

SALMONID RESPONSE TO SUPERFUND REMEDIATION
IN SILVER BOW CREEK, MONTANA

by

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DEDICATION

For Helen.

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ABSTRACT

Uncontrolled disposal of hard rock mining wastes in the Butte hill mining district of Montana, led to the extirpation of fish from Silver Bow Creek throughout the 20th century. Superfund remediation has been ongoing since 1998 and metal concentrations are reduced. However, water quality remains poor due to inadequate treatment of municipal sewage. To assess the effectiveness of Superfund remediation in reestablishing salmonid populations, we evaluated seasonal salmonid abundance and movement in the Silver Bow Creek watershed over a 3-year period. Spatially-continuous abundance surveys were conducted in 34 main stem stream km and each sampled westslope cutthroat trout *Oncorhynchus clarkii lewisi* ($n = 787$) and brook trout *Salvelinus fontinalis* ($n = 1,846$) was PIT-tagged. Movements of PIT-tagged individuals were monitored at seven stationary antenna sites and during six seasonal portable antenna surveys. Monthly synoptic water quality samples were collected. In the main stem, water quality was poor below the wastewater effluent and was characterized by acutely toxic copper concentrations, elevated ammonia levels (e.g., $\text{NH}_3\text{-N} = 2.8$ mg/L), and hypoxia during summer nights (e.g., $\text{DO} = 1.4$ mg/L). Longitudinal abundance of salmonids closely resembled the longitudinal trend in DO. Regression analysis revealed strong associations between salmonid occurrence and abundance with DO (positive) and copper (negative) concentrations during the summer. However, westslope cutthroat trout relative abundance increased between summer and winter in remediated segments that had been hypoxic during the summer. Few brook trout recolonized the remediated main stem during the study period and the wastewater effluent may have deterred brook trout movement. Westslope cutthroat trout moved into remediated segments during the late summer and early fall as hypoxia subsided. The majority of westslope cutthroat trout sampled in the main stem were large-bodied adults (≥ 200 mm TL) contrasting with the predominantly small-bodied counterparts in the tributaries. Despite hypoxia and copper toxicity, recolonization of indigenous westslope cutthroat trout apparently was driven by the reexpression of a fluvial-adfluvial migratory behavior, a pattern that was not possible during the 100-150 years of main stem contamination.

CHAPTER 1

HISTORY OF CONTAMINATION AND REMEDIATION
IN SILVER BOW CREEK, MONTANA

The Comprehensive Environmental Response, Compensation and Liability Act of 1980 (CERCLA or “Superfund”) was established to protect human health and the environment from abandoned hazardous waste sites (USEPA 2012a). As of August 2012, the U.S. Environmental Protection Agency had identified 1,304 National Priority List sites. There are 55 new sites proposed, and 360 sites determined to be “remedied” and removed from the National Priority List (USEPA 2012a). Many of the most severely contaminated sites on the National Priority List are related to hard rock (non-coal) mining contamination. The U.S. Environmental Protection Agency has identified 156 hard rock mining Superfund “mega-sites” where costs associated with remediation of contaminated watersheds are expected to exceed \$50 million (USEPA 2004). Several of these watersheds are located in the intermountain western U.S. and are inhabited by indigenous salmonids. Because salmonids are highly sensitive to metal contamination (Chapman 1978; Hansen et al. 1999; Mebane et al. 2012), reduced population abundance, or complete extirpation from portions of the watershed are common consequences. Some examples of reduced or extirpated salmonid populations from hard rock mining contamination in the intermountain western U.S. include portions of the Animas (Finger et al. 2004), Eagle (Woodling and Albeke 1999), upper Arkansas (IEc 2006), Alamosa (MWH et al. 2005), and South Platte (CDPHE and USEPA 2009) watersheds in

Colorado; the Belle Fourche (Hoyer and Schwickerath 2005) watershed in South Dakota; the Couer d'Alene (Woodward et al. 1997; USFWS et al. 2007) and Salmon (Mebane 1997; NOAA-DAARP 2012) watersheds in Idaho; and the Boulder (Church et al. 2007), Missouri (Hargrave et al. 2000; USEPA 2002; Harper et al. 2009; USEPA 2012b), and Clark Fork (MTNRDP et al. 2005) watersheds in Montana. Environmental damage assessments of contamination in watersheds inhabited by salmonids prominently feature evidence of negative effects to those populations (Morey et al. 2002; Finger et al. 2004; USFWS et al. 2007), and recovery of those populations is frequently cited as a primary goal of site restoration plans (Woodling and Albeke 1999; MTNRDP et al. 2005; USFWS et al. 2007). However, detailed assessment and quantification of the response of salmonid populations to Superfund remedies are either rare or not commonly published.

In this thesis, I attempt to provide a detailed assessment and quantification of the response of salmonid populations to Superfund remediation in Silver Bow Creek, Montana. Superfund remediation of the Streamside Tailings Operable Unit of Silver Bow Creek is nearly complete as of 2012. Recovery of salmonid populations, particularly indigenous westslope cutthroat trout *Oncorhynchus clarkii lewisi*, is a primary restoration goal (MTNRDP et al. 2005; Saffel 2011; MTNRDP 2012). Specific objectives of the thesis research were to quantify the influence of water quality and habitat characteristics on the distribution and abundance of salmonids in the Silver Bow Creek main stem (Chapter 2), and to evaluate fish movement patterns in the Silver Bow Creek stream network and assess the extent of fish recolonization in remediated sections (Chapter 3). Results will be used to inform continuing conservation and management

decisions for restoring and protecting westslope cutthroat trout populations and other indigenous taxa in Silver Bow Creek.

The Clark Fork River Superfund mega-site complex is one of the most infamous contaminated river systems in the USA. In fact, one contaminated site is considered a heritage landmark (the Slag Canyon), and another site (the Berkeley Pit) is advertised as a tourist attraction as one of the largest and most acidic pit-mining lakes in the world (Gammons et al. 2009). The vast majority of the contamination to the Clark Fork River was derived from copper mining in the Silver Bow Creek headwaters encompassed by the Butte Hill mining district in the city of Butte, Montana (Figure 1). For more than a century, from the 1870s through the 1980s, Silver Bow Creek received direct and indirect waste rock, slag (smelting byproduct), contaminated groundwater (Chadwick et al. 1986; Luoma et al. 2005), and raw municipal sewage inputs (Gless and Miller 1973). An estimated 2 million m³ of tailings were deposited directly into the stream (Luoma et al. 2005). A major flood in 1908 redistributed the waste rock and tailings downstream throughout the stream channel and floodplain (Chadwick et al. 1986). Prior to remediation, streamside tailings deposits were distributed along the entire length of the stream (Brook and Moore 1989), and localized deposits reached depths of 5 m (G. Mullen, Montana Natural Resource Damage Program, personal communication). Much of the floodplain was devoid of vegetation (Luoma et al. 2005). Throughout the mining period (c., 1870 to 1982), the stream was utilized primarily as a sewer for the removal of industrial wastes. In 1978, the average daily loads of copper, zinc, and ammonia in the

stream immediately below Butte were estimated at 27, 189, and 357 kg/day respectively (Beuerman and Gleason 1978).

The copper ore deposits mined and smelted in the Butte area were found in the Butte Quartz Monzonite formation dissected by dikes of porphyritic lode ore (Gammons et al. 2009). Underground mining operations began in the area in the 1870s, and open-pit mining continues at reduced levels today in the Continental Pit mine. Today, there are at least 60 vertical shafts to depths of more than 1.6 km and approximately 16,000 km of underground mine workings (see: www.pitwatch.org). In 1955, mining effort shifted from underground to open-pit operations with the initiation of the Berkeley Pit mine. Mining ceased in the Berkeley Pit and the underground mine workings in 1982, and groundwater diversions were discontinued at that time. Contaminated groundwater began to accumulate in the Berkeley Pit and the underground mine workings immediately. Today, the Berkeley Pit lake (volume ~ 216 million m³) and the underground mines contain strongly acidic (pH range = 2.3-2.5) water with extremely high metal concentrations (e.g., total recoverable copper range = 73-137 mg/L, total recoverable iron range = 507-883 mg/L; see: www.pitwatch.org). The Silver Bow Creek headwaters have been entirely absorbed into the Berkeley and Continental Pits and the city of Butte metro sewer system including the Metro Storm Drain (Figure 1).

The first attempt to mitigate environmental damage to Silver Bow Creek occurred in 1972 with lime treatment of mining wastewaters prior to discharge into the creek (Spindler 1977). In 1996, the Yankee Doodle Tailings Pond system was constructed (Figure 1). In 1998, Superfund remediation activities began removing streamside tailings

and reconstructing the stream channel. Also in 1998, the Butte Treatment Lagoons began treating metal contaminated groundwater at the site of two former ore smelters, the Colorado Smelter and the Butte Reduction Works, along Silver Bow Creek (CTEC 2012).

It is not known when fish were extirpated from Silver Bow Creek or when they first returned. The stream was presumed to be uninhabitable due to mining contamination in 1892 and was described as “brown soup” (Everman 1892, cited by Luoma et al. 2005). Fisheries investigations found the stream to be uninhabited by fish in 1986 (Trout et al. 1986). We are unaware of published or unpublished fisheries surveys between 1892 and 1985, and between 1987 and 2001. Although fish sampling effort was low, it seems unlikely that any fish inhabited Silver Bow Creek prior to 1998.

Macroinvertebrates were not collected in Silver Bow Creek until 1975; they were first sampled in sites nearest Butte in 1981 (Chadwick et al. 1986). The first fish (longnose suckers *Catostomus catostomus*, slimy sculpin *Cottus cognatus*, and central mudminnow *Umbra limi*) were collected in 2002 (MTFWP 2009). Remediation contract workers visually observed fish, presumably longnose suckers, in Silver Bow Creek between 1998 and 2002 during initial remediation phases (T. Reilly, Montana Department of Environmental Quality, personal communication).

Prior to remediation, copper and zinc concentrations in Silver Bow Creek regularly exceeded acute water quality standards (USEPA 1986) by an order of magnitude, but since remediation began in 1998, levels have been substantially reduced (Figure 2). In the spring of 2004, three salmonids (two westslope cutthroat trout, and one

brook trout *Salvelinus fontinalis*) were collected for the first time in Silver Bow Creek in an unremediated reach within 1 km of the German Gulch tributary confluence (MTFWP 2009; Figure 1) where populations of both species are relatively robust (Saffel et al. 2011). In remediated stream segments closer to Butte, brook trout and westslope cutthroat trout were first collected in 2007 (MTFWP 2009).

Although metal concentrations in Silver Bow Creek are reduced (Figure 2), water quality continues to be impaired. For example, metal concentrations in the stream become acutely toxic during precipitation events, and these temporary increases have been associated with fish mortality (Selch 2009). However, the spatial extent of the spikes in dissolved metals is not well understood. Additionally, nutrient pollution and hypoxia occur in some sections of Silver Bow Creek, but the effects on fish abundance and behavior have not been evaluated. Watershed nitrogen yield (nitrogen load per watershed area) in Silver Bow Creek was high, even compared to other “developed” watersheds (Plumb 2009). The Butte wastewater treatment plant effluent was the primary source of at least 80% of the nitrogen, nearly all of which was discharged to the stream as ammonia (Plumb 2009). The ammonia-rich wastewater effluent contributed to increased primary productivity downstream, and decomposition of the ammonia led to hypoxic periods during summer nights (Gammons et al. 2011). Understanding what effect Superfund remediation has had on the recovery of fish populations in Silver Bow Creek will require careful evaluation of the influence of each of these water quality factors and those physical habitat characteristics altered by stream channel reconstruction.

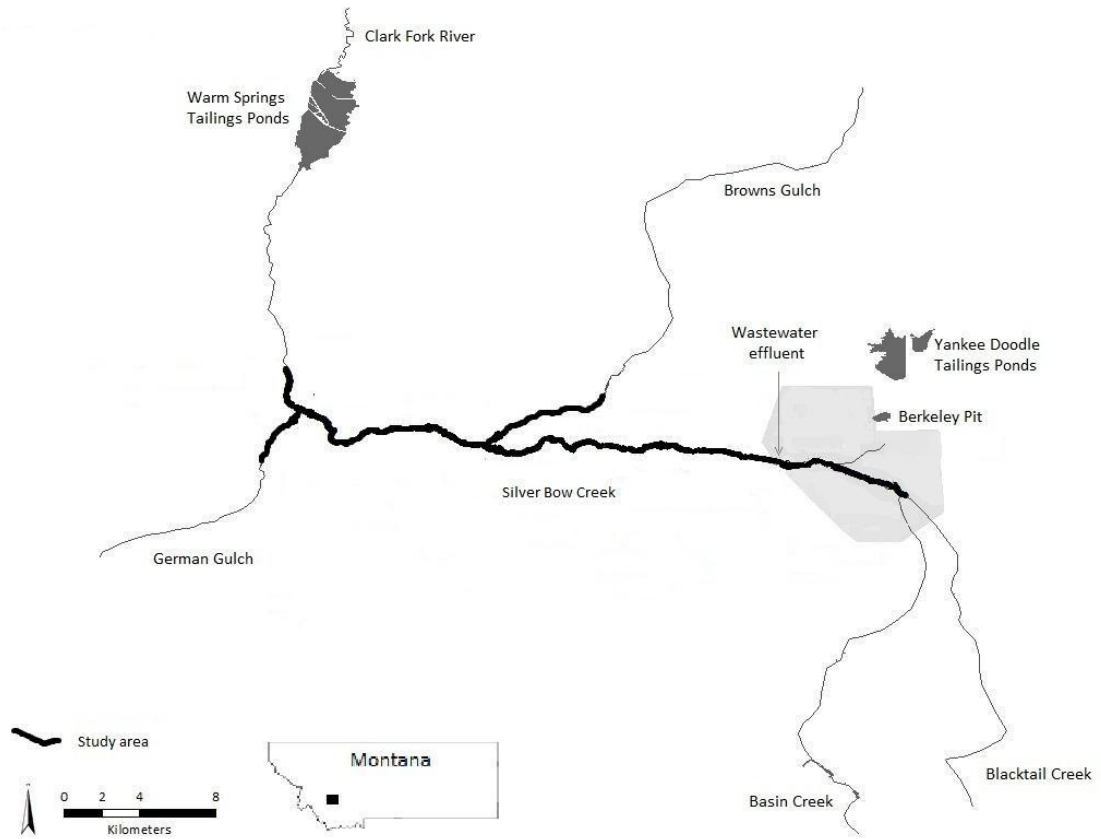


Figure 1.-Silver Bow Creek stream network in southwestern Montana.

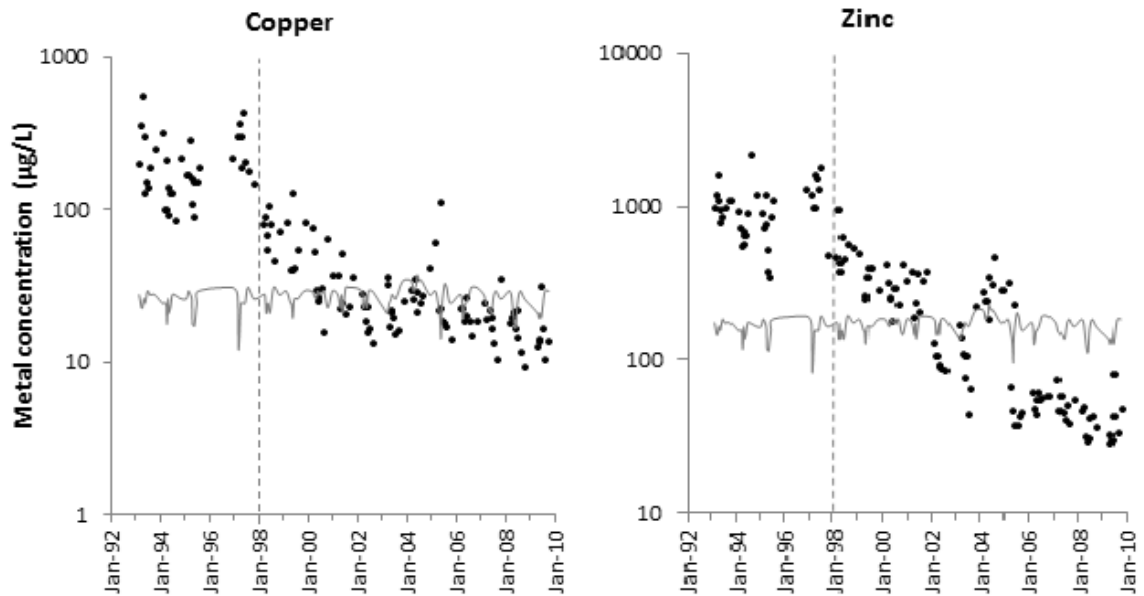


Figure 2.-Total recoverable copper and zinc concentrations ($\mu\text{g/L}$) in Silver Bow Creek, Montana prior to and during Superfund remediation (Source: USGS stream gage 12323250). The solid grey line represents the hardness-based acute standards (USEPA 1986). The dashed grey line represents the approximate beginning date of Superfund remediation in the Streamside Tailings Operable Unit.

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CHAPTER 2

THE INFLUENCE OF EUTROPHICATION-INDUCED HYPOXIA AND COPPER
CONCENTRATION ON SALMONID OCCURENCE AND ABUNDANCE
IN A SUPERFUND REMEDIATED MONTANA STREAM

Abstract

Uncontrolled disposal of hard rock mining wastes in the Butte hill mining district near Butte, Montana led to the extirpation of fishes from Silver Bow Creek throughout the 20th century. Superfund remediation has been ongoing since 1998 and metal concentrations have been substantially reduced, although still exceed aquatic life standards during rain events. Moreover, the stream is influenced by municipal sewage and during midsummer, hypoxia has been observed at night below the effluent. Despite the water quality problems, six fish species, including three sensitive salmonids, are now present in the stream. To evaluate the success of remediation in reestablishing salmonid populations, we conducted spatially-continuous abundance surveys in 34 stream km during the summers of 2010 and 2011. Synoptic water quality samples were collected during the survey periods and habitat surveys were conducted following abundance sampling in 2011. Logistic and zero-altered negative binomial (ZANB) regression models were developed to estimate the likelihood of occurrence and relative abundance of salmonids in relation to water quality and habitat characteristics. In both years, extensive stream portions (≈ 6 km) had low DO (< 5 mg/L), and minimum DO concentrations were < 2.5 mg/L. Ammonia and copper concentrations increased sharply

below the wastewater effluent. The longitudinal abundance of salmonids closely resembled the longitudinal trend in DO during both years and salmonids were almost entirely absent from hypoxic (<4.0 mg/L) sections of the stream. Regression analysis revealed strong associations between salmonid occurrence and abundance with DO (positive) and copper (negative) concentrations. Ammonia concentration, channel cover, and coarse substrate percentages were positively associated with salmonid occurrence or abundance in particular years, however the influence of these variables was less consistent and weaker than that of DO and copper concentration. These findings indicate that remediation has largely succeeded in reestablishing salmonid populations in Silver Bow Creek despite ammonia pollution, hypoxia, and acutely toxic copper concentrations. However, persistent chronic and acute copper concentrations, if not contained, will likely limit the abundance of salmonids in Silver Bow Creek in the future even if ammonia pollution and hypoxia are alleviated.

Introduction

The metal mining industry produces more toxic waste than any other U.S. industry (USEPA 2004), and these wastes are among the most ecologically damaging results of human land use activities (Luoma et al. 2005). The U.S. Environmental Protection Agency has identified 156 hard rock (non-coal) mining Superfund “mega-sites” (individual sites with expected cleanup costs of at least \$50 million) contaminating rivers and streams nationwide and remediation of these sites is estimated to cost as much as \$24 billion (USEPA 2004). Given the continued demand for metals worldwide (WB

2012), the large number of Superfund National Priority List sites currently identified ($n = 1,304$) and the proposed new sites ($n = 55$), environmental remediation projects (removal or isolation of contaminants from the environment, Finger et al. 2007) are likely to become more common in the future (USEPA 2012). Understanding how aquatic organisms respond to changes in the physical habitat and water chemistry of streams and rivers after remediation will be essential to plan and implement future remediation activities. Extirpated or depauperate fish populations in metal contaminated streams and rivers represent substantial economic and recreational losses (Morey et al. 2002), and recovery of affected fish populations is an important goal of environmental restoration (Woodling and Albeke 1999; MTNRDP et al. 2005; Finger et al. 2007). However, despite the substantial costs of remediation and explicit goals to restore fish populations, few projects have explicitly quantified the degree of fish population recovery following remediation (Hornberger et al. 2009). Typically, evaluation of remediation has focused on measurements of response in water quality (Prat et al. 1999), macroinvertebrates (Nelson and Roline 1996), or levels of metal contaminants in fish tissue (Gale et al. 2004).

Streams and rivers contaminated by metals may also receive excessive loads of nutrients from municipal and industrial sewage (Prat et al. 1999). The incentive for local governments to provide expensive secondary and tertiary wastewater treatments to already-contaminated streams may be low. As a result, metal contamination may be exacerbated by inputs of poorly treated municipal sewage, resulting in increased suspended solids, nuisance algal blooms, stream temperature alterations, toxic

pharmaceuticals and other substances, and hypoxia (Warren 1971). Coldwater fishes, such as salmonids, are especially sensitive to hypoxia associated with nutrient pollution.

Salmonids often avoid areas of hypoxia (Poulsen et al. 2011) and high concentrations of toxic metals (Woodward et al. 1997). Hypoxic or metal contaminated stream zones may act as landscape filters to sensitive salmonid species by excluding these fish from affected stream zones (Poff 1997). Thus, observed in sufficient detail over appropriate scales, spatial patterns of salmonid abundance in streams affected by hypoxia or metal contamination may reflect dissolved oxygen (DO) or metal-concentration gradients. However, observing abundance patterns of fishes at broad spatial scales in detail sufficient to relate that variation in abundance to variation in water quality or habitat is challenging, and may require spatially-explicit sampling strategies (Torgersen 1999; Fausch et al. 2002; Gresswell et al. 2006).

Silver Bow Creek near Butte, Montana is an example of a nearly completed stream remediation project. Silver Bow Creek originates in the Butte Hill mining district, the source of metal contamination to the Clark Fork River Superfund mega-site complex (MTDEQ and USEPA 1996; Luoma et al. 2005). Remediation and stream channel reconstruction of Silver Bow Creek has been ongoing since 1998, and by 2010 project activities were complete in the uppermost 20 km of stream. To adequately remove the extensive tailings deposits in the Silver Bow Creek stream channel and floodplain (MTDEQ and USEPA 1996), nearly all stream channel and floodplain soils were removed up to depths of 5 m (G. Mullen, Montana Natural Resource Damage Program, personal communication) and replaced with clean backfill (B. Bucher, CDM Smith, paper

read during conference proceedings of the Montana Natural Resource Damage Program Conference, 2012). Prior to remediation, metal contamination had caused the complete loss of macroscopic life in the Silver Bow Creek stream ecosystem (Chadwick et al. 1986). Since remediation, water quality has improved (Chapter 1; USEPA 2011) and macroinvertebrates and fish, including salmonids, have returned to the stream. However, water quality remains spatially and temporally variable, and the influence of remaining metal contaminants and municipal sewage discharges on the distribution and abundance of fish in the stream is poorly understood. A hypoxic zone in the stream below the Butte municipal wastewater treatment plant effluent related to large ammonia loads in the treatment plant effluent is a special concern (Plumb 2009; Gammons et al. 2011).

There were few fisheries surveys prior to remediation, and there has been no effort to identify specific factors limiting salmonid distribution and abundance in the stream after remediation commenced (MTNRDP et al. 2005). The purpose of this study was to identify how remediation has influenced the water quality and physical habitat of Silver Bow Creek and to assess the relationship of those factors with the occurrence and abundance of salmonids. We conducted extensive (≈ 34 km), spatially-continuous assessments of fish abundance during a 2-year study period. These data were used to examine the effects of physical habitat and water quality on longitudinal salmonid abundance patterns. These data were also used to quantify associations between salmonid occurrence and abundance with the physical and chemical characteristics of the stream.

Methods

The primary intent of this study was to assess the influence of water quality and physical habitat on the distribution and abundance of salmonids in Silver Bow Creek following Superfund remediation. We hypothesized that during the summer, DO and ammonia concentrations would vary substantially over relatively short distances (Plumb 2009; Gammons et al. 2011), and salmonid distribution and abundance would reflect these conditions. However, metal levels and physical habitat characteristics were also likely to vary substantially and could also influence distribution and abundance patterns. Distinguishing the effect of each specific water quality and physical habitat factor, independent of each other factor, required extensive sampling to accommodate the range of water quality and physical habitat conditions. Therefore, to understand the specific influence of each of these factors on the distribution and abundance of salmonids we adopted a spatially-continuous, rather than random, sampling strategy to assess the influence of water quality and physical habitat at the landscape scale (Gresswell et al. 2006).

Study Area

Spatially-continuous sampling for salmonids was conducted in 33.8 km of the Silver Bow Creek main stem (Figure 1). The main stem study area extended from a point 0.3 stream km above the Basin Creek confluence (stream km 0.0) downstream to a small temporary dam (stream km 33.8) built during the initial remediation phase to capture sediments entrained by remediation activities upstream (Figure 1). The study area was

subdivided into 326, 100-m sampling increments which are referred to as “sample sections”. Sample sections were referenced by stream distance (km) downstream from the upstream study area boundary. Fish abundance sampling and physical habitat estimates occurred within each 100-m sample section. Within a subsample of those sections, synoptic water quality measurements were also taken, and these were interpolated between nearest sites to provide estimates for each section. Sample section boundaries were identified in the field at approximately 100-m intervals and more precise thalweg lengths were measured by tape. Abundance sampling in 2010 occurred from 6 to 26 July between stream km 0.0 and 24.2 (239 sample sections), and occurred in 2011 from 6 July to 4 August between stream km 0.0 and 33.3 (326 sample sections). Sampling did not occur in one section of private property (stream km 1.7), in the “Slag Canyon” (stream km 4.0-4.5), or in the impoundment reservoir (stream km 33.4-33.8). Physical habitat surveys were conducted in September 2011. The sample section was the base unit for statistical analyses.

For descriptive purposes, the study area was subdivided into six “segments” (segment length range: 2.5 to 8.2 km) and eight “reaches” based on geomorphic and geochemical differences. Segments were demarcated by tributary junctions ($\geq 15\%$ main stem flow), valley form changes (confined and unconfined), and geochemical stream zones. Reaches were demarcated by segment boundaries, land use changes, and more specific geochemical processes. Segment and reach boundaries were the same in the four lower segments (lower and upper Durant Canyon, Ramsay Flats, and Recovery) but the Hypoxic and Butte segments were each subdivided into separate reaches. The Hypoxic

segment was subdivided into the Degradation and Active Decomposition reaches based on geochemical processes. The Butte segment was subdivided into the Blacktail Creek and Lower Area One Superfund Operable Unit (LAO) reaches based on land use changes (see below).

In the uppermost 6.8 km, Silver Bow Creek passes through the Butte segment (stream km 0.0 to 6.8, Figure 1), comprised of the Blacktail Creek (stream km 0.0 to 4.0) and LAO reaches (stream km 4.0 to 6.8). The Butte segment was classified as a distinct segment because it was upstream from the Butte wastewater treatment plant effluent discharge and therefore water quality was not influenced by the effluent. Throughout the Butte segment base stream discharge was approximately $0.3 \text{ m}^3/\text{s}$, with flows primarily derived from the Blacktail Creek and Basin Creek watersheds and small water gains in the LAO reach. The Blacktail Creek reach was separated from the LAO reach because remediation in the LAO reach has been extensive, whereas in the Blacktail Creek reach, remediation activities have targeted only specific tailings deposits leaving the majority of the floodplain and stream channel undisturbed. The 4.0 km Blacktail Creek reach was influenced by beaver activity and physical habitat was characterized by predominantly fine stream channel sediments (silt and sand) and extensive willow cover above the channel. At stream km 4.0, the Metro Storm Drain enters the main stem (Figure 1) marking the boundary between the Blacktail Creek and LAO reaches within the Butte segment. Discharge of the Metro Storm Drain was negligible except during precipitation events. The ancestral Silver Bow Creek stream channel roughly followed the Metro Storm Drain path into the uplands of the Butte Hill mining district. However, the historic

stream channel and upper headwaters of Silver Bow Creek have now been absorbed into the Yankee Doodle Tailings Pond, the Berkeley Pit, and the Butte storm sewer system. Immediately downstream from the Metro Storm Drain confluence, Silver Bow Creek (which in name begins at the Metro Storm Drain and Blacktail Creek confluence) passes through the 0.5 km Slag Canyon; a narrow, confined (i.e., approximately 10 m in height and width) chasm of compressed slag. Slag Canyon sample sections were not sampled due to concerns of instability in the canyon walls. The Slag Canyon was considered a heritage landmark and the slag walls were not removed during remediation. The walls of the canyon were considered to be chemically inert (C. H. Gammons, Montana Tech University, personal communication). The 2.8 km LAO reach was suspected to be a source of metal contamination to Silver Bow Creek (Selch 2009). Discharge from the Montana Pole treatment plant ($\approx 0.03 \text{ m}^3/\text{s}$) and the Butte Treatment Lagoons ($\approx 0.03 \text{ m}^3/\text{s}$) entered Silver Bow Creek in this reach. Both of these Superfund groundwater mitigation facilities were situated within the historic stream floodplain. The Montana Pole groundwater treatment plant treats ground and surface water contaminated by wood treatment byproducts; specifically pentachlorophenols (PCPs), polynuclear aromatic hydrocarbons (PAHs), dioxins, and furans (USEPA 1993). The Butte Treatment Lagoons treat metal contaminated groundwater from the Butte area with lime and a series of settling ponds (USEPA 2006).

Remediation was extensive in the LAO reach, and in each of the next three downstream segments (the Hypoxic, Recovery, and Ramsay Flats segments), consisting of total removal and replacement of floodplain soils and complete stream channel

reconstruction (Bucher, unpublished). The remediated stream channel design in the LAO reach and all downstream remediated segments was characterized by large meanders with nearly uniform width and depth in riffles and runs, and no side channels. Scour pools occurred along the outside of meanders. Substrate was almost entirely comprised of gravel and sand. The LAO channel reach was designed with a narrower and shallower channel compared to downstream segments due to the smaller flows in this reach. Remediation and re-vegetation progressed downstream from the Slag Canyon and therefore upstream segments had more extensive channel cover from riparian vegetation.

The Butte wastewater treatment plant effluent discharged directly into Silver Bow Creek at stream km 6.8 and marked the boundary between the Butte and Hypoxic segments (Figure 1). The effluent was extremely ammonia-rich and typically contributed about 40% of the stream flow of Silver Bow Creek downstream (Plumb 2009). In the Hypoxic segment (stream km 6.8-14.8, Figure 1) water quality was strongly influenced by ammonia discharged from the wastewater effluent. The Hypoxic segment was divided into two reaches based on prevailing geochemical processes. In the first 3.7 km downstream from the effluent (Degradation reach, stream km 6.8-10.5), ammonia from the wastewater effluent was rapidly transformed into nitrate and biochemical oxygen demand was high (Gammons et al. 2011), although DO levels generally were higher in the Degradation reach compared to the Active Decomposition during summer nights. From approximately stream km 10.5 to 14.8 (Active Decomposition reach), hypoxia was observed during summer nights and the lowest DO levels generally occurred in this reach (Plumb 2009; Gammons et al. 2011). The two reaches of the Hypoxic segment were

named based upon descriptions of similar stream reaches by Warren (1971). The approximate point where DO levels began to recover, (stream km 14.8) marked the boundary between the Hypoxic and Recovery segments (Figure 1). Throughout the Recovery segment, DO levels typically increased and the Recovery segment was bounded downstream by the Browns Gulch tributary confluence at stream km 21.8 (Figure 1).

The Ramsay Flats segment was delineated by the Browns Gulch confluence (stream km 21.6) at the upstream end, and at the downstream end by a valley form change from an unconstrained valley to a constrained valley known as Durant Canyon (stream km 24.6, Figure 1). Brook trout were relatively abundant in lower portions of Browns Gulch, and westslope cutthroat trout were almost entirely absent (J. Naughton, unpublished data), providing a potential source population for brook trout recolonization of the remediated main stem. Prior to remediation, floodplain tailings deposits in the low gradient Ramsay Flats segment were extensive with local deposits up to 5 m deep (G. Mullen, Montana Natural Resource Damage Program, personal communication).

The upper Durant Canyon (stream km 24.6 to 31.3) and lower Durant Canyon (stream km 31.5 to 33.8) segments were separated by the German Gulch tributary confluence at stream km 31.3 (Figure 1). Remediation was in progress in portions of the canyon in both 2010 and 2011, but complete in all upstream reaches. The Durant Canyon segments differed from upstream segments by the more confined valley, the greater coarseness of the stream substrate, and the scarcity of pools and vegetative cover. Pools tended to be deeper, although smaller in area in Durant Canyon segments compared to

upstream segments. Westslope cutthroat trout and brook trout were abundant in the German Gulch tributary (J. Naughton, unpublished data). The study area was bounded downstream by a small temporary dam (stream km 33.8) built during the initial remediation phase to capture sediments entrained by remediation activities upstream (Figure 1).

Water Quality

Synoptic water quality samples were collected to assess longitudinal variation in water chemistry. Water quality was measured at three different intervals during the period of electrofishing surveys. Hourly water quality monitoring devices (Hach Hydromet, Hydrolab MS5 Multiparameter Sonde) placed at three stream locations indicated that minimum DO levels occurred between 0000 and 0630 at each location (Figure 2). Water sampling was conducted during the predawn hours (0200 to 0630) to approximate the minimum daily DO concentrations at 20 sites along the main stem (Figure 1). Sample sites were spaced approximately every 1-4 km, with site locations spaced more closely in the Hypoxic and Recovery segments (Figure 1).

DO concentrations (mg/L) were measured according to the Membrane-Electrode Method (Hauer and Hill 2006; APHA et al. 2012) using a handheld YSI-Professional Plus multimeter (YSI Model 6050000). The instrument was field calibrated prior to each day of sampling. Water samples were also collected for later determination of ammonia and copper concentrations. Samples were collected in stream runs or low-gradient riffles at the stream thalweg by manually filling a 0.5-L Nalgene sampling bottle at a depth of approximately 0.1 m. Samples were then filtered (Whatman 6901-2502 GD/X 25 syringe

filter, 0.2 μm membrane pore size) and divided into two separate pre-rinsed 60-mL Nalgene bottles and placed on ice. Total dissolved ammonia as nitrogen ($\text{NH}_3\text{-N}$) was measured within two days of sample collection by colorimetry (USEPA 2001a) using a portable spectrophotometer (Hach Company, Model DR2700-01) according to the Nessler Method (Hach method 8038, HC 2001; APHA et al. 2012) which is accurate within the range of concentrations between 0.02 and 2.50 mg/L. Water samples with concentrations above this range were diluted with deionized water prior to analysis. Ammonia concentrations in field blanks (i.e., deionized water samples prepared in the same manner as stream water samples) were below detection limits (0.02 mg/L). Estimated precisions based on field and laboratory duplicates were within 10%.

Dissolved copper concentration was measured by Inductively Coupled Plasma-Atomic Emission Spectrometry (ICP-AES) following USEPA Method 200.7 protocol (USEPA 2001b). Water samples were analyzed in March 2012 at the University of Montana Environmental Biogeochemistry Laboratory. Minimum detection limits (i.e., the lowest concentration at which copper could be detected and reliably distinguished from zero) were 0.8 $\mu\text{g/L}$ and practical quantification limits (i.e., the lowest concentration at which copper concentration exceeded the uncertainty associated with the analysis procedure, González and Herrador 2007) were 5.0 $\mu\text{g/L}$. Copper was the primary metal of interest in this study and dissolved, rather than total recoverable, concentrations were considered because dissolved copper may be more bioavailable to fish (USEPA 2007) and has the potential to acutely impact fish (through respiration). Interpolation of water quality from the 20 sample sections to unsampled sections was achieved by averaging the

closest upstream and downstream measurements and weighting by distance (i.e., “mean imputation”, Little and Rubin 1987; Fox 2008).

Physical Habitat

Removal of contamination, rather than habitat restoration, was the intended purpose of Superfund remediation (MTDEQ and USEPA 1996; MTNRDP et al. 2005). As a result, we predicted that the habitat quality of the remediated channel would be poor and specific physical habitat characteristics may be negatively associated with the occurrence and abundance of salmonids. To evaluate this prediction, we quantified four habitat characteristics within each sample section that were influenced directly by remediation and stream channel reconstruction and that have also previously been associated with salmonid abundance (Table 1): percent pool area, percent coarse substrate (Brittain et al. 1993), percent overhead vegetative cover (Butler and Vernon 1968; Romero et al. 2005), and width to depth ratio (Torgersen et al. 1999). Physical habitat data allowed for the relative comparison of specific physical habitat factors to each other as well as to specific water quality factors. Stream habitat surveys were conducted during September 2011.

Habitat variables were derived from calculated proportional area estimates of each individual geomorphic channel units (e.g., pools, riffles, Bisson et al. 2006) within each sample section. Individual channel unit dimensions were estimated visually; at every tenth sampling unit, channel length and width was measured by tape to assess the precision of visual estimates (Moore et al. 1997). Estimated precision of visual estimates were within 20%. The total sample section area was estimated as the sum of all

individual channel unit areas within each section. The percent pool area was calculated as the sum of scour and dam pool unit areas within each section divided by the total section area. Percent coarse substrate (substrate >10 cm diameter) and percent vegetative cover were also estimated visually in each channel unit. The Coarse variable (Table 1) was calculated as the average of all channel unit percent coarse substrate estimates within each section and weighted proportionally by the channel unit area. The percent vegetative cover (Cover, Table 1) was estimated visually for each channel unit and quantified in the same manner as the Coarse variable. The width-depth ratio was measured at the longitudinal midpoint of each channel unit, at the same transect where the wetted width was estimated and the thalweg depth was measured. The WD variable (Table 1) was calculated as the average wetted width to thalweg depth ratio from each channel unit, within each section, weighted in the same manner as the Cover and Coarse variables.

Occurrence and Abundance

Spatially-continuous electrofishing surveys were conducted to describe longitudinal patterns in salmonid distribution and abundance. Salmonid occurrence and abundance were measured in each 100-m sample section using single-pass electrofishing as described by Bateman et al. (2005) using a pulsed-DC, Smith-Root backpack electrofishing unit (Smith-Root, Inc., Model LR-24, Vancouver, WA). Field crew size varied from four to six with a minimum of two persons netting fish. Stream conductivity ranged from 200 to 600 mS/cm. Electrofishing power ranged from 200 to 350 V, from 30 to 35 Hz, and the duty cycle was set at 12%. Mean electrofishing time per 100-m

section was 365 s. All sampled westslope cutthroat trout and brook trout were weighed (nearest 0.1 g) and measured (nearest mm TL) to examine differences in size frequencies between segment cohorts. Electrofishing surveys were discontinued for the day when stream temperature exceeded 17°C. Field crews typically sampled 20-30 sample sections per day.

Statistical Analysis

Regression models were developed to estimate the likelihood of salmonid occurrence and relative abundance as a function of water quality and physical habitat. Pearson correlations among explanatory variables were examined prior to analyses to consider colinearity. Moderate colinearity of dissolved oxygen (DO, Table 1) and ammonia (NH₃, Table 1) was evident in both years ($r = 0.69$ and 0.59 in 2010 and 2011 respectively). However, removing or retaining either DO or NH₃ from global (all-variable) regression models did not alter the coefficient direction or significance ($\alpha = 0.05$) of either variable, and therefore both were retained for consideration in the regression models.

Logistic regression was used to evaluate the influence of specific water quality and physical habitat factors on salmonid occurrence. Logistic regression is commonly applied to model species occurrence in ecological data (Zuur et al. 2007). Zero-altered negative binomial (ZANB) regression was used to evaluate the influence of those same factors on salmonid relative abundance. In both study years, large proportions of the sections sampled had counts of zero salmonids (62% and 43% in 2010 and 2011 respectively). Poisson regression is not recommended for modeling relative abundance

data with zero-inflation (Zurr et al. 2009) and overdispersion (dispersion parameters were >1.6 in both years; Ramsay and Schafer 2002; Fox 2008). Zero-altered regression models remedy zero-inflation by modeling count data in two-parts; a logistic regression modeling a binary response (occurrence) and a zero-truncated regression model for the nonzero counts (Zuur 2009; Santos et al. 2011). ZANB models were fitted with the R software package “pscl” (Zeileis et al. 2008).

Separate logistic and ZANB regression models were developed for both study years for several reasons. First, the overall population size and distribution pattern of main stem salmonids was expected to vary from year to year. Movement data from related research between 2009 and 2011 indicated that a net increase in salmonid abundance likely occurred in main stem sections between 2010 and 2011 through immigration from tributary populations (J. Naughton, unpublished data). Reestablishment of the main stem salmonid populations appears to be a recent phenomenon (MTFWP 2009). Additionally, dissolved copper data was only available for 2011. Finally, Superfund remediation activities in progress in Durant Canyon segments in July 2010 precluded sampling in those segments in July 2010. As a result, sampling occurred in fewer sections in 2010 ($n = 239$) compared to 2011 ($n = 326$).

To assess the relative importance of water quality and physical habitat associated with salmonid occurrence and relative abundance, we defined a priori sets of candidate logistic regression and ZANB regression models that represented combinations of the three water quality (DO, Cu, NH₃) and four physical habitat (Pool, WD, Coarse, Cover) variables (Table 1) including all-variable (global), water quality-only, and habitat-only

models. Schwarz's Bayesian Criterion (BIC) was used to select the most parsimonious of the candidate models (Burnham and Anderson 1998; Fox 2008). BIC was used for model selection rather than Akaike's Information Criteria (AIC) due to the additional penalty BIC enforces for large sample sizes (Ramsay and Schafer 2002; Santos et al 2011).

Spatial autocorrelation is of concern when continuously sampling linear networks (Cliff and Ord 1973; Cressie 1993; Ganio et al. 2005). We evaluated spatial autocorrelation by examining residual plots, semivariograms, and autocorrelation function plots. In residual plots, spatial autocorrelation manifests by patterning or "runs" in the data (Cliff and Ord 1973; Ramsay and Schafer 2002). A semivariogram plots semivariance by sample section and spatial autocorrelation is identified by an increasing trend with lag distance (Cressie 1993; Ganio et al. 2005). Autocorrelation function plotted by lag distance identifies spatial autocorrelation by excessive values outside the 95% confidence intervals (Ramsay and Schafer 2002; Fox 2008). Examination of residual plots, semivariograms, and autocorrelation function plots of the most parsimonious logistic (Figure 3) and ZANB (Figure 4) regression models did not indicate that spatial autocorrelation would bias regression analyses. All statistical analyses were conducted with R statistical software (R Core Development Team 2012). Semivariograms were created with the R software package "geoR" (Diggle and Ribiero Jr. 2007).

The model fit of the most parsimonious (lowest BIC score) logistic regression models were evaluated with the McFadden pseudo R-squared statistic and by receiver-operating characteristic (ROC) plots. The model fit of the most parsimonious ZANB

regression models were evaluated by ROC plots. The McFadden pseudo R-squared statistic, also known as the likelihood ratio index, may be conservative compared to other pseudo R-squared statistics and values between 0.2 and 0.4 indicate a reasonably well fitting model (Hu et al. 2006). ROC plots represent the relation between sensitivity (true positive fraction) and specificity (false positive fraction) in a fitted model and provide a measure of model accuracy or predictive power (Fielding and Bell 1997; Ripley et al. 2005; Al-Chokhachy and Budy 2007). The area under the curve of an ROC plot (AUC) provides a statistical measure of predictive value with 0.5 indicating a model with no predictive value and values approaching 1.0 indicating stronger predictive value. Models with AUC values of 0.7 to 0.9 are considered to have moderate accuracy and those with AUC values greater than 0.9 are considered to have high accuracy (Manel et al. 2001). McFadden pseudo R-squared statistics were calculated with the R software package “pscl” (Jackman 2012). ROC plots were created with the R software package “pROC” (Xavier et al. 2011).

Results

Water Quality and Physical Habitat

Water quality in Silver Bow Creek deteriorated substantially below the Butte wastewater treatment plant effluent. Effluent discharge ($\approx 0.24 \text{ m}^3/\text{s}$) comprised approximately 40% of the downstream flow of Silver Bow Creek ($\approx 0.60 \text{ m}^3/\text{s}$) during the July 2010 and July 2011 sample periods and the effluent appeared to be a primary source of ammonia and copper (e.g., concentrations of $\text{NH}_3\text{-N}$, dissolved Cu, and total

recoverable Cu in the effluent on 19 July 2011 were 18.0 mg/L, 17.9 µg/L, and 36.9 µg/L respectively). Ammonia and copper concentrations in Silver Bow Creek increased immediately below the effluent (Figures 5 and 6). Ammonia concentrations declined rapidly, approaching background levels similar to the Butte segment, by the Ramsay Flats segment (Figures 5 and 6). By contrast, dissolved copper levels remained elevated throughout all downstream segments (Figure 6). Between the Degradation and Active Decomposition reaches of the Hypoxic segment DO decreased by more than 4.0 mg/L within only a few km (Figures 5 and 6). Stream zones with DO concentrations below aquatic life standards for juvenile salmonids (i.e., 5.0 mg/L, USEPA 1986a) were extensive (<6 km) in both years and reached hypoxic levels (e.g., 2.25 mg/L and 2.28 mg/L in 2010 and 2011 respectively, Figures 5 and 6).

Longitudinal trends in physical habitat were evident. Pool area varied substantially from section to section across the study area, but in general pool area was highest in the LAO reach (stream km 4.0-6.8) and in the Ramsay Flats segment (Figure 7). The wetted channel width to thalweg depth ratio (Width:depth) was generally uniform throughout the study area at a ratio of approximately 20:1, with little variation between sections (Figure 7). Coarse substrate percentage was high (typically 50%) in the Durant Canyon segments, but low (typically <20%) in all upstream segments (Figure 7). Cover was absent in the Durant Canyon and Ramsay Flats segments but increased in each upstream segment (Figure 7).

Occurrence and Abundance

Spatial patterns in the relative abundance of salmonids in Silver Bow Creek generally reflected spatial DO patterns in both 2010 (Figure 5) and 2011 (Figure 6) and these patterns were consistent between sample years (Figure 8). Differences in salmonid relative abundances between segments and reaches were substantial. For example, salmonid relative abundance (expressed as mean trout/100m \pm 95% C.I.) in the Blacktail Creek reach (2010 = 10.78 \pm 2.44, 2011 = 11.83 \pm 2.67) exceeded that in the Active Decomposition reach (2010 = 0.02 \pm 0.04, 2011 = 0.16 \pm 0.13) and the Recovery segment (2010 = 0.08 \pm 0.08, 2011 = 0.36 \pm 0.15) by approximately two orders of magnitude (Figure 8). Relative abundance in the Degradation reach (2010 = 0.89 \pm 0.56, 2011 = 1.77 \pm 0.84), and the Ramsay Flats segment (2010 = 2.47 \pm 0.93, 2011 = 1.73 \pm 0.71) were also high compared to the Active Decomposition and Recovery segments, but low compared to the Blacktail Creek segment (Figure 8). In the Durant Canyon segments, sampled only in 2011, relative abundances were similar to the Ramsay Flats segment immediately upstream (Figure 8).

The most parsimonious logistic regression model in 2010 included the DO, Coarse, and Cover variables (Table 2). In 2011, when Cu was included in the suite of variables considered, the most parsimonious logistic regression model included DO, Cu, and Coarse (Table 2). The most parsimonious ZANB regression model included DO and Cover in 2010, but DO, Cu, and NH₃ (water quality-only model) in 2011 (Table 3). All of the most highly ranked logistic and ZANB regression models included DO (Tables 2 and 3). Similarly, in 2011, all of the most highly ranked logistic and ZANB regression

models included Cu (Tables 2 and 3). Habitat only regression models were the least supported by the data and variables quantifying channel geometry (Pool and WD, Table 1) were not retained by any of the most highly ranked models.

In 2010, fish sampling occurred in all segments upstream from the Durant Canyon segments (Figure 1) and salmonids were absent from the majority (62%) of the sections sampled. Of 239 sections sampled in 2010, westslope cutthroat were present in 34 (14%), and brook trout were present in 72 (30%). The most parsimonious logistic regression model for the 2010 data indicated that salmonid occurrence was positively related to DO (p -value <0.0001), Coarse substrate (p -value = 0.0126), and Cover (p -value = 0.0157, Table 4; Figure 9). Dissolved oxygen had a strong positive association with the occurrence of salmonids. A one unit (mg/L) increase in DO (range: 2.25-8.02) was associated with an increased likelihood of salmonid occurrence of 6.18 times after controlling for variation in Coarse and Cover (Table 4). The association of salmonid occurrence with Coarse and Cover was also positive but weak compared to DO. The likelihood of salmonid occurrence associated with a 10% increase in Coarse (range: 0-62%) was 1.87 and for a 10% increase in Cover (range: 0-50%) was 1.65 (Table 4). The model fit well (McFadden pseudo R-squared = 0.529) and accuracy was high (AUC = 0.939, Delong's 95% confidence interval for AUC = 0.901-0.969).

In 2011, fish sampling included all sample sections from 2010 and an additional 87 sections in the upper and lower Durant Canyon segments. Dissolved copper concentration (Cu, Table 1) was added to the array of variables measured and included in the analysis. Salmonids were present in 186 (57%) of the 326 sample sections.

Westslope cutthroat trout were present in 128 (39%) and brook trout were present in 102 (31%) of the sample sections. The most parsimonious logistic regression model indicated that salmonid presence was related to DO (p -value <0.0001), Cu, (p -value <0.0001), and Coarse ($p = 0.0126$, Table 4; Figure 9). DO was again positively associated with salmonid occurrence, but this relationship was not as strong as in 2010. Each one unit (mg/L) increase in DO (range: 2.28-8.84 mg/L) was associated with an increased likelihood of salmonid occurrence of 1.61 times after controlling for variation in Cu and Coarse (Table 4). Each one unit ($\mu\text{g/L}$) increase in Cu (range: 1.0-19.0 $\mu\text{g/L}$) was associated with a decreased likelihood of salmonid occurrence of 0.88 after controlling for variation in DO and Coarse (Table 4). Each 10% increase in Coarse (range: 0-90%) was associated with an increased likelihood of occurrence of 1.31 times after accounting for variation in DO and Cu (Table 4). The model fit reasonably well (McFadden pseudo R -squared = 0.304) and had moderately high accuracy (AUC = 0.851, Delong's 95% confidence interval = 0.809-0.893).

In 2010, westslope cutthroat trout comprised 12% (91/749) of all salmonids sampled. The most parsimonious ZANB regression model (Table 3) indicated that salmonid abundance was positively related to DO (p -value <0.0001) and Cover (p -value = 0.0001, Table 5; Figure 10). The influence of DO on salmonid abundance was particularly strong. Each one unit (mg/L) increase in DO was associated with an increase of 0.94 more salmonids per sample section when Cover was held constant (Table 5). However, the influence of Cover on salmonid abundance was small by comparison. A 10% increase in Cover was associated with an increase of 0.26 salmonids per section

holding DO constant (Table 5). Model accuracy was moderate although the confidence interval was large (AUC = 0.809, Delong's 95% confidence interval = 0.684-0.934).

In 2011, westslope cutthroat trout accounted for 29% (333 of 1152) of all salmonids sampled. The most parsimonious 2011 ZANB regression model (Table 3) indicated that trout abundance was positively related to DO (p -value <0.0001) and NH₃ (p -value <0.0001), and negatively related to Cu (p -value <0.0001, Table 5; Figure 10). Each one unit (mg/L) increase in DO was associated with 0.67 more salmonids per sample section and each one unit (mg/L) increase in NH₃ (range: 0.09-2.76) was associated with an increase of 0.58 salmonids per sample section (Table 5). Each one unit (µg/L) increase in Cu (range: 0.0-19.0) was associated with 0.14 fewer salmonids per sample section (Table 5). Model accuracy was moderately high (AUC = 0.845, Delong's 95% confidence interval = 0.803-0.888).

Salmonid species composition shifted at the wastewater effluent. Upstream from the effluent brook trout dominated the sample catch in both years; 99% (631 of 637) in 2010 and 98% (705 of 716) in 2011. However, westslope cutthroat trout comprised the majority below the effluent; 81% (85 of 105) in 2010 and 74% (322 of 436) in 2011.

Discussion

Superfund remediation activities have removed streamside tailings (MTDEQ and USEPA 1996), capped and removed exposed waste rock piles in the uplands (USEPA 2006), and treated contaminated groundwater (USEPA 2006), resulting in substantial improvement to the water quality of the stream (Chapter 1; USEPA 2011). The presence

of salmonids in the remediated segments of Silver Bow Creek signifies substantial progress toward a fundamental goal articulated in the watershed restoration plan (MTNRDP et al. 2005). Without remediation it is unlikely salmonids would presently occupy any portion of Silver Bow Creek. However, despite water quality improvement, hypoxia from ammonia pollution coupled with copper levels that remain acutely toxic to fish were the primary factors limiting the occurrence and abundance of salmonids in Silver Bow Creek during the summer sample periods of 2010 and 2011. Hypoxia and elevated copper levels affected major portions of the stream, with at least 6 km (18%) of the study area having DO levels below aquatic life standards for streams with salmonids (5 mg/L, USEPA 1986b) and 27 km (80%) affected by acutely toxic copper levels. Both of these factors were of primary importance in determining whether or not salmonids were present in a sample section and their relative abundance.

Dissolved oxygen concentration was a primary factor governing the occurrence and abundance of salmonids in Silver Bow Creek in both 2010 and 2011. All of the most parsimonious logistic and ZANB regression models included DO and the association of DO with salmonid occurrence and abundance was always positive and highly statistically significant. Salmonids are oxygen-sensitive fishes (USEPA 1986b) and avoid hypoxia (Poulsen et al. 2011). Branchial oxygen-sensitive chemoreceptors located in the gill enable fishes to sense the surrounding DO levels (Poulsen et al. 2011) and these chemoreceptors may be involved in avoidance behavior (Schurmann et al. 1998; Skjaeraasen et al. 2008; Herbert et al. 2011). Mild hypoxia is tolerable by adult fishes through the regulation of oxygen consumption and aerobic metabolism (Gamperl and

Driedzic 2009; Perry et al. 2009) although at the cost of reduced growth and swimming ability (USEPA 1986a). In experimental settings, salmonid mortality has been observed at DO concentrations of 3 mg/L for 10 hour exposure periods (USEPA 1986a). In both 2010 and 2011, DO concentrations in Silver Bow Creek dipped well below 3 mg/L and concentrations remained at these levels for up to 10 hours. In this study, salmonids were rarely sampled in sections with DO <6.0 mg/L, and even more rarely sampled in sections with DO <4.0 mg/L.

Hypoxia in Silver Bow Creek was directly related to ammonia discharged into the stream from the Butte wastewater treatment plant effluent. Nitrogen yields (nitrogen loads per watershed area) were high in Silver Bow Creek, even compared to other “developed” watersheds (Plumb 2009). The Butte wastewater effluent was the primary source of at least 80% of the nitrogen in Silver Bow Creek, nearly all of which was discharged to the stream as ammonia (Plumb 2009). The ammonia-rich effluent supported increases in primary productivity (aquatic plant and algae growth) downstream, and photosynthesis rates were high (Gammons et al. 2011). However, at night when no photosynthesis occurred, respiration rates (the oxidative decay of accumulated organic materials) led to hypoxia (Gammons et al. 2011). Between the Degradation and Active Decomposition reaches of the Hypoxic segment, respiration rates were extremely high and DO was rapidly depleted.

Hypoxia in Silver Bow Creek was accompanied by elevated copper levels and dissolved copper concentration was strongly negatively associated with salmonid occurrence and abundance. Total recoverable copper concentrations exceeded acute

toxicity levels and dissolved copper concentrations approached acute toxicity levels in the majority of the study area below the wastewater effluent. Synoptic data from the main stem, combined with the high copper concentrations measured in the effluent, suggest that the Butte municipal wastewater treatment plant may be a primary source of copper to Silver Bow Creek.

Despite high ammonia concentrations the influence of ammonia on salmonid occurrence was negligible and ammonia was not retained in the most parsimonious logistic regression models in either 2010 or 2011. However, ammonia concentration had a statistically significant, but weak, positive association with salmonid abundance in 2011. In comparing the spatial abundance trend in both years, one notable difference was the greater extent and abundance of salmonids in the Degradation reach of the Hypoxic segment where ammonia concentrations were highest in both years. This change largely was responsible for the increased influence of ammonia on abundance in the 2011 ZANB regression model.

Given the high levels of ammonia in Silver Bow Creek, ammonia toxicity may be a potential problem. Ammonia becomes toxic if the proportion of un-ionized ammonia (NH_3) to ionized ammonium (NH_4^+) in solution is high. The proportion of NH_3 to NH_4^+ is positively related to the water temperature and pH (USEPA 1998). Water temperatures in Silver Bow Creek are relatively high during the summer, often exceeding 20°C . During our nighttime synoptic sampling, the pH and temperatures were not sufficient to induce ammonia toxicity at temperatures measured in Silver Bow Creek (USEPA 1998). Photosynthesis during the daytime raises the pH of the water by consuming CO_2 (a weak

acid) and by reducing nitrate to organic nitrogen (Gammons et al. 2011). Therefore, the possibility for ammonia toxicity in Silver Bow Creek during the daytime cannot be discounted. However, we did not observe negative relationships between salmonid occurrence or abundance with ammonia concentration.

Water quality in Silver Bow Creek was highly variable on hourly as well as seasonal time scales. Diel (24-hour) DO and dissolved organic carbon (DOC) swings in Silver Bow Creek were among the highest in the published literature due to the extreme rates of photosynthesis and respiration (Gammons et al. 2011). Metal concentrations in streams are known to fluctuate on diel cycles (Nimmick et al. 2011) and diel fluctuations in the zinc concentration of Silver Bow Creek were among the highest measured in any stream worldwide (D. Nimmick, U.S. Geological Survey, unpublished data). Diel fluctuations in metal concentrations may benefit salmonid survival by providing periodic relief from acute toxicity (Nimmick et al. 2007), and this may be true of DO fluctuations as well. However, extreme diel fluctuations of DO, DOC, zinc, and possibly other metals in Silver Bow Creek provides a highly unstable habitat for salmonids and other fishes

One interesting finding from this study was the shift in salmonid species composition occurring at the Butte wastewater effluent; from predominantly brook trout above to predominantly westslope cutthroat trout below the effluent. Askey et al. (2007) observed a similar phenomenon that occurred below effluent discharges from Calgary, Alberta into the Bow River. I hypothesize that this shift may be related to the water quality conditions below the effluent, specifically, an interaction between species tolerance and interspecific competition. Nonindigenous brook trout populations

commonly displace indigenous westslope cutthroat trout populations in streams where they have been translocated in the intermountain western U.S. (Peterson et al. 2004; Shepard 2010). By modeling occurrence and abundance of salmonids as a combined taxa, rather than each species separately, I focused this study on the influence of water quality on salmonid abundance without the confounding effects of interspecies competition. However, water quality conditions in Silver Bow Creek may provide an opportunity to investigate the dynamics of interspecific competition within the context of these strong water quality gradients.

Important ecological processes influencing the distribution and abundance of organisms occur at broad spatial scales (Wiens 1989) and broad perspectives may be necessary to understand processes occurring at these scales (Turner 2005). Observed in a narrow spatial context (e.g., isolated sample units randomly distributed along a stream network), ecological metrics such as the abundance of a given species in a particular location may be unexplainable without a sufficiently broad perspective. Stream ecologists have begun to incorporate perspectives and theory from landscape ecology into studies of fishes in lotic systems, but the spatial scale at which data is collected must be appropriate to the questions of interest (Torgersen et al. 1999; Fausch et al. 2002; Gresswell et al. 2006). Fausch et al. (2002) recommend that systematic census of coarse-grain habitat and fish assemblage data may be the only way to understand broad-scale processes in stream networks. However, to census a stream or river reach, sample effort within units generally must be reduced, or the overall sampling effort and expense becomes impractical. Bateman et al. (2005) demonstrated that in headwater streams,

single-pass electrofishing counts provide reliable relative abundance estimates at substantially reduced effort compared to depletion (Zippin 1958) or mark-recapture (Otis et al. 1978, cited by Korman et al. 2010) abundance estimates. The reduced effort within sample units allows for increased sample fraction (proportion of sampled units within the sample frame) for the same total effort, as is recommended by Hankin (1984), Hankin and Reeves (1988), and Mitro and Zale (2000). By increasing the sample fraction researchers substantially improve their ability to observe longitudinal patterns of species abundance in streams.

This study provides a rare example of a spatially-continuous stream sampling approach (for other examples see Torgersen et al. 1999; Johnson et al. 2005; Gresswell et al. 2006) at scales sufficiently broad (34 km) to evaluate the influence of water quality factors on the occurrence and abundance of fish. The vast majority of the sample sections in this study area had never been previously sampled. Other sampling efforts in this stream (MTFWP 2009), and in most other streams (Imhof et al. 1996; Fausch et al. 2002; Gresswell et al. 2006), have targeted isolated sample units. In this study, spatially-continuous sampling allowed for detailed observation of broad spatial patterns clearly outlining the effect of hypoxia on salmonid occurrence and abundance. Spatially-continuous sampling also identified the Ramsay Flats segment and the Degradation reach of the Hypoxic segment (neither of which had been previously sampled) as areas where westslope cutthroat trout were particularly abundant in both years, and these areas were crucial in isolating and identifying the influence of DO from other factors.

Management Implications

Salmonid populations have recovered in major portions of Silver Bow Creek despite continued water quality problems related to the wastewater treatment plant effluent and elevated copper concentrations. During the time period in which this study was conducted (2010-2011) the municipal wastewater treatment plant had a poor ability to control ammonia and copper concentrations in its effluent. Silver Bow Creek is small ($\approx 0.6 \text{ m}^3/\text{s}$) relative to the effluent discharge ($\approx 0.24 \text{ m}^3/\text{s}$) and the ammonia caused severe hypoxia downstream. Despite the expenditures involved in the remediation of metal contamination in Silver Bow Creek (over \$100 million, MTNRDP et al. 2005), the severity of the nitrogen-eutrophication problem in the stream has only recently been identified (Plumb 2009) and the geochemical processes causing hypoxia only recently described (Gammons et al. 2011).

However, hypoxia was not the only water quality problem in Silver Bow Creek. Total recoverable copper concentrations were acutely toxic and dissolved copper concentrations were near acutely toxic levels. The majority of the copper appeared to be derived from the wastewater effluent. According to these data, salmonid abundance would increase if the wastewater treatment plant is upgraded in a way that removes copper from the effluent as well as ammonia. At the time of this writing (2012) wastewater treatment plant upgrades were in the planning stages. Based on the data collected during this study, we predict that salmonids would become more widely distributed and more abundant downstream from the wastewater effluent if the ammonia and copper released through the Butte wastewater effluent are substantially reduced.

Seasonally, ammonia and DO were highly variable in Silver Bow Creek due to the lowered water temperatures and lower rates of respiration (Plumb 2009). During the cold months, hypoxia does not occur in Silver Bow Creek (Chapter 3). Based on the variables included in these summertime regression models, the water quality conditions of Silver Bow Creek in winter (high DO and NH₃ concentrations and similar copper concentrations, Chapter 3) would be considerably more favorable for salmonids compared to summer. It seems reasonable to conclude that salmonid abundance in Silver Bow Creek segments below the effluent would be higher during colder months and sampling by Montana Fish Wildlife and Parks in the spring and fall appears to support this conclusion (J. Lindstrom and T. Selch, Montana Fish Wildlife and Parks, unpublished data).

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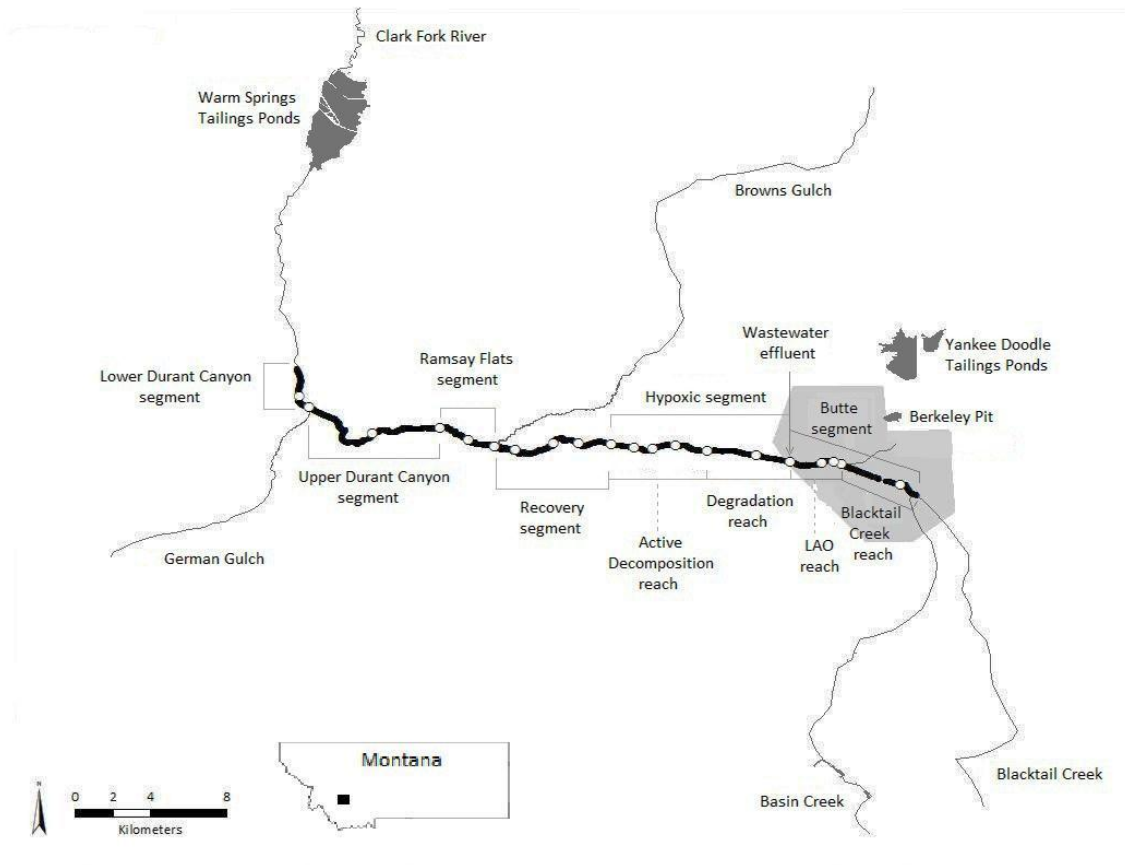


Figure 1.-Silver Bow Creek study area in southwestern Montana. Fish abundance sampling occurred in all sections highlighted by thick black line. White circles indicate the locations of water sample sites. Grey polygon represents urban area of the city of Butte, Montana.

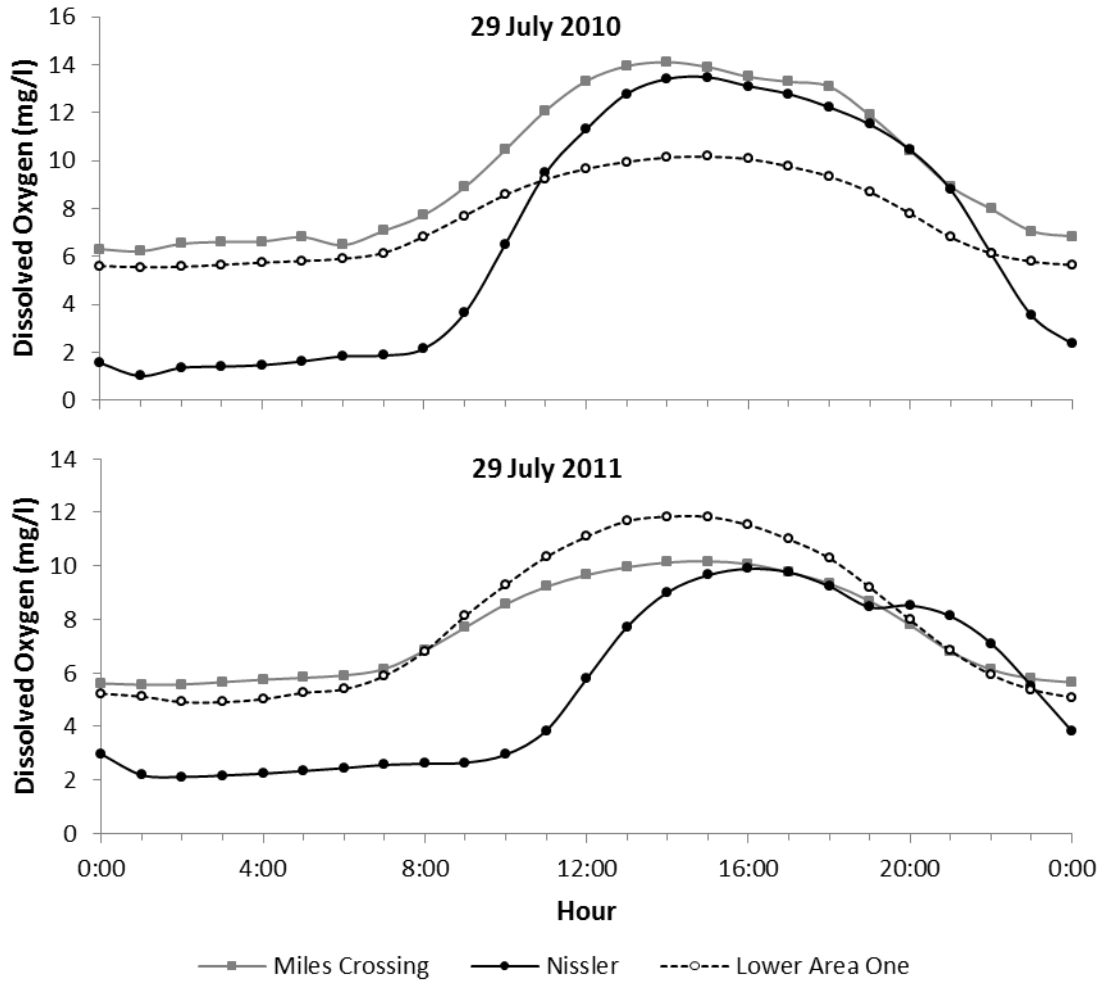


Figure 2.-Hourly DO (mg/L) in Silver Bow Creek, Montana at Miles Crossing, Nissler, and Lower Area One. In relation to the Butte wastewater treatment plant effluent, the sites are 17.7 km downstream (Miles Crossing), 8.0 km downstream (Nissler), and 1.5 km upstream (Lower Area One) from the effluent outfall.

Table 1.-Variables estimated in each sample section of Silver Bow Creek, Montana.

Variable	Method of estimation
Trout	Total catch of westslope cutthroat trout <i>Oncorhynchus clarkii lewisi</i> and brook trout <i>Salvelinus fontinalis</i> .
DO ^a	Minimum dissolved oxygen concentration during the sampling periods in 2010 and 2011 (mg/L).
NH ₃ ^a	Total dissolved ammonia concentration measured on 28 July 2010 and 19 July 2011 (NH ₃ -N, mg/L).
Cu ^a	Total dissolved copper concentration measured on 19 July 2011 (µg/L).
Pool	Percent of section area comprised of pool channel unit types (scour pools and dam pools).
Cover	Visually estimated percent surface area shaded by vegetative cover when sun is directly overhead.
Coarse	Percent of section bottom area with coarse substrate (particle size diameter >10 cm).
WD	Ratio of wetted channel width to thalweg depth of each section.

^a Water quality measurements were made in 20 sample sections. Water quality values were imputed (filled in) to sections with missing data by averaging the closest upstream and downstream measurements and weighting by distance (mean imputation; Little and Rubin 1987).

Table 2.-Model selection results for a candidate set of binomial logistic regression models containing combinations of water quality (dissolved oxygen [DO], dissolved copper [Cu], and ammonia [NH₃] concentrations) and habitat variables (pool area [Pool], coarse substrate [Coarse], vegetative cover [Cover], and the wetted width to thalweg depth ratio [WD]) in relation to the occurrence of salmonids in 2010 ($n = 239$) and 2011 ($n = 326$) sample sections of Silver Bow Creek, Montana. Models were ranked in terms of the difference between their BIC score and the lowest score (Δ BIC).

Model	Number of parameters ^a	BIC	Δ BIC
2010			
DO, Coarse, Cover	4	172.02	0
DO, Coarse	3	172.86	0.84
DO, Cover	3	173.58	1.56
DO, NH ₃ , Coarse, Cover	5	174.13	2.11
DO, NH ₃ , Pool, Coarse, Cover	6	179.02	7
DO	2	179.84	7.82
Water quality: DO, NH ₃	3	183.72	11.7
Global: DO, NH ₃ , Pool, WD, Coarse, Cover	7	184.5	12.48
Habitat: Pool, WD, Coarse, Cover	5	282.54	110.52
2011			
DO, Cu, Coarse	4	333.37	0
DO, Cu	3	334.46	1.09
DO, Cu, Pool, Coarse	5	336.2	2.83
Water quality: DO, Cu, NH ₃	4	340.07	6.7
DO, Cu, Pool, Coarse, Cover	6	341.11	7.74
DO, Cu, NH ₃ , Pool, Coarse, Cover	7	346.25	12.88
Global: DO, Cu, NH ₃ , Pool, WD, Coarse, Cover	8	351.43	18.06
DO	2	360.28	26.91
Cu	2	384.97	51.6
Habitat: Pool, WD, Coarse, Cover	5	403.77	70.4

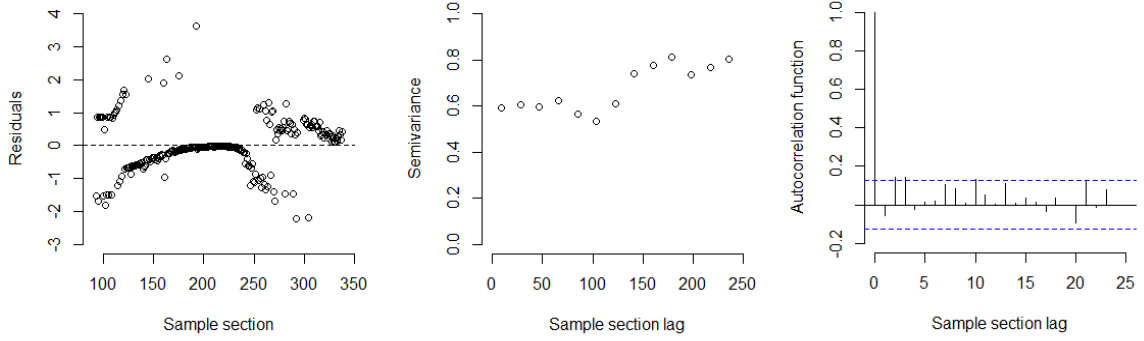
^a Parameters include the model variables and intercept.

Table 3.-Model selection results for the count portion of a candidate set of zero-altered negative binomial (ZANB) regression models containing combinations of water quality (dissolved oxygen [DO], dissolved copper [Cu], and ammonia [NH₃] concentrations) and habitat variables (pool area [Pool], coarse substrate [Coarse], and vegetative cover [Cover] percentage, and the wetted width to thalweg depth ratio [WD]) in relation to the relative abundance of salmonids in 2010 ($n = 239$) and 2011 ($n = 326$) sample sections in Silver Bow Creek, Montana. Models were ranked in terms of the difference between their BIC score and the lowest score (Δ BIC).

Model	Number of parameters ^a	BIC	Δ BIC
2010			
DO, Cover	8	683.61	0
DO, Pool, Cover	9	683.73	0.12
DO, Coarse, Cover	9	684.72	1.11
DO, Pool, Coarse, Cover	10	684.93	1.32
DO, Pool, WD, Coarse, Cover	11	689.17	5.56
DO	7	692.12	8.51
Global: DO, NH ₃ , Pool, WD, Coarse, Cover	12	695.38	11.77
Water quality: DO, NH ₃	8	696.2	12.59
Habitat: Pool, WD, Coarse, Cover	10	736.38	52.77
2011			
Water quality: DO, Cu, NH ₃	9	1230.19	0
DO, Cu, NH ₃ , Coarse	10	1234.73	4.54
DO, Cu, NH ₃ , Coarse, Cover	11	1239.76	9.57
DO, Cu	8	1241.19	11
DO, Cu, NH ₃ , WD, Coarse, Cover	12	1244.68	14.49
Global: DO, Cu, NH ₃ , Pool, WD, Coarse, Cover	13	1250.12	19.93
Cu	7	1267.91	37.72
DO	7	1326.54	96.35
Habitat: Pool, WD, Coarse, Cover	10	1332.53	102.34

^a Parameters include variables from count and logistic portions of each ZANB regression model, the additional theta parameter of the negative binomial model (not shown), and intercepts from each portion of the model. Count portions of each model (shown) include variables from the most parsimonious logistic models (see Table 2).

2010



2011

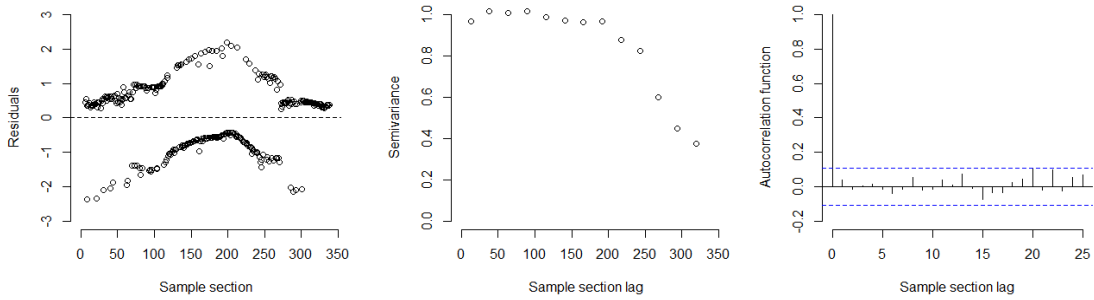
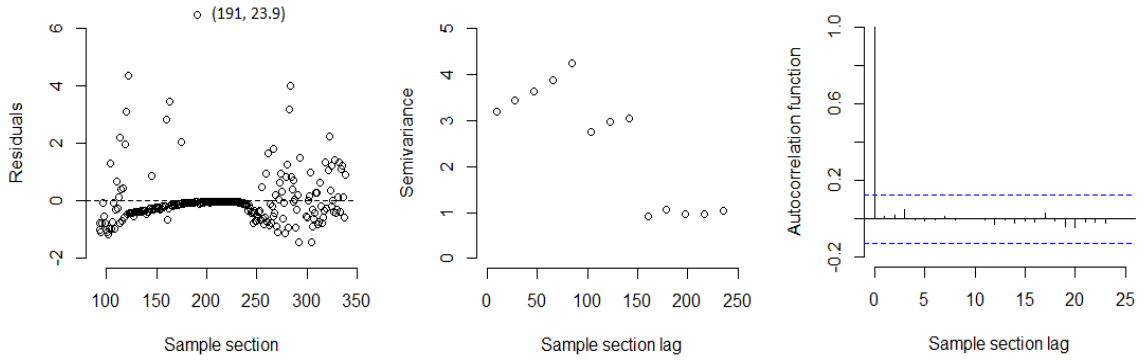


Figure 3.-Residual, semivariance, and autocorrelation function plots by sample section for the most parsimonious logistic regression models of salmonid occurrence in Silver Bow Creek, Montana for 2010 and 2011 sampling. Sample sections are ordered sequentially from the furthest downstream section.

2010



2011

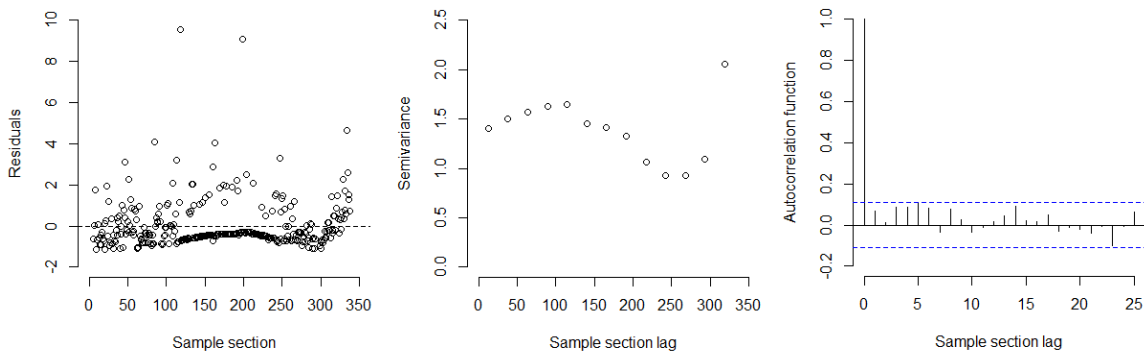


Figure 4.-Residual, semivariance, and autocorrelation function plots by sample section for the most parsimonious ZANB regression models of salmonid abundance in Silver Bow Creek, Montana for 2010 and 2011 sampling. Sample sections were ordered sequentially from the furthest downstream section.

2010

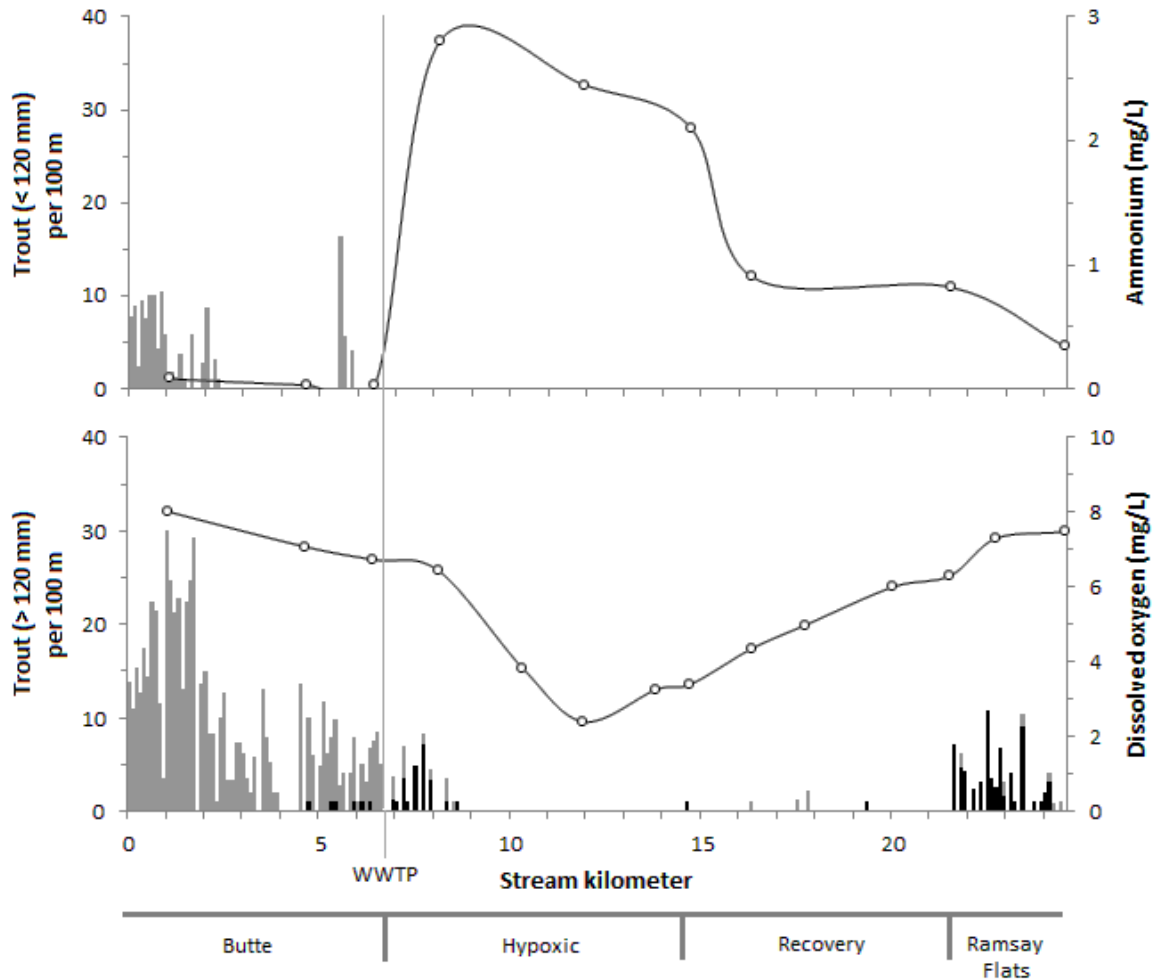


Figure 5.-Longitudinal abundances of westslope cutthroat trout *Oncorhynchus clarkii lewisi* (black bars) and brook trout *Salvelinus fontinalis* (grey bars stacked above black bars), and synoptic ammonia ($\text{NH}_3\text{-N}$), and dissolved oxygen concentrations in Silver Bow Creek, Montana. Fish sampling occurred from 6 to 26 July 2010. All synoptic measurements (white circles) occurred during predawn hours (0200-0630) on 28 July 2010. Vertical line indicates location of the Butte wastewater treatment plant effluent (WWTP). Study area segments are identified in the lower bar. Stream km were measured downstream from the upstream end of the study area.

2011

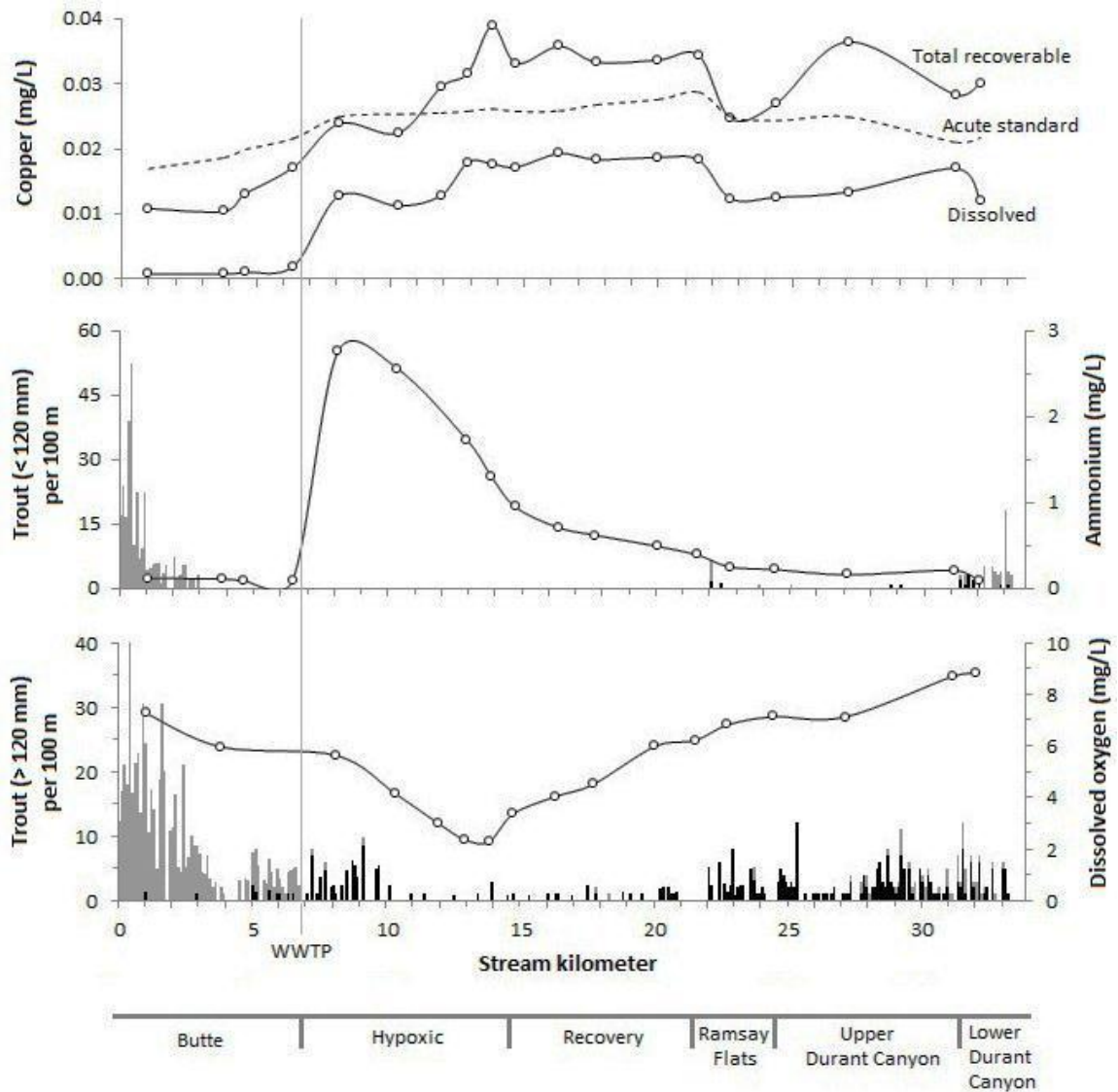


Figure 6.-Longitudinal abundances of westslope cutthroat trout *Oncorhynchus clarkii lewisi* (black bars) and brook trout *Salvelinus fontinalis* (grey bars stacked above black bars), and synoptic copper, ammonia (NH₃-N), and dissolved oxygen concentrations in Silver Bow Creek, Montana. Fish sampling occurred from 7 July to 4 August 2011. Synoptic copper and ammonia sampling occurred on 19 July 2011, and dissolved oxygen on 3 August 2011. All synoptic measurements (white circles) occurred during predawn hours (0200-0630). The hardness-based copper standard is from USEPA (1986a). Vertical line indicates location of the Butte wastewater treatment plant effluent (WWTP). Study area segments are identified in the lower bar. Stream km were measured downstream from the upstream end of the study area.

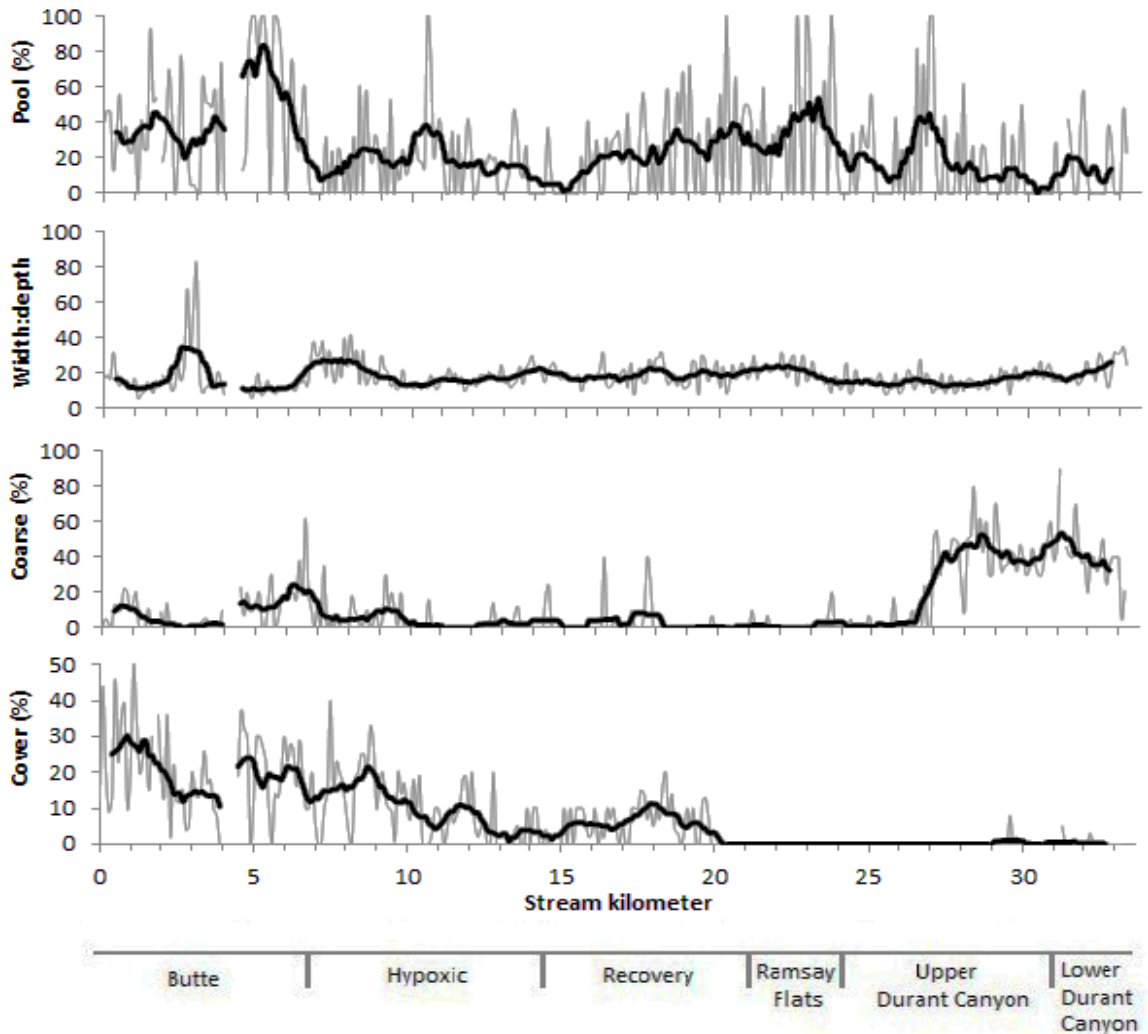


Figure 7.-Longitudinal stream profiles of pool area percentage, width to depth ratio, coarse substrate percentage, and channel cover percentage for Silver Bow Creek, Montana, September 2011. Thin grey lines are data summarized by each 100 m sample section. Black lines are the moving averages by stream km. Study area segments are identified in the lower bar. Stream km were measured downstream from the upstream end of the study area.

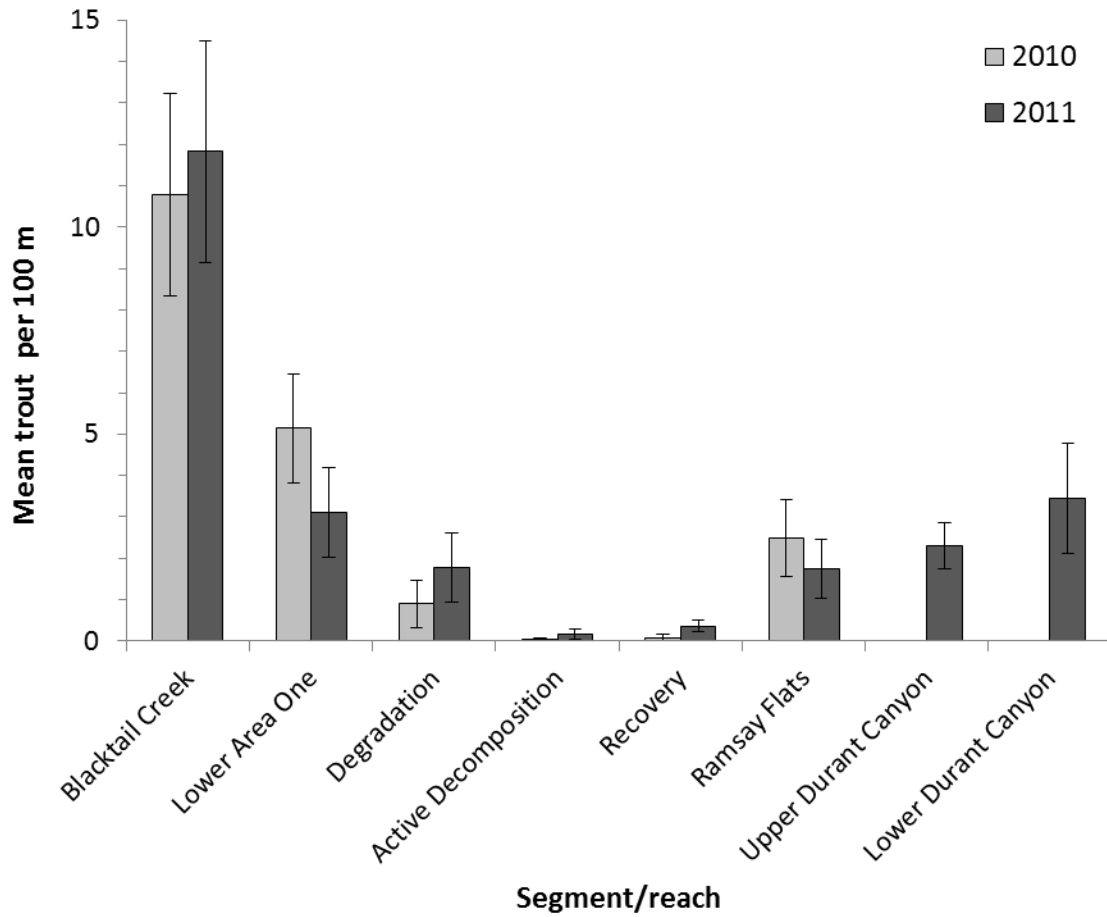


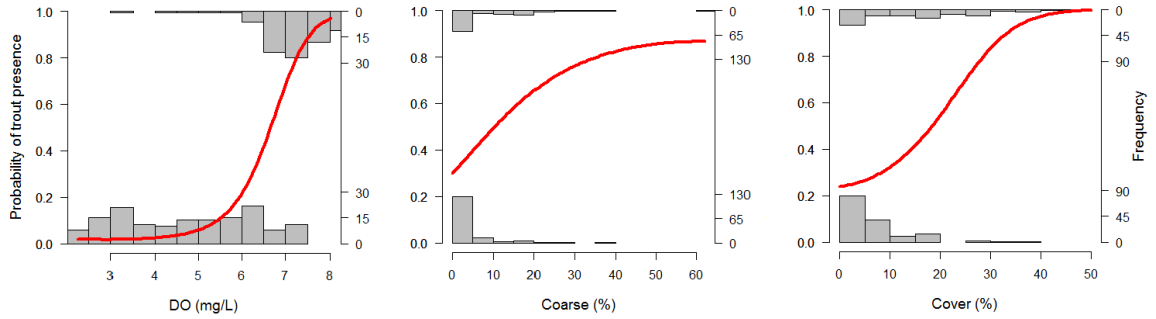
Figure 8.-Relative abundance of salmonids in Silver Bow Creek segments and reaches in July 2010 and July 2011. Error bars represent 95% confidence intervals. The upper and lower Durant Canyon segments were not sampled in 2010.

Table 4.-Coefficient estimates and associated measures for explanatory variables from logistic regression analysis of salmonid occurrence from 2010 ($n = 239$) and 2011 ($n = 326$) sample data, Silver Bow Creek, Montana.

Variable	Coefficient Estimate	Standard error	<i>P</i> -value	Odds ratio estimate	95% CI for odds ratio estimate
2010					
Intercept	-12.78	1.831	< 0.0001		
DO	1.821	0.267	< 0.0001	6.178	(3.838, 10.954)
Coarse	0.063	0.025	0.013	1.065	(1.016, 1.122)
Cover	0.049	0.02	0.016	1.051	(1.011, 1.096)
2011					
Intercept	-0.874	0.809	0.286		
DO	0.473	0.106	< 0.0001	1.605	(1.310, 1.991)
Cu	-0.132	0.027	< 0.0001	0.876	(0.829, 0.920)
Coarse	0.026	0.011	0.013	1.027	(1.006, 1.049)

Note: Dissolved oxygen (DO) units (mg/L) differ from dissolved copper (Cu) units ($\mu\text{g/L}$). Coarse units are the percentage of each sample section substrate area with > 0.1 m substrate particle size and Cover is the percentage of the section surface area shaded at mid-day.

2010



2011

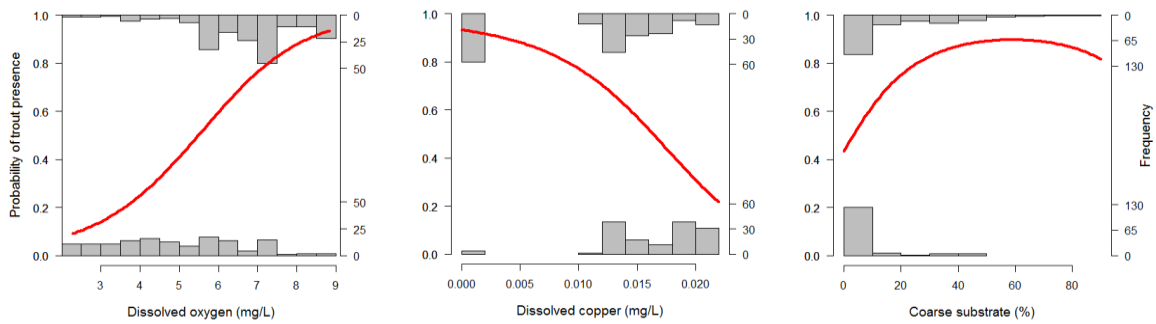


Figure 9.-Relationships between salmonid occurrence and explanatory variables from the preferred logistic regression models in Silver Bow Creek, Montana during 2010 and 2011 sampling.

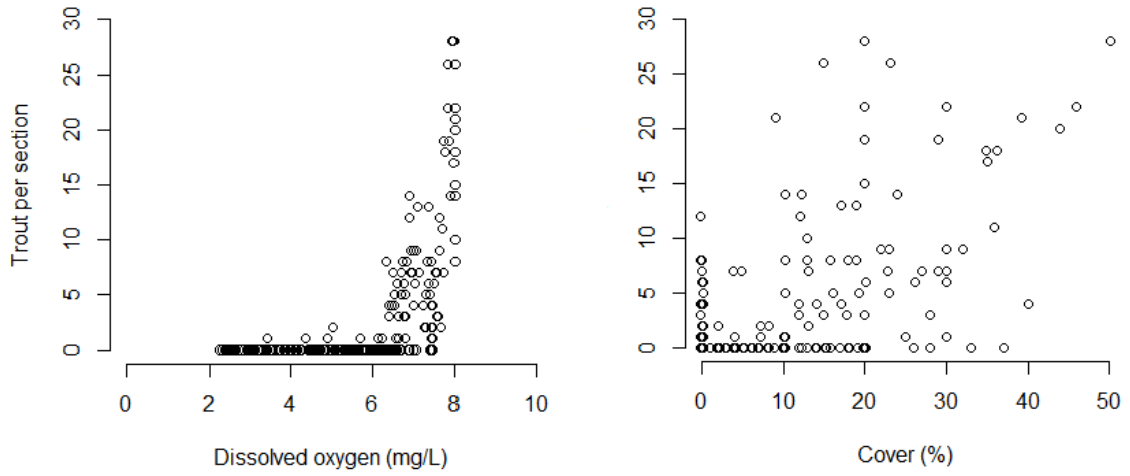
Table 5.-Coefficient estimates, standard errors, 95% confidence intervals, and *P*-values for explanatory variables from zero-altered negative binomial (ZANB) regression analysis of salmonid relative abundance in 2010 (*n* = 239) and 2011 (*n* = 326) sample sections, Silver Bow Creek, Montana.

Variable	Coefficient estimate	Standard error	95% CI for coefficient estimate	<i>P</i> -value
2010				
Intercept	-9.785	0.940		
DO	0.941	0.130	(0.686, 1.196)	< 0.0001
Cover	0.023	0.006	(0.012, 0.035)	0.0001
2011				
Intercept	-6.656	0.676		
DO	0.674	0.096	(0.486, 0.862)	< 0.0001
Cu	-0.140	0.011	(-0.162, -0.118)	<0.0001
NH3	0.575	0.137	(0.306, 0.844)	< 0.0001

Note: The logistic portions of each model (not shown) were modeled by the most parsimonious logistic regression models from each year (see Table 2).

60

2010



2011

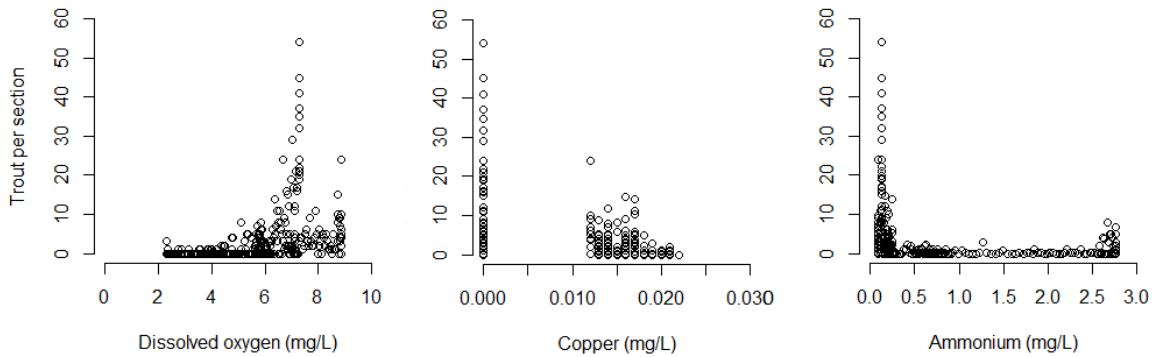


Figure 11.-Plots of salmonid relative abundance and explanatory variables from the most parsimonious ZANB regression models in Silver Bow Creek, Montana during 2010 and 2011 sampling.

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CHAPTER 3

SALMONID RECOLONIZATION OF A SUPERFUND REMEDIATED
MONTANA STREAMAbstract

Uncontrolled disposal of mining wastes in the Butte hill mining district of Montana, led to the extirpation of fish from Silver Bow Creek throughout the 20th century. Superfund remediation has been ongoing since 1998 and metal concentrations are reduced. However, water quality remained poor due to inadequate treatment of municipal sewage. From 2009-2011, we evaluated seasonal salmonid abundance and movement in the Silver Bow Creek watershed to assess the effectiveness of Superfund remediation in reestablishing salmonid populations and to describe the movement patterns associated with recolonizing salmonid populations. Westslope cutthroat trout, *Oncorhynchus clarkii lewisi* ($n = 787$) and brook trout *Salvelinus fontinalis* ($n = 1,846$) were PIT-tagged during three summer sampling periods. Movements of PIT-tagged individuals were monitored at seven stationary antenna sites and individuals were relocated during six seasonal portable antenna surveys. Water quality was poor below the wastewater effluent during the study period and was characterized by acutely toxic copper concentrations, elevated ammonia levels (e.g., $\text{NH}_3\text{-N} = 2.8$ mg/L), and hypoxia during summer nights (e.g., $\text{DO} = 1.4$ mg/L). Westslope cutthroat trout relative abundance increased between summer and winter in remediated segments that had been hypoxic during the summer. Few brook trout recolonized the remediated main stem

during the study period and the wastewater effluent may have deterred downstream brook trout movement. By contrast, westslope cutthroat trout moved into remediated segments during the late summer and early fall as hypoxia subsided. The majority of westslope cutthroat trout sampled in the main stem were large-bodied adults (≥ 200 mm TL) contrasting with the predominantly small-bodied counterparts in the tributaries. Recolonization of westslope cutthroat trout appears to have been driven by the reexpression of a fluvial-adfluvial migratory behavior, a pattern that was not possible during the 100-150 years of mainstem contamination. This population represents a rare case of a salmonid population reexpressing, rather than losing, a migratory life-history behavior. The rapid rate in which this migratory behavior reemerged, probably since 2006, suggests that the migratory capacity of this westslope cutthroat trout population persisted despite changes in the physical habitat template that prevented migration from unaffected tributary source populations.

Introduction

The majority of inland salmonid species exhibit potamodromous (movement between separated freshwater sites, Secor and Kerr 2009) life-history strategies, moving between feeding habitats and spawning and rearing habitats (Northcote 1997). Within watersheds, populations often display distinct life history forms in response to differences in the environmental template (Gresswell et al. 1994; Hutchings 2004). For example, westslope cutthroat trout *Oncorhynchus clarkii lewisi* indigenous to the upper and middle Columbia, South Saskatchewan, and upper Missouri watersheds (Behnke 1992) have

evolved diverse life-history strategies including fluvial, fluvial-adfluvial, and allacustrine forms, and these life-history strategies often evolved in sympatry (Liknes et al. 1988; McIntyre and Rieman 1995).

Westslope cutthroat trout are today threatened by habitat loss, habitat degradation, displacement by nonindigenous species, and hybridization with nonindigenous species (Allendorf and Leary 1988; Muhlfeld 2008; Gresswell 2011). Migratory life-history forms of westslope cutthroat trout and other salmonids are particularly at risk because lowland rivers and streams (i.e., high-order streams, Strahler 1952), where migratory adults feed, have been most directly affected by habitat disturbance and translocations of nonindigenous species (Varley and Gresswell 1988; Kruse et al. 2000; Nelson et al. 2002). In response, a common management remedy has been to isolate indigenous salmonid populations above barriers under the assumption that these isolated populations will persist because of their isolation from nonindigenous species (Hildebrand and Kershner 2000; Peterson et al. 2008; Gresswell and Vondracek 2010). However, this combination of anthropogenic disturbances, translocated nonindigenous species, artificial movement barriers, and isolated indigenous salmonid populations in headwater streams has contributed to the declines in the migratory life-history forms of indigenous salmonids (Nelson et al. 2002; Rieman et al. 2003). As a result, many indigenous salmonid populations are now dominated by small-bodied resident forms, particularly isolated populations (Northcote 1992; Kruse et al. 1997, 2000).

Potamodromous salmonid populations with access to high quality foraging habitats in rivers and lakes are likely to have high proportions of migratory individuals

because of the benefits of migratory behavior (greater size, fitness, and fecundity, Johnsson and Johnsson 1993; Bohlin et al. 2001; Olsson and Greenberg 2011). Because migratory individuals tend to be larger and more fecund than nonmigratory counterparts (Jonsson and Jonsson 1993; Northcote 1997), the loss of migratory life-history forms from a population may reduce the capacity of the population to adapt to changing environmental conditions and ultimately decrease the probability of persistence (Leary et al. 1990; Nelson et al. 2002; Peterson et al. 2008).

In contrast to large migratory forms in large stream and lake networks, many salmonid populations evolved in isolated systems, and persistence depended upon adaptation to the limited available habitats and the adoption of fluvial (stream resident) life-history strategies (Rieman et al. 2003). For example, expansion and contraction of glaciers during the Pleistocene Epoch led to local extirpations followed by recolonization of salmonids whose ancestors survived in refugia (Wood 1995; Smith et al. 2002), and these factors contributed to the diversity of life-history forms observed among cutthroat trout (Gresswell et al. 1994). Recolonization may have been more rapid for those populations that were able to reexpress migratory life-history forms. Thus, each life-history form reflects environmental conditions (Warren and Liss 1980) and migratory forms are expressed when feeding habitats are available (Gross 1987).

Salmonids exhibit a strong tendency to colonize vacant habitat. For example, recolonization of stream sections exposed by glacial retreat during recent climate warming has been described by several anadromous species; sockeye salmon *Oncorhynchus nerka*, pink salmon *Oncorhynchus gorbuscha*, chum salmon

Oncorhynchus keta, coho salmon *Oncorhynchus kisutch*, and Dolly Varden *Salvelinus malma malma* (Burger et al. 1997; Woody et al. 2000; Milner et al. 2000).

Potamodromous salmonids have also demonstrated the capacity to colonize (often termed “invade”) new environments. Examples include the expansion of translocated populations beyond original translocation sites (Kinneson et al. 2001; Quin et al. 2001; Halverson 2010), recolonization after experimental removals (Peterson et al. 2004; Shepard 2010), and observations of colonizing events (Kennedy 1950).

In the last century, extirpations of salmonid populations from portions of watersheds have been attributed to “natural” (geologic and climactic) disturbances such as volcanism (Hawkins and Sedell 1990), wildfire (McMahon and deCalesta 1990; Gresswell 1999), and debris flows (Roghair and Dolloff 2005). However, the majority of salmonid extirpations are directly attributable to anthropogenic disturbance such as mining contamination (some examples include IEC 2006; Finger et al. 2007; CDPHE and USEPA 2009), nutrient pollution (Hansen and Mensberg 1996; Meade 2004; Perrier et al. 2010), and barrier construction blocking access to spawning sites (Nehlsen et al. 1991; Parrish et al. 1994). Where dams have been removed or circumvented, anadromous salmonids have demonstrated the capacity to recolonize former habitat (Brenkman et al. 2008; Kiffney et al. 2009). However, in cases of contamination, extirpations may extend to the watershed-scale (Hansen and Mensberg 1996; Luoma et al. 2005). In some respects, contamination mimics glacial advance where affected portions of a watershed are defaunated for extended periods, but other portions of the watershed are unaffected. Remediation (removal of contaminants, Finger et al. 2007) may provide an opportunity

for salmonid populations surviving in these refugia to recolonize previously contaminated portions of a watershed, if dispersal corridors remain open and contaminant levels are sufficiently reduced. Under these circumstances, migratory life-history forms may be particularly important because remediated portions of the watershed may remain unsuitable for early life stages because of metal toxicity (Sorenson 1991), but foraging habitat and space for adult fish may be abundant (Northcote 1997).

The remediation of Silver Bow Creek in southwest Montana presents the opportunity to observe recolonizing movement patterns of salmonids in a stream network from which they were extirpated for more than a century. The entire main stem of Silver Bow Creek was uninhabitable to fish during the 20th century because of metal contamination from copper mining in the headwaters (Luoma et al. 2005; MTNRDP et al. 2005). However, indigenous populations of westslope cutthroat trout, longnose suckers *Catostomus catostomus*, and slimy sculpin *Cottus cognatus* persisted in tributary refugia (MTFWP 2009). In addition to the indigenous species, translocations of nonindigenous brook trout *Salvelinus fontinalis* and central mudminnow *Umbra limi* successfully established populations in the tributaries (MTFWP 2009). Superfund remediation began in the Silver Bow Creek main stem in 1998, and metal concentrations have been substantially reduced (Chapter 1; USEPA 2011). However, water quality problems remain. The stream is extremely eutrophic, and during the summer, a large portion of the stream becomes hypoxic at night because of ammonia pollution from treated municipal sewage (Plumb 2009; Gammons et al. 2011). Despite these water quality problems, fish (longnose suckers) were observed in the stream for the first time in 2002, four years after

remediation began. Salmonids (brook trout and westslope cutthroat trout) were observed in the stream for the first time in 2004 (MTFWP 2009).

Little fisheries data was collected in Silver Bow Creek prior to remediation and no previous efforts were made to describe the movements of salmonids in response to remediation. The purpose of this study was to describe recolonizing movements of salmonids from undisturbed tributary populations into the remediated main stem, to describe distribution patterns within the main stem, and to evaluate how movements of individuals within the main stem contributed to seasonal changes to distribution patterns. At the time this study was initiated (2009), salmonids had only recently been sampled in the remediated main stem (MTFWP 2009). Throughout the main stem and lower portions of connected tributaries, salmonids were implanted with passive integrated transponders (PIT-tags) and their movements were monitored over a 3-year period. These data were used to assess the relative contribution of each tributary salmonid population to the recolonization of the remediated main stem. Additionally, these data were used to describe changes in seasonal relative abundance in specific portions of the main stem and also to describe important time periods when recolonizing movements occurred.

Methods

Study Area

The Silver Bow Creek watershed is located in southwest Montana and is a tributary to the Clark Fork River within the Columbia River basin. The watershed is

bounded to the south and east by the continental divide, with watershed elevations exceeding 2,438 m (Plumb 2009). Stream elevation in the study area ranged from 1550-1670 m. Mean annual precipitation is 32.5 cm (Plumb 2009). Mean July high temperatures are 27°C and mean low January temperatures are -15°C. Typical base discharge in Silver Bow Creek during the study period was 0.6 m³/s (USGS stream gage 12323250). Maximum annual discharge typically occur during snowmelt periods in the late spring. The maximum measured discharges in the main stem during each year of this study were 3.2, 5.5, and 7.1 m³/s on 6 August 2009, 16 June 2010, and 9 June 2011 respectively at USGS stream gage 12323250. The study area was subdivided into six main stem and two tributary segments (segment length range: 2.5-8.2 km; Figure 1). Segments were identified by changes in valley form (i.e., between confined and unconfined valleys, Frissell et al. 1986), tributary junctions (with ≥15% of main stem discharge), and at the approximate boundaries of geochemical stream zones. Two segments were further subdivided into separate reaches based on land use changes and more specific geochemical processes. See Chapter 2 for detailed description of physical habitat characteristics within each main stem segment.

Water chemistry, physical habitat, and discharge in the Butte segment differed substantially from the main stem due to the location of this segment above the wastewater effluent. The Butte segment was subdivided into two reaches; the Blacktail Creek reach encompassing the uppermost 4 km and the Lower Area One-Superfund Operable Unit (LAO) reach encompassing the lower 3 km of the Butte segment (Figure 1). Base discharge in the Butte segment was approximately 0.3 m³/s during the study period

(USGS stream gage 12323240). Remediation activities in the Blacktail Creek reach targeted specific tailings deposits leaving the majority of the floodplain and stream channel unaltered. By contrast, the majority of the LAO reach was reconstructed during remediation (Figure 1). The Blacktail Creek and LAO reaches were separated at the Metro Storm Drain confluence (Figure 1). The Metro Storm Drain was historically a primary source of metal contamination to Silver Bow Creek (Spindler 1977). Remaining tailings deposits underlying the Metro Storm Drain have been implicated as a continuing source of metal recontamination to Silver Bow Creek (Madison and Metesh 2004; Tucci and Icopini 2012). Historically, the ancestral Silver Bow Creek stream channel roughly followed the Metro Storm Drain path into the uplands of the Butte Hill mining district, but it has now been absorbed into the Yankee Doodle Tailings Pond system, the Berkeley Pit, and the Butte storm sewer system. Within the LAO reach, Silver Bow Creek receives the discharges from the Montana Pole treatment plant ($\approx 0.03 \text{ m}^3/\text{s}$) and the Butte Treatment Lagoons ($\approx 0.03 \text{ m}^3/\text{s}$). Both of these Superfund groundwater mitigation facilities are situated within the historic stream floodplain. The Montana Pole groundwater treatment plant treats ground and surface water contaminated by wood treatment byproducts, specifically pentachlorophenols (PCPs), polynuclear aromatic hydrocarbons (PAHs), dioxins, and furans (USEPA 1993). The Butte Treatment Lagoons treat metal contaminated groundwater from the Butte area with lime and a series of settling ponds (USEPA 2006).

The Silver Bow Creek main stem consisted of five segments downstream from the Butte wastewater treatment plant effluent. The uppermost three main stem segments

below the effluent (the Hypoxic, Recovery, and Ramsay Flats, Figure 1) were remediated as of the beginning of this study in 2009. Remediation consisted of complete floodplain soil removal and channel reconstruction (B. Bucher, CDM Smith, paper read during conference proceedings of the Montana Natural Resource Damage Program Conference, 2012).

Because previous research identified the Butte sewage effluent as the primary source of ammonia to Silver Bow Creek (Plumb 2009) and the cause of downstream hypoxia (Gammons et al. 2011), we subdivided the Hypoxic segment into two reaches (the Degradation and Active Decomposition reaches) based on Warren's (1971) description of longitudinal stream zones in waters receiving organic effluent. In the Degradation reach of Silver Bow Creek (Figure 1), Gammons et al. (2011) described the rapid transformation of ammonia to nitrate and the subsequent depletion of dissolved oxygen (DO) from the stream due to the biochemical oxygen demand from the decomposition of accumulated organic materials. This effect appeared to be strongest during summer nights (Plumb 2009) and resulted in a hypoxic zone approximately 3 km downstream from the effluent outfall (Plumb 2009; Gammons et al. 2011). The boundary of the Degradation and Active Decomposition reach was the approximate point where Plumb (2009) and Gammons et al. (2011) identified hypoxia (DO <5.0 mg/L, USEPA 1986). The Hypoxic and Recovery segments were divided at the approximate point, approximated by Plumb (2009), where DO levels gradually began to recover during the summer, approximately 8 km downstream from the wastewater effluent (Figure 1). The Browns Gulch tributary enters the main stem at the boundary between the Recovery and

Ramsay Flats segments. Our own data indicated that DO levels typically approached background levels similar to the Blacktail Creek reach by the downstream end of the Recovery segment (Figure 1). Prior to remediation, floodplain tailings deposits in the Ramsay Flats segment were extensive, with local deposits up to 5 m deep (G. Mullen, Montana Natural Resource Damage Program, personal communication). The downstream end of the Ramsay Flats segment marked the approximate boundary of the upper Durant Canyon segment.

Downstream from the Ramsay Flats segment, Silver Bow Creek passed through Durant Canyon which is separated into upper and lower segments by the confluence with German Gulch tributary (Figure 1). Remediation was in progress in portions of the upper Durant Canyon segment in both 2010 and 2011. The main stem study area was bounded downstream by a small temporary dam built during the initial remediation phase to capture sediments entrained by remediation activities upstream.

Downstream from the wastewater effluent, two tributaries entered Silver Bow Creek. German Gulch, a high gradient tributary enters Silver Bow Creek at stream km 31.3. Fish were sampled in portions of the lower 4.2 km of German Gulch (Figure 1). Browns Gulch, entering Silver Bow Creek at stream km 21.6, is the largest tributary to Silver Bow Creek by watershed area (Figure 1). During midsummer, Browns Gulch occasionally is dewatered to the extent that no above ground flow reaches Silver Bow Creek at the confluence (anonymous landowner, personal communication), but during the 3 years of this study, Browns Gulch maintained flows at the confluence.

Water Quality

Longitudinal and seasonal variation in water quality was assessed with monthly synoptic water sampling. Water samples were collected approximately every month from June 2010 to December 2011. Minimum daily DO concentrations were estimated by sampling during the predawn hours (0200 to 0630). There were 25 water quality monitoring sites; 20 main stem sites and 5 tributary sites including one in the wastewater effluent outfall. For description of synoptic site locations, sample collection, and analysis methods see Chapter 2.

Fish Sampling and PIT-tagging

To describe the longitudinal relative abundance patterns of each salmonid species in detail, spatially-continuous electrofishing surveys in the Silver Bow Creek main stem segments were conducted in 2010 and 2011. During these electrofishing surveys a large proportion of the individuals present in the main stem were PIT-tagged. To compare the relative contribution of each tributary population to the recolonization of Silver Bow Creek, electrofishing and PIT-tagging operations were conducted in the tributaries in 2009, 2010, and 2011. Each segment was subdivided into 100-m sample sections. Main stem sample sections were referenced by stream distance (km) downstream from the upstream study area boundary. Fish sampling occurred in each sample section of the study area, and many sections were sampled annually over a 3-year period beginning in July 2009. Fish were collected and PIT-tagged in 33.8 km of the Silver Bow Creek main stem and in the lower portions of each tributary; German Gulch (4.2 km) and Browns Gulch (5.8 km; Figure 1). Synoptic water quality measurements were taken in a

subsample of the main stem sections each month from July 2010 until December 2011. Electrofishing occurred during July and August of each year with backpack electrofishing gear. In 2009, sampling effort (16 sampling days, between 6 July and 25 August 2009, 18.3 stream km sampled) was focused primarily on tributaries and the Butte segment. In 2010 (16 sampling days, between 7 July and 12 August 2010, 27.6 stream km sampled) and 2011 (17 sampling days, between 6 July and 4 August 2011, 38.3 stream km sampled), sampling effort was shifted to the main stem segments with a spatially-continuous approach in to investigate longitudinal relative abundance patterns in the main stem.

Single-pass electrofishing surveys (Bateman et al. 2005) were conducted upstream from the start of each 100-m sample section using a pulsed-DC backpack electrofishing unit (Smith-Root, Inc., Model LR-24, Vancouver, WA). Field crew size varied from four to six individuals with a minimum of two persons netting fish. Stream conductivity ranged from 0.170-0.558 $\mu\text{s}/\text{cm}$. Electrofishing power ranged from 200 to 350 V, frequency ranged from 30 to 35 Hz, and the duty cycle was set at 24%. Mean electrofishing time per sample section was 365 s. Crews attempted to capture all salmonids. Each captured salmonid was measured to the nearest millimeter total length (TL) and individuals ≥ 120 mm TL were weighed (± 0.1 g). In order to avoid resampling fish, individuals were released into the same sample section in which they were captured after the electrofishing crew had progressed at least two sample sections upstream. Electrofishing surveys were discontinued for the day if stream temperature exceeded 17°C (CBFWA-PTSC 2009).

Half-duplex 23-mm PIT-tags (Texas Instruments, Plano, TX) were implanted in each salmonid ≥ 120 mm. Surgical procedures were similar to those described by Bateman and Gresswell (2006). Each PIT-tagged salmonid was additionally marked by removing the adipose fin to identify previously tagged individuals that expelled or “shed” their PIT-tag (Bateman et al. 2009).

Stationary Antenna Sites

To monitor the direction and timing of PIT-tagged fish movements, seven stationary antenna sites were established (Figure 1). Antenna sites consisted of a PIT-tag detector (Oregon RFID, Multi-Antenna HDX reader), two stream-width antennas, and a tuning capacitor for each antenna (Oregon RFID, Standard Remote Tuner Board). An 85watt solar panel and 12 V deep cycle battery was used to power each site. Main stem antenna sites were positioned at segment boundaries and immediately above the German Gulch and Browns Gulch confluences (Figure 1). Each antenna site was identified by the main stem sample section that the site was located in (e.g., MS-31.3 = “main stem 31.3”) or the tributary in which the antenna site is located (GG = “German Gulch”, BG = “Browns Gulch”; Figure 1). At each antenna site, a pair of stream-width antennas was positioned approximately 5 m apart in a “swim-through position” (Castro-Santos et al. 1996; Zydlewski et al. 2001; Novick 2005). By pairing antennas, the direction of movement was deduced according to the sequence of antenna detections (Castro-Santos et al. 1996; Zydlewski et al. 2001). Antennas were constructed from fine-threaded copper speaker wire (8-14 AWG). Detection range of each antenna was adjusted or “tuned” with the tuning capacitor circuit board by adjusting the inductance of each

antenna (Oregon RFID 2009). Antennas were considered “in tune” when a PIT-tag, oriented perpendicular to the antenna field, was detected throughout the antenna field at a distance of 0.3 m. Detection records were uploaded every 2 weeks, and antenna tuning was adjusted as necessary. Antenna sites were located in relatively high-velocity channel units (e.g., riffles, Bisson et al. 2006) where fish would be less likely to rest and the frequency of redundant detections was reduced. Redundant detections were defined as detections of the same individual, on the same day, at the same antenna, <5 minutes from the last detection. Redundant detection records were not included in any analyses.

Antenna sites were installed in spring and summer of each year and were maintained until the solar panels could no longer provide sufficient operating power. Antennas were generally operational from July through October and no sites were used during the winter (January-March). Although some antenna sites were installed during the spring, prior to snowmelt runoff, variable flows and weather conditions made operation generally sporadic at all sites until July of each year. The GG and MS-31.3 antenna sites were not operational in 2010, and the MS-14.8 and MS-24.6 sites were not operational in 2009.

Portable Antenna Surveys

Seasonal portable antenna surveys of the main stem segments provided a means to relocate previously PIT-tagged fish and evaluate how the relative abundance patterns of each species varied seasonally in main stem segments. Portable PIT-tag antenna units, similar to those described by Hill et al. (2006), were used to relocate PIT-tagged fish in the Silver Bow Creek main stem segments in 2010 and 2011. Portable antenna units were

adapted from stationary antenna readers and tuners, but powered by 16 V rather than 12 V batteries to increase read range (maximum read range for 23-mm half-duplex PIT tags ≈ 0.8 m). Portable antenna surveys (Cucherousset et al. 2005) progressed downstream with one or two surveyors. Surveys were conducted seasonally between August 2010 and December 2011 in main stem segments where electrofishing occurred during the previous summer. Tributaries were not surveyed except for the lower 1.5 km of German Gulch in September 2011. The May 2011 survey was conducted near the peak of the spring runoff period and, some parts of the Blacktail Creek reach of the Butte segment, and the Ramsay Flats segment were inaccessible because of high water. Portable antenna surveys were generally conducted on consecutive days until completion, with surveyors progressing 2-8 km each day, depending on the frequency of detections.

To compare the longitudinal salmonid abundance patterns from electrofishing surveys with subsequent portable antenna surveys with differing numbers of surveyor, portable antenna survey counts in each sample section were expanded by the estimated relocation efficiency according to the number of surveyors conducting each survey. Relocation efficiency was estimated for one- and two-person portable antenna surveys in three segments; the Hypoxic, Ramsay Flats, and upper Durant Canyon segments. Immediately following the 2011 electrofishing survey of each of these segments, a one-person and then a two-person portable antenna survey was conducted in the same segment. In each segment, the number of PIT-tagged fish present during the portable antenna surveys was known from the number of PIT-tagged fish released into each segment during the preceding electrofishing survey, subtracting the number of fish that

left the segment past the stationary antenna sites bracketing the segment. This estimate of PIT-tagged fish in a segment was used to determine the relocation efficiency of the survey. In addition to PIT-tagged fish present in each segment, PIT-tags ($n = 25$) were also hidden in the substrate of each segment by a third individual who was not involved in the portable antenna surveys. Surveyors were blind to the number of PIT-tags deposited in each segment and to the locations of the PIT-tags, but were aware that PIT-tags had been placed in the segment. Efficiency surveys were not conducted in the Recovery segment because of the scarcity of PIT-tagged fish encountered during the electrofishing survey. No one-surveyor efficiency surveys were conducted in the upper Durant Canyon segment because no one-surveyor surveys occurred in that segment. Efficiency surveys were also not conducted in the Butte and lower Durant Canyon segments because these segments were not bracketed by stationary antenna sites (Figure 1) and movements of individuals out of these segments prior to the portable antenna survey could not be accounted for.

Statistical Analysis

To compare salmonid relative abundance between summer (July 2010, August 2010, and July 2011) and winter (December 2010, April 2011, and December 2011) surveys, a repeated measures Hotelling's T^2 statistic was used. Segments were grouped according to summer time DO levels to test if segments experiencing summer hypoxia (the Hypoxic and Recovery segments) had disproportionately large seasonal differences in mean westslope cutthroat trout relative abundance. Species relative abundance

differences within specific segments, during specific time steps, were compared with paired *t*-tests.

The activity levels of PIT-tagged fish, approximated by the mean number of detections per individual, were compared by species at each antenna site by comparisons of proportions. Antenna sites with higher mean detections per individual were presumed to be located in habitats more frequently utilized by the individuals detected. All of the antenna sites were operational between July and November 2011, and therefore comparisons were restricted to that time period.

The proportions of individual brook trout PIT-tagged in each tributary and subsequently detected by stationary antenna sites in the main stem was compared by tributary. Because the Butte segment was unaffected by the effluent, and because it was separated from downstream main stem segments by the MS-6.6 antenna site (Figure 1), we classified the Butte segment as a tributary for this analysis. Any brook trout that was initially PIT-tagged in Browns Gulch, German Gulch, or in the Butte segment that was subsequently detected at either a main stem or tributary antenna site, was considered to have moved into the main stem. Detections at tributary antenna sites were classified as main stem detections due to the proximity of each tributary antenna site to the main stem (i.e., within 10 m). This analysis was restricted to brook trout because relatively few westslope cutthroat trout were PIT-tagged in Browns Gulch ($n = 7$) and the Butte segment ($n = 23$) relative to German Gulch ($n = 367$). The analysis was also restricted to months in which all tributary antenna sites (BG, GG, MS-6.6; Figure 1) were simultaneously operational (August-November 2009 and July-November 2011).

Net gains and losses of PIT-tagged individuals from specific segments, during particular months, were assessed by the directional movement patterns at stationary antenna sites. For example, if the number of individual PIT-tagged fish moving upstream past an antenna site during a period of time exceeded the number moving downstream, or vice versa, it was inferred that a net increase of PIT-tagged individuals occurred in the segment upstream from the antenna site. The seasonality of these directional movement patterns helped to assess movement patterns.

Electrofishing surveys were conducted at approximately the same time each year (July-August) allowing for comparisons of length-frequency distributions between sample cohorts. To compare length-frequency distributions, the proportion of each sample cohort ≥ 200 mm TL was determined, and this proportion was used to test for equality among cohorts. Westslope cutthroat trout PIT-tagged in German Gulch, but never detected outside of German Gulch (hereafter referred to as “residents”), were compared to those PIT-tagged in German Gulch and later detected in the main stem (hereafter referred to as “migrants”). Similarly, westslope cutthroat trout sampled in the Silver Bow Creek main stem were compared to German Gulch residents.

Results

In total, 2,633 salmonids were PIT-tagged (787 westslope cutthroat trout and 1,846 brook trout) in the Silver Bow Creek stream network over a 3-year period. Of the 787 westslope cutthroat trout PIT-tagged, about half (397) were PIT-tagged in tributaries (including the Butte segment as a tributary), and 64 (16%) of those were relocated in

main stem segments during the study (Table 1). Of the 1,846 brook trout PIT-tagged, 89% (1,648) were PIT-tagged in tributaries, and 111 (7%) of those were relocated in main stem segments during the study (Table 1). Electrofishing surveys in 2010 and 2011 recaptured 15% (63 of 421) of the westslope cutthroat trout and 9% (123 of 1333) of the brook trout previously PIT-tagged. Of those PIT-tagged in the main stem segments, portable antenna surveys relocated 64% (265 of 413) of the westslope cutthroat trout and 52% (702 of 1340) of the brook trout at least once. Stationary antenna sites detected 32% (248 of 787) of all PIT-tagged westslope cutthroat trout and 14% (254 of 1,846) of all PIT-tagged brook trout.

Water Quality

During the summer months (July, August, and September) a stream zone with lower levels of DO developed from approximately stream km 10.0-20.0 (Figure 2 and 3). Hypoxia was most severe in the Active Decomposition reach of the Hypoxic segment during the July and August samples (Figures 2 and 3). The lowest DO levels were typically encountered between stream km 11.0-13.0, in early August (lowest DO levels during each year were 1.41 mg/L on 3 August 2010 and 2.28 mg/L on 3 August 2011). During the hypoxic period (July, August, and September), DO levels typically returned to background levels similar to those in the Butte segment, by the Ramsay Flats segment (Figures 2 and 3).

Ammonia concentrations in the Butte wastewater treatment plant effluent discharging into Silver Bow Creek were high ($\text{NH}_3\text{-N}$ range = 4.00-21.71 mg/L) and 12 of 15 samples exceeded 10.00 mg/L. Synoptic ammonia samples in the Silver Bow

Creek main stem showed the effects of this effluent with all ammonia levels rising from approximately 0.1-0.2 mg/L at sample sites above the effluent, to 1.4-2.8 mg/L at sample sites immediately below the effluent (Figure 4). During all synoptic samples, ammonia levels decreased with downstream distance from the effluent, however, during the colder months the rate at which ammonia levels decreased was lower and levels remained high (≥ 1.18 mg/L) at all downstream sample sites during the winter (December and April) samples (Figure 4).

Total recoverable copper concentrations in the Butte wastewater treatment plant effluent ranged from 0.013-0.037 mg/L, and levels increased in the spring and early summer. All synoptic samples suggested that copper levels were lowest in the Butte segment and steadily increased downstream, beginning at the wastewater effluent point source (Figure 4). Copper levels were the highest during spring runoff (15 May 2011) and lowest in early (28 September 2011) and late fall (2 December 2011; Figure 4). Longitudinally, all synoptic samples revealed a spike in copper levels near stream km 13.0 (Figure 4).

Portable Antenna Relocation Efficiency

Portable antenna surveys were significantly more efficient with two-surveyors than with one-surveyor, but there was no evidence that efficiency differed among segments when the number of surveyors was the same. The proportion of PIT-tags relocated with two-surveyors (49% or 117 of 241) was higher than the corresponding proportion for one-surveyor (30% or 32 of 107). There was strong evidence that the difference in relocation efficiency between one- and two-surveyors was statistically

significant (95% C.I. from 7 to 30%, Chi-square statistic = 9.7693, d.f. = 1, p -value = 0.00180). However, there was no evidence that relocation efficiency differed among segments for the same number of surveyors.

Seasonal Abundance Patterns

In both the July 2010 and July 2011 electrofishing survey, there was a high degree of spatial variability in salmonid abundance, with the lowest abundances occurring in the Recovery and Hypoxic segments (Figures 2 and 3). In each annual electrofishing survey, salmonid occurrence and abundance was strongly positively associated with DO concentration (Chapter 2). In July 2011, when copper concentrations were measured, occurrence and abundance was strongly negatively associated with copper concentration (Chapter 2). In addition to spatial variation within each electrofishing survey, there was evidence that the relative abundance of westslope cutthroat trout (expressed as mean fish per km) increased between the July 2010 and July 2011 electrofishing surveys in some segments, but not in others. Mean westslope cutthroat trout per km increased by 5.7 (95% C.I. from 2.0 to 9.4) in the Hypoxic segment (t -statistic from paired t -test = 3.091, d.f. = 79, one-sided p -value = 0.0014) and by 2.8 (95% C.I. from 1.4 to 4.3) in the Recovery segment (t -statistic from paired t -test = 3.973, d.f. = 67, one-sided p -value < 0.0001) between the July 2010 and July 2011 samples. Increases in mean westslope cutthroat trout per km between summer sampling periods were restricted to segments with low DO; there was no evidence that mean relative abundance of westslope cutthroat trout differed in the Butte and Ramsay Flats segments between July 2010 and July 2011. Mean brook trout per km did not differ between the July 2010 and July 2011

electrofishing surveys in any segment. No comparisons were made for the Durant Canyon segments between years because these segments were not sampled in July 2010.

Comparison of summer and winter relative abundance showed seasonal changes in relative abundance in hypoxic (Hypoxic and Recovery) segments, but no change in normoxic (Ramsay Flats and Butte) segments. Mean westslope cutthroat trout per km differed between the hypoxic and the normoxic segments (F -statistic from repeated measures Hotelling's T^2 statistic = 6.94, d.f. = 2, 237, p -value = 0.0012). Hypoxic segments had 2.93 fewer westslope cutthroat trout per km in the summer compared to the winter (95% C.I. from 0.36 to 5.50). By comparison, there was essentially no difference in mean westslope cutthroat trout per km in the normoxic segments between summer and winter (0.08 fewer westslope cutthroat trout per km during the summer in the normoxic segments, 95% C.I. from -4.78 to 4.94).

Spatial Movement Patterns

There was strong evidence that the proportion of PIT-tagged brook trout that moved into the main stem differed among tributaries (Chi-square statistic = 79.59, d.f. = 2, p -value <0.0001). The observed proportions of brook trout PIT-tagged in each tributary that were subsequently detected moving into the main stem were 9% (27 of 300) in German Gulch, 21% (43 of 207) in Brown Gulch, and 3% (35 of 1,142) in the Butte segment.

The mean number of detections per individual was significantly higher for both salmonid species at the BG site compared to the GG antenna site. For westslope cutthroat trout, there was strong evidence that the mean number of detections per

individual was greater at BG compared to GG (one-sided t -statistic from two sample t -test = 4.9159, d.f. = 56, p -value <0.0001) with an estimated 3.45 more detections per individual at the BG site (95% C.I. from 2.05 to 4.86). For brook trout, there was moderate evidence that the mean number of detections per individual was greater at BG compared to GG (one-sided t -statistic from Welch two sample t -test = 2.2624, d.f. = 10.034, p -value = 0.0235) with an estimated 11.34 more detections per individual at the BG site (95% C.I. from 0.18 to 22.49). Two brook trout at the BG antenna site were considered outliers and removed from the analysis; one was detected 185 times and the other detected 85 times. The confidence interval associated with brook trout at the BG antenna was large due to the small number of individuals detected ($n = 11$, after removing the two outliers). In contrast to these apparent differences between tributary sites, there was no evidence that the mean number of detections per individual differed for westslope cutthroat trout between main stem antenna sites and evidence of differences between main stem sites for brook trout were inconclusive (F -statistic from ANOVA = 2.252, p -value = 0.0679).

Temporal Movement Patterns

In October and November 2009, more individual westslope cutthroat trout moved out of German Gulch compared to those that moved into German Gulch resulting in a net gain of 9 individuals in the Silver Bow Creek main stem (Figure 5). A similar trend was evident in August and September of 2011, when a net of 6 westslope cutthroat trout moved out of German Gulch and into the main stem (Figure 5). Whereas most directional movements of individual westslope cutthroat trout at the German Gulch

antenna site were downstream, or out of German Gulch and into Silver Bow Creek, monthly directional movement frequencies of westslope cutthroat trout at the Browns Gulch antenna site were generally equal between upstream and downstream (Figure 5). In July 2010, 6 more westslope cutthroat trout moved upstream into Browns Gulch than moved downstream; however, 5 more individuals moved downstream the following month (Figure 5). Brook trout tended to move upstream into German Gulch during August and September of both 2009 and 2011, but at other times the directional movements of individuals roughly balanced out (Figure 6). During the month of October 2009, 44 brook trout moved downstream out of Browns Gulch and into the Silver Bow Creek main stem, and most of these individuals (41) moved during 5-day period between 17 and 21 October 2009 (Figure 6).

At all main stem antenna sites, individual westslope cutthroat trout moved more frequently upstream than downstream during the months of August, September, and October, and this trend was particularly striking in 2011 (Figure 7). In contrast, individual brook trout generally moved upstream and downstream at main stem antenna sites with approximately equal frequency (Figure 8). Of the 44 brook trout that moved out of Browns Gulch in October 2009 (Figure 9), 34 moved upstream into the Recovery segment, and 10 moved downstream into the Ramsay Flats segment (Figure 6).

Length-frequency Distributions

In the Silver Bow Creek main stem, 79% of the westslope cutthroat trout PIT-tagged ($n = 413$) were ≥ 200 mm TL, compared to just 29% of those PIT-tagged in German Gulch that were never detected in the main stem ($n = 315$). The 52 westslope

cutthroat trout PIT-tagged in German Gulch that were detected in the main stem also had a substantially higher proportion (66%) of individuals ≥ 200 mm TL compared to those that did not move into the main stem (29%).

There was strong evidence that German Gulch resident westslope cutthroat trout had a smaller proportion (estimated difference = 0.37, 95% C.I. = 0.22-0.52) of individuals ≥ 200 mm TL compared to migrants (Chi-squared = 24.6732, d.f. = 1, one-sided p -value < 0.0001 ; Figure 7). Because only seven westslope cutthroat trout were PIT-tagged in Browns Gulch, no similar comparison was made for westslope cutthroat trout in Browns Gulch (Figure 7). Compared to German Gulch residents, westslope cutthroat trout PIT-tagged in Silver Bow Creek also had a substantially higher proportion (estimated difference = 0.51, 95% C.I. = 0.45-0.58) of individuals ≥ 200 mm TL (Chi-squared = 190.2566, d.f. = 1, one-sided p -value < 0.0001 ; Figure 7).

For brook trout, there was strong evidence that the proportion of individuals ≥ 200 mm TL was greater for migrants compared to residents (estimated difference = 0.45, 95% C.I. = 0.27-0.64) in German Gulch (Chi-squared = 24.9111, d.f. = 1, one-sided p -value < 0.0001). However, there was no evidence that these proportions differed in Browns Gulch (Figure 7). There was strong evidence that the proportions of brook trout ≥ 200 mm TL differed between individuals PIT-tagged in Silver Bow Creek (proportion = 0.53), German Gulch residents (proportion = 0.29), and Browns Gulch residents (proportion = 0.44; Chi-squared = 67.2981, d.f. = 2, two-sided p -value < 0.0001 ; Figure 7).

Discussion

These data suggest that westslope cutthroat trout were in the process of recolonizing Silver Bow Creek during the study period (July 2009 to December 2011); however, summer hypoxia appeared to influence both spatial and seasonal movement patterns. Prior to the beginning of this study in 2009, only 13 westslope cutthroat trout had been sampled in Silver Bow Creek, and only 2 prior to 2007 (MTFWP 2009). In this study, sampling effort was substantially more extensive than previous sampling efforts (MTFWP 2009). In total, we captured and PIT-tagged 413 westslope cutthroat trout in the Silver Bow Creek main stem. Despite hypoxic conditions during the summer in 2010 and 2011, westslope cutthroat trout relative abundance increased in hypoxic main stem segments between 2010 and 2011. Although these data represent only a two-year comparison, these results are consistent with the findings of longer term relative abundance trends in Silver Bow Creek observed by Montana Fish Wildlife and Parks biologists (J. Lindstrom, Montana Fish Wildlife and Parks, unpublished data). We observed that the late summer and early fall appeared to be an important time period in which westslope cutthroat trout moved into main stem segments, particularly into those segments that were hypoxic during the summer. Detection records at main stem antennas indicated that PIT-tagged westslope cutthroat trout moved predominantly in the upstream direction at the downstream boundaries of the Hypoxic and Recovery segments during the late summer and early fall (August to October). By late fall when portable antenna surveys were conducted (December), relative abundance of PIT-tagged westslope

cutthroat trout was significantly greater in those segments that had been hypoxic during the summer.

Westslope cutthroat trout did not appear to have established a distinct population within the Silver Bow Creek main stem. Rather, the presence of westslope cutthroat trout in remediated main stem segments appeared to be a result of the establishment, or reexpression, of fluvial-adfluvial behavior among individuals from tributary source populations, most likely the German Gulch tributary. We did not find evidence that westslope cutthroat trout had successfully spawned in the Silver Bow Creek main stem. No juvenile (<120 mm TL) westslope cutthroat trout were sampled in main stem segments with the exception of a small number of individuals sampled near the German Gulch ($n = 13$) and Browns Gulch ($n = 3$) confluences in 2011 (Chapter 2). Lower Browns Gulch and the Butte segment also did not appear to provide westslope cutthroat trout source populations to the Silver Bow Creek main stem. Few westslope cutthroat trout were PIT-tagged in Browns Gulch ($n = 7$) and the Butte segment ($n = 23$), and no juveniles were sampled in either of those segments. Based on the relative scarcity of westslope cutthroat trout in sampled portions of the Browns Gulch and Butte segments, German Gulch appears to provide the most likely source population of westslope cutthroat trout to Silver Bow Creek in this study area.

Length-frequency distributions of westslope cutthroat trout sampled in German Gulch and in the Silver Bow Creek main stem suggested that the sampled portions of German Gulch were primarily inhabited by juveniles, small-bodied residents, or both, whereas Silver Bow Creek was primarily inhabited by large-bodied adults. A qualitative

analysis of the westslope cutthroat trout length-frequency distribution of German Gulch suggests that there may be a missing year-class. In the German Gulch length-frequency distribution, there were three progressively smaller modes at 120, 200, and 360 mm. However, there were few individuals between 240 and 300 mm, a size range that would likely represent another age class modal peak. In the Silver Bow Creek main stem, the length-frequency distribution was approximately normal and centered on 260 mm, corresponding with the size group that appears to be missing in German Gulch. These findings suggest that during this study, a relatively large proportion of the westslope cutthroat trout inhabiting the German Gulch study area may have migrated into Silver Bow Creek prior to reaching larger sizes (e.g., 240 mm TL or more).

During this study, Silver Bow Creek was highly eutrophic due to the effects of ammonia-rich effluent from the Butte wastewater treatment plant (Plumb 2009). The ammonia from the wastewater effluent has been implicated as the cause of the hypoxia occurring during the summer (Gammons et al. 2011). After controlling for variation in DO and copper concentration, our data indicated that salmonid abundance increased in relation to ammonia concentration (Chapter 2). The eutrophic conditions in Silver Bow Creek may provide a rich feeding environment for migratory adult westslope cutthroat trout, and similar findings have been observed for rainbow trout *Oncorhynchus mykiss* (Askey et al. 2011). Synoptic sampling indicated that during winter, DO and ammonia levels increased, but copper levels decreased. All of these conditions were related to increased salmonid abundance during the summer (Chapter 2). Extrapolation of relationships identified spatially between water quality factors and salmonid abundance

during the summer to seasonal relationships between relative abundance and those same water quality factors is speculative. Even so, the increases in the relative abundance of salmonids observed during the winter in previously hypoxic segments and the spatial and seasonal movement patterns that corresponded with these local increases in abundance, suggests that seasonal variation in water quality is largely responsible for the seasonal variation in westslope cutthroat trout abundance.

Brook trout do not appear to be recolonizing remediated main stem segments as rapidly as westslope cutthroat trout. Brook trout were rare in the remediated main stem segments downstream from the Butte wastewater effluent (Ramsay Flats, Recovery, and Hypoxic segments) compared to westslope cutthroat trout. In total, only 36 brook trout were captured and PIT-tagged in those segments compared to 194 westslope cutthroat trout. Of the brook trout PIT-tagged in each tributary, there were large differences in the proportions of those individuals that were later detected in the main stem. Brook trout PIT-tagged in Browns Gulch (which enters Silver Bow Creek between the Ramsay Flats and Recovery segments where brook trout were rarely sampled), were detected moving into the main stem in the highest proportion (21% or 43 of 207). By contrast, only 3% (34 of 1,142) of the brook trout PIT-tagged in the Butte segment were later detected in the main stem below the effluent. Although the Butte segment would seem to represent a strong potential source population for brook trout to recolonize remediated Silver Bow Creek segments, movements of individuals from the Butte segment into remediated main stem segments downstream were rare. Brook trout apparently tolerate the habitat conditions of the post-remediation stream channel because they have recolonized and are

now the most abundant salmonid species in the remediated LAO reach of the Butte segment. However, high levels of ammonia and copper in the main stem below the effluent may have deterred brook trout movement downstream.

The number of westslope cutthroat trout PIT-tagged in German Gulch ($n = 367$) greatly exceeded the number PIT-tagged in Browns Gulch ($n = 7$), however, overall the number of individuals detected at each tributary antenna site was nearly equal (Browns Gulch = 54, German Gulch = 55), and the number of detections per individual was higher at the Browns Gulch antenna. This finding suggests that, although lower Browns Gulch does not appear to provide rearing habitat for juveniles, it may provide refuge habitat for adult westslope cutthroat trout inhabiting the main stem.

Anthropogenic activities over the past few centuries have altered the biotic (e.g., species assemblages) and abiotic (e.g., flow and temperature regimes, water quality) habitat template (*sensu* Southwood 1977) of lotic fishes (Hobbs et al. 2009). Environmental restoration and remediation of many of these severely degraded and contaminated systems, such as Silver Bow Creek, will be necessary in many instances to recover lost populations in many of these streams. In these instances, understanding the behavioral mechanisms (e.g., movement patterns) of fishes in response to environmental restoration is critical to increase the likelihood of success and improve planning for continuing and future restoration and remediation activities. Published descriptions of natural (i.e., not facilitated by translocations) fish recolonization in streams are rare, particularly when populations have been extirpated from a stream or river for extended periods (e.g., a century or more in Silver Bow Creek). The capacity of salmonids to

recolonize streams or stream reaches after relatively short periods of time, such as a few years, is comparatively well documented (Peterson et al. 2004; Shephard 2010).

However, descriptions of stream recolonization by salmonids after extended time periods have rarely been directly observed and described. Rather, evidence of recolonization has generally been limited to genetic analyses (McCusker et al. 2000; Milner et al. 2000; Woody et al. 2000) which may not provide as much insight into the behavioral mechanisms involved.

Several studies have documented declines in the prevalence of migratory life-history forms in salmonid populations (Elliot 1987; Northcote 1992; Nelson et al. 2002), and declines may occur in as little as 20-30 years (Morita et al. 2000). However, examples of salmonid populations exhibit shifts toward the reexpression of migratory (e.g., fluvial-adfluvial) life-history patterns, as has apparently occurred in the German Gulch westslope cutthroat trout population, are rare. Recent declines in the prevalence of migratory forms have been attributed to the relative scarcity of high quality feeding areas or the lack of access to those feeding areas (Nelson et al. 2002). Prior to remediation it is unknown what proportion of westslope cutthroat trout in the German Gulch population exhibited migratory behaviors; however it is highly unlikely that individuals that migrated out of German Gulch survived given the toxicity levels in Silver Bow Creek prior to remediation (Spindler 1977; Luoma et al. 2005). The rapid rate in which westslope cutthroat trout have begun to move into Silver Bow Creek, and the predominantly large size of those individuals, suggests that during this study, Silver Bow Creek was utilized by migratory adults as feeding habitat (Northcote 1997).

Westslope cutthroat trout are a species of special concern in Montana (MTFWP 2007) and are indigenous to the Silver Bow Creek watershed (Behnke 1992). As of 2008, the population inhabiting German Gulch was estimated to be 0.3% hybridized with rainbow trout *Oncorhynchus mykiss* (J. Lindstrom and R. Leary, Montana Fish Wildlife and Parks, unpublished data). The most likely source of rainbow trout to Silver Bow Creek are the Warm Springs Tailings Ponds where rainbow trout have been stocked, located approximately 14 km downstream from the German Gulch-Silver Bow Creek confluence. Historically, the German Gulch westslope cutthroat trout population may have been isolated from hybridization by the toxicity of Silver Bow Creek. However, during this study 6 rainbow trout were sampled in Silver Bow Creek. As water quality in Silver Bow Creek has improved, it is likely that rainbow trout inhabiting the Warm Springs Ponds will continue to colonize Silver Bow Creek. To deter hybridization between rainbow trout and westslope cutthroat trout, fisheries managers may need to consider strategies to isolate the Silver Bow Creek-German Gulch westslope cutthroat trout population from further hybridization.

Our primary research objective was to determine if recolonization of the Superfund remediated Silver Bow Creek main stem had occurred and to describe the related movement patterns. Despite 100-150 years of contamination, individuals in the German Gulch westslope cutthroat trout population appear to have rapidly reexpressed a migratory life-history behavior, and this behavior seems to have played a primary role in facilitating the recolonization of remediated segments. In contrast, brook trout have appeared reluctant recolonize remediated main stem segments. Brook trout may have

been avoiding specific water quality conditions in the main stem, specifically, ammonia pollution, hypoxia, metal toxicity, or all of those. Planned wastewater treatment plant upgrades in Butte are expected to reduce the ammonia loads in the main stem, and in turn alleviate hypoxia during the summer. Taken in context, the wastewater treatment plant upgrades will likely represent one more powerful alteration to this ecosystem. Since 1998, conditions in Silver Bow Creek have changed rapidly with the remediation and removal of all previous floodplain and stream channel materials, the complete reconstruction of a new stream channel and floodplain, and the substantial reduction of metal contaminant levels (Chapter 2). Since remediation began, a shift in the macroinvertebrate assemblage of Silver Bow Creek has been noted, from predominantly metal tolerant to predominantly nutrient tolerant taxa (Sullivan 2010). Reducing ammonia loads in the wastewater effluent will likely substantially alter the ecology of the stream yet again, and at that time further investigation into the response of salmonids to remediation as well as the wastewater plant upgrades may be warranted. Data collected during this study will provide a valuable point of comparison to evaluate the success of the wastewater plant upgrades. In addition, the response of salmonids to remediation will likely need to be reevaluated in the context of lowered ammonia levels and increased summer DO levels. At that time, the relationships between salmonid abundance and metal concentrations as well as physical habitat characteristics may be quite different than those observed during this study. Prior to remediation, Silver Bow Creek was a profoundly metal contaminated ecosystem from which fish are believed to have been extirpated for at least a century (Luoma et al. 2005). The presence of westslope cutthroat

trout in Silver Bow Creek during this study highlights a remarkable success for environmental remediation and underscores the resilience of indigenous salmonids, as well as the capacity of this species to adapt to changing environmental conditions.

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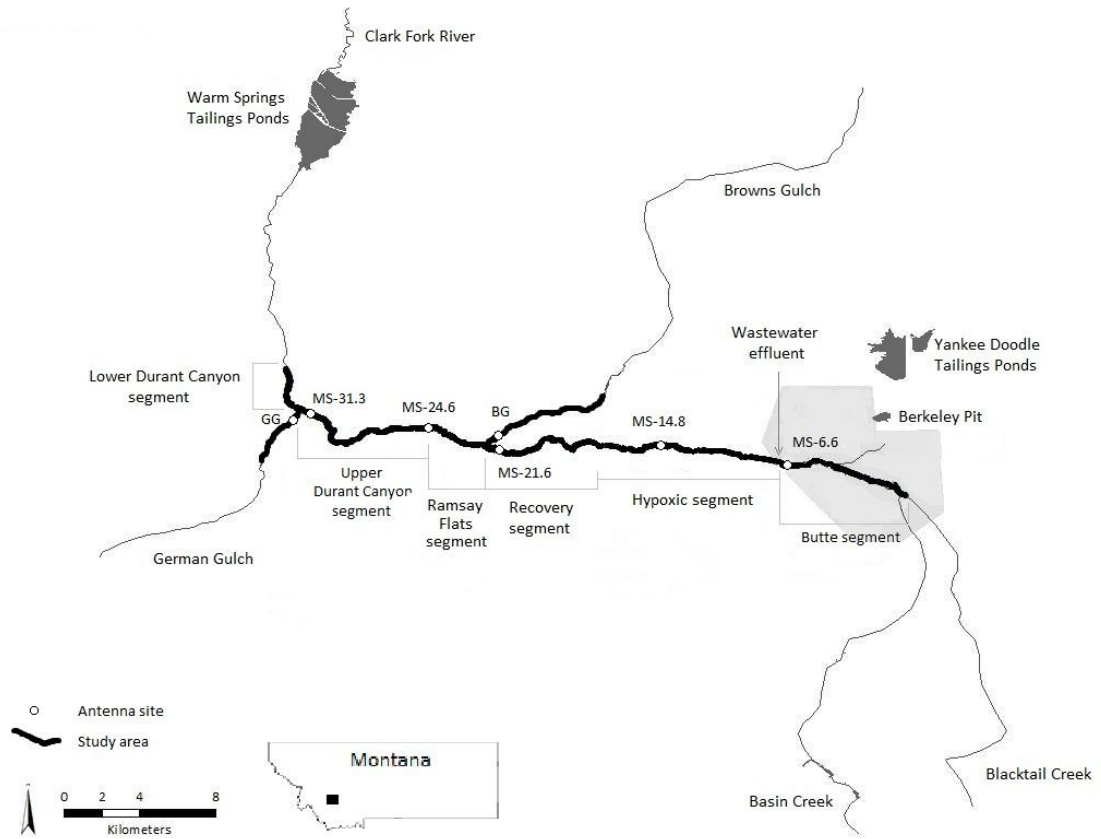


Figure 1.-Silver Bow Creek study area in southwestern Montana.

Table 1.-Proportion of individuals PIT-tagged that were subsequently relocated in the Silver Bow Creek main stem, between July 2009 and December 2011. Relocation methods included electrofishing recaptures, portable antenna relocations, and stationary antenna site detections. The Butte segment was treated as a tributary in this table. Main stem segments are those downstream from the Butte segment. See Figure 1 for segment locations.

Species and original PIT-tagging segment	Number PIT-tagged	Number relocated in main stem segments	Percent relocated in main stem segments
Westslope cutthroat trout			
German Gulch	367	52	14
Browns Gulch	7	4	57
Butte	23	8	35
Main stem (below Butte)	390	314	81
Brook trout			
German Gulch	300	31	10
Browns Gulch	206	50	24
Butte	1,142	30	3
Main stem (below Butte)	198	137	69

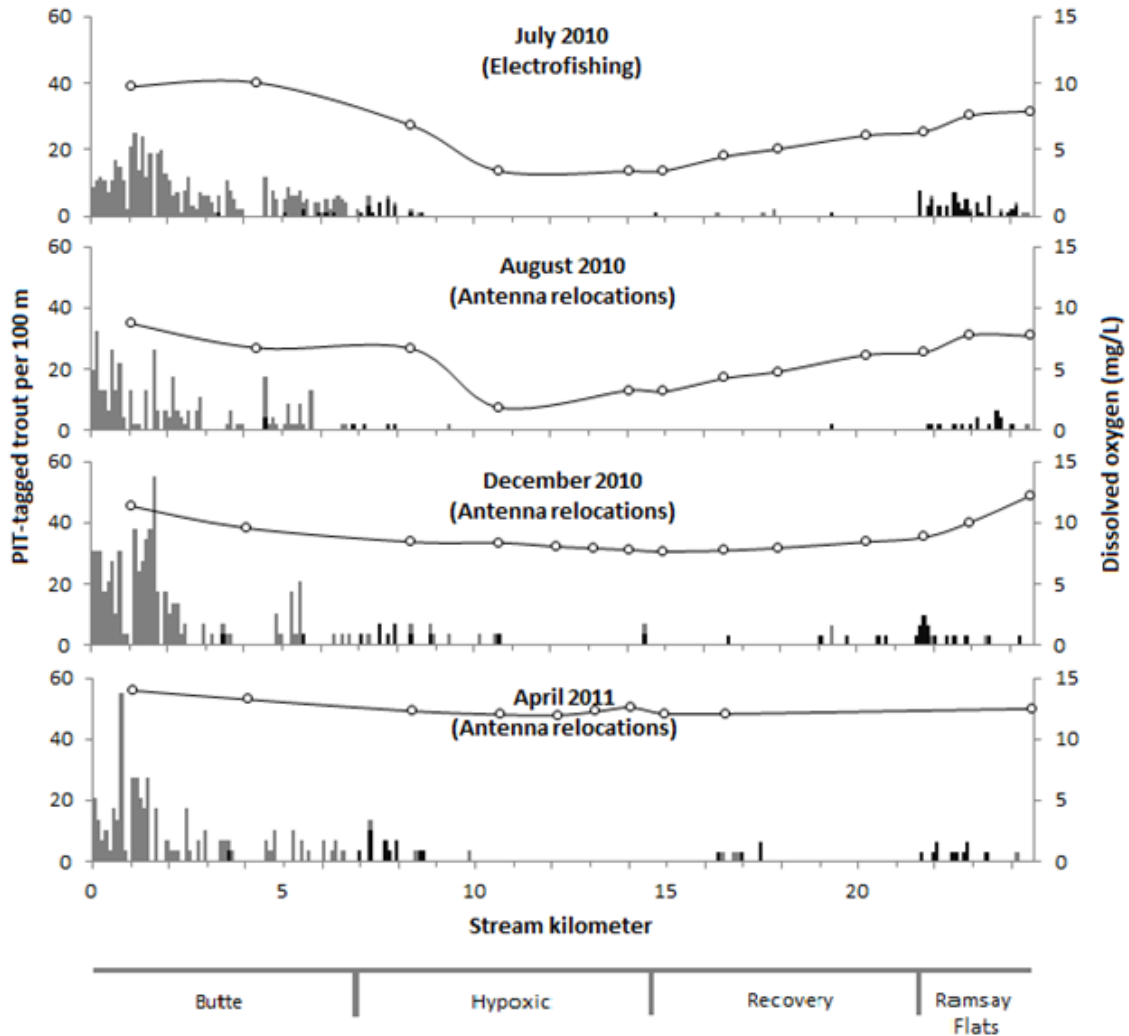


Figure 2.-Distribution of PIT-tagged trout during July 2010 electrofishing and three subsequent seasonal portable antenna surveys in Silver Bow Creek and synoptic nighttime DO concentrations during each survey. Westslope cutthroat trout *Oncorhynchus clarkii lewisi* are represented by black bars and brook trout *Salvelinus fontinalis* are represented by grey bars (stacked above black bars). Trout abundances were expanded in each portable antenna survey by the estimated efficiency of each segment. All synoptic measurements (white circles) occurred during predawn hours (0200-0630). Study area segments are identified in the lower bar. Stream km were measured downstream from the upstream end of the study area.

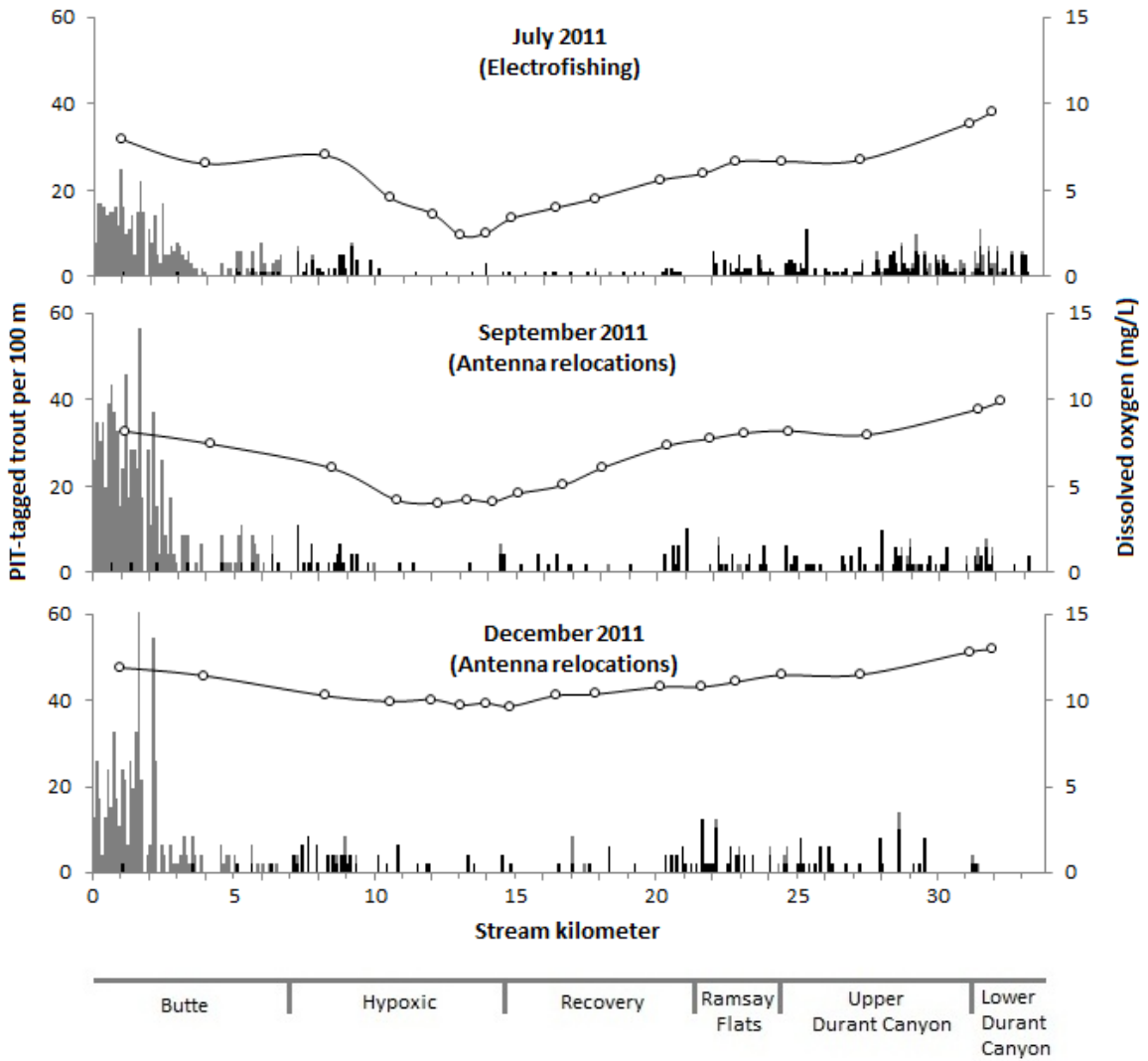


Figure 3.-Distribution of PIT-tagged trout during July 2011 electrofishing and subsequent seasonal portable antenna surveys in Silver Bow Creek with synoptic nighttime DO concentrations during each survey. Westslope cutthroat trout *Oncorhynchus clarkii lewisi* are represented by black bars and brook trout *Salvelinus fontinalis* are represented by grey bars (stacked above black bars). Trout abundances were expanded in each portable antenna survey by the estimated efficiency of each segment. All synoptic measurements (white circles) occurred during predawn hours (0200-0630). Study area segments are identified in the lower bar. Stream km were measured downstream from the upstream end of the study area.

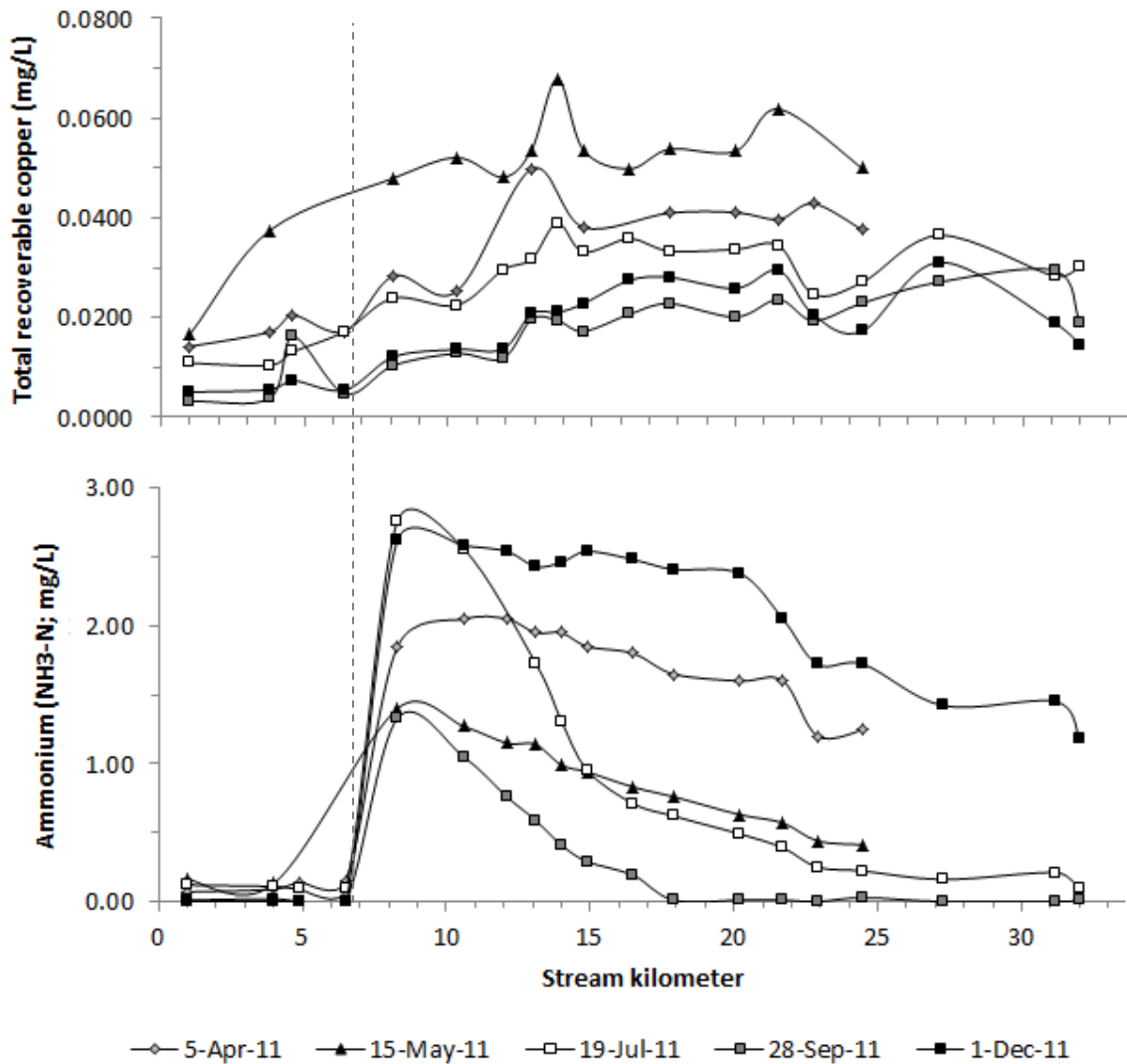


Figure 4.-Synoptic copper and ammonia in Silver Bow Creek, Montana between April 2011 and December 2011. The location of the Butte wastewater treatment plant effluent is identified by the dashed vertical line. Stream kilometers are measured downstream from the upstream end the study area.

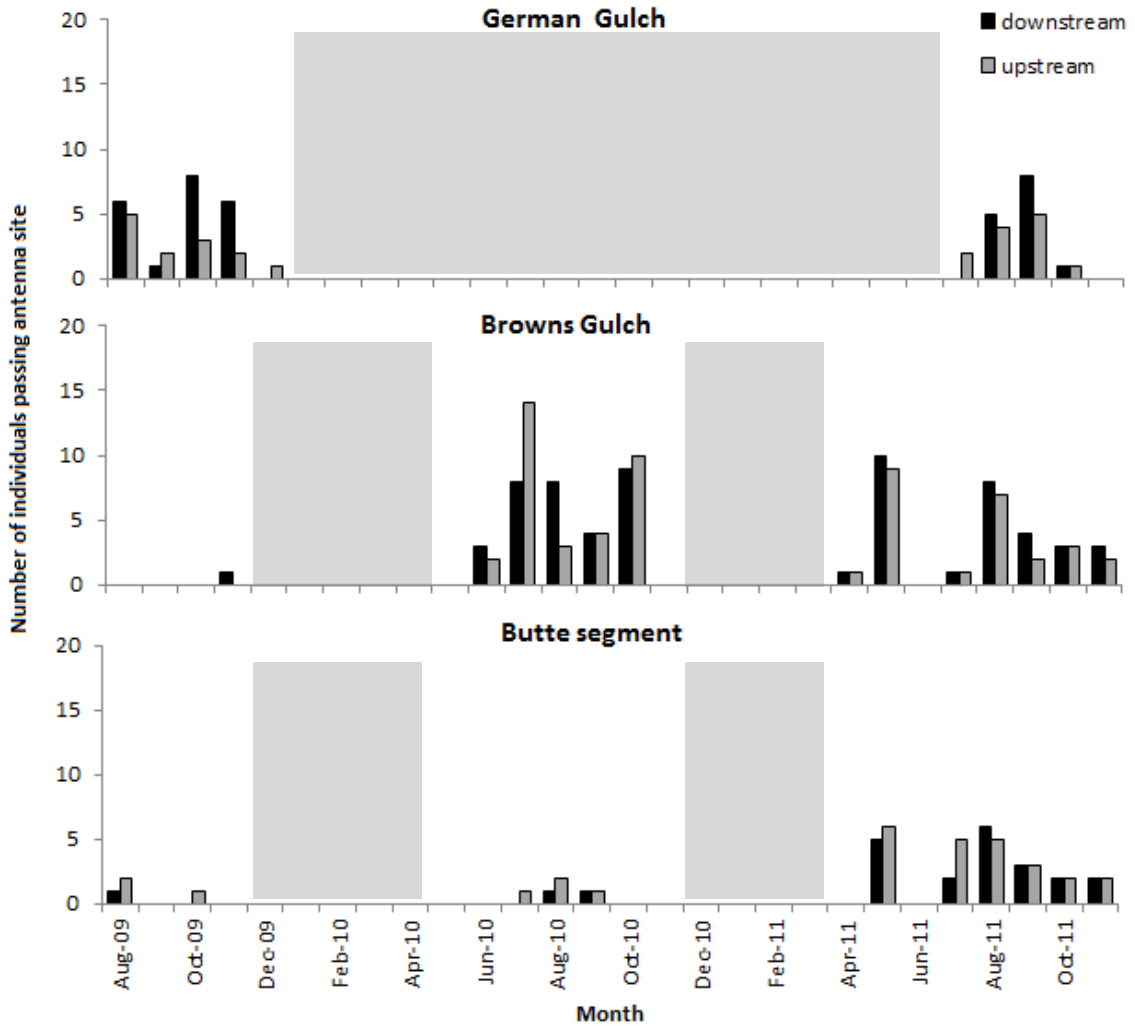


Figure 5.-Number of individual PIT-tagged westslope cutthroat trout *Oncorhynchus clarkii lewisi* moving into (grey bars) and out of (black bars) Silver Bow Creek tributaries by month from August 2009 to December 2011 based on stationary antenna site detections. Periods when antennas were inoperable are shaded.

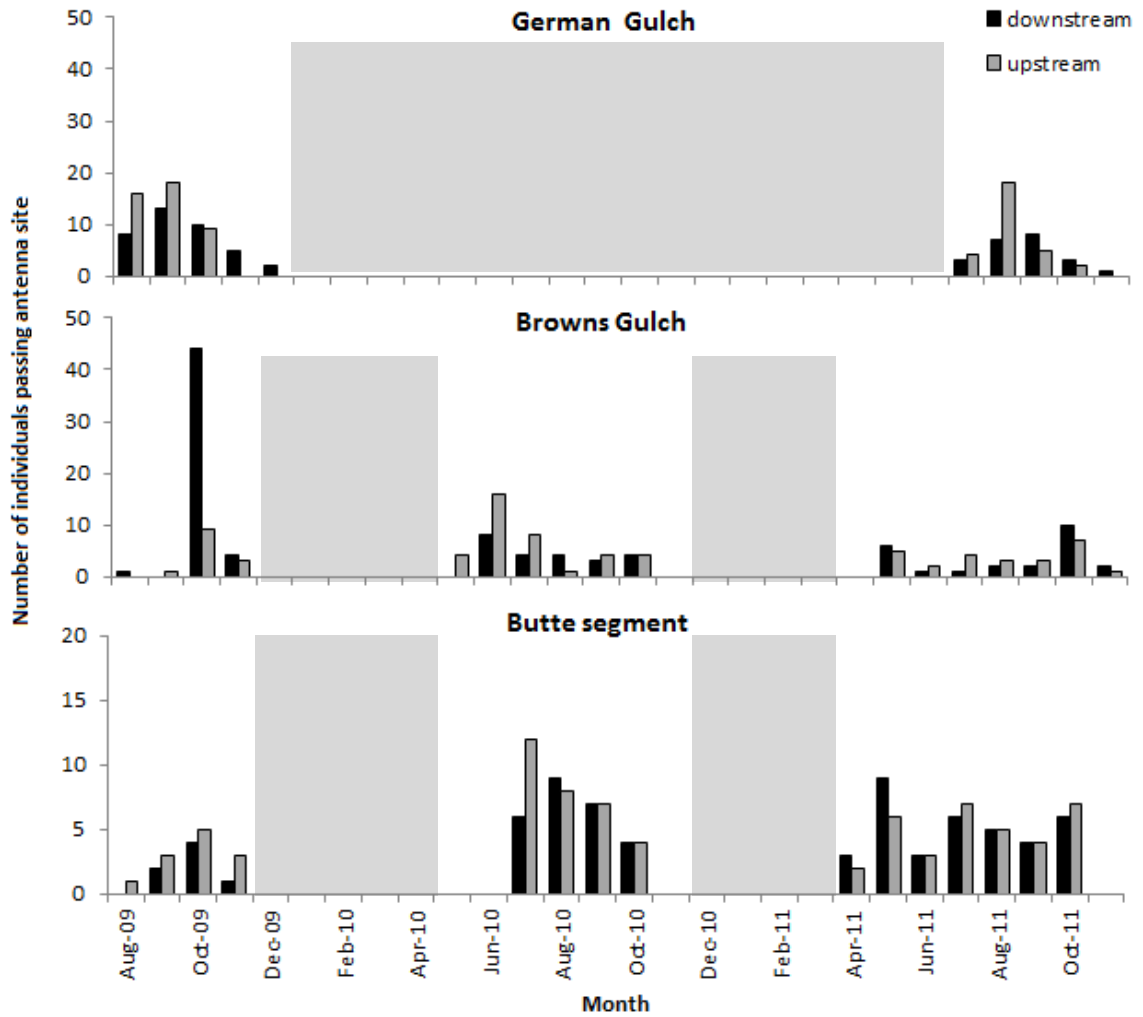


Figure 6.-Number of individual PIT-tagged brook trout *Salvelinus fontinalis* moving into (grey bars) and out of (black bars) Silver Bow Creek tributaries by month from August 2009 to December 2011 based on stationary antenna site detections. Periods when antennas were inoperable are shaded.

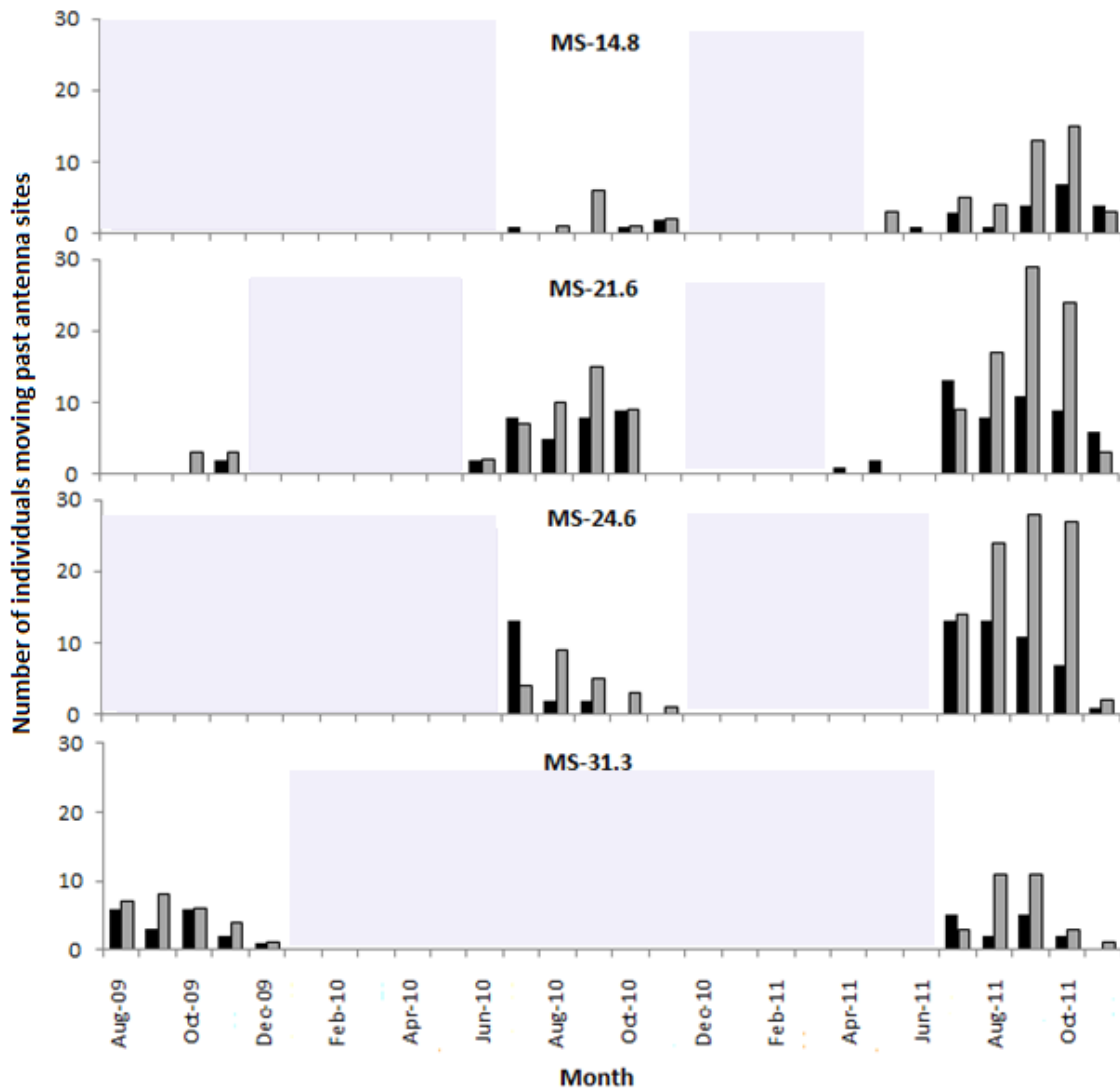


Figure 7.-Number of individual PIT-tagged westslope cutthroat trout *Oncorhynchus clarkii lewisi* moving upstream (grey bars) and downstream (black bars) past antenna sites in the Silver Bow Creek main stem by month. Periods when antennas were inoperable are shaded. See Figure 2 for the locations of each antenna site.

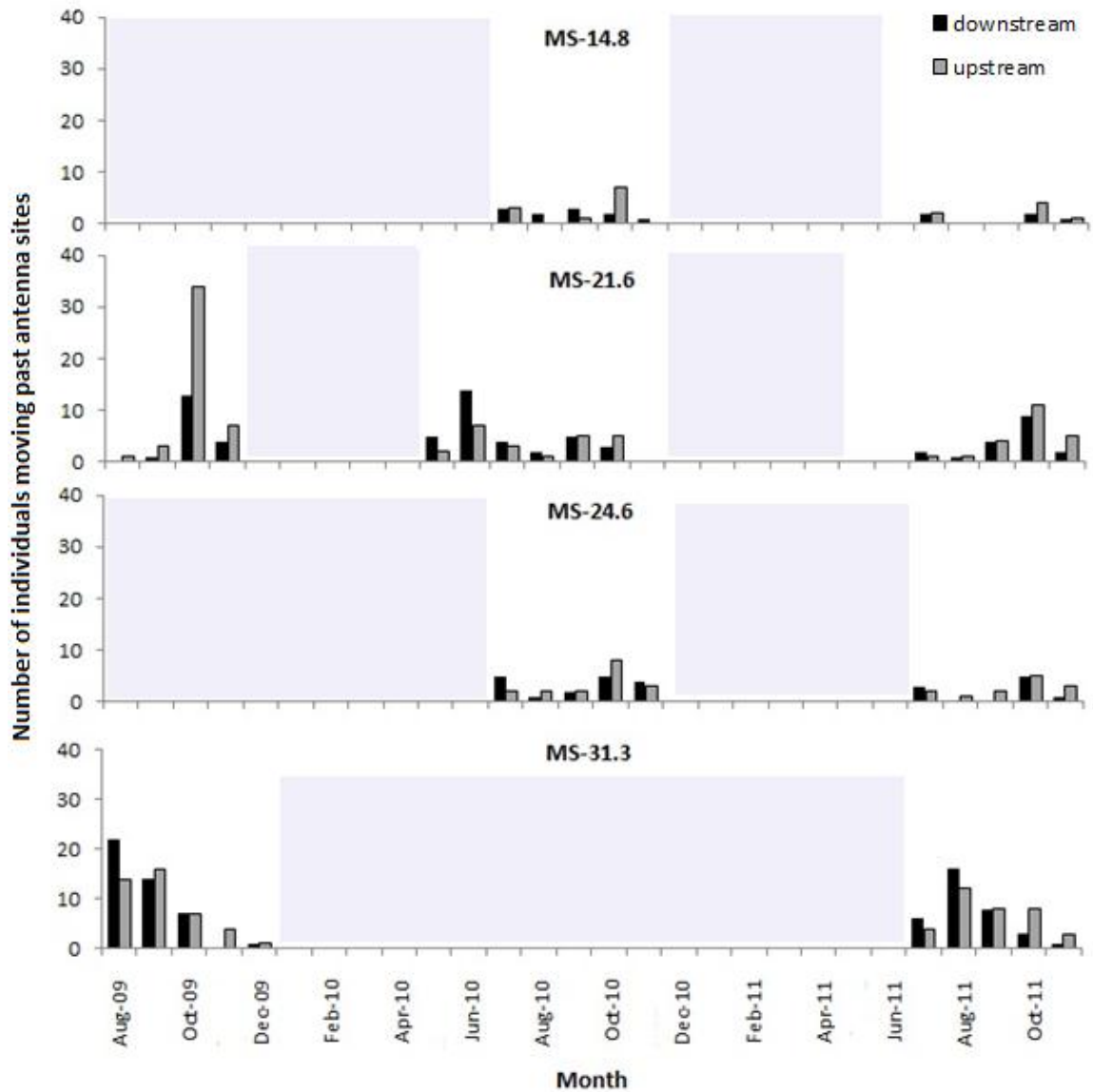


Figure 8.-Number of individual PIT-tagged brook trout *Salvelinus fontinalis* moving upstream (grey bars) and downstream (black bars) past antenna sites in the Silver Bow Creek main stem by month. Periods when antennas were inoperable are shaded. See Figure 2 for the locations of each antenna site.

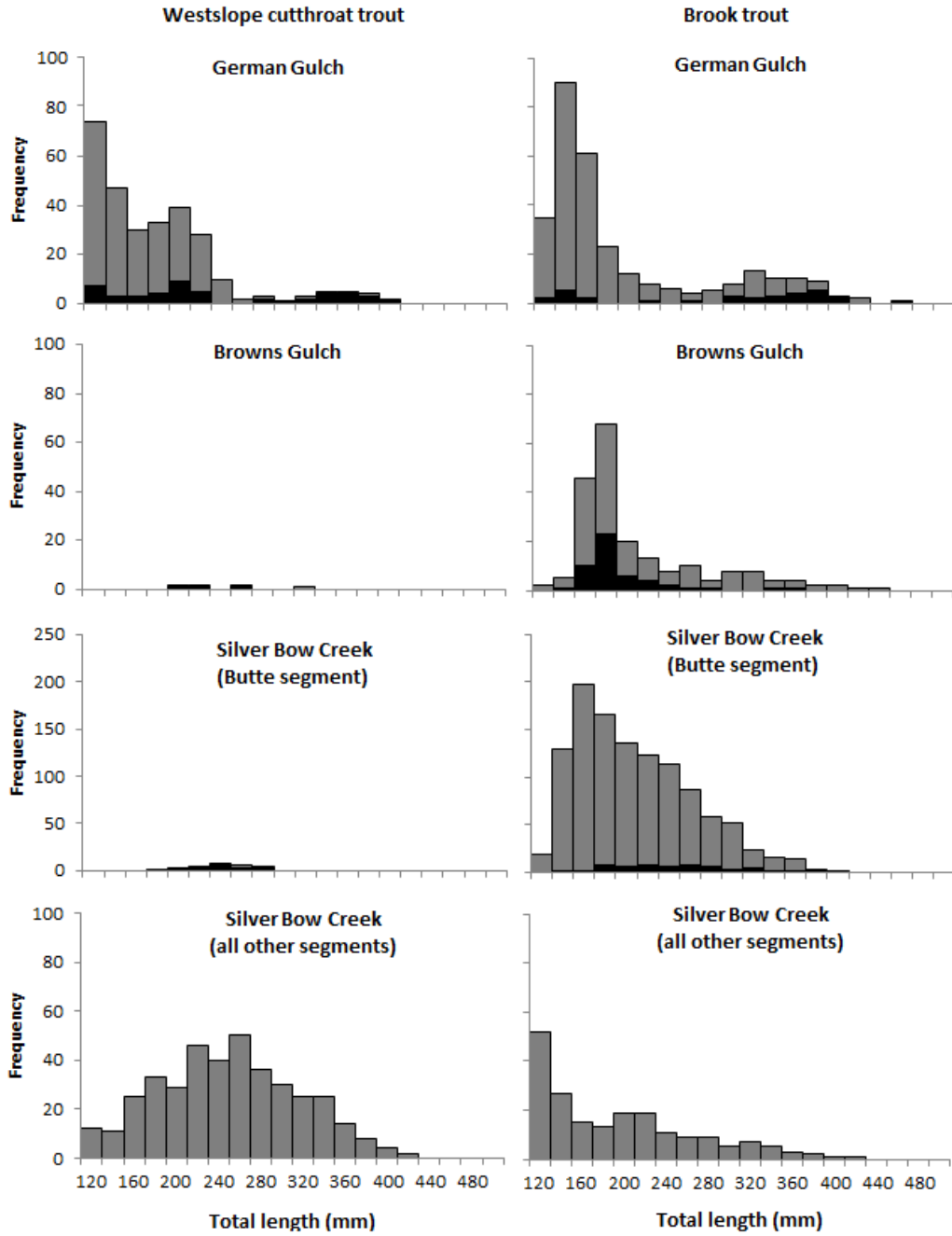


Figure 9.-Length-frequency distributions of westslope cutthroat trout *Oncorhynchus clarkii lewisi* and brook trout *Salvelinus fontinalis* PIT-tagged in the Silver Bow Creek stream network between 2009 and 2011. Black bars represent tributary fish that moved into Silver Bow Creek.

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CHAPTER 4

FISHERY RESTORATION RECOMMENDATIONS FOR
SILVER BOW CREEK, MONTANA

The Silver Bow Creek Watershed Restoration Plan (MTNRDP et al. 2005) states, “the ultimate goal of the remedial and restoration actions is to improve Silver Bow Creek to a condition that supports a self-reproducing fishery for trout species”. Remedial activities have largely succeeded in improving water quality by lowering metal concentrations (Chapter 1). To adequately remove the massive tailings deposits in the Silver Bow Creek stream channel and floodplain (MTDEQ and USEPA 1996), nearly all stream channel and floodplain soils were removed (up to depths of 5 m) and replaced with clean backfill (B. Bucher, CDM Smith, paper read during conference proceedings of the Montana Natural Resource Damage Program, Riparian Restoration in a Contaminated Environment, 2012). Consequently reconstruction of the entire stream channel was necessary (Bucher, unpublished). In this respect, the remediation of Silver Bow Creek is unprecedented in both the scale of contamination and the scale of remedial response.

Salmonids are sensitive to water quality (Chapman 1978; Hansen et al. 1999; Mebane et al. 2012) and demonstrate clear preferences for specific habitat types (Chisholm and Hubert 1986; Heggenes et al. 1991; Young 1998), but, how salmonids would respond to habitat alteration on this scale was unknown. The presence of westslope cutthroat trout *Oncorhynchus clarkii lewisi* and five other fish species in remediated portions of Silver Bow Creek partially demonstrates the suitability of this

remediated stream channel for fish. However, remediation activities were not intended to restore the stream habitat to pre-mining conditions but rather to remove immediate contamination and isolate the stream channel from sources of recontamination. To achieve the goal of returning the stream to “a condition supporting a self-reproducing fishery for trout,” the watershed restoration plan anticipated the need for further restoration actions following remediation (MTNRDP et al. 2005). Therefore, in addition to providing an assessment of the overall recovery of fish populations in the watershed, this thesis was largely intended to provide recommendations for Montana Fish Wildlife and Parks, the Montana Natural Resource Damage Program, and the Montana Department of Environmental Quality to prioritize further restoration actions that will most likely benefit westslope cutthroat trout and other indigenous fishes.

This investigation of the response of fish populations to Superfund remediation in Silver Bow Creek focused on two primary questions. First, are longitudinal salmonid distribution and abundance patterns influenced by water quality and physical habitat characteristics (Chapter 2)? To prioritize restoration actions, it was necessary to identify and quantify which specific factors currently limit the abundance of salmonids in the stream. Superfund remediation activities primarily influenced the physical habitat and heavy metal concentration in Silver Bow Creek. However, water quality in Silver Bow Creek is impaired by ammonia pollution and hypoxia in addition to heavy metal contamination, and therefore, in this analysis it was critical to account for those factors as well. Results indicated that in the 33.8 km main stem study area in 2010 and 2011, salmonid distribution and abundance during the summertime was most strongly related to

dissolved oxygen (DO) concentration (positive) and copper concentration (negative). By contrast, relationships between physical habitat characteristics and salmonid distribution and abundance were either not statistically significant (for pool area and width-depth ratio), or were positive (for channel cover and coarse substrate percentages) but not consistently statistically significant among different years or among models with different response variables (i.e., distribution and abundance models). From this investigation it appears that hypoxia and copper are limiting the distribution and abundance of salmonids in Silver Bow Creek. Ammonia levels, although high in Silver Bow Creek, were not negatively related to either distribution or abundance. However, the ammonia pollution is directly related to hypoxia, which does appear to limit salmonid distribution and abundance.

We attempted to determine if fish populations from uncontaminated tributaries in the Silver Bow Creek stream network moved into remediated portions of the system, suggesting that recolonization was occurring (Chapter 3). From 2009 to 2011, we observed several fish movement patterns related to fish recolonization of Silver Bow Creek. For example, westslope cutthroat trout from the German Gulch tributary, and from main stem segments near the German Gulch confluence, appeared to be moving into remediated Silver Bow Creek stream segments during the late summer and early fall as hypoxia decreased in relation to declining nitrification as water temperature dropped. The majority of the westslope cutthroat trout that moved into, or were collected in, the main stem were large-bodied adults in contrast to the predominantly small-bodied westslope cutthroat trout of German Gulch. Mainstem individuals appeared to remain in

the main stem for extended periods. The greater size of the westslope cutthroat trout in the main stem suggested that main stem segments were utilized as feeding habitat for migratory adults. Because it was unlikely that the large-bodied westslope cutthroat trout sampled in the main stem originally hatched there, the presence of these individuals suggests that individuals from tributary populations (most likely German Gulch) reexpressed a migratory life history behavior. Brook trout *Salvelinus fontinalis* in Blacktail Creek did not often move downstream past the Butte wastewater treatment plant effluent into main stem segments where ammonia and copper concentrations sharply increased.

Perhaps the most notable finding of this research has been the rapid rate at which westslope cutthroat trout in the German Gulch population adapted to the availability of feeding habitat in the remediated Silver Bow Creek main stem. These individuals appear to use the main stem for security and feeding, and then returning to German Gulch to spawn (fluvial-adfluvial migratory behavior; Varley and Gresswell 1988; Northcote 1997). Curiously, the number of brook trout in the main stem has remained low. Westslope cutthroat trout were approximately four times more abundant than brook trout in main stem segments below the effluent. In contrast, brook trout were more common than westslope cutthroat in the remediated Lower Area One reach. Physical characteristics of this section are similar to downstream main stem segments, but because it is upstream of the effluent confluence, ammonia and copper concentrations are low, and DO is high.

Findings from this study suggest that the first restoration priority to increase salmonid abundance in Silver Bow Creek is to alleviate hypoxia by reducing ammonia concentrations in the Butte wastewater treatment plant effluent. Hypoxia during the summertime limits the abundance of salmonids in Silver Bow Creek. Fortunately, wastewater treatment plant upgrades are now in the planning phase. To improve public access to the remediated stream corridor and to highlight the progress in rehabilitating Silver Bow Creek, construction of a 40 km bicycle path along Silver Bow Creek between the cities of Butte and Anaconda, Montana is in progress. Reducing the levels of ammonia in the stream will not only improve conditions for salmonids but will also improve the area aesthetically by reducing the algal blooms that occur during late summer. Throughout the Hypoxic and Recovery segments (Chapter 2, Figure 1), aquatic plant and algae growth is so extensive that the stream stage exceeds bankfull during late summer because of water displacement by plant and algal growth. Additionally, the smell of ammonia in Silver Bow Creek is strong and reducing the ammonium levels would eliminate this odor.

One unintended consequence of reducing the ammonia loads in the wastewater effluent is that brook trout populations may expand into main stem segments currently dominated by westslope cutthroat trout. It is unclear why brook trout populations from the Butte segment have not recolonized the remediated main stem segments downstream. It seems plausible that brook trout are not tolerant of the excessive ammonia concentrations below the effluent. However, this hypothesis is speculative. Silver Bow Creek has undergone dynamic changes in the last decade, and it is difficult to predict the

response of fishes to further modification of water quality. Because the wastewater effluent currently exerts a strong influence on the ecology of the system, a thorough reevaluation of the fish population response to wastewater treatment plant upgrades would be useful. As DO and ammonia levels approach a more normal range of variation, the relationships between salmonid distribution and abundance in relation to copper concentrations and physical habitat characteristics of the stream may be substantially different.

Reducing copper concentrations is another critical restoration priority for improving conditions for salmonids in Silver Bow Creek. Superfund remediation has reduced copper concentrations from pre-remediation levels (Chapter 2, Figure 2); however, copper continues to have negative effects on salmonid distribution and abundance. Synoptic sampling demonstrated that both dissolved (Chapter 2, Figure 8) and total recoverable copper (Chapter 3, Figure 14) increased below the effluent confluence and remained elevated downstream throughout the remainder of the main stem study area. Therefore, it may be prudent for wastewater treatment plant upgrades to incorporate removal of copper from effluent discharging into Silver Bow Creek.

A third restoration priority is the identification and alleviation of metal contamination sources in the Butte Hill area. Over the long term, reduction of copper concentrations in Silver Bow Creek may be the most important factor for restoring Silver Bow Creek salmonid populations to a “self-sustaining” condition. At base flow, the Butte wastewater effluent contributes approximately 40% of the main stem flows (Plumb 2009), and elevated copper concentrations in the effluent affect main stem copper levels.

However, during periods of high stream discharge (e.g., spring snowmelt runoff and summer precipitation events), metal concentrations increase in all parts of the main stem, and the source does not appear to be the wastewater effluent. For example, our synoptic water samples yielded elevated copper concentrations in Silver Bow Creek during spring snowmelt runoff (15 May 2011) when discharge ($1.8 \text{ m}^3/\text{s}$) was approximately three times greater than base flow ($0.56 \text{ m}^3/\text{s}$) conditions (28 September 2011 and 2 December 2011; Chapter 3, Figure 14). Selch (2009) found that even minor precipitation events in the watershed can lead to spikes in metal (and ammonia) concentrations in Silver Bow Creek, and spikes were associated with increased westslope cutthroat trout mortality.

Genetic analysis of westslope cutthroat trout in the German Gulch-Silver Bow Creek population indicates that the population is 0.3% hybridized with rainbow trout (J. Lindstrom and R. Leary, Montana Fish Wildlife and Parks, unpublished data), and it is classified as a “core conservation population” for cutthroat trout management (UDWR et al. 2000). Conservation of unique ecological and behavioral traits specific to populations is an important goal of the fisheries management divisions of the states of Colorado, Idaho, Montana, Nevada, New Mexico, Utah, and Wyoming (UDWR et al. 2000). Because westslope cutthroat trout migrants from German Gulch have apparently resulted in the reexpression of a fluvial-adfluvial life-history type in Silver Bow Creek, a fourth restoration priority would be to protect this population from further hybridization with rainbow trout.

Because the Warm Springs Ponds contains a population of rainbow trout, the German Gulch-Silver Bow Creek westslope cutthroat trout is at risk from further

hybridization unless the rainbow trout are removed or the westslope cutthroat population is isolated from rainbow trout colonization. Peterson et al. (2008) examined the costs and benefits of isolating indigenous salmonid populations to protect them from colonization (or “invasion”) by nonindigenous salmonids and determined that the size and quality of the habitats isolated strongly influenced the potential benefit of the intentional isolation strategy. Similarly Hildebrand and Kershner (2000) determined that to ensure the persistence of isolated cutthroat trout populations, the most important consideration was the overall size (stream length) of isolated habitat (i.e., at least 25 km of habitat for low-density cutthroat trout populations). If a barrier was placed in Silver Bow Creek below the German Gulch confluence, the total stream length inhabited by salmonids that would be isolated, would exceed 25 km by an order of magnitude. However, if this strategy were to be implemented, connectivity between German Gulch and Silver Bow Creek would be necessary to insure movement of individuals among components of the stream network. Some westslope cutthroat trout from German Gulch use Silver Bow Creek to complete their life history, and therefore, maintaining connectivity between the spawning and rearing habitats in German Gulch and the feeding habitats in Silver Bow Creek will be critical to persistence and expansion of the fluvial-adfluvial life-history strategy.

Finally, the lower portion of Browns Gulch is frequently dewatered during the summer (Saffel et al. 2011). We observed that, although westslope cutthroat trout were not frequently sampled in Browns Gulch, they frequently moved between this tributary and Silver Bow Creek. This observation suggests that westslope cutthroat trout may be using lower Browns Gulch as temporary refuge habitat to avoid highly variable water

quality in the main stem. Therefore, maintaining adequate flows to insure connectivity between the two habitat areas may promote persistence of salmonids in the stream network.

Since remediation began in 1998, conditions in Silver Bow Creek have changed rapidly. If planned wastewater treatment plant upgrades are successful, and ammonia loads in the wastewater effluent are reduced, the ecology of the stream is likely to be strongly altered. At that time further investigation into the response of salmonids to remediation as well as the wastewater plant upgrades may be warranted. Data collected during this study will provide a valuable point of comparison to evaluate the success of the wastewater plant upgrades. In addition, the response of salmonids to remediation will likely need to be reevaluated in the context of lowered ammonia levels and increased summer DO levels. At that time, the relationships between salmonid abundance and metal concentrations as well as physical habitat characteristics may be quite different than those observed during this study.

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