution of pearl dace and 11 in the inferred historical distribution of chrosomid dace).

All captured fish were placed in buckets with portable aerators. I anesthetized fish in batches with tricaine methanesulfonate (MS-222) at the effective dose for cyprinids (100-200 mg/L; Coyle et al. 2004; Mercy et al. 2013) before processing. I identified fish to species when possible, and I collected up to 10 voucher specimens of each species when identification was uncertain (unless fewer than 10 individuals were collected, in which case all individuals were photographed). Chrosomid dace vouchers were collected (see below) and identified in the laboratory. Voucher specimens were euthanized with an overdose of MS-222 (1000 mg/L; AVMA 2013) and preserved in a 10% buffered

formalin solution. I allowed non-vouchered fish to recover in 19-L plastic buckets before returning them to the stream reach where they were collected.

Habitat. Habitat was sampled as prescribed by the Montana Prairie Fish and Habitat Sampling Protocol (Bramblett 2003; Appendix B). I measured physical habitat characteristics at 11 evenly-spaced transects (i.e., at 30-m intervals) along a reach after fish collection. I recorded substrate size, wetted width to the nearest 0.1 m, and depth to the nearest 0.01 m at five evenly-spaced locations along each transect. Substrate sizes were classified as fines (< 2 mm), sand (0.06 mm to 2 mm), fine gravel (2 mm to 16 mm), coarse gravel (16 mm to 64 mm), cobble (64 mm to 250 mm), boulder (250 mm to 4,000 mm), hardpan (consolidated fine substrate), bedrock (> 4,000 mm), wood, or other (Lazorchark et al. 1998). I photographed upstream and downstream-facing views at the upper terminus, midpoint, and lower terminus of each site.

Water chemistry was sampled in the pool nearest to the center of the reach. Water temperature (°C), conductivity (µS/cm), and dissolved oxygen concentration (mg/L) were measured with a Yellow Springs Institute model 85 meter (YSI); pH was measured with an Oakton pH pen; and turbidity (NTUs) was measured with a LaMotte 2020 turbidity meter. Water temperature, conductivity, and dissolved oxygen concentration were not measured at 24 sites in 2016 when the YSI meter was being repaired.

Chrosomid Dace Vouchers

Hybrid dace have physical characteristics that are intermediate between Northern Redbelly Dace and Finescale Dace (New 1962; Schlosser et al. 1998; Mee and Rowe

2010). However, considerable morphological variation exists in both parental species and their hybrid (Goddard et al. 1989). Northern Redbelly Dace and hybrid dace differ in internal characteristics such as intestine complexity and pharyngeal tooth counts, and differ subtly in external characteristics such as mouth size and shape (New 1962; Goddard et al. 1989; Mee and Rowe 2010). Therefore, distinguishing chrosomid dace is difficult under field conditions, suggesting that my putative field identifications needed laboratory confirmation.

The number of chrosomid dace vouchered at each site was based on a tradeoff between maximizing the probability of detecting hybrid dace while minimizing the number of individuals sacrificed from each site. I determined a priori that I would not sacrifice more than 50% of individuals collected at a site to mitigate negative effects of sampling on chrosomid dace populations. I used data from the Montana Prairie Stream Fish Survey (Bramblett 2010) to determine that the average number of chrosomid dace captured from a 300-m reach was 100 (95% CI = 63 to 135). The probability of detecting hybrid dace depends on the proportion of hybrids in the population and the number of voucher specimens examined. To the best of my knowledge, no data are available to estimate the proportion of hybrid dace in chrosomid dace populations, so I assumed low proportions of hybrid dace.

The number of chrosomid dace vouchers necessary to detect at least one hybrid dace varied depending on the total number vouchered and the proportion of hybrid dace in the population (Figure 3). I used a cumulative binomial equation to calculate the total number of fish (n) necessary to voucher to result in a 90% or 95% probability (x) of

vouchering a hybrid dace, if the proportion of hybrid dace (p) was relatively low at 0.10, 0.05, or 0.01 (Equation 1).

$$x = 1 - (1 - p)^n \qquad (1)$$

If the proportion of hybrid dace is greater than 0.1 at either probability, or 0.05 at 90% probability, less than half of the individuals would have to be sacrificed from an average collection of chrosomid dace from a 300-m reach (Figure 3). More than half would have to be sacrificed if the proportion of hybrid dace was 0.01 at either probability, or if the proportion was 0.05 at 95% probability.

Therefore, if more than 20 chrosomid dace were collected, I vouchered up to half of the individuals but no more than 50 total. If fewer than 20 chrosomid dace were collected, I photographed each individual and returned them to the stream to mitigate negative effects of voucher collection on small chrosomid dace populations. However, if any chrosomid dace died from sampling trauma, they were vouchered regardless of the total number collected.

I vouchered chrosomid dace non-randomly. I putatively identified chrosomid dace in the field based on size and mouth shape (New 1962; Joswiak and New 1989), and I targeted potential hybrid dace by vouchering the largest specimens and specimens with straight mouths. Therefore, the likelihood of vouchering a hybrid dace was probably higher than if specimens were vouchered at random.

I examined intestinal characteristics and pharyngeal tooth counts to identify 186 chrosomid dace voucher specimens to taxon in the laboratory. Each specimen was measured (total length) and assigned a unique ID. Each whole specimen, its mouth, and

its intestinal tract were photographed. The intestinal complexity was recorded, and a tentative identification was assigned (New 1962). A separate datasheet was used to record pharyngeal arch identifications so that tentative identifications based on intestinal characteristics would not bias pharyngeal tooth counts. Pharyngeal arches were removed by Ridewood dissection (Bemis et al. 2004), stained with alizarin red (Taylor 1967), and observed under a dissecting microscope to determine pharyngeal tooth counts. I identified a specimen as a hybrid dace if it had a second row of pharyngeal teeth on either pharyngeal arch (Goddard et al. 1989; Mee and Rowe 2010) or had a short intestine with few loops (New 1962).

Voucher specimen identifications made with the different methods mostly agreed. The laboratory identification agreed with the field identification in 73.7% of specimens. The overall error rate in laboratory identification appeared to be low, but was higher for hybrid dace than for Northern Redbelly Dace. The tooth count could not be determined in 13.4% of specimens because of decalcified or broken pharyngeal arches; therefore, these identifications were based entirely on intestinal complexity. The intestinal identifications disagreed with the pharyngeal tooth identifications in 7.0% of specimens. Most of these specimens (6.5%) had simple intestines (indicating hybrid dace), but no second row of pharyngeal teeth (indicating Northern Redbelly Dace). Observer error, given the small size and fragility of the pharyngeal arches, was the probable cause of these discrepancies. Only one specimen (0.5%) was identified as a Northern Redbelly Dace as judged by its intestinal characteristics and as a hybrid dace as judged by its tooth count.

<u>Analyses</u>

<u>Influential Non-native Predators</u>. Non-native fishes were considered influential predators if they were classified as piscivorous in the literature, collected at many sites, and the individuals collected were large enough to be piscivorous. Only Northern Pike and non-native trout (i.e., Brook Trout *Salvelinus fontinalis*, Brown Trout *Salmo trutta*, and Rainbow Trout *Oncorhynchus mykiss*) were classified as influential non-native predators. I determined the effects of non-native predators on pearl dace and chrosomid dace by quantifying their co-occurrence at the stream scale. Co-occurrence of pearl dace or chrosomid dace with non-native predators was compared to the co-occurrence expected under the null hypothesis of a G-test with a Williams correction (Gotelli and Ellison 2013) to determine if co-occurrence was significantly different than predicted by chance. Only Northern Pike and non-native trout (i.e., Brook Trout *Salvelinus fontinalis*, Brown Trout *Salmo trutta*, and Rainbow Trout *Oncorhynchus mykiss*) were classified as influential non-native predators.

<u>Estimates of Changes in Pearl Dace and Chrosomid Dace Distributions</u>. I estimated the percent declines in distributions of pearl dace and chrosomid dace with four different methods (sampled-only, liberal, conservative, non-native predator) to assess the inherent uncertainty in the estimates caused by the unknown status of pearl dace and chrosomid dace in streams they occupied historically but that I did not sample. Percent declines (d) were calculated by dividing the inferred number of currently occupied historical streams (c) by either the number of historical streams sampled (pearl dace $h = 25$; chrosomid dace $h = 73$) or the total number of historical streams (pearl dace $h = 30$;

chrosomid dace $h = 119$), subtracting the quotient from one and multiplying by 100 (Equation 2).

$$d = \left(1 - \frac{c}{h}\right) \times 100 \qquad (2)$$

Each method used a different set of assumptions to infer the number of currently occupied historical streams (Table 3).

The sampled-only method considered a historical stream to be currently occupied by pearl dace or chrosomid dace if the taxon was collected there in 2015 to 2017; the number of currently occupied streams was divided by the number of historical streams sampled (Table 3). The sampled-only method made no inferences about the historical streams that were not sampled in 2015 to 2017. The sampled-only method resulted in moderate estimates of percent decline in distribution.

The liberal method considered a historical stream to be currently occupied by pearl dace or chrosomid dace if they were collected there in 2015 to 2017; the number of currently occupied historical streams was divided by the total number of historical streams. The liberal method assumed that pearl dace or chrosomid dace were absent from historical streams that were not sampled in 2015 to 2017. Therefore, the liberal method resulted in the highest estimates of percent decline in distribution.

The conservative method considered a historical stream to be currently occupied by pearl dace or chrosomid dace if they were collected there in 2015 to 2017, or if the historical stream was not sampled in 2015-2017; the total was divided by the total number of historical streams (Table 3). The conservative method assumed that pearl dace or chrosomid dace were still present in the historical streams that were not sampled in

2015 to 2017. Therefore, the conservative method resulted in the lowest estimates of percent decline in distribution.

The non-native predator method was similar to the conservative method in that it considered a historical stream to be currently occupied by pearl dace or chrosomid dace if they were collected there in 2015 to 2017, or if the historical stream was not sampled in 2015-2017. However, the non-native predator method used other collections of influential non-native predators (MFISH 2016) to better inform the estimate. The non-native predator method assumed that pearl dace or chrosomid dace were still present in the historical streams that were not sampled in 2015 to 2017 unless influential non-native predators were collected by others in the historical stream (Table 3); the inferred number of currently occupied historical streams was divided by the total number of historical streams. The non-native predator method resulted in moderate estimates of percent decline in distribution.

Habitat Comparison. I compared habitat and water quality between streams with dace and without dace to ensure that all streams sampled had habitat for dace. Habitat data (mean width, mean depth, proportion fine substrates, water temperature, $\log_e$ turbidity, pH, dissolved oxygen concentration, $\log_e$ conductivity) were summarized by site and averaged across streams to compare streams where pearl dace and chrosomid dace were and were not collected. Proportion fine substrates was calculated by dividing the number of occurrences of fines and sand substrates at a site by the total number of substrate measurements (Mullen 2007). Turbidity and conductivity data were log transformed because they were not normally distributed.

Each habitat variable was compared visually (Figure 4) between pearl dace and

non-pearl dace streams in the inferred historical distribution of pearl dace and with

hypothesis tests between chrosomid dace and non-chrosomid dace streams in the inferred

historical distribution of chrosomid dace. I did not compare pearl dace streams to non-

pearl dace streams with hypothesis tests because of unequal sample sizes and violation of

the equal variance assumption. Habitats of chrosomid dace and non-chrosomid dace

streams were compared with t-tests if the data were normally distributed or with

Wilcoxon rank-sum tests if the data were not normally distributed. I tested for

homogeneity of variances of habitat variables from chrosomid dace and non-chrosomid

dace streams with Bartlett's test if the data were normally distributed or with Levene's

test if the data were not normally distributed.

Results

Current Distributions of Pearl Dace and Chrosomid Dace in Montana

Pearl dace were probably present at eight sites distributed on five streams in

Montana in 2017. I collected pearl dace at eight sites distributed among five streams in

2015-2016 clustered in three widely-separated groups (Figures 5-8). However, pearl dace

were not collected from one site (on Eagle Creek, Daniels County) when it was

resampled in 2016 (Figure 6). Additional sampling by others in 2017 clarified the current

distribution of pearl dace in Montana. Pearl dace were still present in Snake Creek, Bean

Creek, and Plentywood Creek in 2017 (S. Dalbey, Montana Fish, Wildlife and Parks,

personal communication). The Blackfeet Environmental Office found another population

of pearl dace in the Middle Fork Milk River in 2017 (Figure 8; A. Gilham, U. S. Fish and Wildlife Service, personal communication). Therefore, pearl dace were still present in at least five streams in Montana in the autumn of 2017.

Chrosomid dace were probably present at 44 sites distributed on 41 streams in Montana in 2017. I collected Northern Redbelly Dace at 43 sites distributed on 40 streams in 2015-2016 that were widely dispersed throughout their inferred historical distribution (Figure 9). However, no Northern Redbelly Dace were collected from the Little Missouri drainage. I collected hybrid dace at 19 sites (44.2% of Northern Redbelly Dace sites), distributed on 19 streams (46.3% of Northern Redbelly streams), which were widely distributed throughout their inferred historical distribution (Figure 9). The proportion of hybrid dace ranged from 0.03 to 0.67 at the sites where they were collected (mean = 0.130; SE = 0.032); however, these estimates are fairly rough because only 73.7% of laboratory identifications of voucher specimens matched putative field identifications. Additional sampling in 2017 clarified the current distribution of Northern Redbelly Dace in Montana. The Blackfeet Environmental Office found another population of Northern Redbelly Dace in the Middle Fork Milk River during the summer of 2017 (A. Gilham, personal communication). Voucher specimens were not collected to evaluate hybrid dace presence in the Middle Fork Milk River. Therefore, Northern Redbelly Dace were still present in at least 41 streams and hybrid dace were present in at least 19 streams in 2015-2017.

Influential Non-native Predators

Northern Pike and non-native trout were widely distributed in the inferred

historical distributions of pearl dace (Figure 10) and chrosomid dace (Figure 11) in

Montana, but occurred in different regions. Northern Pike were collected from 33 sites

(43.4%) in the inferred historical distribution of pearl dace and from 48 sites (37.2%) in

the inferred historical distribution of chrosomid dace, whereas non-native trout were

collected from 21 sites (16.3%) in the inferred historical distribution of pearl dace and

from 27 sites (20.9%) in the inferred historical distribution of chrosomid dace. However,

Northern Pike were collected almost exclusively from the eastern half and non-native

trout from the western half of the inferred historical distributions of pearl dace and

chrosomid dace (Figure 12), and Northern Pike and non-native trout never co-occurred.

Northern Pike were mostly collected from prairie streams originating from the

Northwestern Glaciated Plains or Northwestern Great Plains Ecoregions. Non-native

trout were mostly collected from mountain foothill streams originating from the Canadian

Rockies or Northern Rockies Ecoregions located along the Rocky Mountain Front or

within isolated mountain ranges.

Northern Pike and non-native trout were widespread in streams with historical

collections of pearl dace and chrosomid dace streams, but I found almost no overlap

between pearl dace and influential non-native predators (Table 4). I collected non-native

predators from 14 of the 23 sampled pearl dace historical streams (60.9%). However,

non-native predators were collected at only one (Eagle Creek) of six sampled historical

streams with pearl dace (16.7%), significantly fewer than expected by chance (G-test: G_1

= 7.68, P = 0.006). I collected both pearl dace and Northern Pike from Eagle Creek in July 2015, but I collected Northern Pike and no pearl dace when I resampled it in September 2016. Northern Pike were also collected from two nearby sites on Eagle Creek where I did not collect pearl dace (Figure 7). Therefore, pearl dace may have been extirpated from Eagle Creek during 2015 or 2016. If pearl dace were extirpated from Eagle Creek between July 2015 and September 2016, then the current overlap between influential non-native predators and pearl dace is even lower than estimated from my data. I collected non-native predators from 25 of the 73 sampled chrosomid dace historical streams (34.2%), but non-native predators were collected at only 8 of 41 sampled historical streams with chrosomid dace (19.5%), which was significantly fewer than expected by chance (G-test: G_1 = 25.9, P < 0.001).

Changes in Pearl Dace and Chrosomid Dace Distribution

The current distributions of pearl dace and chrosomid dace in Montana are smaller than the inferred historical distributions. Pearl dace were collected from 5 of 23 sampled historical streams, not including Eagle Creek where they may have been extirpated. Chrosomid dace were collected from 42 of the 73 sampled historical streams, including 6 of the 8 sampled historical streams in the upper Musselshell River drainage. Therefore, I estimated that pearl dace distribution in Montana streams has decreased 63.3 to 83.3% (Table 5), and chrosomid dace distribution in Montana streams has decreased 32.0% to 67.2% (Table 5).

Habitat Comparison

Median water temperature, $\log_e$ turbidity, dissolved oxygen concentration, and conductivity differed slightly between pearl dace streams and non-pearl dace streams (Figure 4). However, the sample sizes of pearl dace streams were probably too small to capture the natural variation in the habitat variables or to yield accurate estimates of the parameters. I only detected a difference in the proportion fine substrates between chrosomid dace and non-chrosomid dace streams (Table 6). Chrosomid dace streams had a 0.17 greater mean proportion of fine substrates (95% CI = 0.04 to 0.29) than streams without chrosomid dace (Wilcoxon rank-sum test: $W_{88} = 1365$; $P = 0.012$). However, the equal variance assumption was violated for proportion of fine substrates (Levene's-test: $F_{1, 88} = 6.43$, $P = 0.013$), so these results may not be reliable.

Discussion

Pearl dace are at extreme risk and chrosomid dace are at moderate risk of extirpation from Montana. The known extant populations of pearl dace in Montana are widely separated and extirpation is apparently ongoing as evidenced by the Eagle Creek collections. Chrosomid dace are still widely distributed throughout some drainages in Montana (i.e., upper Musselshell, middle Milk), uncommon in others (i.e., Judith, Poplar, Big Muddy, Lower Yellowstone), and may have been extirpated from the Little Missouri River drainage.

Non-native predators appear to be the biggest threat to pearl dace and chrosomid dace in Montana. Non-native predators are widespread and most remaining dace

populations are at high risk of future invasion by these predators. Observational studies such as this one cannot determine causality; however, two lines of evidence suggest that predation by non-native predators was the primary cause of the declines of pearl dace and chrosomid dace. First, I sampled sites with high likelihood of pearl dace or chrosomid dace occurrence, but I collected them in relatively few locations even though the habitat and water quality at most sites were probably suitable (Figure 4, Table 6; Bestgen 1989; Cunningham 1995, 2006; Cunningham and Hickey 1996; Stasiak 2006; Felts 2013). Conversely, I collected non-native predators from many sites and from many streams with historical records where I did not collect pearl dace or chrosomid dace. The low co-occurrence of pearl dace and chrosomid dace with non-native predators, particularly in streams where dace were once present, suggests that Northern Pike and non-native trout caused extirpation of pearl dace and chrosomid dace from these streams or reduced them to non-detectable abundances. The low co-occurrence probably resulted from predation, but could have resulted from a less probable mechanism (e.g., competition, disease spillover). Second, Pearl dace and chrosomid dace almost never co-occur with large piscivorous fish. In fact, the absence of large-bodied piscivorous fish was described as a critical component of suitable pearl dace and chrosomid dace habitat in both lakes and streams (Scott and Crossman 1973; Becker 1983; Bestgen 1989; Cunningham 2006; Stasiak 2006). Pearl dace and chrosomid dace may face occasional predation by Creek Chub *Semotilus atromaculatus*, and sometimes co-occur with small native Brook Trout (Stasiak 1972, 2006; Becker 1983; Cunningham 2006), but they almost never co-occur with large-bodied piscivorous fish (Bestgen 1989; Chapleau et al. 1997; Whittier et al.

1997; Findlay et al. 2000; Trumpickas et al. 2011; Aiken et al. 2012). This seeming inability of pearl dace and chrosomid dace to co-occur with native large-bodied predatory fish suggests that they would also be sensitive to the presence of these predators where they have been introduced.

Northern Pike threaten pearl dace and chrosomid dace in the eastern half of their inferred historical distributions and non-native trout threaten them in the western half. The geographic pattern probably resulted from differential dispersal of Northern Pike and non-native trout from stocking locations. Northern Pike were probably able to disperse throughout the large prairie rivers (i.e., Milk River, Missouri River, Yellowstone River) from reservoirs and the St. Mary Canal (Mogen and Best 2011), from where they were able to colonize small prairie drainages far upstream and downstream of stocking locations. Non-native trout probably cannot tolerate the variable conditions of most prairie streams, and were thus limited in dispersal and largely restricted to those streams originating in mountain ranges and tailwaters below hypolimnetic release reservoirs. Pearl dace and chrosomid dace occur in both prairie and mountain foothill streams in Montana and are therefore exposed to predation by two different large, non-native predators in two different habitat types where they occur.

Pearl dace and chrosomid dace may be especially vulnerable to non-native predators in prairie streams because biological interactions may be stronger in intermittent than in connected reaches. Predators are particularly efficient in the isolated pools of intermittent reaches where prey fish are confined (Power et al. 1985; Schlosser 1987; Lohr and Fausch 1996; Labbe and Fausch 2000) and cannot emigrate as they do in

other systems when confronted with predation (He and Kitchell 1990; Kitchell et al. 1994; Fraser et al. 1995). Introduced Northern Pike quickly reduced and eliminated species in systems where water was less limiting and refuge was more available (He and Kitchell 1990; Kitchell et al 1994; DeBates et al. 2003; Byström et al. 2007; Haught and von Hippel 2011; Nicholson 2015). I would therefore expect them to have a particularly strong effect on sensitive native fishes such as pearl dace and chrosomid dace in intermittent streams. Therefore, pearl dace and chrosomid dace are probably at greater risk from predation in prairie streams than in mountain foothill streams.

Chrosomid dace are probably temporarily secure upstream of a series of diversion dams on the Musselshell River, which are thought to be a barrier to Northern Pike movement (M. Ruggles, Montana Fish, Wildlife and Parks, personal communication). Davis Diversion Dam is the first significant barrier to upstream fish movement, followed by the Stella Diversion Dam, and the Parrot Diversion Dam (M. Ruggles, personal communication). However, Davis Diversion probably limits native Sauger dispersal (M. Ruggles, personal communication), probably resulting in a sub-optimal length of river available for spawning migrations (*sensu* Jaeger et al. 2005). The Stella and Parrot Diversion Dams are probably the most reliable barriers that minimize the negative effects of fragmentation on native fish assemblages, and providing fish passage at dams upstream and downstream of these dams will probably benefit native fish conservation. Non-native trout were stocked widely throughout the Musselshell River drainage, including in many historical chrosomid dace streams (MFISH 2016). However, non-native trout in the Musselshell River drainage are somewhat restricted in distribution,

which may be the result of low physiological tolerance to downstream water quality and habitat conditions. Isolation of the upper Musselshell River drainage by diversion dams may have helped to conserve prairie fishes such as chrosomid dace, especially because the drainage is large and relatively intact such that dispersal is still possible among local populations.

Habitat degradation is probably the next biggest threat to pearl dace and chrosomid dace in Montana, following non-native predators. Predator presence explained the most variation in minnow species richness in lakes in the eastern United States, but minnow species richness declined with increasing anthropogenic activity in the absence of predators (Whittier et al. 1997). Often, effects of non-native predators are amplified or facilitated by habitat degradation and the interaction between the two can result in further homogenization of fish assemblages (Clavero and Garcia-Berthou 2006; Ricciardi and MacIsaac 2011; Turek et al. 2013). Although I did not explicitly study anthropogenic stressors, agriculture, grazing, and oil extraction are common throughout the distributions of pearl dace and chrosomid dace in Montana, and probably contribute to stream dewatering, increased sediment and nutrient inputs, and fragmentation from road crossings. However, the effects of habitat degradation on pearl dace and chrosomid dace in Montana have not been examined.

Climate change will probably decrease the amount of habitat for pearl dace and chrosomid dace. The turn of the century drought from 2000-2010 was probably the most intense in the upper Missouri drainage in the last 1,200 years and was closely linked to increased temperature over the region (J. Martin, Montana State University, personal

communication). Drought intensities in the Missouri River basin in Montana will probably increase because temperature increases are expected to continue (IPCC 2014). If future warming and drying of prairie streams occurs, this will probably contribute to increased intermittency in prairie streams, which may exacerbate the effects of other stressors and further reduce the distributions of pearl dace and chrosomid dace in Montana, (Bertrand et al. 2009).

I am confident that I detected pearl dace and chrosomid dace at sites where they were present in sufficient numbers for a population to persist. However, I could not quantify my detection probability because of time constraints and lack of resources, so I cannot be certain that I detected pearl dace and chrosomid dace sites where they were present in low abundances. Additionally, the four sampling gears I used probably had different detection probabilities. To account for this, I sampled a length of stream that was sufficient when seining, and more than sufficient when electrofishing, to capture all species in Wyoming prairie streams (Patton et al. 2000). I still may have failed to detect pearl dace and chrosomid dace at sites where they were in very low abundance, but these sites probably do not harbor stronghold populations. Finally, I may have missed a population of pearl dace or chrosomid dace elsewhere in the drainages I sampled. Further fish collections may discover additional populations of pearl dace and chrosomid dace.

In conclusion, pearl dace are currently rare and at high risk of extirpation from Montana and chrosomid dace are currently uncommon and at risk of extirpation from most drainages in Montana, although they appear to be secure in the upper Musselshell drainage. Northern Pike and non-native trout were probably the main causes of the

reductions in pearl dace and chrosomid dace distributions in Montana; further expansion of these non-native predators may result in further extirpations. Habitat degradation probably also played a role in the decline of pearl dace and chrosomid dace and future climate change may exacerbate the effects of other stressors.

Management Implications and Recommendations

Securing the remaining populations of pearl dace and chrosomid dace is critical for their continued persistence in Montana. Without conservation action, pearl dace may be extirpated from Montana and chrosomid dace may be extirpated from additional drainages. Exclusion or removal of non-native predators from remaining pearl dace and chrosomid dace populations will afford fisheries managers the best chance of conserving these taxa. My recommendations are to 1) determine which populations are secure from invasion of non-native predators and which populations may need more immediate conservation action, 2) identify and sample potential locations of additional pearl dace and chrosomid dace populations, 3) consider the feasibility of removing non-native predators, constructing barriers to secure existing populations or to prevent re-invasion, and reintroducing pearl dace and chrosomid dace to historical drainages, 4) maintain the Stella Diversion and Parrot Dam in the Musselshell drainage as barriers to upstream Northern Pike movement, 5) assess the risk of stocking of Northern Pike and non-native trout in waterbodies containing or connected to those containing pearl dace and chrosomid dace, and 6) Implement a long term monitoring program of pearl dace and chrosomid dace populations to assess population trends.

Addressing the threat of habitat degradation will allow fisheries managers to further secure these sensitive populations. My recommendations are to 1) work with land owners and change management on public land to distribute stock tanks and exclude cattle from riparian areas in streams with pearl dace and chrosomid dace, 2) limit new road development near streams with pearl dace and chrosomid dace, and 3) limit new oil extraction activities on public land near streams with pearl dace and chrosomid dace.

Finally, managers might consider potential negative effects of isolation (Fausch et al. 2009; Carim et al. 2016) on secured populations of pearl dace and chrosomid dace. Barriers can impede further invasion of non-native species (Freeman et al. 2002; Jackson and Pringle 2010) and managers construct barriers for the explicit purpose of isolating threatened populations from non-native species (Thompson and Rahel 1998; Fausch et al. 2009; Rahel 2013), but barriers also limit dispersal, which is probably important for the long-term persistence of pearl dace and chrosomid dace (*sensu* Fausch and Bestgen 1997). Dispersal of individuals among local populations allows for recolonization after local extirpation (Hanski and Simberloff 1997). Isolation of pearl dace and chrosomid dace populations may secure them from non-native predators, but it may leave them vulnerable to extirpation from stochastic events (Sheldon 1988; Lesica and Allendorf 1995) and increase their risk of inbreeding depression, genetic drift, and increased homozygosity (Meffe 1986). Therefore, I recommend following the guidelines in Meffe (1986) to maximize long-term genetic health of isolated populations and increasing the number of pearl dace populations through reintroductions to buffer against extirpations from stochastic events.

Table 1. Sources of pearl dace collection data used to create the inferred historical distribution in Montana. Collections are listed by source, drainage, and then waterbody. Those collections associated with a voucher specimen are indicated with Yes. Catalog numbers for voucher specimens in museums are included in parenthesis after the source name. Museum abbreviations follow American Society of Ichthyologists and Herpetologists standards (Sabaj 2016).

Source	Drainage	Waterbody	Voucher
A. Gilham, Montana State University, personal communication	Marias	Badger Creek	
		Birch Creek	
		Blacktail Creek	
		Coldfeet Coulee Spring Creek	
		Cut Bank Creek	
		Deep Creek	
		Greasewood Creek	
		Two Medicine River	
		Unnamed Tributary	
		Willow Creek	
	St. Mary	Bird Creek	
		St. Mary River	
		Willow Creek	
	Upper Milk	Dry Fork Milk River	
		Middle Fork Milk River	
		South Fork Milk River	
Bramblett 2010	Big Muddy	Big Muddy Creek	Yes
		Eagle Creek	Yes
	Middle Milk	Assiniboine Creek	Yes
Fishnet2 (UMMZ 173924)	Big Muddy	Ator Creek	Yes
Fishnet2 (ROM 17778)	Frenchman	Cypress Lake Drainage Canal	Yes
Fishnet2 (USNM 369861.5284)		Frenchman Creek	Yes

Table 1 continued

Fishnet2 (ROM 17787)		Unknown; Small Stream Crossing Hwy 21, 9 Mi. S. of Cypress Hills Parks	Yes
Fishnet2 (ROM 17792)		Unknown; Small Stream Crossing Hwy 21, 12 Mi. S. of Cypress Hills Park	Yes
Fishnet2 (UA 3340)	Lodge	Bare Creek	Yes
Fishnet2 (UA 3566)		Lodge Creek	Yes
Fishnet2 (UA 3567)		Unnamed tributary to Lodge Creek	Yes
Fishnet2 (UMMZ 173923; UMMZ 173925)	Lower Missouri section	Wolf Creek	Yes
Fishnet2 (UW 003383)	Marias	Two Medicine river	Yes
Fishnet2 (KUI 21501)	Middle Milk	Snake Creek	Yes
Fishnet2 (CMNFI 1970-0191.10)	Poplar	Lost Child Creek	Yes
Fishnet2 (OS 6209; UMMZ 138315; UMMZ 218997)	St. Mary	Moran's Bathtub	Yes
Henderson and Peter 1969	Middle Milk	Middle Creek	
	St. Mary	St. Mary River	
Hubbs 1942	St. Mary	Moran's Bathtub	
Lee et al. 1980	Frenchman	Frenchman Creek	
	St. Mary	St. Mary River	
McCulloch et al. 1998	Battle	Battle Creek	
	Frenchman	Caton Creek	
		Conglomerate Creek	
		Farwell Creek	
	Lower Milk	Morgan Creek	
MFISH	Battle	Salmo Reservoir	
	Beaver	Larb Creek	
	Big Muddy	Antelope Creek	

39

Table 1 continued

		Boxelder Creek	
	Lower Missouri section	East Shotgun Creek	
	Middle Milk	Bean Creek	
		Clear Creek	
		Snake Creek	
	Poplar	Poplar River	
Mogen and Best 2011	St. Mary	St. Mary Diversion	
MSU museum (MSUB 6854)	Battle	Salmo Reservoir	
MSU museum (MSUB 3101)	Big Muddy	Ator Creek	Yes
MSU museum (MSUB 1797; MSUB 1800; MSUB1805)		Smoke Creek	Yes
MSU museum (MSUB 4050; MSUB 7519)		Whitetail Creek	Yes
MSU museum (No specimen, just entry on collection card)	Fort Peck Reservoir	Carpenter Creek	
MSU museum (MSUB 4131)	Lower Milk	Porcupine Creek	Yes
MSU museum (MSUB 1812; MSUB 1840: MSUB 1961; MSUB 4052)	Lower Missouri section	Wolf Creek	Yes
MSU museum (MSUB 3069; MSUB 6779)	Middle Milk	Lodge Creek	Yes
MSU museum (MSUB 7509)		Paradise Canal	
MSU museum (MSUB 6836)		Snake Creek	Yes
MSU museum (MSUB 1821; MSUB 1841)	Poplar	Poplar River	Yes
Nelson and Paetz 1970	St. Mary	St. Mary River	
	Lodge	Lodge Creek	
Nelson and Paetz 1992	Lodge	Lodge Creek	
		Bare Creek	
Schultz 1941	Marias	Two Medicine River	
Willock 1969	Frenchman	Unspecified tributaries in Cypress Hills	

Table 2. Sources of chrosomid dace collection data used to create the inferred historical distribution in Montana. Collections are listed by source, drainage, and waterbody. Those collections associated with a voucher specimen are indicated with Yes. Catalog numbers for voucher specimens in museums are included in parenthesis after the source name. Museum abbreviations follow American Society of Ichthyologists and Herpetologists standards (Sabaj 2016).

Source	Drainage	Waterbody	Voucher
A. Gilham, Montana State University, personal communication	Marias	Birch Creek	
		Cut Bank Creek	
		Willow Creek	
	Upper Milk	Fox Creek	
		Middle Fork Milk River	
Barfoot 1993	Upper Little Missouri	Little Beaver Creek	Yes
Bramblett 2010	Judith	Wolf Creek	
	Lower Milk	Crow Creek	
	Lower Missouri section	East Fork Little Porcupine Creek	
		Little Porcupine Creek	Yes
	Lower Musselshell	Fords Creek	
	Middle Milk	Assiniboine Creek	
		Snake Creek	
	Middle Missouri section	Eagle Creek	
		Siparyann Creek	
	Upper Missouri section	Flat Creek	
Bramblett et al 2005	Judith	Sage Creek	
	Lower Missouri section	Little Porcupine Creek	
	Upper Musselshell	South Fork Big Coulee Creek	
Cope 1879	Middle Milk	Battle Creek	
Fishnet2 (CMNFI 1961-0203.1)	Battle	Battle Creek	Yes
Fishnet2 (CMNFI 1967-0637.3)	Frenchman	Belanger Creek	Yes

Table 2 continued

Fishnet2 (ROM 17803)		Lonepine Creek	Yes
Fishnet2 (UAIC 11164.02)	Judith	Ross Fork Creek	Yes
Fishnet2 (UAIC 11165.02)		Sage Creek	Yes
Fishnet2 (CMNFI 1961-0205.2)	Lodge	Lodge Creek	Yes
Fishnet2 (UMMZ 173879)	Lower Missouri section	Cow Creek	Yes
Fishnet2 (UMMZ 173878)		Wolf Creek	Yes
Fishnet2 (CMNFI 1967-0670.4)	Marias	Cutbank Creek	Yes
Fishnet2 (UMMZ 218996)		Lower Two Medicine Lake	Yes
Fishnet2 (CMNFI 1967-0622.2)	Middle Milk	Milk River	Yes
Fishnet2 (CMNFI 1970-0191.3)	Poplar	Lost Child Creek	Yes
Fishnet2 (CMNFI 1966-0334.3)	Upper Milk	Police Creek	Yes
Fishnet2 (CMNFI 1966-0405.1)		Tributary to the Milk River	Yes
Fishnet2 (ROM 17779)		Cypress Lake Drainage Canal	Yes
Fishnet2 (UMMZ 135880)	Upper Musselshell	Musselshell River	Yes
Elser 1980	Lower Yellowstone	Box Elder Creek	
		Cottonwood Creek	
		Deer Creek	
		Morgan Creek	
		North Fork Fox Creek	
		Smith Creek	
		Thirteenmile Creek	
		War Dance Creek	
Henderson and Peter 1969	Middle Milk	Grant Creek	
	St. Mary	St. Mary River	
Hunziker et al. 2015	Lower Missouri section	Missouri River	
Lee et al. 1980	Frenchman	Frenchman Creek	

Table 2 continued

McCulloch et al 1998		Caton Creek
		Conglomerate Creek
		Farwell Creek
	Lower Milk	Morgan Creek
	Middle Milk	Battle Creek
MFISH	Big Muddy	Ator Creek
		Boxelder Creek
		Middle Fork Eagle Creek
		Plentywood Creek
	Judith	Boyd Creek
		Buffalo Creek
		Coyote Creek
		Hamilton Coulee
		Indian Creek
		Ross Fork Creek
		Salt Creek
	Lower Milk	Beaver Creek
	Lower Missouri section	Remuda Creek
		West Fork Remuda Creek
	Lower Musselshell	Horsethief Coulee
		Johnson Coulee
		Lodgepole Creek
		Log Gulch
		North Fork McDonald Creek
		Surenough Creek
		Tyler Creek

Table 2 continued

		Yellow Water Creek
	Lower Yellowstone	Clear Creek
	Marias	Bullhead Creek
		Miners Coulee
		Sheep Creek (oxbow pond)
	Middle Milk	Bean Creek
		Bullhook Creek
		Clear Creek
		East Fork Stinky Creek
		Garland Creek
		Little Boxelder Creek
		Lodge Creek
		Red Rock Coulee
	Middle Milk	Stinky Creek
		Woody Island Coulee
	Middle Missouri section	Arrow Creek
		Cow Creek
		Little Sandy Creek
		Possum Run Creek
		Spring Creek
	Middle Musselshell	Parrot Creek
	Poplar	Coal Creek
		Manternach Coulee
	Sun	Adobe Creek
		Big Coulee
		Blackfoot Coulee

44

Table 2 continued

		Muddy Creek tributary	
	Teton	Blindhorse Creek	
		Gamble Coulee	
		Muddy Creek	
		Spring Coulee	
	Upper Missouri section	Big Willow Creek	
		Castner Coulee	
		Huff Creek	
		Keaster Creek	
	Upper Musselshell	American Fork	
		Antelope Creek	
		Blake Creek	
		Careless Creek	
		Currant Creek	
		Hopley Creek	
		Lebo Creek	
		Little Careless Creek	
		Milton Creek	
		Mud Creek	
		Painted Robe Creek	
		Roberts Creek	
		Simmons Creek	
		Swimming Woman Creek	
MSU museum (MSUB MSUB 1787; MSUB 1795; MSUB1806)	Big Muddy	Smoke Creek	Yes
MSU museum (MSUB 7522; MSUB 4050)		Whitetail Creek	Yes

Table 2 continued

MSU museum (MSUB 6599; MSUB 6766)	Gallatin	Central Park Pond	Yes
MSU museum (MSUB 6855)		Farm Pond	Yes
MSU museum (MSUB 6863)		Nixon Gulch	Yes
MSU museum (MSUB 1813)	Judith	Beaver Creek	Yes
MSU museum (MSUB 7420)		Big Spring Creek	Yes
MSU museum (MSUB 7579)		Little Casino Creek	Yes
MSU museum (MSUB 7606)	Lower Milk	Big Warm Creek	Yes
MSU museum (MSUB 4048; MSUB 3081; MSUB 1854; MSU B1811)		Cherry Creek	Yes
MSU museum (MSUB 7612)		Little Warm Creek	Yes
MSU museum (MSUB 4131)		Porcupine Creek	Yes
MSU museum (MSUB 3091)	Lower Missouri section	Cow Creek	Yes
MSU museum (MSUB 4120; 1986)		Redwater River	Yes
MSU museum (MSUB1791; MSUB 1812; MSUB 1831; MSUB 1840; MSUB 1935; MSUB 1961; MSUB 4052; MSUB 5255)		Wolf Creek	Yes
MSU museum (MSUB 1801; MSUB 1951; MSUB 2015)	Lower Musselshell	McDonald Creek	Yes
MSU museum (MSUB 6098)	Lower Yellowstone	Box Elder Creek	Yes
MSU museum (MSUB 6946)	Madison	Darlington Ditch	Yes
MSU museum (MSUB 7417)		Madison River	Yes
MSU museum (MSUB 7421; MSUB 7422)		Sheep Creek	Yes
MSU museum (MSUB 4049)	Middle Milk	Battle Creek	Yes
MSU museum (MSUB 1852)		Beaver Creek	Yes
MSU museum (MSUB 1939; MSUB 1998)		Big Sandy Creek	Yes
MSU museum (MSUB 3069; MSUB 6776)		Lodge Creek	Yes
MSU museum (MSUB 1849)		Peoples Creek	Yes

Table 2 continued

MSU museum (MSUB1866)		Whitewater Creek	Yes
MSU museum (MSUB 4044)	Middle Missouri section	Eagle Creek	Yes
MSU museum (MSUB 3082; MSUB 4542; MSUB 4545)		Fort Peck Reservoir	Yes
MSU museum (MSUB 7563)	Middle Musselshell	Musselshell River	Yes
MSU museum (MSUB 7241)	Teton	Eureka Reservoir	Yes
MSU museum (MSUB 7593)	Upper Milk	Red River	Yes
MSU museum (MSUB 7039)	Upper Musselshell	Big Coulee Creek	Yes
MSU museum (MSUB 7038)		Fish Creek	Yes
MSU museum (MSUB 6806; MSUB 7566)		Musselshell River	Yes
MSU museum (MSUB 7029)		South Fork Big Coulee Creek	Yes
Ostovar 2012	Middle Missouri section	Telegraph Creek	
Nelson and Paetz 1970	St. Mary	St. Mary River	
	Upper Milk	Milk River	
Nelson and Paetz 1992	St. Mary	Tributary to St. Mary River, not labelled	
	Middle Milk	Bare Creek	
		Lodge Creek	
Stash 2001	Middle Milk	Milk River	
Willock 1969	Frenchman	Unspecified tributaries in Saskatchewan	
Willock 1969	Milk	Unspecified tributaries in Saskatchewan and Alberta	

Table 3. Variable definitions of four estimation methods used to estimate changes in pearl dace and chrosomid dace distributions.

Estimation method	Inferred number of currently occupied historical streams (*c)*	Number of historical streams (*h*)	Estimates of decline (*d*)
Sampled-only	Pearl dace or chrosomid dace collected in 2015-2017	Number of sampled historical streams	Moderate
Liberal	Pearl dace or chrosomid dace collected in 2015-2017	Total number of historical streams	High
Conservative	Pearl dace or chrosomid dace collected in 2015-2017, or historical stream not sampled in 2015 - 2017	Total number of historical streams	Low
Non-native predator	Pearl dace or chrosomid dace collected in 2015-2017 or historical stream not sampled in 2015 – 2017, unless influential non-native predator collected by others	Total number of historical streams	Moderate

.

Table 4. Co-occurrence of pearl dace and chrosomid dace with two non-native predators at sites in the inferred historical distributions of pearl dace and chrosomid dace, 2015-2017. Non-native trout consisted of Brook Trout, Brown Trout, Rainbow Trout, or a combination of two or all three species. Individual rows represent the number of sites in each 6-digit hydrologic unit code (HUC 6) and the sum of all sites across all drainages. Northern Pike and non-native trout were never collected at the same site.

HUC 6	Basin name	Total sites sampled with fish present	Dace taxa present, predator not collected	Dace taxa and predator present	Predator present, dace taxa not collected	Neither predator nor PEDA collected
**Pearl dace and Northern Pike**
100100	Saskatchewan	1	1	0	0	0
100302	Marias	20	0	0	0	20
100500	Milk	33	4	0	4	25
100600	Missouri-Poplar	22	2	1	11	8
Sum		*76*	*7*	*1**	*15*	*53*
**Pearl dace and non-native trout**
100100	Saskatchewan	1	1	0	0	0
100302	Marias	20	0	0	10	10
100500	Milk	33	4	0	7	22
100600	Missouri-Poplar	22	3	0	0	19
Sum		*76*	*8†*	*0*	*17*	*51*
**Chrosomid dace and Northern Pike**
100100	Saskatchewan	1	1	0	0	0
100301	Upper Missouri	5	3	0	0	2
100302	Marias	21	4	0	0	17
100401	Missouri-Musselshell	13	4	0	0	9

Table 4 continued

100402	Musselshell	13	8	0	0	5
100500	Milk	38	16	1	4	17
100600	Missouri-Poplar	26	5	0	12	9
101000	Lower Yellowstone	6	2	0	2	2
101102	Little Missouri	6	0	0	2	4
	Sum	*129*	*43*	*1*	*20*	*65*
	Chrosomid dace and non-native trout					
100100	Saskatchewan	1	1	0	0	0
100301	Upper Missouri	5	2	1	1	1
100302	Marias	21	4	0	11	6
100401	Missouri-Musselshell	13	3	1	3	6
100402	Musselshell	13	7	1	2	3
100500	Milk	38	17	0	7	14
100600	Missouri-Poplar	26	5	0	0	21
101000	Lower Yellowstone	6	2	0	0	4
101102	Little Missouri	6	0	0	0	6
	Sum	*129*	*41*	*3*	*24*	*61*

*Pearl dace were collected in 2015, but were not collected in 2016 and may have been extirpated (Eagle Creek). †Pearl dace may have been extirpated from 1 site (Eagle Creek).

Table 5. Estimated percent declines in distribution of pearl dace and chrosomid dace in Montana, using four different methods. The number of historical streams currently occupied by pearl dace or chrosomid dace (c) was inferred following a different set of assumptions for each method. The percent declines were calculated by dividing the number of currently occupied historical streams (c) by h (either the number of historical streams sampled [$N_{pearl} = 24$, $N_{chrosomid} = 70$] or the total number of historical streams [$N_{pearl} = 30$, $N_{chrosomid} = 119$]), subtracting the quotient from 1, and multiplying by 100.

	Estimation method			
Variable	Sampled only	Liberal	Conservative	Non-native predator
Pearl dace				
Occupied historical streams (c)	5	5	11	9
Total historical streams (h)	23	30	30	30
Percent decline (d)	78.3	83.3	63.3	70.0
Chrosomid dace				
Occupied historical streams (c)	41	41	85	71
Total historical streams (h)	73	125	125	125
Percent decline (d)	43.8	67.2	32.0	43.2

52

Table 6. Comparison of eight habitat variables at streams where chrosomid dace were collected ($N = 45$) and streams where chrosomid dace were not collected ($N = 81$). Habitat variables that were normally distributed were analyzed with a t-test (T) and habitat variables that were not normally distributed were analyzed with a Wilcoxon rank-sum test (W).

Habitat variable	Test statistic	p-value
Mean width	$W_{91} = 903$	0.573
Mean depth	$T_{79} = -0.124$	0.902
Proportion fine substrates	$W_{88} = 1245$	0.022[1]
Water temperature	$T_{52} = -0.579$	0.565
Log turbidity	$T_{60} = 0.383$	0.703
pH	$W_{93} = 943$	0.727
Dissolved Oxygen	$T_{55} = -1.164$	0.249
Log conductivity	$T_{61} = -0.207$	0.836

[1] Significant at an alpha level of 0.05; however, the equality of variance assumption was violated (Levene's-test: $F_{1, 88} = 6.43$, $P = 0.013$)

Figure 1. Inferred historical distribution of pearl dace in Montana based on vouchered (solid squares), published (solid circles), and unpublished (open circles) collections. Gray lines demarcate delineated sections (labelled). Unreliable historical collections are also labelled. The dashed yellow line demarcates the southern boundary of the Northwest Glaciated Plains Ecoregion.

53

Figure 2. Inferred historical distribution of chrosomid dace in Montana based on vouchered (solid squares), published (solid circles), and unpublished (open circles) collection records. Gray lines demarcate delineated sections (labelled).

Figure 3. The calculated probability of vouchering at least one hybrid dace based on the number of chrosomid dace voucher specimens collected for three different proportions of hybrid dace. The dashed line shows the 90% probability cutoff and the dotted line shows the 95% probability cutoff.

Figure 4. Variability of eight habitat variables at streams in the inferred historical distribution of pearl dace where pearl dace were (*N* = 5) and were not collected (*N* = 44). Boxplots show the median, outliers, and interquartile range of habitat variables and numbers denote sample sizes. Habitat variables could not be compared statistically because of the highly unequal sample sizes, resulting in violations of the equal variance assumption.

Figure 5. Pearl dace collecions, 2015-2016. The solid black circles show where pearl dace were collected and the red diagonal cross shows Eagle Creek, where pearl dace were apparently extirpated between July 2015 and September 2016. The gray polygon shows the inferred historical distribution of pearl dace in Montana.

Figure 6. Inset of Snake Creek drainage. Collections are displayed as follows: black circled diagonal cross, fish but no pearl dace or influential non-native predators; solid black circles, pearl dace; red inverted triangles, collections of influential non-native predators from the Montana Fisheries Information System. The 10-digit hydrologic unit codes (HUC 10) are outlined in grey.

Figure 7. Inset of upper Big Muddy Creek drainage. Collections are displayed as follows: black circled diagonal crosses, fish but no pearl dace or influential non-native predators; solid black circles, pearl dace; red diagonal cross, pearl dace apparently extirpated; red upright triangles, influential non-native predators; red inverted triangles, collections of influential non-native predators from the Montana Fisheries Information System. The 10-digit hydrologic unit codes (HUC 10) are outlined in grey.

Figure 8. Inset of Willow Creek drainage in the inferred historical distribution of pearl dace in Montana. Collections are displayed as follows: solid black circles, pearl dace; solid black squares, pearl dace collected by the Blackfeet Environmental Office in 2017; red upright triangles, influential non-native predators. The 10-digit hydrologic unit codes (HUC 10) are outlined in grey.

Figure 9. Chrosomid dace collections, 2015-2016. The black circles show where only Northern Redbelly Dace were collected, and the black and white circles show where both Northern Redbelly Dace and hybrid dace were collected. The gray polygon shows the inferred historical distribution of chrosomid dace in Montana.

Figure 10. All sites sampled in the inferred historical distribution of pearl dace in Montana, 2015 and 2016. Collections are displayed as follows: open circles, no fish; black circled diagonal crosses, fish but no pearl dace or influential non-native predators; solid black circles, pearl dace; red diagonal cross, pearl dace apparently extirpated; red triangles, influential non-native predators. The gray polygon shows the inferred historical distribution of pearl dace. Sites were jittered a maximum of 12 km, with most sites moving < 5 km.

Figure 11. All sites sampled in the inferred historical distribution of chrosomid dace in Montana, 2015 and 2016. Collections are displayed as follows: open circles, no fish; black circled diagonal crosses, fish but no chrosomid dace or influential non-native predators; solid black circles, chrosomid dace; black and red circles, chrosomid dace and influential non-native predators; red triangles, influential non-native predators. Sites were jittered a maximum of 25 km, with most sites moving < 5 km.

Figure 12. Collections of non-native predators from 140 sites in the inferred historical distribution of chrosomid dace, including 88 sites in the inferred historical distribution of pearl dace, 2015-2016.

EFFECTS OF INVASIVE NORTHERN PIKE ON MONTANA PRAIRIE FISH

ASSEMBLAGES

Introduction

Non-native predatory fishes threaten freshwater biodiversity globally. Predatory game fishes were and continue to be stocked widely to provide recreational and commercial fishing opportunities, despite strong evidence of their negative effects on aquatic communities (Rahel 2000; Eby et al. 2006; Gherardi 2007; Vitule et al. 2009; Fausch and García-Berthou 2013). Introduced piscivores were implicated in the extinction of multiple freshwater fishes (Ogutu-Ohwayo 1993; Patankar et al. 2006) and were associated with reduced native fish species richness and abundance in both lentic (Chapleau et al. 1997; Whittier et al. 1997; Findlay et al. 2000; MacRae and Jackson 2001) and lotic systems (Rincon et al. 1990; Simon and Townsend 2003; Spurgeon et al. 2014; Sepulveda et al. 2015).

Northern Pike *Esox lucius* are one such widely-stocked game fish that negatively affects native fish assemblages. Northern Pike are voracious predators that can reduce and extirpate multiple native fishes in a single system (Eklöv and Hamrin 1989; Beaudoin et al. 1999; Sepulveda et al. 2013). Unsurprisingly, introduced Northern Pike were associated with low abundances or extirpations of native fishes in lentic (Tonn and Magnuson 1982; He and Kitchell 1990; Kitchell et al. 1994; Patankar et al. 2006; Byström et al. 2007; Haught and von Hippel 2011) and lotic systems (Rincon et al. 1990;

Sepulveda et al. 2015), including in two prairie drainages (Labbe and Fausch 2000; Spurgeon et al. 2014).

Northern Pike are not native to Montana outside of the Saskatchewan River drainage (Brown 1971; Holton and Johnson 2003), but were widely introduced in the state (MFISH 2016). Northern Pike subsequently dispersed from stocking locations and are now widespread throughout most drainages containing and connected to those stocked reservoirs (MFISH 2016). Despite speculation that Northern Pike would decrease native fish species richness in prairie streams of Montana (Stagliano 2005; R. Bramblett, Montana State University, personal communication), the extent to which Northern Pike have affected Montana prairie stream fish assemblages has not been evaluated.

Relatively little is known about Northern Pike in prairie streams. Prairie streams have highly variable flow, turbidity, and water temperature (Dodds et al. 2004), which are not typically associated with Northern Pike habitat in their native range (Scott and Crossman 1973; Becker 1983). Despite this, Northern Pike were collected from prairie streams in Montana (Guzevich 1993; MFISH 2016), Colorado (Labbe and Fausch 2000), South Dakota (Harland and Berry 2004), and Nebraska (Harland and Berry 2004; Spurgeon et al. 2014), and probably occur in prairie streams in other states where they have been stocked. Reduced native species richness and occurrence in prairie streams that contained Northern Pike were briefly noted by Labbe and Fausch (2000) and Spurgeon et al. (2014), but neither investigation focused on differences in fish assemblage between sites with and without Northern Pike; my study is the first to do so. My objectives were to (1) evaluate whether prairie streams with and without non-native Northern Pike had

different native species richness, and (2) evaluate whether prairie streams with and

without non-native Northern Pike had different fish assemblages.

Methods

Study Area

The study area consisted of fourth-order and smaller prairie streams in the Milk

River drainage, Missouri River drainage in Montana from the Fort Peck dam to the North

Dakota border (hereafter Missouri River drainage), lower Yellowstone River drainage

(from the Bighorn River confluence to the North Dakota border), and Little Missouri

River drainage in Montana (Figure 13). Ninety-one of the streams are located in the

Northwestern Glaciated Plains Ecoregion and 97 are located in the Northwestern Great

Plains Ecoregion (Omernik and Griffith 2014), both of which have prevalent agricultural

activity and some oil development. The potential native fish species pool is similar in the

two ecoregions; however, pearl dace are found only in the Northwestern Glaciated Plains

Ecoregion (Chapter 1) and Creek Chub are found mostly in the Northwestern Great

Plains Ecoregion (Holton and Johnson 2003) but also occur in some tributaries of the

Missouri River in the Northwestern Glaciated Plains Ecoregion.

Data Acquisition and Processing

I gathered fish collection data from six sources with fish collection records across

the Northwestern Glaciated Plains and Northwestern Great Plains Ecoregions in Montana

using multiple gear types (Table 7). Some sources reported catch per unit effort (CPUE);

however, I only used presence-absence data to preclude gear and effort biases.

68

I winnowed the dataset a priori to ensure that the data were reasonably

independent and represented the study area of interest. I restricted my scope of interest to

fourth-order and smaller prairie streams from drainages where Northern Pike were widely

introduced or widely collected (MFISH 2016). I removed sites that were within 5 km of a

confluence with a river (fifth-order and larger; Mullen et al. 2011). If repeat observations

were made at a site, I only included the most recent collection record. I then selected one

site per HUC 10 (10-digit hydrologic unit code; USGS 2015) to account for spatial

autocorrelation among collections; sites with Northern Pike were preferentially selected

over sites without Northern Pike, but the selections were otherwise random. The

winnowing process resulted in 49 sites with and 139 sites without Northern Pike.

Analyses

Native Species Richness. I examined whether native species richness differed

between 49 sites with and 139 sites without non-native Northern Pike, after accounting

for watershed area, with a Poisson regression from the car package (Fox and Weisburg

2011) in R version 3.4.3 (R Development Core Team 2017). Watershed area was

included as a covariate because fish species richness increases with watershed area

(Fausch et al. 1984; Karr et al. 1986). The 24 native fish taxa were categorized as native

minnows ($N = 10$) or native non-minnows ($N = 14$; Table 8), and I ran separate Poisson

regressions on the species richness of each category. Three species from the *Hybognathus*

genus, and two taxa from the *Chrosomus* genus, were combined for the native minnow

species richness analysis (i.e., hybognathid minnows and chrosomid dace) because of

potential uncertain identifications to the species level. I did not detect an interaction

between the presence of Northern Pike and watershed area in either regression and therefore did not include an interaction term in the reported models. The assumptions for both Poisson regressions seemed to be reasonably met; I found low multicollinearity based on variance inflation factors, suggesting that the pike presence and watershed area variables were reasonably independent (Garson 2012), and low overdispersion based on the ratio of the residual deviance to the residual degree of freedom being close to one, suggesting that my sites were reasonably independent (Hinde and Demétrio 1998).

Native Fish Assemblages. I examined whether different native fish assemblages existed between sites with and without Northern Pike and identified species that best distinguished between these fish assemblages with a discriminant function analysis (DFA) from the Mass (Venables and Ripley 2002) and ade4 packages (Dray and Dufour 2007) in R version 3.4.3 (R Development Core Team 2017). I did not have a large enough sample size to run a DFA on all 24 native taxa, for which I needed at least 72 sites with and 72 sites without Northern Pike (3 × the number of discriminating taxa; Williams and Titus 1988; McGarigal et al. 2000), but I only had 49 sites with Northern Pike. Therefore, I ran separate DFAs on native minnow taxa and on native non-minnow species because the minimum sample size requirements were met ($N_{minnow} = 3 \times 10 = 30$, $N_{non\text{-}minnow} = 3 \times 14 = 42$). I calculated misclassification rates for both DFAs with a leave-one-out (jackknife) validation approach because it results in unbiased prediction errors compared to the common resubstitution approach (Olden and Jackson 2002).

Results

Native Species Richness

Mean species richness of native minnows was 52% lower (95% CI = 38% to 63%) at sites with Northern Pike than at sites without Northern Pike ($\chi^2{}_1$= 33.4; $P <$ 0.001; Figure 14), after accounting for the effect of watershed area. I did not detect a difference in mean species richness of native non-minnows between sites with and without Northern Pike ($\chi^2{}_1$= 0.267; P = 0.604; Figure 15). Mean native minnow species richness and mean native non-minnow species richness both increased with increasing watershed area (native minnows: $\chi^2{}_1$= 45.1, $P <$ 0.001; native non-minnows: $\chi^2{}_1$= 45.6, P < 0.001), after accounting for the presence of Northern Pike.

Native Fish Assemblages

Native minnow taxa correctly discriminated 77.1% of sites overall, 49.0% of sites with Northern Pike, and 87.1% of sites without Northern Pike based on jackknife validation (Figure 16). Sites were best distinguished based on presence of Fathead Minnows, which were more common at sites without than with Northern Pike; no other native minnow taxa were strongly associated with the canonical axis (Table 9).

Native non-minnow species correctly discriminated 78.7% of sites overall, 24.5% of sites with Northern Pike, and 97.8% of sites without Northern Pike based on jackknife validation (Figure 17). Sites were best distinguished based on presence of Shorthead Redhorse, which were more common at sites with than without Northern Pike, and the presence of Brook Stickleback, which were more common at sites without than with

Northern Pike (Table 10); however, neither of these, nor any other native non-minnow, was strongly associated with the canonical axis.

Discussion

Northern Pike predation probably caused the reduced native minnow species richness I observed in prairie streams in Montana with Northern Pike. However, causality cannot be determined from an observational study such as this one because the observed patterns could result from uncontrolled confounding variables such as a difference in habitat preferences between some native minnows and Northern Pike. Nevertheless, two strong lines of evidence implicate Northern Pike predation as the causal factor. First, predation is an important driver of fish assemblage structure (Juanes et al. 1996), and non-native predators can have particularly strong effects on aquatic communities (Ogutu-Ohwayo 1990; He and Wright 1992; Vitule et al. 2009; Cucherousset and Olden 2011). Moreover, Northern Pike are known for reducing small fish abundance and richness, even in systems with complex habitat and abundant water (He and Kitchell 1990; Kitchell et al. 1994; Haught and von Hippel 2011; Nicholson et al. 2015; Sepulveda et al. 2015). Second, biotic interactions seem to be particularly strong in intermittent pool environments common in prairie streams (Power et al. 1985; Matthews 1988; Statzner et al. 1988). Prey fish in isolated refugia pools are highly vulnerable to predators (Power et al. 1985; Schlosser 1987; Lohr and Fausch 1996; Labbe and Fausch 2000) and are unable to find sufficient refuge or emigrate, as they can in perennial systems (He and Kitchell 1990; Kitchell et al. 1994; Fraser et al. 1995). Therefore, Northern Pike probably reduce

native fish abundances and richness more quickly in intermittent reaches than perennial reaches of prairie streams, or than in systems with more refuge and water (He and Kitchell 1990; Kitchell et al. 1994; Nicholson et al. 2015).

Northern Pike predation seemingly affects native minnows more strongly than native non-minnows in Montana prairie streams. Northern Pike are capable of extirpating small-bodied fishes because they can consume all age classes thereof (Tonn et al. 1993; Chapleau et al. 1997; Nicholson et al. 2015), but are unlikely to completely extirpate large-bodied fishes because some individuals are of a size sufficient to escape predation (Tonn et al. 1993). The native minnows included in this study were all small-bodied (rarely achieving total lengths > 200 mm, Holton and Johnson 2003), making them vulnerable to Northern Pike predation. In contrast, most of the native non-minnows were large-bodied (often achieving total lengths > 300 mm, Holton and Johnson 2003) and more rare than native minnows. These two factors probably explains the lack of a relationship between Northern Pike presence and native non-minnow species richness.

Native fishes that spend part of their time in large rivers are probably somewhat resilient to Northern Pike predation effects. Prairie rivers probably have more available refuge than prairie streams, thereby allowing prey fishes to more easily avoid detection by non-native predators in rivers than in streams (*sensu* Juanes et al. 1996). Additionally, prairie rivers probably have fewer Northern Pike than many of the lakes and streams where they have reduced and eliminated native fishes. Therefore, native fish populations in large rivers may act as source populations to sink populations in streams with Northern Pike (*sensu* Woodford and McIntosh 2010). Source-sink dynamics may help explain the

coexistence of some potentially sensitive native fishes and Northern Pike in prairie streams.

Sites with Northern Pike had a somewhat different fish assemblage than sites without Northern Pike, but limitations of the dataset made this difference difficult to detect, as evidenced by the overlap between sites with and without Northern Pike along the canonical axes. First, the model had more information about the sites without than with Northern Pike because of the disparate sample sizes, and the full variation of Northern Pike sites may not have been captured in the data. Therefore, the misclassification rate for sites with Northern Pike was higher than for sites without for the native minnow analysis.

Second, the limited observations of some native fishes (e.g., Northern Pearl Dace, Smallmouth Buffalo, Sauger) probably did little to distinguish between sites with and without Northern Pike. The model had little information about the types of streams these rarely-observed fishes occur in, and their few observations probably obscured the model results. If more observations of these fishes were made, I would expect to see more discrimination between sites with and without Northern Pike.

Third, the model did not account for covariates that could help explain the presence of certain native fishes (e.g., watershed area, ecoregion [*sensu* Stagliano 2005], time since invasion). For example, the native minnow and native non-minnow species richness analyses showed that watershed area explains variation in native species richness, after accounting for the presence of Northern Pike. However, DFAs cannot

include information about covariates. Therefore, some of the overlap between sites with and without pike may have resulted from the unexplained variation from covariates.

Fourth, the use of coarse presence-absence data for Northern Pike and native fishes probably resulted in a simplified understanding of the relationship between the two. A species was considered present even if only a few individuals remained at a site; however, the species could have been at significant risk of subsequent extirpation. Additionally, the effect of a non-native species is often correlated with its abundance (Parker et al. 1999); therefore, I expect that some sites with Northern Pike that overlap with sites without Northern Pike on the canonical axes probably had low abundances of Northern Pike. Much less overlap would be expected if abundance data had been used for this analysis.

Finally, detection probabilities may have differed among collectors, who used different gear types and crews over 40-year time span. Five sources used the Montana Prairie Fish and Habitat Sampling Protocol or a similar protocol, but the 37 sites sampled by Elser et al. (1980) were sampled using a wide variety of sampling methods that could have resulted in variable detection probabilities of certain species resulting from gear selectivity.

The severe consequences resulting from Northern Pike introductions should warrant caution when considering future stocking or removal of barriers to Northern Pike passage. Moreover, removal of Northern Pike may not evoke unassisted recolonization by prey fishes, at least in lentic ecosystems (He and Kitchell 1990; Kitchell et al. 1994; Nicholson et al. 2015). Direct management actions may be necessary to reverse effects of

Northern Pike. Additionally, complete removal of already established populations of non-native species is often impossible (Cucherousset and Olden 2011; but see Kruse et al. 2013), and attempted control is expensive, unending, and only effective on local scales (Fausch and Garcia-Berthou 2013). Therefore, preventing further expansion and limiting new introductions of Northern Pike are probably the most viable management tools to limit their effect on native prairie fish assemblages.

In conclusion, my findings suggest that Northern Pike predation reduced native minnow species richness in the prairie streams they invaded in Montana. Additional studies incorporating abundance data may provide a more robust assessment of the negative effects of Northern Pike on native minnows, and may find potentially significant effects on native non-minnow species as well. Further expansion of Northern Pike will negatively affect the native prairie fish assemblages in the newly-invaded drainages and may lead to the extirpation of sensitive species (e.g. Northern Pearl Dace, Chapter 1). Exclusion of Northern Pike from drainages where they have not yet invaded will enhance maintenance of native prairie minnow assemblages in those drainages.

Table 7. Sources of compiled fish collection data used in the Poisson regression and discriminant function analyses.

Source	Number of sites	Sampling method[a]	Years of data collection
Elser et al. 1980	37	M	1976-1980
Bramblett et al. 2005	19	S1	1999-2001
Bramblett 2010	99	S2	2002-2007
Davis et al. 2010	3	S2	2005-2006
Mullen et al. 2011	5	S2	2005-2007
Chapter 1	25	S2*	2015-2016

[a] M = used electrofishing boats, boat-mounted mobile anode electrofishers, bank electrofishers, gillnets, seines, baited fish traps, or dipnets, or a combination of several gears to sample unknown reach lengths; S1 = used seines, dipnets, or both seines and dipnets to sample 40 times the max stream width or 150 m minimum to 500 m maximum; S2 = used seines, dipnets, or both seines and dipnets to sample 300 m following the Montana Prairie Fish and Habitat Sampling Protocol; S2* = used seines, dipnets, both seines and dipnets, or backpack electrofishers to sample 300 m following the Montana Prairie Fish and Habitat Sampling Protocol.

Table 8. Family, scientific and common names, number of sites, and frequency of occurrence (%) of 10 native minnow and 14 native non-minnow taxonomic groups collected from 188 sites on 4th order and smaller prairie streams in the Milk, Missouri, lower Yellowstone, and Little Missouri River drainages, 1976-2016.

Family	Scientific name	Common name	Number of sites	%
		Native minnows		
Cyprinidae	*Chrosomus eos* and *C. eos* × *C. neogaeus*	Northern Redbelly Dace and Northern Redbelly × Finescale Dace hybrid (chrosomid dace)	14	7.4
	Couesius plumbeus	Lake Chub	63	33.5
	Hybognathus argyritis, H. hankinsoni, and *H. placitus*	Western Silvery Minnow, Brassy Minnow, and Plains Minnow (hybognathid minnows)	63	33.5
	Margariscus nachtriebi	Northern Pearl Dace	2	1.1
	Notropis atherinoides	Emerald Shiner	3	1.6
	Notropis stramineus	Sand Shiner	33	17.6
	Pimephales promelas	Fathead Minnow	145	77.1
	Platygobio gracilis	Flathead Chub	22	11.7
	Rhinichthys cataractae	Longnose Dace	39	20.7
	Semotilus atromaculatus	Creek Chub	30	16.0
		Native non-minnows		
Hiodontidae	*Hiodon alosoides*	Goldeye	6	3.2
Catostomidae	*Carpiodes carpio*	River Carpsucker	13	7.0
	Catostomus catostomus	Longnose Sucker	3	1.6
	Catostomus commersonii	White Sucker	99	52.7
	Catostomus platyrhynchus	Mountain Sucker	8	4.3
	Ictiobus bubalus	Smallmouth Buffalo	1	0.5
	Ictiobus cyprinellus	Bigmouth Buffalo	1	0.5
	Moxostoma acrolepidotum	Shorthead Redhorse	20	10.6
Ictaluridae	*Ictalurus punctatus*	Channel Catfish	11	5.9
	Noturus flavus	Stonecat	12	6.4
Lotidae	*Lota lota*	Burbot	1	0.5

Table 8 continued

Lotidae	*Lota lota*	Burbot	1	0.5
Gasterosteidae	*Culaea inconstans*	Brook Stickleback	54	28.7
Percidae	*Etheostoma exile*	Iowa Darter	15	8.0
	Sander canadensis	Sauger	1	0.5

Table 9. Standardized canonical coefficients from the discriminant function analysis of sites with and without Northern Pike based on native minnow taxa. Minnow taxa were negatively correlated with sites with Northern Pike if they had negative coefficients, and positively correlated with sites with Northern Pike if they had positive coefficients.

Taxa	Standardized canonical coefficient	Number of sites where present	Number of sites with Northern Pike
Fathead Minnow	-0.655	145	25
Lake Chub	-0.277	63	10
Hybognathid minnows	-0.262	63	7
Creek Chub	-0.156	30	4
Chrosomid dace	-0.085	14	1
Sand Shiner	-0.027	33	8
Flathead Chub	0.038	22	7
Pearl Dace	0.181	2	1
Longnose Dace	0.302	39	12
Emerald Shiner	0.457	3	3

Table 10. Standardized canonical coefficients from the discriminant function analysis of sites with and without Northern Pike based on native non-minnow species. Non-minnow species were negatively correlated with sites with Northern Pike if they had negative coefficients, and positively correlated with sites with Northern Pike if they had positive coefficients.

Taxa	Standardized canonical coefficients	Number of sites where present	Number of sites with Northern Pike
Brook Stickleback	-0.476	54	5
River Carpsucker	-0.369	13	4
Longnose Sucker	-0.299	3	1
Channel Catfish	-0.072	11	4
Mountain Sucker	-0.006	8	4
Sauger	0.008	1	1
Stonecat	0.057	12	7
Iowa Darter	0.091	15	6
White Sucker	0.204	99	34
Goldeye	0.230	6	5
Burbot	0.236	1	1
Smallmouth Buffalo	0.310	1	1
Bigmouth Buffalo	0.342	1	1
Shorthead Redhorse	0.492	20	13

Figure 13. Locations of the 139 sites without Northern Pike (blue circles) and 49 sites with Northern Pike (orange triangles) used in the Poisson regression and discriminant function analyses Sites were located in the Milk River, Missouri River (downstream of Fort Peck dam), lower Yellowstone River (downstream of Bighorn River confluence), and Little Missouri drainages.

Figure 14. Modeled relationships of mean native minnow species richness and watershed area and associated 95% confidence bands at sites without ($N = 139$) and with Northern Pike ($N = 49$) derived by Poisson regression. Mean native minnow species richness at sites without Northern Pike was not plotted above 10 (the maximum number of native minnow species) and was not extrapolated past the largest observed watershed area of sites without Northern Pike.

Figure 15. The modeled relationship from the Poisson regression of native non-minnow species richness and watershed area at sites without Northern Pike ($N = 139$) and at sites with Northern Pike ($N = 49$). Mean native non-minnow species richness and associated 95% confidence bands are shown. Mean native non-minnow species richness at sites without Northern Pike was not extrapolated past the largest observed watershed area of sites without Northern Pike.

Figure 16. Differentiation of sites without Northern Pike ($N = 139$) and sites with Northern Pike ($N = 49$) along the canonical axis (LD 1) from the discriminant function analysis for native minnows. The canonical axis was most associated with the presence of Fathead Minnows, with Fathead Minnows more common at sites on the left and less common at sites on the right (see Table 9).

Figure 17. Differentiation of sites without Northern Pike (N = 139) and sites with Northern Pike (N = 49) along the canonical axis (LD 1) from the discriminant function analysis for native non-minnows. The canonical axis was most associated with the presence of Shorthead Redhorse and Brook Stickleback, where Shorthead Redhorse were more common at sites on the right and Brook Stickleback were more common at sites on the left (see Table 10).

REFERENCES CITED

Aiken, J. K., C. S. Findlay, and F. Chapleau. 2012. Long-term assessment of the effect of introduced predatory fish on minnow diversity in a regional protected area. Canadian Journal of Fisheries and Aquatic Sciences 69:1798–1805.

Allan, J. D., and A. S. Flecker. 1993. Biodiversity conservation in running waters: identifying the major factors that threaten destruction of riverine species and ecosystems. BioScience 43:32–43.

Angers, B., and I. J. Schlosser. 2007. The origin of *Phoxinus eos-neogaeus* unisexual hybrids. Molecular Ecology 16:4562–4571.

AVMA (American Veterinary Medical Association). 2013. AVMA guidelines for the euthanasia of animals: 2013 edition. American Veterinary Medical Association, Schaumburg, Illinois. Available: https://www.avma.org/kb/policies/documents/euthanasia.pdf. (January 2018).

Barfoot, C. 1993. Longitudinal distribution of fishes and habitat in Little Beaver Creek, Montana. Master's thesis. Montana State University, Bozeman.

Beaudoin C. P., W. M. Tonn, E. E. Prepas, and L. I. Wassenaar. 1999. Individual specialization and trophic adaptability of Northern Pike (*Esox lucius*): an isotope and dietary analysis. Oecologia 120:386–396.

Becker, G. C. 1983. Fishes of Wisconsin. The University of Wisconsin Press, Madison.

Bemis, W. E., E. J. Hilton, B. Brown, R. Arrindell, A. M. Richmond, C. D. Little, L. Grande, P. L. Forey, and G. J. Nelson. 2004. Methods for preparing dry, partially articulated skeletons of Osteichthyans, with notes on making Ridewood dissections of the cranial skeleton. Copeia 2004:603–609.

Berkman, H. E., and C. F. Rabeni. 1987. Effect of siltation on stream fish communities. Environmental Biology of Fishes 18:285–294.

Bertrand, K. N., K. B. Gido, W. K. Dodds, J. N. Murdock, and M. R. Whiles. 2009. Disturbance frequency and functional identity mediate ecosystem processes in prairie streams. Oikos 118:917–933.

Bestgen, K. R. 1989. Distribution and notes on the biology of *Phoxinus eos* (Cyprinidae) in Colorado. The Southwestern Naturalist 34:225–231.

Bramblett, R. G. 2003. Fish and habitat sampling protocol for prairie streams. Montana Cooperative Fisheries Research Unit, Bozeman.

Bramblett, R. G. 2010. Synthesis of Montana prairie stream fish surveys, 1999-2007.

Report to Montana Department of Fish, Wildlife and Parks, Helena.

Bramblett, R. G., T. R. Johnson, A. V. Zale, and D. G. Heggem. 2005. Development and evaluation of a fish assemblage index of biotic integrity for northwestern Great Plains streams. Transactions of the American Fisheries Society 134:624–640.

Bramblett, R. G., and A. V. Zale. 2004. Status of Pearl Dace (*Margariscus margarita*) in Montana. Available: http://www.fisheriessociety.org/AFSmontana/PearlDace.html (May 2015).

Brown, C. 1971. Fishes of Montana. Big Sky Books, Bozeman, Montana.

Byström, P., J. Karlsson, P. Nilsson, T. Van Kooten, J. Ask, and F. Olofsson. 2007. Substitution of top predators: effects of pike invasion in a subarctic lake. Freshwater Biology 52:1271–1280.

Carim, K. J., L. A. Eby, C. A. Barfoot, and M. C. Boyer. 2016. Consistent loss of genetic diversity in isolated Cutthroat Trout populations independent of habitat size and quality. Conservation Genetics 17:1363–1376.

Chapleau, F., C. S. Findlay, and E. Szenasy. 1997. Impact of piscivorous fish introductions on fish species richness of small lakes in Gatineau Park, Québec. Ecoscience 4:259–268.

Clavero, M., and E. Garcia-Berthou. 2006. Homogenization dynamics and introduction routes of invasive freshwater fish in the Iberian peninsula. Ecological Applications 16:2313–2324.

Cope, E. D. 1879. A contribution to the zoology of Montana. The American Naturalist 13:432–441.

Coyle, S. D., R. M. Durborow, and J. H. Tidwell. 2004. Anesthetics in aquaculture. Southern Regional Aquaculture Center (SRAC) Publication 3900. Available: https://srac.tamu.edu/index.cfm/event/viewAllSheets/. (January 2018).

Cross, F. B., R. E. Moss, and J. T. Collins. 1985. Assessment of dewatering impacts on stream fisheries in the Arkansas and Cimarron Rivers. University of Kansas Museum of Natural History, Lawrence. Kansas Fish and Game Commission Nongame Wildlife Contract 46.

Cucherousset, J., and J. D. Olden. 2011. Ecological impacts of non-native freshwater fishes. Fisheries 36:215–230.

Cunningham, G. R. 1995. Life history traits and reproductive ecology of the Pearl Dace,

Margariscus margarita (Pisces: Cyprinidae), in the Sandhills of Nebraska. Master's thesis. University of Nebraska, Omaha.

Cunningham, G. R. 2006. Pearl Dace (*Margariscus margarita*): A technical conservation assessment. Report to the U.S. Forest Service, Rocky Mountain Region, Species Conservation Project, Golden, Colorado.

Cunningham, G. R., and S. M. Hickey. 1996. Habitat parameter determination and threatened fish inventory of Sandhills streams. Report to the Nebraska Natural Heritage Program. Nebraska Game and Parks Commission, Lincoln.

Davis, N. K. 2004. Extent and timing of Laurentide glacial lake Musselshell, central Montana. Master's thesis. Montana State University, Bozeman.

Davis, W. N., R. G. Bramblett, and A. V. Zale. 2010. Effects of coalbed natural gas development on fish assemblages in tributary streams of the Powder and Tongue rivers. Freshwater Biology 55:2612-2625.

DeBates, T. J., C. P. Paukert, and D. W. Willis. 2003. Fish community responses to the establishment of a piscivore, Northern Pike (*Esox lucius*), in a Nebraska Sandhill lake. Journal of Freshwater Ecology 18:353-359.

Dodds, W. K., K. Gido, M. R. Whiles, K. M. Fritz, and W. J. Matthews. 2004. Life on the edge: the ecology of Great Plains prairie streams. BioScience 54:205–216.

Dray, S., and A. B. Dufour. 2007. The ade4 package: implementing the duality diagram for ecologists. Journal of Statistical Software 22:1-20.

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

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

Eklöv, P., and S. F. Hamrin. 1989. Predatory efficiency and prey selection: interactions between pike *Esox lucius*, perch *Perca fluviatilis* and rudd *Scardinus erythrophthalmus*. Oikos 56:149–156.

Elser, A. A., M. W. Gorges, and L. M. Morris. 1980. Distribution of fishes in southeastern Montana. Report to the Montana Department of Fish, Wildlife and Parks, Helena.

ESRI (Environmental Systems Research Institute). 2014. ArcGis Release 10.2.2. Redlands, California.

Fausch, K. D., and K. R. Bestgen. 1997. Ecology of fishes indigenous to the central and southwestern Great Plains. Pages 131–166 *in* F. L. Knopf and F. B. Samson, editors. Ecology and conservation of Great Plains vertebrates. Springer-Verlag, New York.

Fausch, K. D., and E. García-Berthou. 2013. The problem of invasive species in river ecosystems. Pages 193-216 *in* S. Sabater and A. Elosegi, editors. River conservation: challenges and opportunities. Fundación BBVA, Bilbao, Spain.

Fausch, K. D., J. R. Karr, and P. R. Yant. 1984. Regional application of an index of biotic integrity based on stream fish communities. Transactions of the American Fisheries Society 113:39–55.

Fausch, K. D., B. E. Rieman, J. B. Dunham, M. K. Young, and D. P. Peterson. 2009. Invasion versus isolation: trade-offs in managing native salmonids with barriers to upstream movement. Conservation Biology 23:859–870.

Felts, E. 2013. Ecology of glacial relict fishes in South Dakota's Sandhills region. Master's thesis. South Dakota State University, Brookings.

Felts, E. A., and K. N. Bertrand. 2014. Conservation status of five headwater stream specialists in South Dakota. The American Midland Naturalist 172:131–159.

Findlay, C. S., D. G. Bert, and L. Zheng. 2000. Effect of introduced piscivores on native minnow communities in Adirondack lakes. Canadian Journal of Fisheries and Aquatic Sciences 57:570–580.

Fishnet2. 2017. Accessed through the Fishnet2 Portal. Available: www.fishnet2.net (November 2017).

Fraser, D. F., J. F. Gilliam, and T. Yip-Hoi. 1995. Predation as an agent of population fragmentation in a tropical watershed. Ecology 76:1461–1472.

Freeman, R., W. Bowerman, T. Grubb, A. Bath, G. Dawson, K. Ennis, and J. Giesy. 2002. Opening rivers to Trojan fish. The ecological dilemma of dam removal in the Great Lakes. Conservation in Practice 3:35–40.

Fox, J., and S. Weisberg. 2011. An R Companion to Applied Regression, 2nd Edition. Sage, Thousand Oaks, California.

Garson, G. D. 2012. Testing statistical assumptions. Statistical Associates Publishing, Asheboro, North Carolina.

Gherardi, F. 2007. Biological invaders in inland waters: an overview. Pages 3-26 *in* F. Gherardi, editor. Biological invaders in inland waters: profiles, distribution and threats, Volume 2. Springer, Dordrecht, The Netherlands.

Gidmark, N. J., and A. M. Simons. 2014. Cyprinidae: carps and minnows. Pages 354–450 *in* M. L. Warren, Jr., and B. M. Burr, editors. Freshwater fishes of North America volume 1: Petromyzontidae to Catostomidae. John Hopkins University Press, Baltimore, Maryland.

Goddard, K. A., O. Megwinoff, L. L. Wessner, and F. Giaimo. 1998. Confirmation of gynogenesis in *Phoxinus eos-neogaeus* (Pisces: Cyprinidae). Journal of Heredity 89:151–157.

Goddard, K., R. Dawley, T. Dowling, and J. Bogart. 1989. Origin and genetic relationships of diploid, triploid, and diploid-triploid mosaic biotypes in the *Phoxinus eos-neogaeus* unisexual complex. Pages 268–280 *in* R. M. Dawley and J. P. Bogart, editors. Evolution and ecology of unisexual vertebrates. New York State Museum, Albany.

Goertz, D. E., and R. D. Phoenix. 2015. Range extensions for Northern Redbelly Dace (*Chrosomus eos*), Fathead Minnow (*Pimephales promelas*), and Iowa Darter (*Etheostoma exile*) in Ontario, Canada. The Canadian Field-Naturalist 129:237–244.

Gotelli, N. J., and A. M. Ellison. 2013. A primer of ecological statistics, 2nd edition. Sinauer Associates, Sunderland, Massachusetts.

Gould, W. R., and C. J. D. Brown. 1968. Fishes of the genus *Semotilus* (Cyprinidae) from Montana. Copeia 1968:628–629.

Guzevich, J. W. 1993. The relationship of physical habitat to the distribution of Northern Pike and Walleye in two Montana prairie streams. Master's thesis. Montana State University, Bozeman.

Hanski, I., and D. Simberloff. 1997. The metapopulation approach, its history, conceptual domain and application to conservation. Pages 5–26 *in* I. Hanski and M. E. Gilpin, editors. Metapopulation biology. Academic Press, London.

Harland, B., and C. R. Berry. 2004. Fishes and habitat characteristics of the Keya Paha River, South Dakota-Nebraska. Journal of Freshwater Ecology 19:169–177.

Haught, S., and F. A. von Hippel. 2011. Invasive pike establishment in Cook Inlet basin lakes, Alaska: diet, native fish abundance and lake environment. Biological Invasions 13:2103–2114.

92

He, X., and J. F. Kitchell. 1990. Direct and indirect effects of predation on a fish community: a whole-lake experiment. Transactions of the American Fisheries Society 119:825–835.

He, X., and R. A. Wright. 1992. An experimental study of piscivore-planktivore interactions: population and community responses to predation. Canadian Journal of Fisheries and Aquatic Sciences 49:1176–1183.

Henderson, N. E., and R. E. Peter. 1969. Distribution of fishes of southern Alberta. Journal of the Fisheries Research Board of Canada 26:325–338.

Hinde, J., and C. G. B. Demétrio. 1998. Overdispersion: models and estimation. Computational Statistics & Data Analysis 27:151-170.

Hoagstrom, C. W., and C. R. Berry. 2006. Island biogeography of native fish faunas among Great Plains drainage basins: basin scale features influence composition. American Fisheries Society, Symposium 48, Bethesda, Maryland.

Hoagstrom, C. W., J. E. Brooks, and S. R. Davenport. 2011. A large-scale conservation perspective considering endemic fishes of the North American plains. Biological Conservation 144:21–34.

Holton, G., and H. Johnson. 2003. A field guide to Montana fishes, 3rd edition. Montana Fish, Wildlife and Parks, Helena.

Hubbs, C. L. 1942. Sexual dimorphism in the cyprinid fishes, *Margariscus* and *Couesius*, and alleged hybridization between these genera. Occasional Papers of the Museum of Zoology, University of Michigan 468:1–6.

Hunziker, J., T. Haddix, and L. Holte. 2015. 2014 Annual report Pallid Sturgeon population assessment and associated fish community monitoring for the Missouri River: segment 2. Annual report to the U.S. Army Corps of Engineers, Missouri River Recovery Program, Omaha, Nebraska.

IPCC (Intergovernmental Panel on Climate Change). 2014. Climate change 2014: synthesis report. Contribution of working groups I, II, and III to the fifth assessment report of the IPCC, Geneva, Switzerland.

Jackson, C. R., and C. M. Pringle. 2010. Ecological benefits of reduced hydrologic connectivity in intensively developed landscapes. BioScience 60:37–46.

Jackson, D. A., K. M. Somers, and H. H. Harvey. 1992. Null models and fish communities: evidence of nonrandom patterns. The American Naturalist 139:930–951.

Jaeger, M. E., A. V. Zale, T. E. McMahon, and B. J. Schmitz. 2005. Seasonal movements, habitat use, aggregation, exploitation, and entrainment of Saugers in the lower Yellowstone River: an empirical assessment of factors affecting population recovery. North American Journal of Fisheries Management 25:1550–1568.

Jelks, H. L., S. J. Walsh, N. M. Burkhead, S. Contreras-Balderas, E. Diaz-Pardo, D. A. Hendrickson, J. Lyons, N. E. Mandrak, F. McCormick, J. S. Nelson, S. P. Platania, B. A. Porter, C. B. Renaud, J. J. Schmitter-Soto, E. B. Taylor, and M. L. Warren. 2008. Conservation status of imperiled North American freshwater and diadromous fishes. Fisheries 33:372–407.

Joswiak, G. R., and J. G. New. 1989. Multivariate analysis of the hybrid *Phoxinus eos × Phoxinus neogaeus* (Pisces: Cyprinidae): reevaluation of univariate analysis. Canadian Journal of Zoology 67:1637–1641.

Juanes, F., J. A. Buckel, and F. S. Scharf. 1996. Feeding ecology of piscivorous fishes. Pages 267–283 *in* P. J. Hart and J. D. Reynolds, editors. Handbook of Fish Biology and Fisheries, Volume 1. Blackwell Publishing, Malden, Massachusetts.

Karr, J. R., K. D. Fausch, P. L. Angermeier, P. R. Yant, and I. J. Schlosser. 1986. Assessing biological integrity in running waters: a method and its rationale. Illinois Natural History Survey Special Publication #5, Champaign.

Kitchell, J. F., L. A. Eby, X. He, D. E. Schindler, and R. A. Wright. 1994. Predator-prey dynamics in an ecosystem context. Journal of Fish Biology 45:209–226.

Kreutzenberger, K., F. Leprieur, and S. Brosse. 2008. The influence of the invasive Black Bullhead *Ameiurus melas* on the predatory efficiency of pike *Esox lucius* L. Journal of Fish Biology 73:196–205.

Kruse, C. G., P. Clancey, S. Barndt, K. Patten, and B. Shepard. 2013.Setting the stage for conservation success: large-scale watershed renovation and re-introduction of cutthroat trout in the Rocky Mountain region of the USA. Pages 26-32 *in* P. S. Soorae, editor. Global re-introduction perspectives: 2013. IUCN/SSC Re-introduction Specialist Group and Environmental Agency, Abu Dhabi, United Arab Emirates. Available: www.iucnsscrsg.org.

Labbe, T. R., and K. D. Fausch. 2000. Dynamics of intermittent stream habitat regulate persistence of a threatened fish at multiple scales. Ecological Applications 10:1774–1791.

Lazorchark, J. M., D. J. Klemm, and D. V Peck. 1998. Environmental monitoring and assessment program-surface waters: field operations and methods for measuring the ecological condition of wadeable streams. USEPA report 620/R-94/004F.

Washington, D.C.

Lee, D., C. Gilbert, C. Hocutt, R. Jenkins, D. McAllister, and J. Stauffer, Jr. 1980. Atlas of North American freshwater fishes. North Carolina Museum of Natural History, Raleigh.

Lesica, P., and F. W. Allendorf. 1995. When are peripheral populations valuable for conservation? Conservation Biology 9:753–760.

Lohr, S. C., and K. D. Fausch. 1996. Effects of Green Sunfish (*Lepomis cyanellus*) predation on survival and habitat use of Plains Killifish (*Fundulus zebrinus*). The Southwestern Naturalist 41:155–160.

MacRae, P. S. D., and D. A. Jackson. 2001. The influence of Smallmouth Bass (*Micropterus dolomieu*) predation and habitat complexity on the structure of littoral zone fish assemblages. Canadian Journal of Fisheries and Aquatic Sciences 58:342–351.

Matthews, W. J. 1988. North American prairie streams as systems for ecological study. Journal of the North American Benthological Society 7:387–409.

McCulloch, B., J. Duncan, and R. Keith. 1998. Fish survey of the Saskatchewan portion of the Missouri River basin. Blue Jay 56:107–115.

McGarigal, K., S. Cushman, and S. Stafford. 2000. Multivariate statistics for wildlife and ecology research. Springer, New York.

Mee, J. A., C. Chan, and E. B. Taylor. 2013. Coexistence of sperm-dependent asexuals and their sexual hosts: The role of differences in fitness-related traits. Environmental Biology of Fishes 96:1111–1121.

Mee, J. A., and L. Rowe. 2010. Distribution of *Phoxinus eos*, *Phoxinus neogaeus*, and their asexually-reproducing hybrids (Pisces: Cyprinidae) in Algonquin Provincial Park, Ontario. PLoS ONE 5(10):1–6.

Mee, J. A., and E. B. Taylor. 2012. The cybrid invasion: widespread postglacial dispersal by *Phoxinus* (Pisces: Cyprinidae) cytoplasmic hybrids. Canadian Journal of Zoology 90:577–584.

Meffe, G. K. 1986. Conservation genetics and the management of endangered fishes. Fisheries 11(1):14–23.

Mercy, T. V. A., V. Malika, and S. Sajan. 2013. Use of tricaine methanesulfonate (MS-222) to induce anaesthesia in *Puntius denisonii* (Day, 1865) (Teleostei:

Cypriniformes: Cyprinidae), a threatened barb of the Western Ghats, India. Journal of Threatened Taxa 5:4414–4419.

MFISH (Montana Fisheries Information System). 2016. Montana Fish, Wildlife and Parks, Helena. Available: http://fwp.mt.gov/fishing/mFish/. (January 2016).

Mogen, J., and E. Best. 2011. Fish entrainment at the Saint Mary Diversion, Montana. Report to the U.S. Bureau of Reclamation, Montana Area Office, Billings.

Montana field guide. 2016. Montana Natural Heritage Program and Montana Fish, Wildlife and Parks. Available: http://fieldguide.mt.gov/speciesDetail.aspx?elcode=AFCJB54020. (January 2017).

Morey, N. M. 2004. New distributional record of the northern redbelly dace in the northern Great Plains. The Prairie Naturalist 36:257–260.

Mullen, J. A. 2007. Spatiotemporal variation in fish assemblages in Montana prairie streams. Master's thesis. Montana State University, Bozeman.

Mullen, J. A., R. G. Bramblett, C. S. Guy, A. V. Zale, and D. W. Roberts. 2011. Determinants of fish assemblage structure in northwestern Great Plains streams. Transactions of the American Fisheries Society 140:271–281.

NatureServe. 2011. NatureServe Explorer: An online encyclopedia of life version 7.1. NatureServe, Arlington, Virginia. Available: http://explorer.natureserve.org. (January 2018).

Nelson, J. S., and M. J. Paetz. 1970. The fishes of Alberta. The University of Alberta Press, Edmonton.

Nelson, J. S., and M. J. Paetz. 1992. The fishes of Alberta, 2nd edition. The University of Alberta Press, Edmonton.

New, J. G. 1962. Hybridization between two Cyprinids, *Chrosomus eos* and *Chrosomus neogaeus*. Copeia 1962:147–152.

Nicholson, M. E., M. D. Rennie, and K. H. Mills. 2015. Apparent extirpation of prey fish communities following the introduction of Northern Pike (*Esox lucius*). Canadian Field-Naturalist 129:165–174.

Oakes, R. M., K. B. Gido, J. A. Falke, J. D. Olden, and B. L. Brock. 2005. Modelling of stream fishes in the Great Plains, USA. Ecology of Freshwater Fish 14:361–374.

Ogutu-Ohwayo, R. 1990. The reduction in fish species diversity in lakes Victoria and

Kyoga (East Africa) following human exploitation and introduction of non-native fishes. Journal of Fish Biology 37:207–208.

Ogutu-Ohwayo, R. 1993. The effects of predation by Nile perch, *Lates niloticus* L., on the fish of Lake Nabugabo, with suggestions for conservation of endangered endemic cichlids. Conservation Biology 7:701–711.

Olden, J. D., and D. A. Jackson. 2002. A comparison of statistical approaches for modelling fish species distributions. Freshwater Biology 47:1976–1995.

Omernik, J. M., and G. E. Griffith. 2014. Ecoregions of the conterminous United States: evolution of a hierarchical spatial framework. Environmental Management 54:1249–1266.

Ostovar, K. 2012. Recolonization of fish in the Telegraph Creek basin. Report to the American Prairie Reserve, Bozeman, Montana.

Parker, I. M., D. Simberloff, W. M. Lonsdale, K. Goodell, M. Wonham, P. M. Kareiva, M. H. Williamson, B. Von Holle, P. B. Moyle, J. E. Byers, and L. Goldwasser. 1999. Impact: toward a framework for understanding the ecological effects of invaders. Biological Invasions 1:3–19.

Patankar, R., F. A. Von Hippel, and M. A. Bell. 2006. Extinction of a weakly armoured threespine stickleback (*Gasterosteus aculeatus*) population in Prator Lake, Alaska. Ecology of Freshwater Fish 15:482–487.

Patton, T. M., W. A. Hubert, F. J. Rahel, and K. G. Gerow. 2000. Effort needed to estimate species richness in small streams on the Great Plains in Wyoming. North American Journal of Fisheries Management 20:394–398.

Power, M. E., W. J. Matthews, and A. J. Stewart. 1985. Grazing minnows, piscivorous bass, and stream algae: dynamics of a strong interaction. Ecology 66:1448–1456.

Quist, M. C., F. J. Rahel, and W. A. Hubert. 2005. Hierarchical faunal filters: An approach to assessing effects of habitat and nonnative species on native fishes. Ecology of Freshwater Fish 14:24–39.

R Development Core Team. 2017. R 3.4.3: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna.

Rahel, F. J. 2000. Homogenization of fish faunas across the United States. Science 288:854–856.

Reynolds, J. B., and A. L. Kolz. 2012. Electrofishing. Pages 305–361 *in* A. V. Zale, D. L.

Parrish, and T. M. Sutton, editors. Fisheries Techniques, 3rd edition. American Fisheries Society, Bethesda, Maryland.

Ricciardi, A., and H. J. MacIsaac. 2011. Impacts of biological invasions on freshwater ecosystems. Pages 211–224 *in* D. M. Richardson, editor. Fifty years of invasion ecology: the legacy of Charles Elton. Wiley-Blackwell, Hoboken, New Jersey.

Rincon, P. A., J. C. Velasco, N. Gonzalez-Sanchez, and C. Pollo. 1990. Fish assemblages in small streams in western Spain: the influence of an introduced predator. Fundamental and Applied Limnology 118:81–91.

Sabaj, M. H. 2016. Standard symbolic codes for institutional resource collections in herpetology and ichthyology: an online reference. Version 6.5 (16 August 2016). Available: http://www.asih.org/resources/standard-symbolic-codes-institutional-resource-collections-herpetology-ichthyology.

Sabater, S., A. Elosegi, and D. Dudgeon. 2013. River conservation: going against the flow to meet global challenges. Pages 15–35 *in* S. Sabater and A. Elosegi, editors. River conservation: challenges and opportunities. Publidisa, Bilbao, Spain.

Sala, O. E., F. S. Chapman, III, J. J. Armesto, E. Berlow, J. Bloomfield, R. Dirzo, E. Huber-Sanwald, L. F. Huenneke, R. B. Jackson, A. Kinzig, R. Leemans, D. M. Lodge, H. A. Mooney, N. L. Poff, M. T. Sykes, B. H. Walker, M. Walker, and D. H. Wall. 2000. Global biodiversity scenarios for the year 2100. Science 287:1770–1775.

Samson, F., and F. Knopf. 1994. Prairie conservation in North America. BioScience 44:418-421.

Schlosser, I. J. 1987. The role of predation in age- and size- related habitat used by stream fishes. Ecology 68:651–659.

Schlosser, I. J., M. R. Doeringsfeld, J. F. Elder, and L. F. Arzayus. 1998. Niche relationships of clonal and sexual fish in a heterogeneous landscape. Ecology 79:953–968.

Schultz, L. 1941. Fishes of Glacier National Park. U. S. Department of the Interior Conservation Bulletin, Number 22, Washington, D.C.

Scott, W., and E. Crossman. 1973. Freshwater fishes of Canada. Galt House Publishing, Oakville, Ontario.

Sepulveda, A. J., D. S. Rutz, A. W. Dupuis, P. A. Shields, and K. J. Dunker. 2015. Introduced Northern Pike consumption of salmonids in southcentral Alaska.

Ecology of Freshwater Fish 24:519–531.

Sepulveda, A. J., D. S. Rutz, S. S. Ivey, K. J. Dunker, and J. A. Gross. 2013. Introduced Northern Pike predation on salmonids in southcentral Alaska. Ecology of Freshwater Fish 22:268–279.

Sheldon, A. L. 1988. Conservation of stream fishes: patterns of diversity, rarity, and risk. Conservation Biology 2:149–156.

Simon, K. S., and C. R. Townsend. 2003. Impacts of freshwater invaders at different levels of ecological organisation, with emphasis on salmonids and ecosystem consequences. Freshwater Biology 48:982–994.

Spurgeon, J. J., R. H. Stasiak, G. R. Cunningham, K. L. Pope, and M. A. Pegg. 2014. Status of native stream fishes within selected protected areas of the Niobrara River in western Nebraska. Great Plains Research 24:71–78.

Stagliano, D. M. 2005. Aquatic community classification and ecosystem diversity in Montana's Missouri River watershed. Report to the Bureau of Land Management. Montana Natural Heritage Program, Helena, Montana.

Stash, S. W. 2001. Distribution, relative abundance, and habitat associations of Milk River fishes related to irrigation diversion dams. Master's thesis. Montana State University, Bozeman.

Stasiak, R. H. 1972. The morphology and life history of the finescale dace, *Pfrille neogaea*, in Itasca State Park, Minnesota. PhD Dissertation, University of Minnesota, Minneapolis.

Stasiak, R. H. 2006. Northern Redbelly dace (*Phoxinus eos*): a technical conservation assessment. Report to the U.S. Forest Service, Rocky Mountain Region, Species Conservation Project, Golden, Colorado.

Statzner, B., J. A. Gore, and V. H. Resh. 1988. Hydraulic stream ecology: observed patterns and potential applications. Journal of the North American Benthological Society 7:307-360

Tallman, R. F., and J. H. Gee. 1982. Intraspecific resource partitioning in a headwaters stream fish, the Pearl Dace *Semotilus margarita* (Cyprinidae). Environmental Biology of Fishes 7:243–249.

Taylor, W. R. 1967. An enzyme method of clearing and staining small vertebrates. Proceedings of the United States National Museum 122(3596):1–17.

Thompson, P. D., and F. J. Rahel. 1998. Evaluation of artificial barriers in small Rocky Mountain streams for preventing the upstream movement of Brook Trout. North American Journal of Fisheries Management 18:206–210.

Tonn, W. M., and J. J. Magnuson. 1982. Patterns in the species composition and richness of fish assemblages in northern Wisconsin lakes. Ecology 63:1149–1166.

Tonn, W. M., C. A. Paszkowski, and I. J. Holopainen, 1992. Piscivory and recruitment: mechanisms structuring prey populations in small lakes. Ecology, 73:951-958.

Trumpickas, J., N. E. Mandrak, and A. Ricciardi. 2011. Nearshore fish assemblages associated with introduced predatory fishes in lakes. Aquatic Conservation: Marine and Freshwater Ecosystems 21:338–347.

Turek, K. C., M. A. Pegg, and K. L. Pope. 2013. Review of the negative influences of non-native salmonids on native fish species. Great Plains Research 23:39–49.

USGS. 2015. National Hydrography Dataset version 2.2. Available: http://nhd.usgs.gov/.

Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell, and C. E. Cushing. 1980. The River Continuum Concept. Canadian Journal of Fisheries and Aquatic Sciences 37:130–137.

Venables, W. N., and B. D. Ripley. 2002. Modern applied statistics with S, 4th edition. Springer, New York.

Vitule, J. R. S., C. A. Freire, and D. Simberloff. 2009. Introduction of non-native freshwater fish can certainly be bad. Fish and Fisheries 10:98–108.

Vorosmarty, C. J., P. B. McIntyre, M. O. Gessner, D. Dudgeon, A. Prusevich, P. Green, S. Glidden, S. E. Bunn, C. A. Sullivan, C. R. Liermann, and P. M. Davies. 2010. Global threats to human water security and river biodiversity. Nature 467:555–561.

Whittier, T. R., D. B. Halliwell, and S. G. Paulsen. 1997. Cyprinid distributions in Northeast U.S.A. lakes: evidence of regional-scale minnow biodiversity losses. Canadian Journal of Fisheries and Aquatic Sciences 54:1593–1607.

Williams, B. K., and K. Titus. 1988. Assessment of sampling stability in ecological applications of discriminant analysis. Ecology 69:1275–1285.

Woodford, D. J., and A. R. McIntosh. 2010. Evidence of source–sink metapopulations in a vulnerable native galaxiid fish driven by introduced trout. Ecological Applications 20:967–977.

APPENDICES

APPENDIX A

FISH COLLECTIONS FROM ALL SITES

SAMPLED, 2015-2016

TABLE A.1. Fish collections from all sites sampled in 2015 and 2016.

HUC 6	Stream	Priority[a]	Site	Latitude	Longitude	Date	Sample gear[b]	Species	Number collected
St. Mary	Willow Creek	1	1	48.99790	-113.1679	Aug 12, 2016	B1	*Chrosomus eos*	3
								Couesius plumbeus	28
								Margariscus nachtriebi	24
								Pimephales promelas	280
								Rhinichthys cataractae	68
								Catostomus commersonii	168
								Culaea inconstans	26
								Cottus bondi	4
Upper Missouri	Adobe Creek	3	1	47.47623	-111.7835	June 08, 2016	B1	*Chrosomus eos*	12
								Chrosomus eos × *C. neogaeus*	5
								Couesius plumbeus	9
								Pimephales promelas	33
								Rhinichthys cataractae	34
								Catostomus commersonii	32
								Culaea inconstans	7
	Big Coulee	3	1	47.52963	-111.9226	June 08, 2016	S1	*Chrosomus eos*	1

Table A.1 Continued

							Couesius plumbeus	19
							Pimephales promelas	6
							Rhinichthys cataractae	510
							Catostomus catostomus	2
							Catostomus commersonii	15
							Catostomus platyrhynchus	3
Castner Coulee	3	1	47.37121	-111.5020	Aug 14, 2016	NA	No fish	NA
Elk Creek	4	1	47.44270	-112.4466	Aug 14, 2016	B1	*Rhinichthys cataractae*	25
							Catostomus commersonii	6
							Salmo trutta	11
							Cottus bondi	61
Muddy Creek tributary	3	1	47.69225	-111.8973	June 08, 2016	B1	*Chrosomus eos*	14
							Chrosomus eos × *C. neogaeus*	1
							Couesius plumbeus	89
							Rhinichthys cataractae	203
							Catostomus commersonii	11
							Oncorhynchus mykiss	1

Table A.1 Continued

								Culaea inconstans	3
	Spring Coulee	4	1	47.71840	-110.6442	August 14, 2016	S1	Couesius plumbeus	6
								Cyprinus carpio	16
								Notropis stramineus	509
Marias	Badger Creek	1	1	48.48240	-112.5837	August 08, 2016	B2, B3	Rhinichthys cataractae	13
								Catostomus catostomus	6
								Catostomus commersonii	3
								Catostomus platyrhynchus	3
								Oncorhynchus mykiss	2
								Cottus bondi	5
			2	48.32750	-112.9757	August 09, 2016	B1, B2	Oncorhynchus mykiss	11
								Prosopium williamsoni	5
								Salvelinus fontinalis	5
								Cottus bondi	35
	Basin Coulee	2	1	48.24841	-110.9760	July 20, 2016	NA	No fish	NA
	Blacktail Creek	1	1	48.24900	-112.7897	August 13, 2016	B1	Couesius plumbeus	65
								Rhinichthys cataractae	151
								Catostomus catostomus	2

Table A.1 Continued

							Catostomus commersonii	41
							Cottus bondi	148
Bullhead Creek	2	1	48.37410	-112.1423	July 23, 2016	B1	*Couesius plumbeus*	7
							Rhinichthys cataractae	14
							Catostomus commersonii	4
							Culaea inconstans	7
							Perca flavescens	11
Cameron Reservoir	2	1	48.84657	-111.4267	July 21, 2016	M	*Pimephales promelas*	692
Cartwright Coulee	2	1	48.26800	-112.5279	July 22, 2016	NA	No fish	NA
Cold Feet Coulee	1	1	48.50264	-112.6719	August 11, 2016	B2	*Chrosomus eos*	104
							Chrosomus eos × C. neogaeus	14
							Couesius plumbeus	9
							Hybognathus hankinsoni	30
							Pimephales promelas	3
							Rhinichthys cataractae	49
							Catostomus commersonii	64
Dry Fork Marias River	2	1	48.22507	-112.0156	July 23, 2016	S1	*Chrosomus eos*	2

Table A.1 Continued

							Species	Count
							Couesius plumbeus	88
							Hybognathus hankinsoni	1
							Pimephales promelas	520
							Rhinichthys cataractae	6
							Catostomus commersonii	171
							Culaea inconstans	137
Little Badger Creek	2	1	48.35759	-113.0027	August 09, 2016	B2	*Rhinichthys cataractae*	32
							Catostomus catostomus	1
							Catostomus platyrhynchus	8
							Oncorhynchus clarkii × mykiss	1
							Oncorhynchus mykiss	4
							Salvelinus fontinalis	33
							Cottus bondi	128
Miners Coulee	2	1	48.78487	-111.4935	July 22, 2016	B1	*Pimephales promelas*	1
							Culaea inconstans	3
Railroad Creek	2	1	48.42253	-113.2106	August 10, 2016	B2	*Salvelinus fontinalis*	9
							Cottus bondi	21
Sheep Creek	2	1	48.20447	-112.4948	July 23, 2016	S1	*Chrosomus eos*	3

Table A.1 Continued

							Species	Count
							Pimephales promelas	16
South Fork Two Medicine River	2	1	48.40550	-113.1545	August 10, 2016	B1, B2	*Rhinichthys cataractae*	1
							Catostomus catostomus	1
							Oncorhynchus clarkii × mykiss	2
							Oncorhynchus mykiss	23
							Cottus bondi	4
Spring Coulee	3	1	47.89809	-112.0554	July 24, 2016	B1	*Couesius plumbeus*	5
							Hybognathus argyritis	1
							Hybognathus hankinsoni	23
							Pimephales promelas	12
							Catostomus commersonii	25
							Salmo trutta	2
							Culaea inconstans	11
Two Medicine River	1	1	48.48426	-112.2283	August 08, 2016	B1, B2	*Platygobio gracilis*	15
							Rhinichthys cataractae	35
							Catostomus catostomus	3

Table A.1 Continued

					Catostomus platyrhynchus	10
					Oncorhynchus mykiss	4
					Prosopium williamsoni	18
					Lota lota	5
					Cottus bondi	41
					Sander vitreus	1
2	48.44450	-112.2606	August 08, 2016	B2, B3	*Rhinichthys cataractae*	10
					Catostomus catostomus	3
					Catostomus platyrhynchus	3
					Oncorhynchus mykiss	2
					Prosopium williamsoni	1
					Lota lota	2
					Cottus bondi	2
3	48.48378	-112.5850	August 08, 2016	B2, B3	*Rhinichthys cataractae*	13
					Catostomus catostomus	3
					Catostomus platyrhynchus	29
					Oncorhynchus mykiss	4

Table A.1 Continued

							Cottus bondi	1
		4	48.42650	-112.9910	August 10, 2016	B1, B2	*Rhinichthys cataractae*	1
							Catostomus platyrhynchus	1
							Cottus bondi	3
West Fork Willow Creek	2	1	48.46068	-111.4244	July 22, 2016	NA	No fish	NA
		2	48.54963	-111.6134	July 22, 2016	NA	No fish	NA
Willow Creek	1	1	48.51280	-113.1284	August 10, 2016	B2	*Rhinichthys cataractae*	14
							Salvelinus fontinalis	32
							Cottus bondi	12
		2	48.55540	-113.0909	August 10, 2016	B2	*Rhinichthys cataractae*	36
							Catostomus catostomus	1
							Catostomus commersonii	13
							Salvelinus fontinalis	6
							Cottus bondi	56
		3	48.65622	-112.7456	August 10, 2016	B2	*Couesius plumbeus*	11
							Hybognathus hankinsoni	2
							Pimephales promelas	4
							Rhinichthys cataractae	5

Table A.1 Continued

								Catostomus commersonii	23
								Culaea inconstans	23
								Cottus bondi	2
			4	48.56168	-113.0213	August 11, 2016	B2	*Chrosomus eos*	3
								Couesius plumbeus	12
								Rhinichthys cataractae	61
								Catostomus commersonii	13
								Culaea inconstans	3
								Cottus bondi	98
Missouri-Musselshell	Arrow Creek	3	1	47.40659	-110.1986	August 15, 2016	S3	*Couesius plumbeus*	3
								Hybognathus argyritis	1
								Platygobio gracilis	5
								Rhinichthys cataractae	57
								Catostomus platyrhynchus	10
	Beaver Creek	3	1	46.96043	-109.5513	August 16, 2016	B1	*Catostomus catostomus*	4
								Catostomus commersonii	31
								Salmo trutta	23
								Cottus bondi	106
	Buffalo Creek	3	1	46.85268	-109.7632	July 29, 2016	B1	*Couesius plumbeus*	12

Table A.1 Continued

							Rhinichthys cataractae	98
							Catostomus commersonii	5
							Salvelinus fontinalis	7
							Cottus bondi	14
Carpenter Creek	4	1	47.75128	-107.1838	July 25, 2015	S1	*Pimephales promelas*	1180
Indian Creek	3	1	47.23447	-109.6835	July 30, 2016	B1	*Couesius plumbeus*	1
							Cyprinus carpio	1
							Pimephales promelas	22
							Rhinichthys cataractae	316
							Catostomus catostomus	23
							Catostomus commersonii	18
Little Casino Creek	3	1	47.05830	-109.4254	August 16, 2016	B1	*Pimephales promelas*	1
							Rhinichthys cataractae	39
							Catostomus catostomus	4
							Catostomus commersonii	15
							Oncorhynchus mykiss	6
							Salmo trutta	102

Table A.1 Continued

							Cottus bondi	1
Possum Run Creek	3	1	47.41063	-109.9581	August 15, 2016	B1	*Chrosomus eos*	6
							Chrosomus eos × C. neogaeus	1
							Couesius plumbeus	258
							Pimephales promelas	20
							Rhinichthys cataractae	95
							Catostomus commersonii	12
							Salvelinus fontinalis	1
							Culaea inconstans	3
Ross Fork Creek	3	1	46.82183	-109.7859	July 30, 2016	B1	*Chrosomus eos*	19
							Chrosomus eos × C. neogaeus	1
							Couesius plumbeus	41
							Pimephales promelas	1
							Rhinichthys cataractae	104
							Catostomus commersonii	21
		2	46.96735	-109.7995	August 12, 2015	S1	*Couesius plumbeus*	57
							Cyprinus carpio	4
							Hybognathus hankinsoni	5

Table A.1 Continued

							Species	Count
							Pimephales promelas	37
							Rhinichthys cataractae	22
							Catostomus catostomus	4
							Catostomus commersonii	136
Sage Creek	3	1	47.12667	-110.0392	August 15, 2016	B1	*Couesius plumbeus*	7
							Rhinichthys cataractae	17
							Catostomus commersonii	2
Siparyann Creek	3	1	47.62080	-108.5331	July 26, 2015	S2	*Chrosomus eos*	4
							Hybognathus argyritis	10
							Hybognathus hankinsoni	2
							Hybognathus placitus	29
							Hybognathus spp.	159
							Notropis stramineus	10
							Pimephales promelas	77
							Platygobio gracilis	115
							Rhinichthys cataractae	20

Table A.1 Continued

							Catostomus catostomus	7
							Catostomus commersonii	153
							Culaea inconstans	20
	Timber Creek	4	1	47.81280	-107.3501	June 23, 2016	S1	Cyprinus carpio	3
							Hybognathus argyritis	22
							Hybognathus placitus	1
							Pimephales promelas	14
							Catostomus commersonii	13
	Wolf Creek	3	1	47.37299	-109.7577	July 30, 2016	S1	Chrosomus eos	7
							Couesius plumbeus	119
							Pimephales promelas	71
							Rhinichthys cataractae	27
							Catostomus commersonii	84
							Culaea inconstans	15
Musselshell	American Fork	3	1	46.37518	-109.7953	June 04, 2016	S1	Chrosomus eos	7
							Couesius plumbeus	11
							Pimephales promelas	1
							Rhinichthys cataractae	259

Table A.1 Continued

							Catostomus catostomus	15
							Catostomus commersonii	20
							Catostomus platyrhynchus	2
							Salmo trutta	8
Box Elder Creek	4	1	47.10083	-108.3699	July 31, 2016	S1	*Cyprinus carpio*	1
							Notropis hudsonius	1
							Notropis stramineus	73
							Platygobio gracilis	5
							Rhinichthys cataractae	1
							Catostomus commersonii	59
							Moxostoma macrolepidotum	7
							Lepomis cyanellus	12
							Micropterus dolomieu	99
Careless Creek	3	1	46.34307	-109.2185	August 17, 2016	S3	*Couesius plumbeus*	25
							Pimephales promelas	1
							Platygobio gracilis	72
							Rhinichthys cataractae	53
							Catostomus commersonii	47

Table A.1 Continued

							Catostomus platyrhynchus	27
							Moxostoma macrolepidotum	11
							Noturus flavus	15
Currant Creek	3	1	46.49125	-109.1456	August 11, 2015	B1	Chrosomus eos	114
							Chrosomus eos × C. neogaeus	13
							Couesius plumbeus	156
							Hybognathus hankinsoni	75
							Pimephales promelas	218
							Rhinichthys cataractae	24
							Catostomus commersonii	116
Daisy Dean Creek	4	1	46.58807	-110.3267	June 05, 2016	B1	Salvelinus fontinalis	10
Ford Creek	3	1	47.12846	-108.7881	July 14, 2016	B1	Chrosomus eos	1
							Chrosomus eos × C. neogaeus	2
							Pimephales promelas	30
							Rhinichthys cataractae	121
							Catostomus commersonii	8
Lebo Creek	3	1	46.38132	-109.7976	June 04, 2016	S1	Chrosomus eos	10

Table A.1 Continued

							Chrosomus eos × *C. neogaeus*	1
							Couesius plumbeus	4
							Pimephales promelas	2
							Rhinichthys cataractae	150
							Catostomus catostomus	15
							Catostomus commersonii	73
							Catostomus platyrhynchus	19
Little Careless Creek	3	1	46.63104	-109.4737	August 12, 2015	B1	*Chrosomus eos*	25
							Chrosomus eos × *C. neogaeus*	3
							Couesius plumbeus	11
							Pimephales promelas	2
							Rhinichthys cataractae	4
							Catostomus commersonii	15
McDonald Creek	4	1	47.03176	-108.5979	July 31, 2016	S1	*Couesius plumbeus*	2
							Notropis stramineus	128
							Pimephales promelas	7

Table A.1 Continued

							Rhinichthys cataractae	127
							Catostomus commersonii	47
							Catostomus platyrhynchus	18
Mud Creek	3	1	46.57530	-110.3425	June 05, 2016	B1	*Chrosomus eos*	1
							Couesius plumbeus	1
North Fork McDonald Creek	3	1	47.05956	-109.2039	July 15, 2016	B1	*Chrosomus eos*	2
							Couesius plumbeus	11
							Pimephales promelas	2
							Rhinichthys cataractae	454
							Catostomus commersonii	67
Painted Robe Creek	4	1	46.18640	-108.9886	August 17, 2016	S1	*Chrosomus eos*	407
							Chrosomus eos × *C. neogaeus*	41
							Couesius plumbeus	237
							Hybognathus hankinsoni	131
							Rhinichthys cataractae	4
							Catostomus commersonii	34

Table A.1 Continued

	Simmons Creek	3	1	46.08498	-109.5891	July 28, 2016	B1	*Couesius plumbeus*	66
								Rhinichthys cataractae	268
								Catostomus commersonii	25
								Catostomus platyrhynchus	5
								Salvelinus fontinalis	41
Milk	Assiniboine Creek	1	1	48.45884	-107.8664	June 03, 2015	S1	*Chrosomus eos*	38
								Chrosomus eos × *C. neogaeus*	5
								Chrosomus spp.	40
								Cyprinus carpio	2
								Pimephales promelas	5
								Catostomus commersonii	4
								Ameiurus melas	3
								Esox lucius	17
								Culaea inconstans	1
								Lepomis macrochirus	1
	Battle Creek	2	1	48.88575	-109.3904	July 02, 2016	S1	*Catostomus commersonii*	26
								Notropis hudsonius	2
								Catostomus commersonii	30

Table A.1 Continued

Bean Creek	1	1	48.43418	-109.1910	July 03, 2016	B1	*Chrosomus eos*	19
							Chrosomus eos × *C. neogaeus*	1
							Margariscus nachtriebi	1
							Pimephales promelas	44
							Rhinichthys cataractae	16
							Catostomus commersonii	26
							Culaea inconstans	1
		2	48.39427	-109.3096	September 06, 2016	B1	*Chrosomus eos*	8
							Couesius plumbeus	13
							Margariscus nachtriebi	4
							Rhinichthys cataractae	189
							Catostomus commersonii	59
							Culaea inconstans	4
		3	48.28785	-109.3821	September 06, 2016	B1	*Pimephales promelas*	180
							Catostomus commersonii	69
							Culaea inconstans	2
Beaver Creek	2	1	47.90546	-108.1628	July 26, 2015	S2	*Chrosomus eos*	112

Table A.1 Continued

							Species	Count
							Hybognathus hankinsoni	16
							Pimephales promelas	170
							Catostomus commersonii	201
							Ameiurus melas	2
							Culaea inconstans	45
							Etheostoma exile	2
		2	48.02430	-107.8899	July 24, 2015	S1	*Catostomus commersonii*	1
							Ameiurus melas	1
							Esox lucius	1
Beaver Creek	2	1	48.50412	-109.7849	July 02, 2015	S1	*Couesius plumbeus*	3
							Pimephales promelas	9
							Rhinichthys cataractae	628
							Catostomus catostomus	4
							Catostomus commersonii	362
							Catostomus platyrhynchus	71
							Oncorhynchus mykiss	1
Big Warm Creek	2	1	48.05950	-108.3147	July 24, 2015	S1	*Chrosomus eos*	8
							Couesius plumbeus	15

Table A.1 Continued

							Pimephales promelas	26
							Rhinichthys cataractae	35
							Catostomus commersonii	20
							Noturus flavus	5
Bullhook Creek	2	1	48.51109	-109.6483	July 05, 2016	B1	*Chrosomus eos*	21
							Chrosomus eos × *C. neogaeus*	2
							Pimephales promelas	84
							Culaea inconstans	63
Cherry Creek	2	1	48.35679	-106.6375	June 09, 2015	D, S1	*Pimephales promelas*	1415
							Esox lucius	3
		2	48.48973	-106.6338	June 09, 2015	NA	No fish	NA
Clear Creek	1	1	48.51100	-109.4090	June 30, 2015	S1	*Couesius plumbeus*	22
							Pimephales promelas	51
							Rhinichthys cataractae	251
							Catostomus commersonii	46
							Catostomus platyrhynchus	37
							Oncorhynchus mykiss	2

Table A.1 Continued

Cottonwood Creek	2	1	48.73415	-108.1784	June 07, 2015	S1	*Couesius plumbeus*	74
							Hybognathus placitus	6
							Pimephales promelas	83
							Rhinichthys cataractae	1
							Catostomus commersonii	74
							Culaea inconstans	4
Dibble Creek	2	1	48.80630	-107.7129	June 08, 2015	D	*Chrosomus eos*	40
							Chrosomus eos × *C. neogaeus*	5
							Hybognathus hankinsoni	396
							Pimephales promelas	217
							Culaea inconstans	35
							Etheostoma exile	1
Dohrs Creek	2	1	48.96300	-111.1796	July 21, 2016	S1	*Pimephales promelas*	67
East Fork Stinky Creek	2	1	48.66643	-107.4091	June 22, 2016	D, S1	*Pimephales promelas*	27
							Catostomus commersonii	10
							Culaea inconstans	48
Garland Creek	2	1	48.56148	-107.8971	June 07, 2015	D	*Chrosomus eos*	83

Table A.1 Continued

							Chrosomus eos × _C. neogaeus_	4
							Pimephales promelas	365
							Culaea inconstans	11
Little Boxelder Creek	2	1	48.52130	-109.5247	June 30, 2015	S2	_Chrosomus eos_	6
							Couesius plumbeus	1
							Hybognathus hankinsoni	2
							Pimephales promelas	42
							Rhinichthys cataractae	61
							Catostomus commersonii	39
							Culaea inconstans	2
		2	48.30510	-109.5518	July 06, 2016	S2	_Couesius plumbeus_	1
							Rhinichthys cataractae	1
							Catostomus commersonii	3
							Salvelinus fontinalis	3
Little Sage Creek	2	1	48.84895	-110.9903	July 21, 2016	B1	_Salvelinus fontinalis_	6
Little Warm Creek	2	1	47.98949	-108.3148	July 23, 2015	D, S1	_Chrosomus eos_	15
							Couesius plumbeus	12

124

Table A.1 Continued

							Hybognathus hankinsoni	2
							Hybognathus placitus	1
							Pimephales promelas	12
							Rhinichthys cataractae	7
							Catostomus commersonii	2
							Culaea inconstans	5
Lodge Creek	1	1	48.77293	-109.4022	July 03, 2016	M	*Notemigonus crysoleucas*	59
							Pimephales promelas	3
							Catostomus commersonii	1
							Ameiurus melas	122
							Lepomis cyanellus	3
							Lepomis macrochirus	5
							Perca flavescens	1
North Fork Dodson Creek	1	1	48.49424	-108.0095	June 05, 2015	D	No fish	NA
Peoples Creek	2	1	48.23284	-109.2046	August 08, 2015	B1	*Rhinichthys cataractae*	32
							Catostomus commersonii	2

Table A.1 Continued

							Catostomus platyrhynchus	1
							Salvelinus fontinalis	1
Porcupine Creek	1	1	48.31941	-106.3984	July 27, 2015	S1	*Notropis stramineus*	15
							Pimephales promelas	11
							Rhinichthys cataractae	12
							Catostomus commersonii	976
							Ameiurus melas	32
							Esox lucius	17
		2	48.20722	-106.3821	July 11, 2015	S1	*Notropis stramineus*	196
							Pimephales promelas	27
							Rhinichthys cataractae	541
							Catostomus commersonii	277
							Esox lucius	5
							Etheostoma exile	1
Redrock Coulee	2	1	48.62882	-109.3885	June 29, 2015	D	*Chrosomus eos*	4
							Pimephales promelas	41
							Culaea inconstans	6
Sage Creek	2	1	48.54227	-110.2786	July 05, 2016	B1	No fish	NA

Table A.1 Continued

Salmo Reservoir	2	1	48.61886	-109.1669	July 02, 2016	M	*Lepomis macrochirus*	43
Snake Creek	1	1	48.37608	-109.2139	July 03, 2016	S1	*Chrosomus eos*	147
							Chrosomus eos × C. neogaeus	4
							Couesius plumbeus	1
							Hybognathus hankinsoni	23
							Margariscus nachtriebi	1
							Pimephales promelas	87
							Rhinichthys cataractae	9
							Catostomus commersonii	38
							Culaea inconstans	6
		2	48.41924	-109.1815	September 06, 2016	B1	*Chrosomus eos*	63
							Couesius plumbeus	6
							Hybognathus hankinsoni	2
							Margariscus nachtriebi	6
							Pimephales promelas	36
							Rhinichthys cataractae	12

Table A.1 Continued

							Catostomus commersonii	23
							Culaea inconstans	68
							Etheostoma exile	1
South Fork Milk River	1	1	48.65110	-113.3203	August 09, 2016	B1	*Salvelinus fontinalis*	33
							Cottus bondi	37
		2	48.74532	-113.1808	August 09, 2016	B1	*Rhinichthys cataractae*	1
							Catostomus commersonii	10
							Salvelinus fontinalis	10
							Cottus bondi	84
Spring Coulee	2	1	48.42671	-109.8682	July 01, 2016	S1	*Ameiurus melas*	2
Stinky Creek	2	1	48.66836	-107.4229	June 22, 2016	D, S1	*Chrosomus eos*	1
							Pimephales promelas	215
							Catostomus commersonii	22
							Culaea inconstans	108
Whitewater Creek	2	1	48.94114	-107.8450	June 06, 2015	S1	*Hybognathus hankinsoni*	2
							Pimephales promelas	1460
							Catostomus commersonii	36
							Culaea inconstans	183
							Etheostoma exile	10

Table A.1 Continued

			2	48.85193	-107.7181	June 06, 2015	S1	*Chrosomus eos*	1
								Pimephales promelas	1236
								Catostomus commersonii	6
								Culaea inconstans	45
								Etheostoma exile	5
	Woody Island Coulee	2	1	48.94901	-108.5393	July 01, 2015	S1	*Chrosomus eos*	1
								Couesius plumbeus	3
								Pimephales promelas	84
								Catostomus commersonii	6
								Culaea inconstans	2
Missouri-Poplar	Antelope Creek	1	1	48.69813	-104.3853	July 09, 2015	S1	*Ameiurus melas*	2
			2	48.69233	-104.4270	June 20, 2016	D, S3	*Ameiurus melas*	2
								Esox lucius	1
	Big Muddy Creek	1	1	48.87505	-104.9217	June 19, 2016	S3	*Couesius plumbeus*	1
								Cyprinus carpio	3
								Pimephales promelas	7
								Rhinichthys cataractae	1
								Catostomus commersonii	34

Table A.1 Continued

							Ameiurus melas	76
							Esox lucius	2
Boxelder Creek	1	1	48.81430	-104.5343	July 09, 2015	S2	Culaea inconstans	179
Charlie Creek	4	1	48.09588	-104.8754	June 21, 2016	S1	Notropis stramineus	2
Coal Creek	2	1	48.95566	-105.8217	June 18, 2016	B1	Catostomus commersonii	7
							Esox lucius	1
		2	48.95618	-105.9493	June 18, 2016	B1	Esox lucius	20
Cow Creek	3	1	47.70002	-105.5215	July 13, 2016	B1	Cyprinus carpio	7
							Culaea inconstans	7
							Lepomis cyanellus	5
							Etheostoma exile	5
Eagle Creek	1	1	48.78108	-105.0394	July 08, 2015	S1	Pimephales promelas	15
							Platygobio gracilis	100
							Rhinichthys cataractae	13
							Catostomus commersonii	80
							Ameiurus melas	3
							Culaea inconstans	6
		2	48.73754	-105.1356	September 14, 2016	B1	Catostomus commersonii	1
							Ameiurus melas	8
							Esox lucius	2
		3	48.72482	-105.1086	September 14, 2016	B1	Cyprinus carpio	3

Table A.1 Continued

							Catostomus commersonii	41
							Esox lucius	1
							Culaea inconstans	1
		4	48.76402	-105.1739	July 08, 2015	D, S1	Margariscus nachtriebi	4
							Catostomus commersonii	17
							Esox lucius	3
							Culaea inconstans	5
					September 14, 2016	D, S1	Hybognathus hankinsoni	1
							Pimephales promelas	9
							Catostomus commersonii	8
							Esox lucius	8
Little Porcupine Creek	2	1	48.17454	-106.0748	June 04, 2015	S1	Chrosomus eos	78
							Chrosomus eos × C. neogaeus	11
							Couesius plumbeus	661
							Hybognathus hankinsoni	279
							Pimephales promelas	125
							Rhinichthys cataractae	499

Table A.1 Continued

							Catostomus commersonii	77
							Etheostoma exile	83
		2	48.36674	-106.0826	July 10, 2015	S2	Chrosomus eos	10
							Chrosomus eos × C. neogaeus	1
							Couesius plumbeus	131
							Hybognathus argyritis	2
							Hybognathus hankinsoni	34
							Pimephales promelas	402
							Catostomus commersonii	100
							Culaea inconstans	9
							Etheostoma exile	9
Manternach Coulee	2	1	48.79388	-105.3024	July 08, 2015	D, S1	Hybognathus hankinsoni	3
							Pimephales promelas	17
							Catostomus commersonii	4
							Culaea inconstans	4
Middle Fork Eagle Creek	2	1	48.79181	-105.0581	September 14, 2016	B1	Chrosomus eos	12
							Hybognathus hankinsoni	37

Table A.1 Continued

							Pimephales promelas	70
							Catostomus commersonii	8
							Culaea inconstans	16
							Etheostoma exile	1
Plentywood Creek	1	1	48.83428	-104.7177	June 19, 2016	D, S1	*Margariscus nachtriebi*	1
							Pimephales promelas	49
							Culaea inconstans	7
		2	48.79317	-104.6636	September 15, 2016	B1	*Couesius plumbeus*	8
							Cyprinus carpio	131
							Hybognathus hankinsoni	8
							Pimephales promelas	40
							Catostomus commersonii	17
							Ameiurus melas	3
							Etheostoma exile	3
							Perca flavescens	1
		3	48.82530	-104.6855	September 15, 2016	S1	*Chrosomus eos*	64
							Cyprinus carpio	2
							Hybognathus hankinsoni	482

Table A.1 Continued

							Margariscus nachtriebi	31
							Pimephales promelas	140
							Platygobio gracilis	1
							Catostomus commersonii	143
							Ameiurus melas	45
							Culaea inconstans	32
							Etheostoma exile	26
Poplar River	1	1	48.69955	-105.4173	June 18, 2016	M, S3	*Notropis hudsonius*	2
							Pimephales promelas	10
							Rhinichthys cataractae	1
							Catostomus commersonii	2
							Esox lucius	1
Redwater River	3	1	47.30625	-105.7669	July 13, 2016	D, S3	*Cyprinus carpio*	2
							Hybognathus hankinsoni	291
							Hybognathus placitus	4
							Notropis stramineus	7
							Pimephales promelas	33
							Culaea inconstans	20
							Lepomis cyanellus	8

Table A.1 Continued

							Etheostoma exile	31
Smoke Creek	1	1	48.36134	-104.7538	August 10, 2015	S1	Catostomus commersonii	10
							Esox lucius	8
West Fork Remuda Creek	3	1	47.91666	-105.9211	June 21, 2016	S1	Chrosomus eos	11
							Chrosomus eos × C. neogaeus	1
							Couesius plumbeus	159
							Cyprinus carpio	3
							Hybognathus hankinsoni	165
							Hybognathus placitus	60
							Notropis stramineus	60
							Pimephales promelas	286
							Rhinichthys cataractae	24
							Catostomus catostomus	5
							Catostomus commersonii	38
							Ameiurus melas	28
							Culaea inconstans	3
Whitetail Creek	1	1	48.88807	-105.1016	June 19, 2016	B1	Catostomus commersonii	5
							Esox lucius	2

Table A.1 Continued

	Wolf Creek	1	1	48.13076	-105.7157	August 10, 2015	S1	*Rhinichthys cataractae*	4
								Catostomus commersonii	16
								Esox lucius	4
Lower Yellowstone	Clear Creek	3	1	47.00889	-104.9558	July 11, 2016	D, S1	*Chrosomus eos*	21
								Hybognathus hankinsoni	1
								Pimephales promelas	48
								Rhinichthys cataractae	9
								Semotilus atromaculatus	197
								Catostomus commersonii	36
								Fundulus kansae	1
								Culaea inconstans	2
								Lepomis cyanellus	3
	Deer Creek	3	1	47.18066	-104.8120	June 16, 2016	S2	*Semotilus atromaculatus*	282
								Catostomus commersonii	163
								Lepomis cyanellus	2
								Lepomis macrochirus	1
	Morgan Creek	3	1	47.27747	-104.6548	July 12, 2016	B1	*Chrosomus eos*	1

Table A.1 Continued

Basin	Stream			Latitude	Longitude	Date	Code	Species	Count
								Rhinichthys cataractae	27
								Semotilus atromaculatus	114
	North Fork Fox Creek	3	1	47.65020	-104.3036	June 17, 2016	B1	*Catostomus commersonii*	2
								Esox lucius	3
	Second Hay Creek	4	1	47.87404	-104.1691	July 12, 2016	S1	No fish	NA
	Smith Creek	3	1	47.42756	-104.3122	June 17, 2016	S1	*Cyprinus carpio*	3
								Notropis atherinoides	179
								Notropis stramineus	41
								Pimephales promelas	54
								Catostomus commersonii	3
								Moxostoma macrolepidotum	8
								Esox lucius	1
								Lepomis cyanellus	4
	Thirteenmile Creek	3	1	47.37461	-104.7062	June 15, 2016	B1	*Rhinichthys cataractae*	21
								Semotilus atromaculatus	61
								Catostomus commersonii	10
Little Missouri	Boxelder Creek	4	1	45.49382	-104.4533	October 01, 2016	S1	*Cyprinus carpio*	324

Table A.1 Continued

						Hybognathus argyritis	1
						Hybognathus hankinsoni	2
						Hybognathus placitus	1
						Notropis atherinoides	60
						Notropis stramineus	385
						Pimephales promelas	92
						Catostomus commersonii	5
						Moxostoma macrolepidotum	2
						Ameiurus melas	254
						Lepomis cyanellus	101
						Perca flavescens	3
		2	45.37829	-104.6730	October 01, 2016	NA	No fish	NA
Little Beaver Creek	3	1	46.01361	-104.4283	September 30, 2016	B1	*Cyprinus carpio*	2
						Pimephales promelas	18
						Semotilus atromaculatus	19
						Catostomus commersonii	7
						Ameiurus melas	34
						Noturus flavus	1

Table A.1 Continued

					Esox lucius	2
					Lepomis cyanellus	42
					Etheostoma exile	2
2	46.04815	-104.3700	September 29, 2016	B1	*Cyprinus carpio*	40
					Notropis stramineus	40
					Semotilus atromaculatus	56
					Catostomus commersonii	31
					Moxostoma macrolepidotum	6
					Ameiurus melas	4
					Noturus flavus	16
					Lepomis cyanellus	30
					Etheostoma exile	1
3	46.20135	-104.0964	September 29, 2016	S1	*Cyprinus carpio*	14
					Hybognathus argyritis	1
					Hybognathus placitus	33
					Notropis stramineus	427
					Pimephales promelas	20
					Platygobio gracilis	21
					Rhinichthys cataractae	40

Table A.1 Continued

					Carpiodes carpio	1
					Catostomus commersonii	68
					Moxostoma macrolepidotum	8
					Ictalurus punctatus	13
					Noturus flavus	1
					Lepomis cyanellus	2
4	46.06778	-104.3358	September 29, 2016	S2	*Cyprinus carpio*	1
					Notropis stramineus	13
					Semotilus atromaculatus	2
					Catostomus commersonii	10
					Moxostoma macrolepidotum	1
					Ameiurus melas	55
					Noturus flavus	11
					Lepomis cyanellus	52
					Etheostoma exile	1
5	46.03086	-104.4108	September 30, 2016	B1	*Cyprinus carpio*	3
					Notropis stramineus	40
					Pimephales promelas	38
					Semotilus atromaculatus	90

Table A.1 Continued

								Catostomus commersonii	73
								Moxostoma macrolepidotum	2
								Ameiurus melas	21
								Esox lucius	1
								Lepomis cyanellus	79
								Etheostoma exile	2
Stagville Draw	4	1	45.82326	-104.5271	October 01, 2016	B1		No fish	NA

[a] See methods. [b] B1 = Smith-Root LR-24 backpack electrofisher; B2 = Smith-Root LR-20B backpack electrofisher; B3 = Smith-Root model 12-B backpack electrofisher; D = Dipnet(s); M = 30 Promar model TR-501 polyethylene collapsible minnow traps; S1 = 6.1 m × 1.8 m seine with 0.64-cm bar mesh; S2 = 4.6 m × 1.8 m seine with 0.64-cm bar mesh; S3 = 6.1 m × 1.8 m bag seine with 0.64-cm bar mesh; NA = dry, photographs taken.

APPENDIX B

FISH AND HABITAT SAMPLING PROTOCOL

FOR PRAIRIE STREAMS

Robert G. Bramblett

Montana Cooperative Fishery Research Unit

January, 2003

1. **Site location**.-Locate the sampling site using GPS for random sites, or by convenience for non-random sites. The GPS location will be the center of the reach, this is where you place the "F" flag (see Step 2). If the site is dry, shift the reach up or downstream to capture the most wetted channel possible on the parcel of land where you have permission for sampling.

2. **Laying out the sample reach**.-Lay out a 300 m sample reach using a measuring tape and a set of 11 pin flags (labeled A-K). Follow the curves in the stream channel with the measuring tape; do not cut across curves. To avoid spooking fish, walk along the bank, not in the stream. Place a flag every 30 m. The "A" flag will be at the downstream end, the "K" flag will be at the upstream end of the reach. The "F" flag will go in the center of the reach.

3. **Block nets**.-Place block nets (these can be old seines, 1/4" mesh) at the upstream (K flag) and downstream (A flag) ends of the sample reach if the water in the channel is continuous, deeper than 25 cm, and relatively clear. This prevents fish from leaving the sample reach.

4. **Seining**.-Select the seine based on the size of the stream to be sampled. The seine length to be used should be approximately equal to or slightly greater than the stream width, and the seine height should be about 1.5 to 2 times greater than the depth of the stream. Dip nets can be used in very shallow, small habitats. Seining

begins at the upstream end (K flag) and proceeds downstream to the A flag. Seining

is performed by two people, one on each end of the seine. In pools, the seine is

pulled down the stream channel, using the shore and other natural habitat features

as barriers. Begin with the seine rolled up on each seine braille. The seine is

typically set perpendicular to shore and hauled downstream parallel to shore. As

you proceed, let out enough seine so that the seine forms a "U" shape, but not so

much that the net is hard to control. Adjust the length of the seine by rolling or un-

rolling net on the seine braille. The speed of seining should be fast enough to

maintain the "U" shape, but not so fast that the floats become submerged, or that the

seine's lead line come way up off the bottom of the stream. If rocks or other snags

are on the bottom, the seine can be lifted off the bottom for a moment to avoid the

snag, or one of the netters can bring the seine around the snag to avoid it, all the

while maintaining the forward progress of the seine. Similarly, areas of dense

aquatic vegetation can be avoided. It is important not to stop the forward progress,

because fish will swim out of the seine. It is better to avoid a snag while keeping

moving than to become snagged, which will allow fish to escape. In "snaggy"

waters, keep more of your seine rolled up for better control.

Proceed downstream while seining. In narrow streams, the entire channel width is

spanned with the seine. In wider streams, one person walks along the shore, while

the other wades through the channel. The length of each seine haul will depend on

the natural features of the stream channel and shoreline, but seine hauls should not

normally be more than 60 or 90 m long. Side channel bars or the end of a standing pool are good areas to haul out or "beach" the seine. Where a large bar or end of a standing pool is present both netters can simply run the net up on the shore. In streams with steep banks or lack of obvious seine beaching areas the "snap" technique can be used. At the end of the haul, the person near shore stops, while the person farthest out turns into shore, quickly, until the seine is up against the bank. The two netters then walk away from each other, taking the slack out of the seine, and keeping the seine's lead line up against the bank.

In riffles, with moderate to fast current, the "kick seine" technique can be used. The seine is held stationary in a "U" shape, while the other team member disturbs the substrate immediately upstream of the net. Then the net is quickly "snapped" out of the water by both team members using an upstream scooping motion.

Seine the entire 300 m reach, covering the linear distance at least once. If part of the 300 m is dry, just skip it. If the stream is much wider than your seine, do extra seine hauls in the large pools to cover the extra width. Sample all habitat types (shoreline, thalweg, side channels, and backwaters).

After each seine haul, place fish in a bucket. If the water is warm, or you have captured many fish, place fish in a fish bag to keep them alive until seining is completed. If you have to work up fish before seining is completed, release

processed fish in an area that has already been seined, as far away from the area

remaining to be seined as possible (or outside of the block nets). Large fish such as

northern pike, common carp, white sucker, shorthead redhorse, or channel catfish,

can be measured, given a small clip to the lower caudal fin and released

immediately.

5. **Processing captured fish**.-Record the species of each fish captured, and measure

    20 "randomly" selected fish to the nearest millimeter, total length. If the species of

    fish is unknown, try to at least record it as Unknown type 1, Unknown type 2, etc.

    Keep track of and record the minimum and maximum length of each species.

    For each species, preserve a subsample of at least 10 individuals per site to serve as

    voucher specimens. Record a small letter "v" next to the recorded length of the fish

    that is vouchered to allow for later validation. For Hybognathus spp., voucher up to

    20 individuals per site. Kill the fish to be vouchered by placing them in a small

    bucket or 1000 ml nalgene jar with an overdose solution of MS-222. After fish

    processing is completed, drain the MS-222 solution and place the fish in a 1000 ml

    nalgene jar with a 10% solution of formalin (in clear water, if possible). For

    specimens longer than 150 mm, an incision should be made on the right ventral side

    of the abdomen after death, to allow fixative to enter the body cavity. The volume

    of formalin solution should be approximately equal to the twice the volume of fish

    tissue to be preserved, and the fish volume should be considered water when

concentrations are determined. For example, if the fish take up 250 ml of the 1000

ml volume, you need about 500 ml of 10 % formalin solution (75 ml formalin and

425 ml water) in the 1000 ml nalgene jar. If necessary, use a second jar to

accommodate all of the specimens. Use safety glasses and gloves when pouring

formalin. Do not let the fish "cook" in the sun for a while and preserve them later,

do it as soon as possible. Label all jars inside and out with Site, Site Number,

Lat/Long, Date, Collectors names. Use pencil on Write-In-the-Rain or high rag

paper for inside labels (just put the label right in with the fish), use a sticker label on

the outside, cover it with clear (ScotchPad high performance packing tape pad

3750-P). Fish specimens should be left in formalin solution for at least 2-7 days.

Fish specimens must have formalin solution soaked out before being handled

extensively. Specimens should be soaked in water for at least 2 days, and water

should be changed at least four times during this period. After soaking out the

formalin, the fish specimens should be placed in either 70% ethanol or 40%

isopropanol for long-term storage.

6. **Habitat survey**.-Channel width, depth of water, and substrate will be measured at

11 transects perpendicular to the stream channel (located at Flags A-K), and along

the thalweg in 10 thalweg intervals between transects (deepest part of channel).

Stream width is measured to the nearest 0.1 m, depth is measured to the nearest cm,

and substrate sizes and codes are on the data sheet. One person will be in the stream

taking measurements while the other records data. Record the Latitude and

Longitude (in digital degrees) of the F flag, the stream name, site number, the date,

the flow status (flowing, continuous standing water, or interrupted standing water)

and the names of the crew members on the data sheet. Take photographs of the site,

capturing as much of the sampling reach as possible. Make sure the date feature on

the camera is turned on, to allow for later identification of site photographs.

Transects.-Start on the left bank (facing downstream) at Flag A. Measure and

record the wetted width of the channel to the nearest 0.1 m. Measure and

record (separated by a comma on the data sheet) five equally spaced depth and

substrate measurements across the wetted stream channel:

1. Left Bank-5 cm from the left bank;

2. Left Center-halfway between the Center and the Left Bank;

3. Center-center of the wetted stream;

4. Right Center-halfway between the Center and the Right Bank;

5. Right Bank-5 cm from the right bank

Thalweg.-Begin by recording the depth and substrate 3 m upstream of the

transect, in the deepest part of the channel (thalweg). Proceed up the thalweg

to Flag B, recording depth and substrate every 3 m along the thalweg. You

will record a total of 10 depths and substrates between each pair of transects.

If the stream channel is dry, record a 0 for depth, and record the substrate. The

last thalweg measurement point should fall on the next upstream transect. The

3 m interval can be estimated, and it is helpful if the data recorder helps to

keep the person in the stream from "squeezing" or "stretching" the thalweg measurements.

Repeat this procedure until all 11 transects and 10 thalweg intervals are completed.

Gear List

- 20', x 6' x ¼" heavy delta seines

- 15' x 4' x ¼" heavy delta

- 30' x 6' x ¼" heavy delta (or delta) with 6' x 6' x 6' bag

- Fish bags: nylon diver's bags, ¼" mesh 18" x 30"

- Mudders – 109.00 at Ben Meadows

- Block nets, Tent stakes

- Stream Conductivity meter

- Thermometer

- Turbidity meter (LaMotte, Ben Meadows 224805, $795.00-might try the "transparency tube" Ben Meadows 224196, $52.95)

- Waders (breathable waders are essential for this work-Cabelas has them for about $100/pair), hip boots are usually too low

- Lug sole wading boots (Cabelas)

- Habitat pole (I make habitat poles out of 1.0" OD PVC pipe. 1.5 m long including caps. Score the pipe every 10 cm with a pipe cutter, then use a Sharpie to mark rings around the pole at the scores, and label the pole 10,

20, 30, etc. 5 cm marks are made between the 10 cm rings, you can visually estimate between the 5 cm marks to get to the nearest cm. Spray or brush a Urethane finish on the pole or your marks will come off fast with sunscreen and bug dope.)

- Metric 30 m tape (Ace Hardware actually carries a tape with metric on one side)

- Measuring boards, one short 300 mm (half a 6" PVC works well for Hybognathus "fin flotation", one long, ~0.5-1 m, or you can just use a meter stick for the odd big fish)

- Hand lens

- Small 1 gallon red bucket from Ace Hardware for doping fish

- 5 gallon buckets

- MS-222

- Labels and tape pads for fish samples

- 1000 ml Nalgene jars o Formalin (buffered is great, but more expensive- I throw a Rolaids in each jar of fish to neutralize the acidity)

- Clip board

- 11 Pin flags labeled A-F