MADISON RIVER DRAINAGE 2188 PROJECT MONITORING REPORT 2021

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Introduction

Montana Fish, Wildlife & Parks (FWP) monitors the Madison River fishery to determine the potential effects from the operations at Hebgen and Madison dams on fisheries in the Madison River Drainage. This work is funded through an agreement with NorthWestern Energy (NWE), the owner and operator of the dams. The agreement between FWP and NWE is designed to assist NWE in meeting the terms and conditions of the Federal Energy Regulatory Commission (FERC) license issued to NWE in 2000 to operate hydropower systems on the Madison and Missouri rivers (FERC 2000). This includes Hebgen and Madison dams (Figure 1), as well as seven dams on the Missouri River collectively referred to by FERC as the 2188 Project. The 2188 license details requirements NWE must follow for the operation of the dam and hydropower facilities on the Madison and Missouri Rivers.

NWE entered a 10-year Memorandum of Understanding (MOU) with state and federal resource management agencies to provide annual funding to implement FERC license requirements for the protection, mitigation, and enhancement (PM&E) of fisheries, recreation, and wildlife resources. The MOU established Technical Advisory Committees (TACs) to collectively allocate annual funding to implement PM&E programs and the provisions of the 5-year fisheries and wildlife PM&E plans using adaptive principles. The Madison Fisheries Technical Advisory Committee (MadTAC) comprised of representatives from NWE, FWP, the U.S. Fish & Wildlife Service (USFWS), the U.S. Forest Service (USFS), and the U.S. Bureau of Land Management (BLM) is responsible for the allocation of funds to address fisheries issues related to operations of the Hebgen and Madison Dams under the 2188 license.

This report summarizes work completed by FWP in 2021 with funding provided by the MadTAC to address requirements of the FERC 2188 license, specifically Articles 403, 408, 409, 412, and 419 that pertain to the Madison river fishery. Work included 1) fish abundance estimates in the Madison River, 2) assessment of fish populations in the three mainstem impoundments: Hebgen Reservoir, Quake Lake, and Ennis Reservoir, 3) conservation and restoration of Arctic Grayling populations, 4) conservation and restoration of Westslope Cutthroat Trout populations, 5) enhancement and restoration of tributaries, 6) participation in a flushing flow evaluation, 7) statistical evaluation of habitat types on fish abundances, 8) assistance with a microchemistry study to evaluate tributary and mainstem spawning contributions to the Madison River fisheries.

Study Area

The Madison River originates in Yellowstone National Park at the confluence of the Gibbon and Firehole rivers and flows north for 180 miles through Southwest Montana to its confluence with the Missouri River near Three Forks. The Madison transitions from a narrow, forested river valley in the headwaters to a broad valley bounded by the Madison and Gravelly mountain ranges south of Ennis. North of Ennis the river flows through a steep canyon for 11 miles before it transitions into a broad alluvial valley bottom where it joins the Jefferson and Gallatin rivers, forming the Missouri River (Figure 1).

Two dams impound the Madison River; Hebgen Dam forms Hebgen Reservoir and the Madison Dam forms Ennis Reservoir (Figure 1). Hebgen Reservoir is operated as a water storage facility to control inflow to the downstream Madison Dam, which is a power generating facility. Madison and Hebgen dam operations are coordinated to provide year-round flows at or above the required minimum flow of 1100 cubic feet per second (cfs) and mitigate thermal issues in the Madison River below Madison Dam (Figure 1).

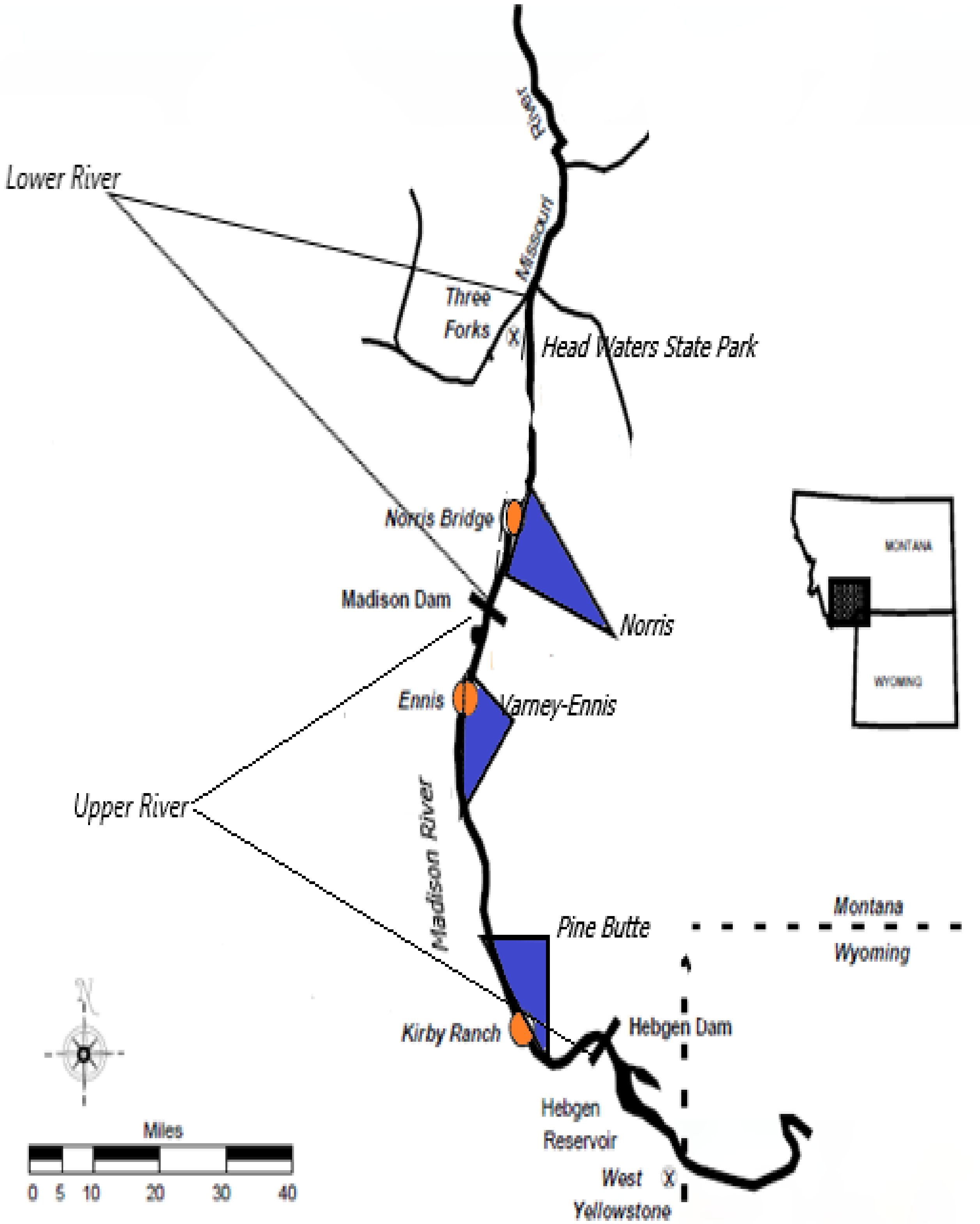

Figure 1. Locations of NWE dams on the Madison River (FERC Project 2188) and delineation of the upper and lower Madison River. FWP annual abundance estimate sections are shown in blue and NWE monitoring sites in orange.

Monitoring and Projects

Article 403-River Discharge: Article 403 of the Project 2188 FERC license specifies operational conditions, including minimum and maximum instream flows in various sections of the Madison River. Specifically, NWE must maintain a minimum flow of at least 150 cfs in the Madison River below Hebgen Dam (gage no. 6-385) and limit the change in outflow from Hebgen to no more than 10% per day. Additionally, a minimum flow of 600 cfs on the Madison River at Kirby Ranch (USGS gage no. 6-388) and 1100 cfs on the Madison River at gage no. 6-410 below the Madison Dam must be maintained. Flows at Kirby Ranch are limited to a maximum of 3500 cfs under normal conditions to minimize erosion of the Quake Lake outlet. License requirements also require the establishment of the permanent flow gauge at Kirby Ranch. FWP and NWE monitor river flows to avoid deviations from operational conditions.

Deviations from Article 403 operational conditions occurred below Hebgen Dam and at Kirby Ranch on November 30, 2021. The deviations were the result of a broken component on the Hebgen Dam gate, which caused the gate to fall and reduce flows from 648 cfs to 228 cfs in 45 minutes. NWE staff increased outflows to 248 cfs 12 hours later where they remained for about 31 hours until the gate could be raised. The abrupt change in discharge resulted in a deviation from the condition that limits changes in the outflow from Hebgen Dam to no more than 10% per day. Additionally, because flows out of Hebgen were 248 cfs or less for about 31 hours, flows at the downstream Kirby Gage decreased below the minimum 600 cfs flow requirement to 395 cfs for about 48 hours.

The rapid reduction of river stage in the Madison River between Hebgen Dam and Quake Lake stranded and killed adult and juvenile fish as well as exposed Brown Trout and Mountain Whitefish redds. FWP, NWE, and volunteers from the public completed a fish salvage operation on December 1st in the affected reaches. Stranding occurred downstream of Quake Lake but was primarily limited to juvenile fish in overwintering habitats (e.g., side channels) upstream of Kirby Bridge that became disconnected from the river as stage dropped. Although no stranded adult fish were observed in this stretch of river, the change in river stage dewatered numerous Brown Trout redds in important spawning areas (Byorth 1999; Downing 2002; Figures 4 and 5). Between Hebgen Dam and the Quake Lake inlet, an estimated 3.4 acres of nearshore spawning habitat may have been exposed (Figure 2). Although that reach of the Madison River is predominantly a single thread channel, the gate failure demonstrated the potential effect of reduced river stage on redds in near-shore habitats (Figure 3). Exposed near-shore habitat was not quantified for the reach between the Raynolds FAS and Kirby Bridge.

NWE and FWP will monitor fish populations to assess the effects of gate failure over the next five years. NWE additionally proposed, in consultation with MadTAC, immediate mitigation options to address the impacts to the fishery caused by the gate failure.

Feet=204+.0329(cfs)

648 cfs=225.31ft

216 cfs=211.10ft

Difference=14.22ft

Acres=(10560ft*14.22ft)/43560ft^2

3.44 acres of exposed near shore habitat

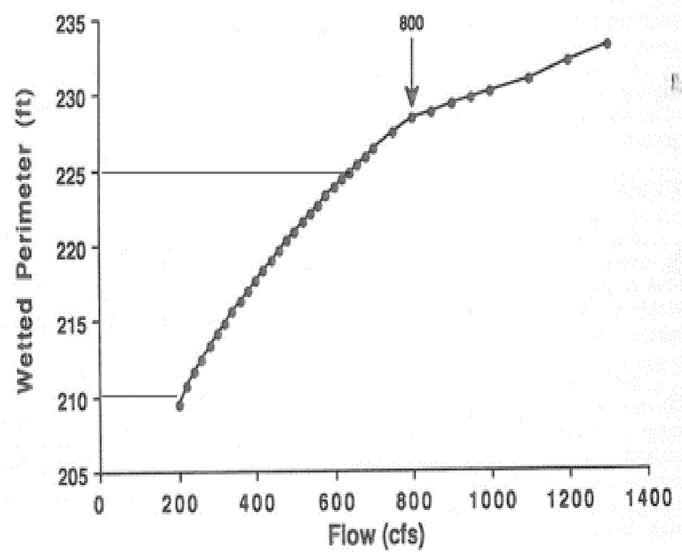

Figure 2. Wetted perimeter of the Madison River between Hebgen Dam and Quake Lake. The area of exposed near shore habitat is estimated from the following equation: Feet = 204 + 0.0329 (cfs).


Figure 3. A dewatered Brown Trout redd near the bank in the Madison River between Hebgen Dam and Quake Lake.

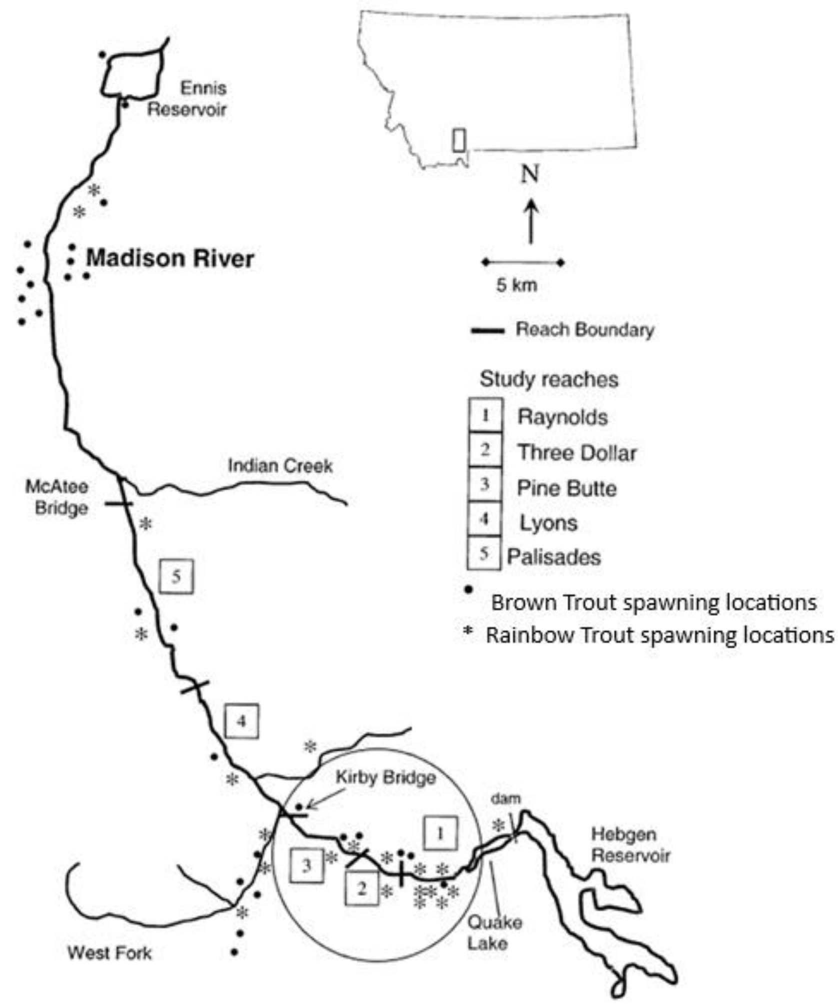

Figure 4. Spawning areas of Brown and Rainbow Trout (Byorth 1999; Downing 2002). The area of concern is in the circle. Brown Trout spawning locations are represented by black dots and Rainbow Trout spawning locations are represented by asterisks. Numbered squares identify reaches delineated by Downing (2002).


Figure 5. Partially dewatered Brown Trout Redd in a side channel near the Kirby Bridge.

Article 408-1) Effects of Project Operations on Hebgen Reservoir Fish Populations: FWP monitors the Hebgen Reservoir fish assemblage with annual spring gill netting surveys for the purpose of assessing the effects of project operations (Figure 6). Significant changes in the fish assemblage would warrant a review of and potential change to project operations to address identified issues.

The mean catch-per-unit-effort (C/f) of total trout in Hebgen Reservoir was about 20 trout/net in 2021, which was slightly above the long-term average (Figure 7). The C/f of Brown Trout decreased about 21% to 14.8 trout/net while Rainbow Trout decreased 12% to 5.2 trout/net, which were below the management goals for each species (Brown Trout management goal = 15.5 fish/net; Rainbow Trout = 7.5 fish/net). However, the mean lengths of Brown and Rainbow Trout increased to 459 mm (≈ 18") and 433 mm (≈ 17"), respectively, which were above the long-term averages. Eighty-five percent of the Brown Trout captured in gill nets were ≥ 406 mm [≈ 16"], which exceeded the management goal of 75%. Sixty-six percent of the Rainbow Trout captured were ≥ 406 mm, which met the management goal.

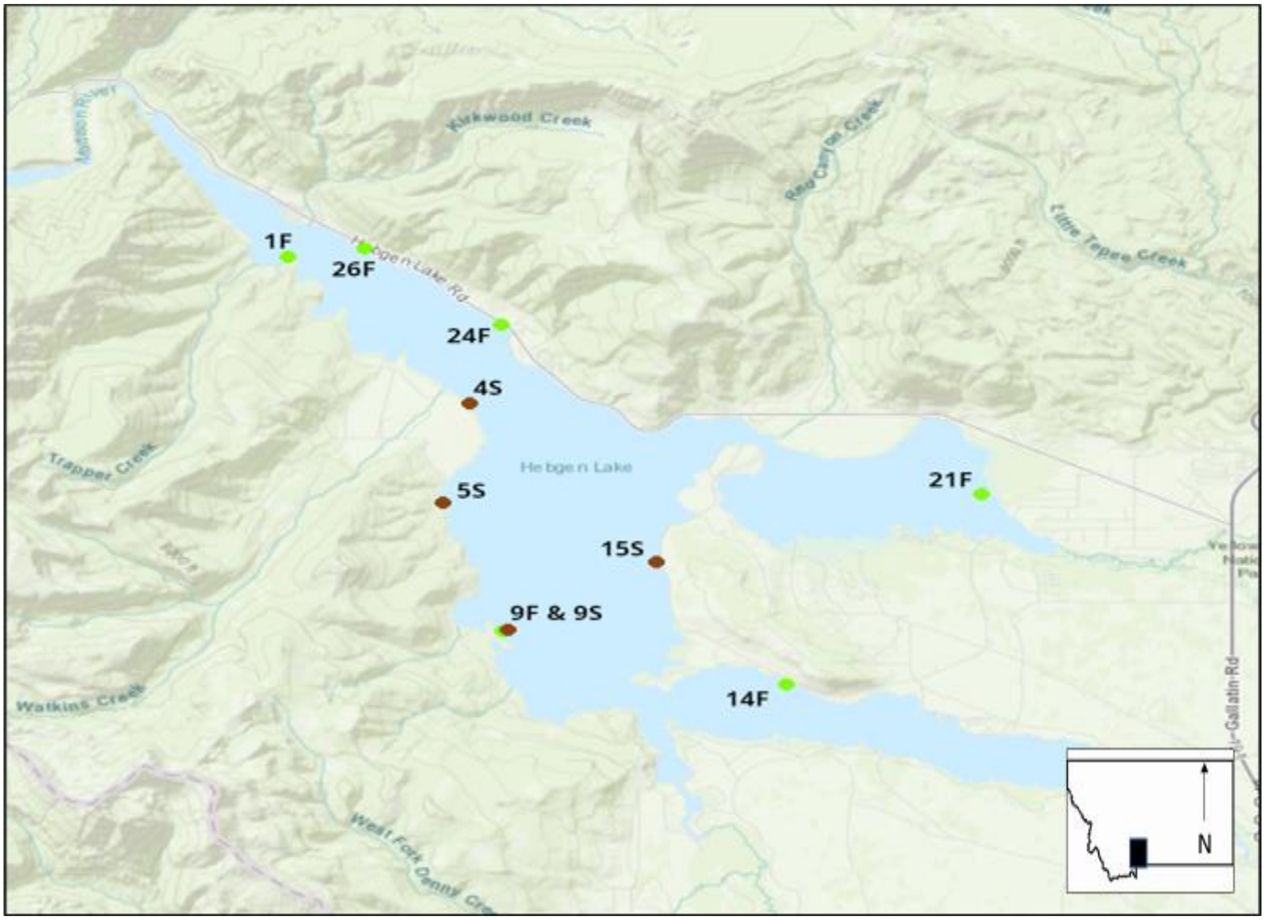

Figure 6. Hebgen Reservoir gill net locations and names. Brown and green circles are sinking (N = 4) and floating (N = 6) gill nets, respectively.

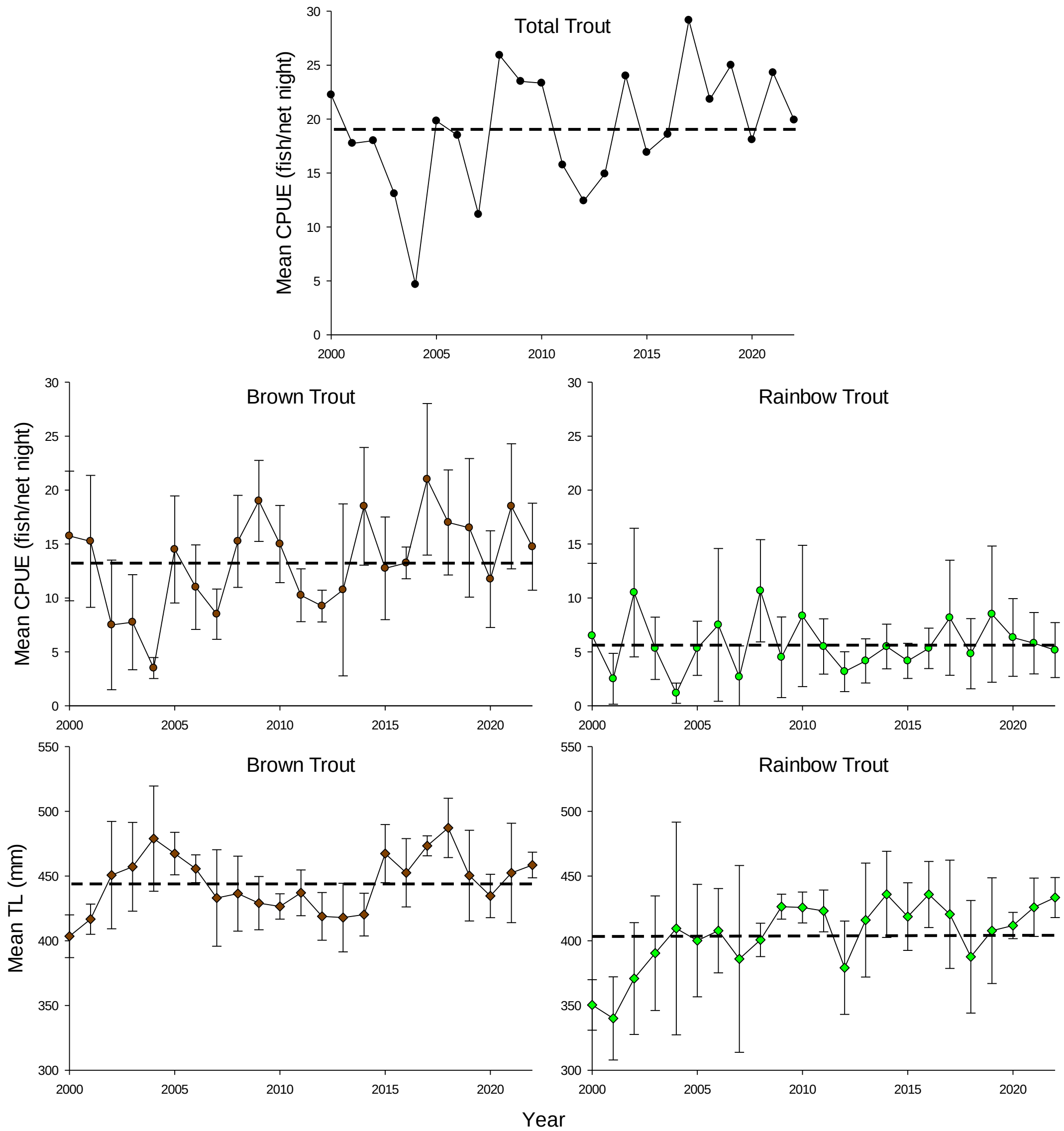

Figure 7. Mean catch-per-unit-effort (CPUE) of total, Brown, and Rainbow Trout captured in Hebgen Reservoir from 2000 to 2022. Total trout abundances represent all trout captured in four sinking gill nets and six floating gill nets. Brown and Rainbow Trout CPUE were limited to either sinking or floating gill nets, respectively. Mean total lengths were calculated using all Brown and Rainbow Trout captured each year. Dashed lines are the long-term averages (2000-2022) and error bars are the 95% confidence intervals.

Article 412–1) Effects of Project Operations on Ennis Reservoir Fish Populations: FWP has historically monitored the Ennis Reservoir fish assemblage with biannual fall gill netting surveys on odd years. New gill net locations were established in 2021 to provide better coverage of the reservoir while eliminating gill net sets in shallow habitats that reduced capture efficiencies. Sampling will occur annually for at least five consecutive years to provide data that can be used to establish management goals for the Rainbow and Brown Trout fisheries. Although FWP will assess long-term trends using data collected with the new sampling approach, much uncertainty will exist with such comparisons until additional data using the new gill net sets are available.

Taking that into consideration, the mean C/f of total trout, Brown Trout, and Rainbow Trout, remain below the long-term averages (Figure 8). However, the mean lengths of Brown Trout (398 mm [≈ 15.5”]) and Rainbow Trout (387 mm [≈ 15.0”]) increased above the long-term averages for both species.

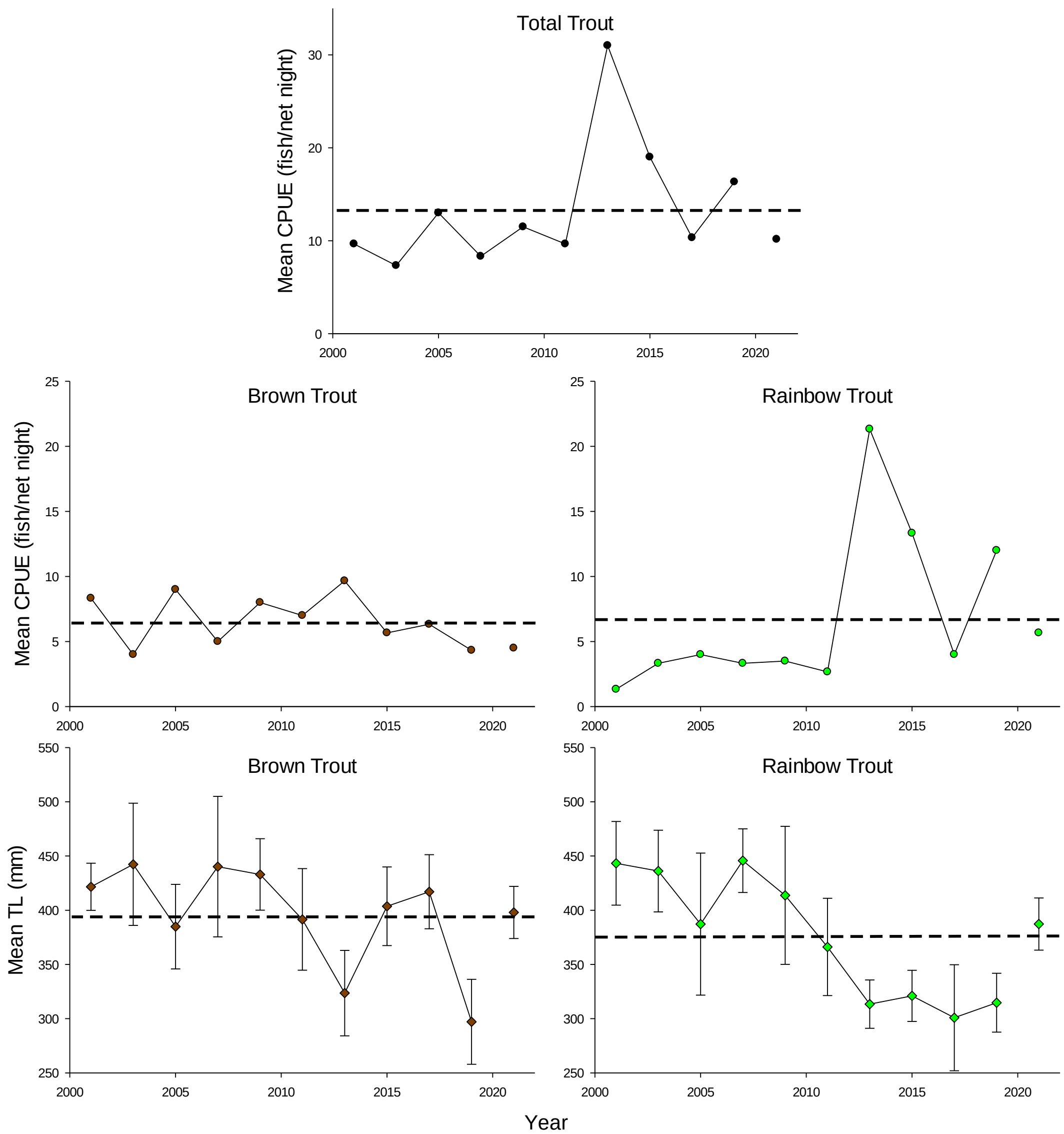

Figure 8. Mean catch-per-unit-effort (CPUE) of total, Brown, and Rainbow Trout captured in gill nets set in Ennis Reservoir from 2001 to 2021. Brown and Rainbow mean CPUE and were calculated using all nets set each year. Mean total lengths were calculated using all Brown and Rainbow Trout captured each year. Dashed lines are long-term averages (2001-2021) and error bars are 95% confidence intervals for mean lengths.

408-3) Reservoir Draw Down Effects on Fish: The interactions between Hebgen Reservoir elevation and operations, trophic status, and the trout populations have been assessed annually by FWP from 2006-2020. Sampling occurred in June, July, and August because these months correspond with the emigration of juvenile trout from natal tributaries to Hebgen Reservoir and their recruitment to the fishery may be influenced by conditions in the reservoir at the time of emigration (Watschke 2006; Clancey and Lohrenz 2007, Clancey and Lohrenz 2008, Clancey and Lohrenz 2009). Reservoir elevation may influence juvenile trout growth and recruitment by altering the amount of habitat along shoreline and zooplankton abundances. Fluctuating reservoir elevations can impoverish the plankton assemblage through the loss of nutrients, which could limit forage for juvenile trout until they can switch to macroinvertebrates or piscivory (Axelson 1961; Haddix and Budy 2005). Hebgen Reservoir has a full pool elevation of 6534.87 feet (msl) and operational standards require NWE to maintain reservoir elevations between 6530.26 feet and 6534.87 feet from June 20 through October 1 and reach full pool elevation by late June or early July. Given the narrow operational range, reservoir conditions are similar among years. As a result, no relationships have been detected between trophic status, zooplankton abundance, or trout and zooplankton abundances. Therefore, limnological sampling, based upon FWP recommendations and input from NWE, will occur every other year or when reservoir elevations fall outside of normal operational ranges.

FWP did not conduct limnological sampling in 2021. However, developing extreme drought conditions resulted in Hebgen pool elevations dropping below normal operational ranges. On 51 occasions, during the summer of 2021, operational changes were made to provide for thermal mitigation in the lower river. Consequently, Hebgen pool elevation dropped below the 6530.26 feet elevation minimum by July 28,2021 and resulted in a 7.0-ft decrease in elevation from June 20 to October 1, 2021.

408-4) Monitor the Effects of Modified Project Operations on Upper Madison River Fish Populations- Madison River Fisheries Assessment: FWP estimated Rainbow and Brown Trout abundances using mark-recapture sampling in three long-term monitoring sections in the Madison River (Pine Butte, Varney, and Norris) to evaluate the influence of modified project operations at Hebgen and Madison dams on the trout fisheries. Although only the influence of project operations are reported here, other potential population drivers (i.e., angling pressure, disease, etc.) are hypothesized to be influential and are being evaluated elsewhere. Trout were collected by electrofishing from a drift boat mounted mobile anode system (Figure 9). Fish captured in the initial trip (marking run) were weighed in grams and measured to the nearest millimeter, marked with a fin clip, observed for hooking scars, and released to redistribute. FWP conducted a second trip (recapture run) about a week later to examine trout for marks administered during the marking run, record lengths of marked fish, as well as document lengths and weights of unmarked fish. Length-specific mark-recapture log-likelihood closed population abundance estimates were generated and standardized to stream mile for Brown and Rainbow Trout using an R-based proprietary FWP fisheries database and analysis tool.

Figure 9. Mobile anode electrofishing (shocking) in the Norris section of the Madison River.

FWP developed management goals for total trout abundances (trout ≥ 252 mm [≈ 10”]) and size structure (percentages of trout ≥ 252 mm that are also ≥ 402 mm (≈ 16”]) for each of the long-term sampling sections using the 66th percentiles of data collected over the past 20 years. The abundance goals for the Pine Butte, Varney, and Norris sections are 2300, 1200, and 2500 trout/mile, respectively. The following are the size structure goals for proportion of fish ≥ 402 mm in each section: Pine Butte – 25%, Varney – 35%, and Norris – 15%. Evaluating PM&E (Protection, Mitigation, and Enhancement) activities and management actions (e.g., flushing flows) in the context of these goals provides a better understanding of how they influence the Madison River trout fishery relative to other potential population drivers. However, difficult sampling conditions in the fall led to unreliable estimates of Brown Trout in the Pine Butte and Varney sections (note the large confidence intervals associated with each estimate in Figure 10). These issues may preclude inference about abundance of Brown Trout in the upper Madison River, which also confounds our ability to determine whether management goals were achieved in those sections. Therefore, the discussion of management goals will be limited to the Norris Section.

Upper Madison River Rainbow Trout abundances were below average in Pine Butte and above average in Varney. In 2021, estimated abundance of Rainbow Trout ≥ 152 mm (≈ 6”) decreased about 22% in the Pine Butte Section to 1,685 trout/mile, which was below the long-term average (Figure 10). The decreased abundance of Rainbow Trout in Pine Butte appeared to be a result of poor recruitment of small fish, which is evidenced in length-frequency histograms by the relatively low number of Rainbow Trout < 252 mm (≈ 10”; Figure 11). Estimated abundances of Rainbow Trout decreased about 17% to 1,995 trout/mile in the Varney Section. However, abundances of Rainbow Trout in Varney remain well-above the long-term average as 2021 provided the second highest abundance estimate in that section in over 20 years. Similar to 2020, many small Rainbow Trout (< 252 mm) were captured in the Varney Section (Figure 12), which may lead to relatively high abundances of large Rainbow Trout the next several years.

Below average abundances of Brown and Rainbow Trout occurred in the lower Madison River. The total estimated abundance of trout in the Norris Section during the spring of 2022 was 1907 trout/mile, which was 24% below the management goal. The estimated abundance of Rainbow Trout in the Norris Section decreased 8% to 1301 trout/mile while Brown Trout increased 14% to 523 trout/mile, which are below the long-term averages for both species (Figure 13). The estimated abundance of Westslope Cutthroat Trout decreased by 16% to 82 trout/mile. Fifteen percent of trout ≥ 252 mm captured in the Norris Section were also ≥ 402 mm, which achieved the management goal for that section. However, the truncated length-frequency histograms of both populations the last two years (Figure 13) indicate survival of juvenile and adult Rainbow and Brown Trout have decreased in the lower Madison River relative to the size structures that supported both populations in the 2000s and 2010s.

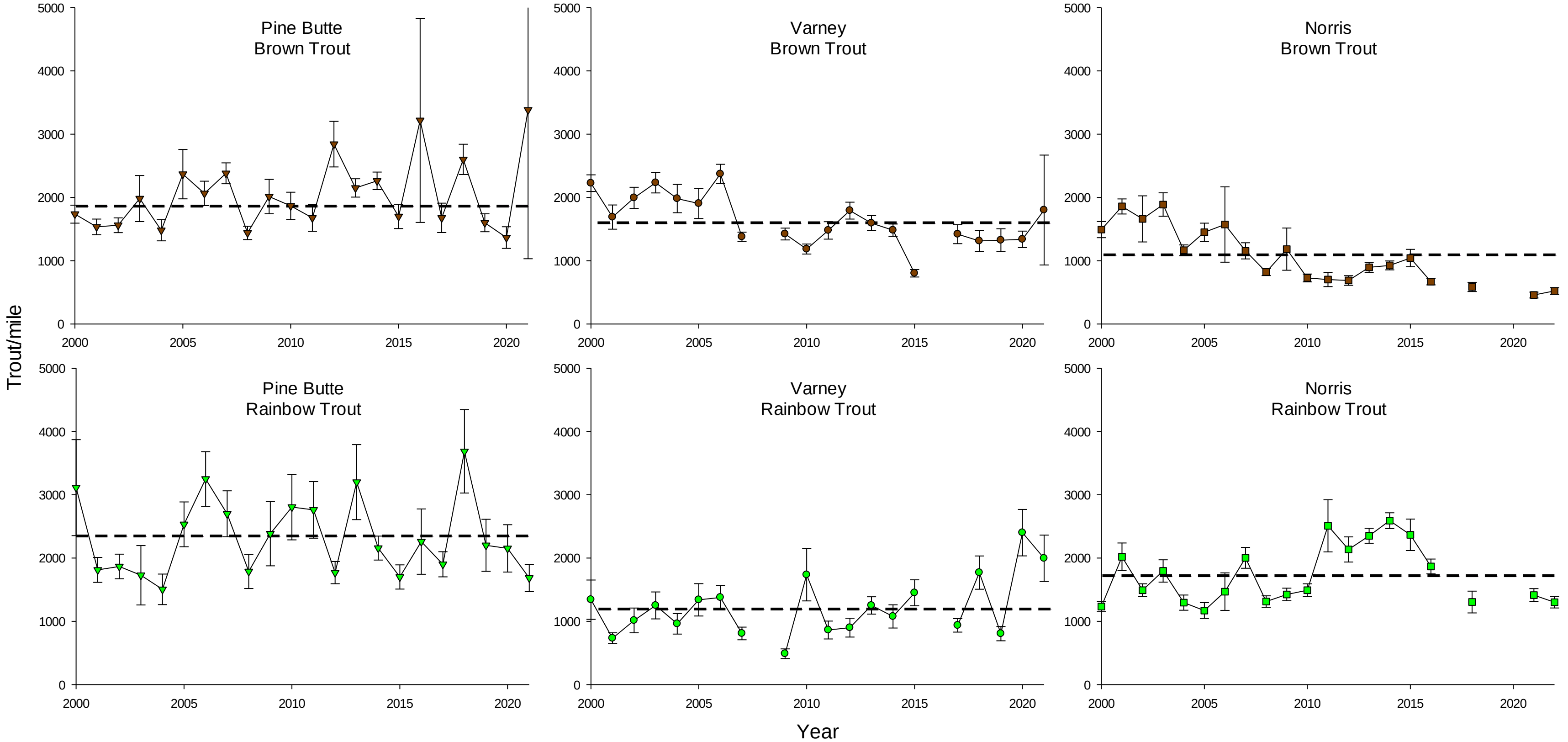

Figure 10. Estimated abundances of Brown and Rainbow Trout ≥ 152 mm (≈ 6") captured in the three long-term sampling sections of the Madison River. Dashed lines are the long-term averages (2000-2022) and error bars are the 95% confidence intervals.

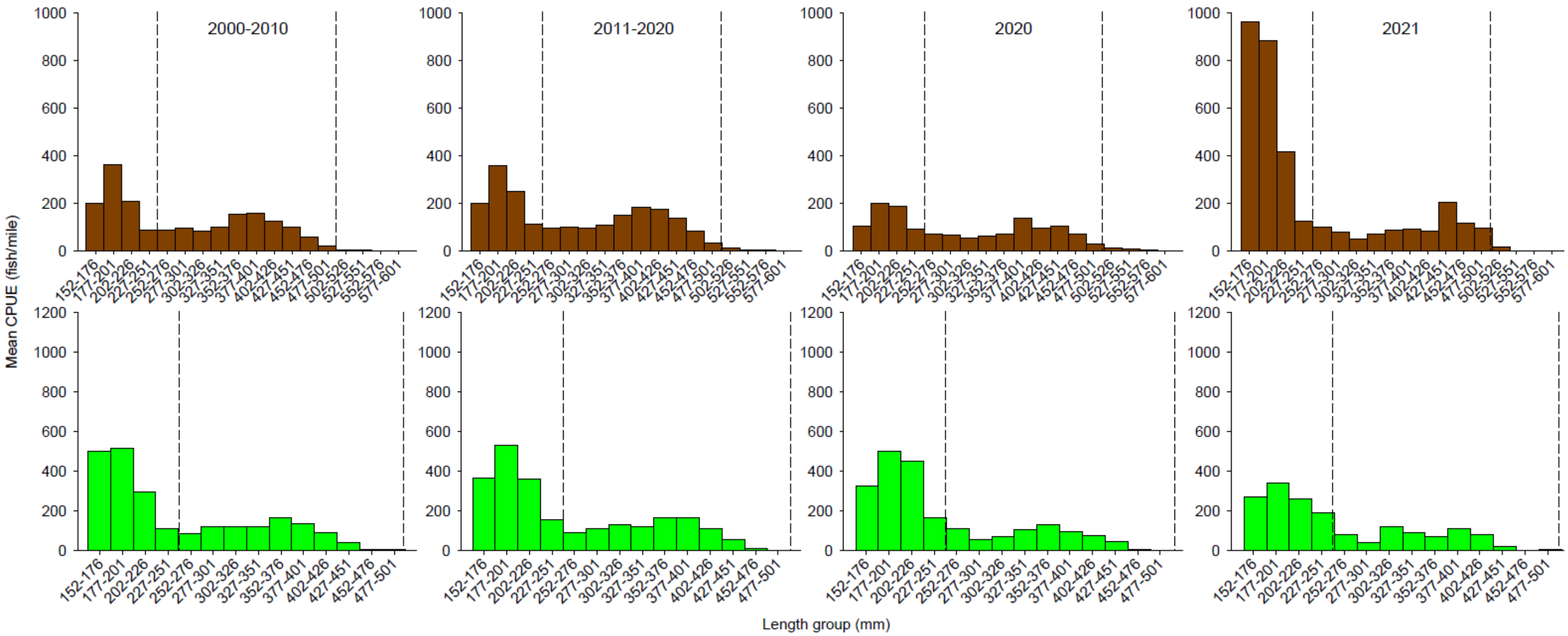

Figure 11. Length frequency histograms of Brown (brown bars) and Rainbow Trout (green bars) ≥ 152 mm (≈ 6") captured in the Pine Butte Section of the Madison River. Dashed lines delineate 10" and 20".

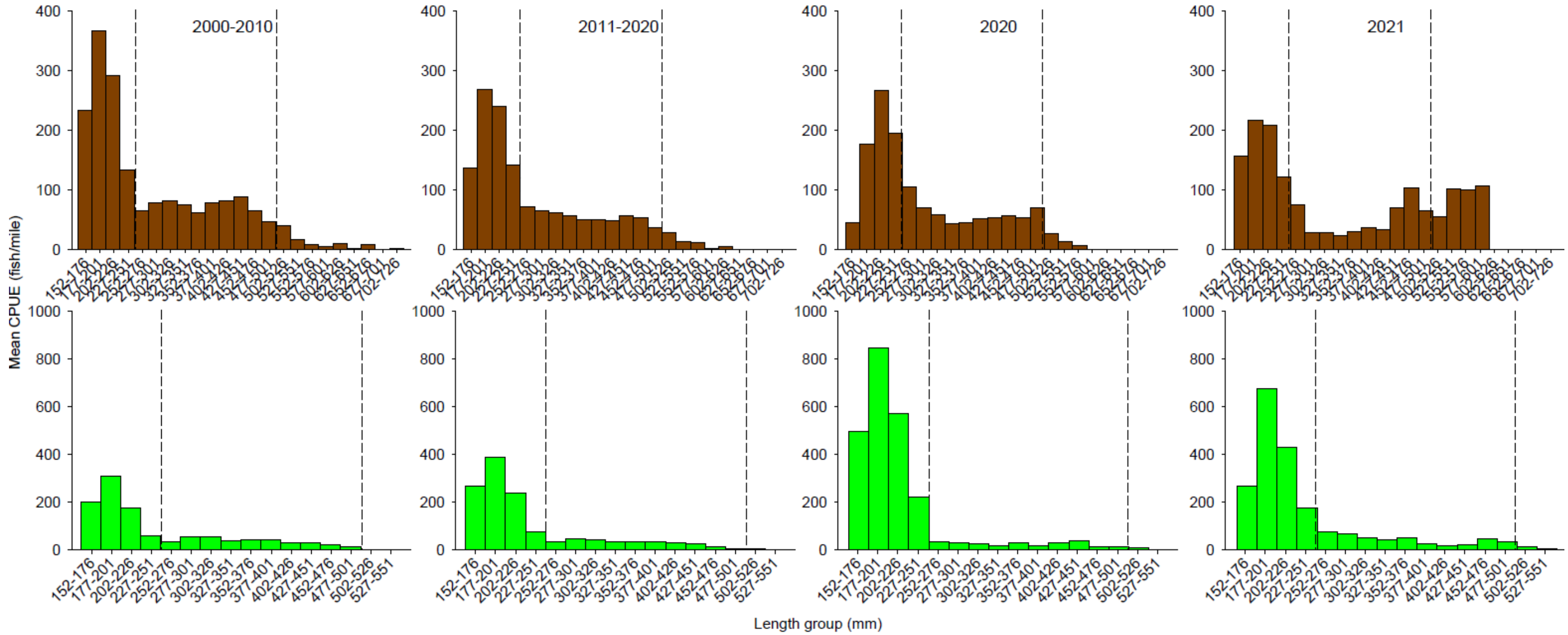

Figure 12. Length frequency histograms of Brown (brown bars) and Rainbow Trout (green bars) ≥ 152 mm (≈ 6”) captured in the Varney Section of the Madison River. Dashed lines delineate 10” and 20”.

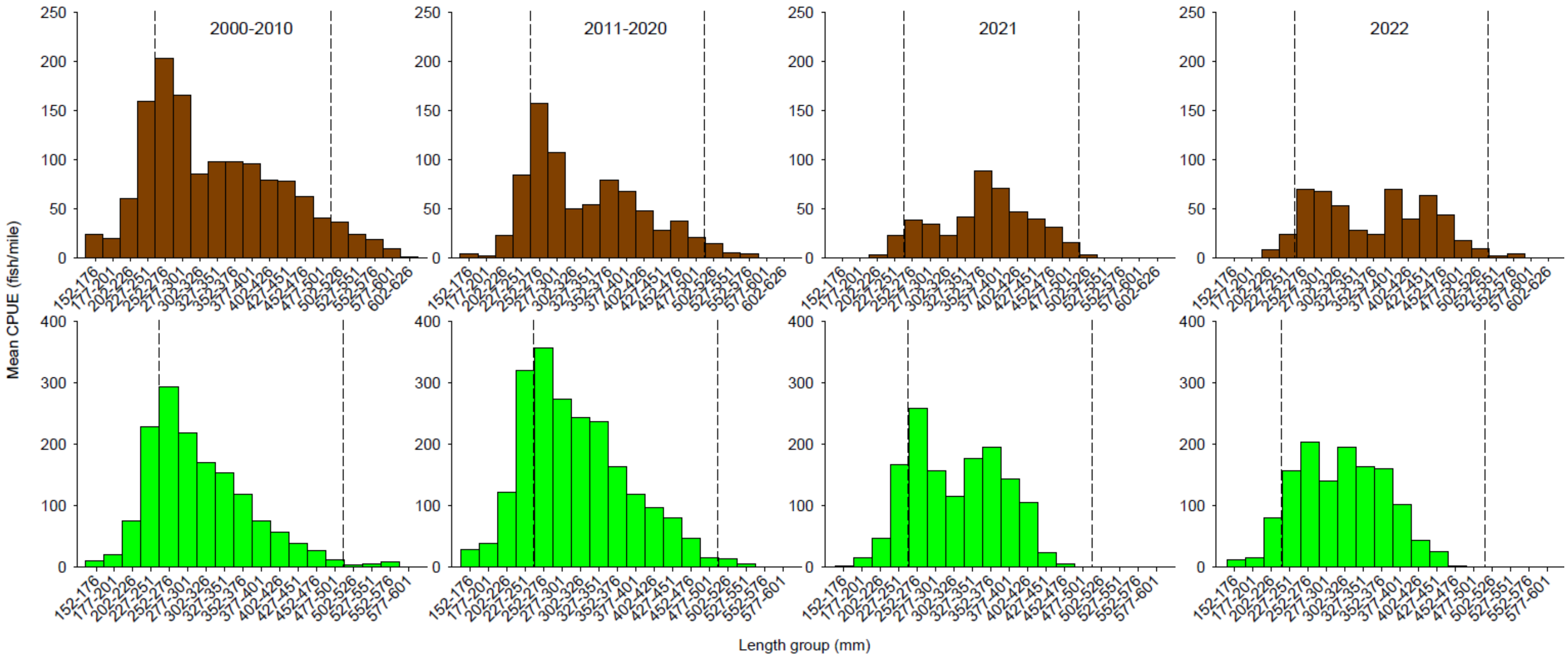

Figure 13. Length frequency histograms of Brown (brown bars) and Rainbow Trout (green bars) ≥ 152 mm (≈ 6”) captured in the Norris Section of the Madison River. Dashed lines delineate 10” and 20”.

408-7) Monitor Species of Special Concern; Madison Artic Grayling; Westslope Cutthroat Trout: Opportunities to recover, conserve, and expand native fish distributions are regularly pursued by FWP and partner agencies. NWE is committed to implementing PM&E measures under Articles 408, 409, 412 of the 2188 FERC License from Hebgen Reservoir to Three Forks Montana to mitigate adverse effects to native fish species associated with Madison Project operations (FERC 2000).

Arctic Grayling: Arctic Grayling reintroduction occurred in several Madison River tributaries between 2014 and 2020. Introductions were carried out by placing embryos in remote site incubators (RSI; Figure 14) and allowing them to hatch and fry to enter the stream. To date, 939,200 eggs have been placed in Madison River tributaries. Hatching success of embryos and fry emigration out of RSIs in tributary streams has been good to fair every year introductions took place except for the 2017 in Blaine Spring Creek, although poor recruitment was observed (Table 1). In 2021, 250,000 eggs from the Green Hollow and Axolotl Lake Big Hole Arctic Grayling genetic reserve brood ponds were divided into Black Sands Spring Creek (150,000) and Moore Creek (100,000) to determine whether higher stocking rates resulted in improved recruitment (Figure 15). During autumn electrofishing surveys, no young-of-year Arctic Grayling were observed in Black Sands Springs or Moore creeks. However, the quality of eggs used for introductions in 2021 was inferior to past years. Eye-up at the Big Timber Hatchery was estimated to be as low as 70% (FWP personal communication, 2021). Introductions will be discontinued in Moore Creek. While there has been limited success in recovering young-of-year grayling in Moore Creek following emigration from RSIs, they have failed to recruit to older age classes. Additionally, access to Moore Creek has been restricted due to a change in land ownership. Arctic Grayling introduction efforts for the next 3-5 consecutive years will focus on Hebgen Reservoir and its tributaries where FWP plans to introduce 1,000,000 eggs and fry from populations of primarily Madison ancestry.


Figure 14. Remote site incubators used to hatch Arctic Grayling eggs in Black Sands Springs in 2021.

Table 1. Arctic Grayling introduction sites. Site, year, quantity of eggs introduced and egg survival and emigration success.

Site	Year	# eggs	Egg survival and emigration
West Fork Madison Upper	2014	1200	Poor
West Fork Madison Middle Spring	2014	10,000	Good
	2015	30,000	Good
	2016	5000	Good
Lake Creek	2014	13,000	Good
	2015	27,000	Good
	2016	5000	Good
Upper O’Dell Creek Grainger Ranch	2015	36,000	Good
	2017	32,000	Good
	2018	60,000	Good
	2019	15,000	Good
O’Dell Creek Longhorn Ranch	2019	45,000	Good
Blaine Spring Creek	2015	15,000	Fair
	2016	5000	Fair
	2017	1000	Poor
	2018	42,000	Fair
	2019	10,000	Fair
	2020	150,000	Fair
Moore’s Creek	2015	5000	Fair
	2016	5000	Fair
	2017	20,000	Fair
	2020	150,000	Fair
	2021	100,000	Fair
Denny Creek	2017	5000	Good
	2018	2000	Good
Black Sands Spring	2021	150,000	Fair

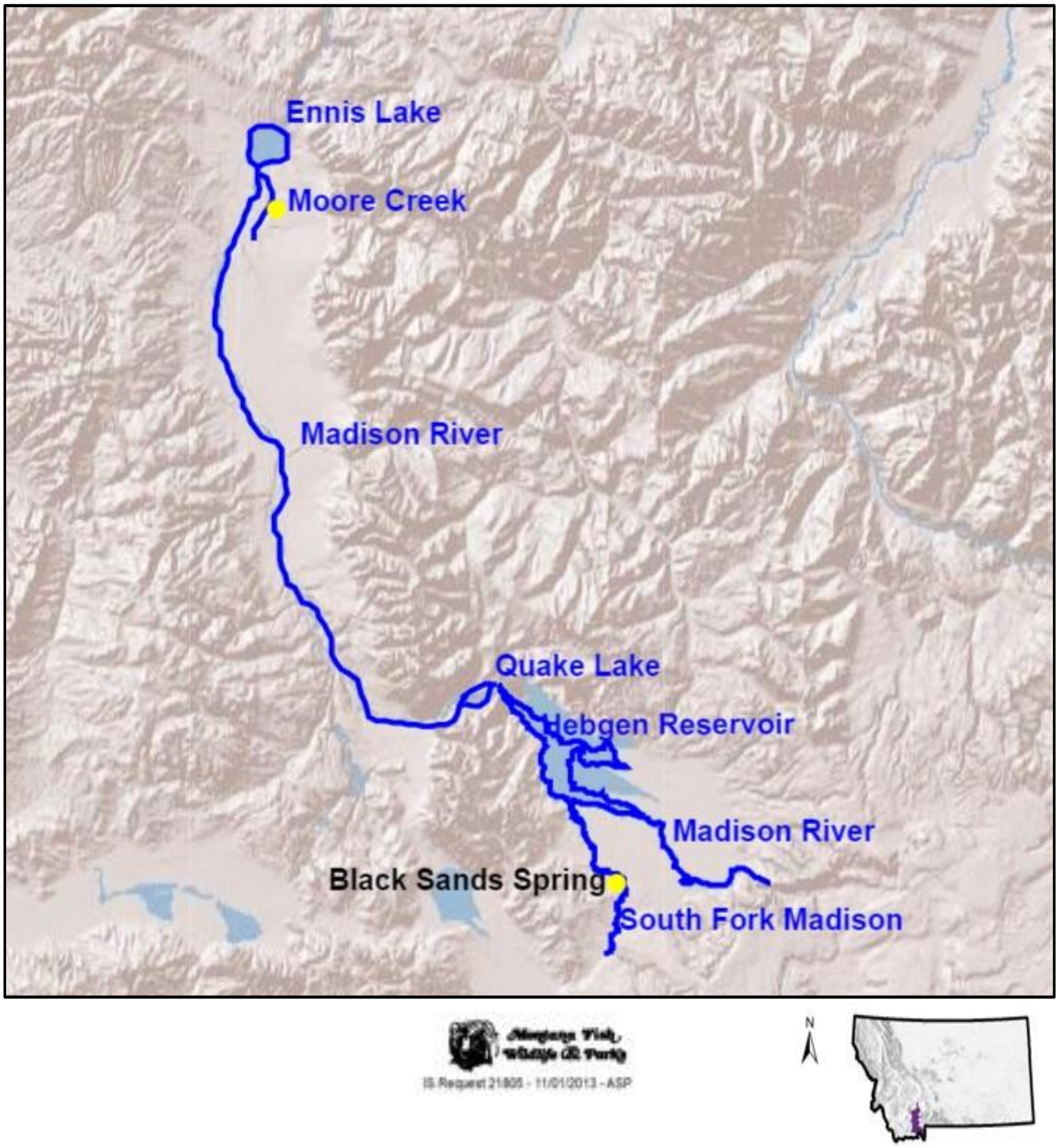

Figure 15. 2021 Arctic Grayling introduction sites Moore and Blaine Springs creeks.

North Fork Spanish Creek-Chiquita Lake: Funding was granted for the construction of a fish barrier on the North Fork Spanish Creek in the Gallatin Drainage by the MadTAC in 2018. The intent of the North Fork Spanish Creek project was to remove non-native trout from 17 miles of stream habitat and two alpine lakes with the intent to reestablish WCT and Arctic Grayling. Typically, funds are restricted to projects in the Madison Drainage; however, an exception to the allocation of funding was made because of limited opportunities and the difficulties of establishing Arctic Grayling populations within the Madison River Basin.

In 2019, Chiquita Lake was treated with the fish toxicant CFT Legumine to remove non-native fishes. The toxicant was applied to the waters of Chiquita Lake from a raft by a two-person crew. The raft was rowed in a grid pattern across the lake while chemical was dispersed from a plastic pesticide tank equipped with a small electric pump. The pump moved the chemical through an array of perforated hoses that were suspended below the water surface. Complete removal was confirmed through the use of gillnets and environmental DNA (eDNA) sampling in 2021. FWP restocked Chiquita Lake with 3666 Arctic Grayling fry of primarily Madison ancestry in 2021. This population will be monitored and managed to ensure it meets long-term conservation goals.

Westslope Cutthroat Trout: FWP's Statewide Fisheries Management Plan calls for the protection and reintroduction of WCT with less than 10% hybridization by non-native fish (i.e., conservation populations) to 20% of historically occupied waters (Montana Statewide Fisheries Management Program and Guide 2018). The MadTAC has granted funding to FWP to pursue these conservation efforts under Articles 408, 409, and 412 of the 2188 project FERC license. WCT PM&E activities in 2021 included completion of the Wall Creek fish migration barrier, assessment of the Tepee Creek barrier, wild fish transfer of WCT from Last Chance Creek into Ruby Creek, and feasibility assessments of Madison River tributaries for fish migration barrier construction to protect WCT conservation populations.

The Wall Creek barrier was constructed over a three-month period in fall 2021. A pre-construction meeting between FWP staff, project engineers, and contractors was held at the construction site on June 9, 2021 to discuss and agree upon material specifications and a construction schedule. Initially, barrier construction was to begin the third week of July, 2021. However, construction was delayed until August 23, 2021 to mitigate the risk of fire caused by construction activities during the extremely hot and dry conditions that predominated July and much of August. Construction site preparation, which consisted of primitive road improvements, clearing and grubbing, and rerouting of the stream to dewater the construction area was completed September 30 (Figure 16). Excavation of the barrier footprint was completed September 22 and the barrier footers were formed and poured the first week of October (Figure 17). Inclement weather during October prohibited concrete trucks from accessing the site because of deteriorating road conditions. Consequently, the final pour for the barrier structure did not occur until November 16 when the ground had frozen. Wall Creek was diverted back to its channel and over the completed barrier on November 22 (Figure 18). The Wall Creek barrier secures 7.5 miles of stream occupied by WCT of 95% genetic purity from invasion by non-native fishes. FWP will continue to monitor and report on the WCT population and performance of the barrier.

Figure 16. Road improvements and barrier site excavation on Wall Creek.

Figure 17. Wall Creek concrete barrier forms.

Figure 18. Completed Wall Creek barrier in November 2021.

Evaluation of the Tepee Creek fish barrier was equivocal and further analysis is needed to develop direction for this project. The Tepee Creek fish migration barrier is a natural waterfall that was improved to create a 12-ft vertical drop in 2019 by a Forest Service explosives crew. In 2020, FWP initiated evaluation of the Tepee Creek barrier to determine the potential for upstream fish passage. On July 15 and July 28, 2020, FWP collected 90 trout above the Tepee Creek barrier by electrofishing. Trout were marked with fin clip and released below the barrier. On July 21, 2021, FWP and CGNF personnel surveyed above the Tepee Creek barrier for the presence of marked fish that were released below the barrier in 2020. The survey was conducted by two crews using backpack electro-fishers in tandem. No marked fish were captured or observed; however, low water conductivity greatly reduced the electrofishing effectiveness and results of the survey do not definitively evaluate the effectiveness of the barrier to prevent upstream fish migration. FWP and CGNF have identified several issues that would likely compromise the effectiveness of the barrier. A pinch point occurs directly downstream of the barrier where debris could collect and cause the formation of a pool of sufficient depth for fish to jump over the barrier. Additionally, areas of reduced stream velocity and drop appear to be developing because of fractures in the rock on river left at the barrier site. WCT recovery efforts in Tepee Creek have been suspended pending a decision among partner agencies on the value of pursuing modifications to the barrier.

Creation of the Ruby Creek WCT population continued with translocation of fish from Last Chance Creek to improve genetic diversity. The Ruby Creek WCT restoration project was initiated in 2012 with the removal of nonnative Rainbow Trout. Ruby Creek was confirmed to be fishless by eDNA sampling in 2015. Since 2015, 94 genetically pure, aboriginal Madison WCT from McClure and Last Chance creeks have been introduced into Ruby Creek with 71 of those fish coming from McClure Creek. FWP and Yellowstone National Park personnel transferred 13 pure, aboriginal Madison WCT from Last Chance Creek to Ruby Creek on July 8, 2021. Fish from Last Chance Creek were collected with a backpack electro-fisher, measured to the nearest millimeter, and a fin clip for genetic analysis was taken from each fish. Fish were placed in an aerated cooler for transport after processing. Fish were placed in a net and allowed to acclimate to the temperature of the Ruby Creek for about 10 minutes. Although few Last Chance Creek trout have been introduced, their genetic contribution to the Ruby Creek population is greater than expected (Fuerstein 2021; Figure 19). FWP anticipates the 2021 introduction of Last Chance trout will continue to improve genetic diversity and increase the fitness of the population. FWP plans to evaluate Ruby Creek WCT distribution, reproductive status, and density in the summer of 2022.

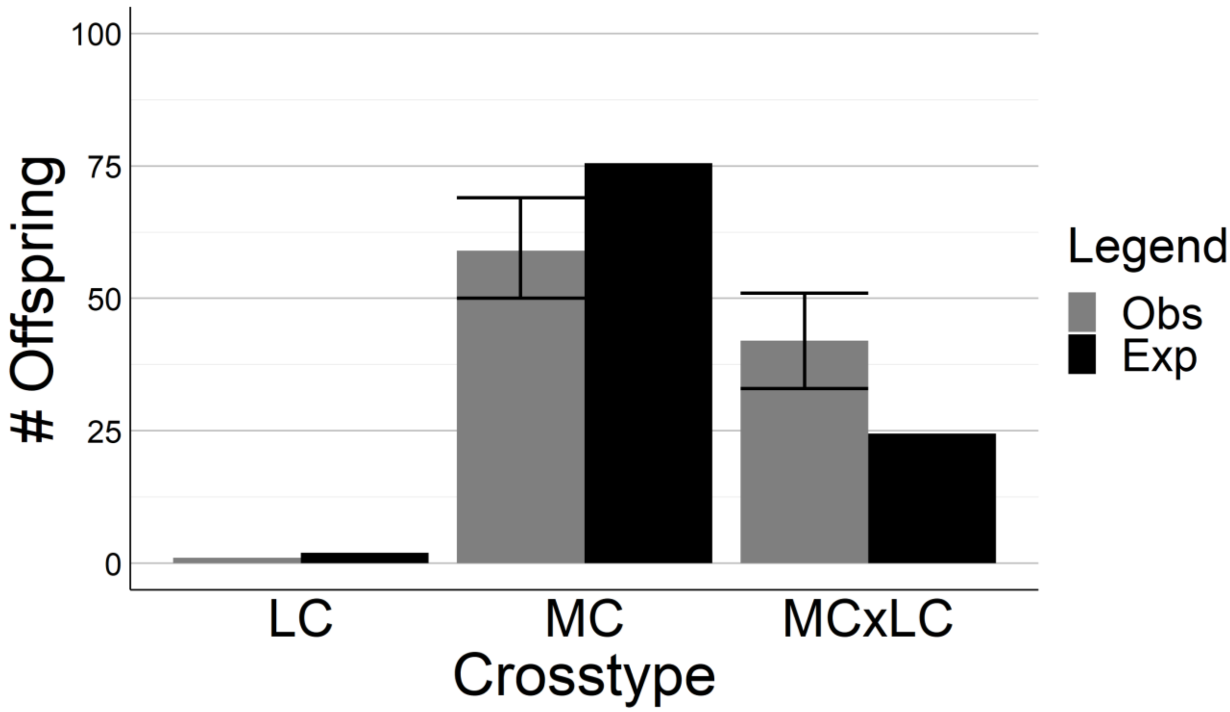

Figure 19. Ruby Creek Westslope Cutthroat Trout introduced into Ruby Creek and the genetic contribution of donors. Gray bars are the observed frequencies of offspring by crosstype. Black bars are the expected frequencies. Error bars are 95% confidence intervals (Feurstein 2021).

Article 409- 3) Fish habitat enhancement both in mainstem and tributary streams: Previous and potential future habitat enhancement activities in the mainstem Madison River and its tributaries were evaluated in 2022. The influence of habitat features (boulders, islands, side channels) in the mainstem Madison River on fish abundances were evaluated using arial imagery and historic electrofishing data. We found no evidence that addition of boulder and side channels will influence overall abundances of Madison River trout > 10”; however, increasing side channel or island density may increase abundances of large trout > 16”. Riparian enhancement on South Meadow Creek shows continued willow recruitment. Habitat restoration in the upper reaches of O’dell Creek between 2005 and 2009 narrowed stream channels, increased stream sinuosity, lowered streambank elevation, and increased stream channel water surface elevations. It appears these restoration activities ultimately enhanced conditions for and increased abundance of large adult fish after initially improving abundances of younger fish. These assessments are described in more detail below.

Associations between Madison River habitat types and fish abundances: The influence of habitat features (boulders, islands, side channels) in the mainstem Madison River on fish abundances was evaluated using arial imagery and historic electrofishing data. Addition of boulders or other mainstem habitat features have been routinely suggested to improve Madison River trout abundances. Habitat or cover (e.g., boulders, large woody debris, undercut banks) have been correlated to trout abundance (Binns and Eiserman 1979; Varley and Gresswell 1988; Molony 2001). Cover provides refuge from predators as well as thermal and velocity heterogeneity. To determine the potential benefits of addition of mainstem habitat features to the Madison River, FWP examined the effects of three habitat covariates (boulders, islands, and side channels) on trout abundances for fish ≥ 10” and ≥ 16” in the Pine Butte, Varney, and Norris sections. Twenty years of data (2000-2020) for each of the monitoring reaches was sorted by sub-stops. Sub-stops were pooled for analysis if sub-stops were combined in some years. For example, if sub-stop A was consistently stopped at but B was often passed by, then A and B were pooled and considered one sub-stop each year. Abundance estimates for fish ≥ 10” and ≥ 16” within each sub-stop were calculated using Chapman’s estimator to initially assess variation among years and sub-stops. Covariates within sub-stops were enumerated using satellite imagery provided by Google Earth (Figure 20). We counted boulders and measured side channel length and main channel length using the measurement tools provided in the Google Earth program. Total channel length (*TCL*) was calculated by adding the main channel length (MCL) of each sub-stop section to side channel length (*SCL*) of each sub-stop section $TCL = MCL + SCL$. Densities (habitat feature/mile) for each covariate were calculated by dividing the number of observed features by *TCL*, $habitat\ feature/mile = \#\ of habitat\ features\ with\ in\ a\ sub\ stop/TCL$. These metrics were sorted and compiled for use in a statistical model to determine covariate effects on abundances (Table 2).

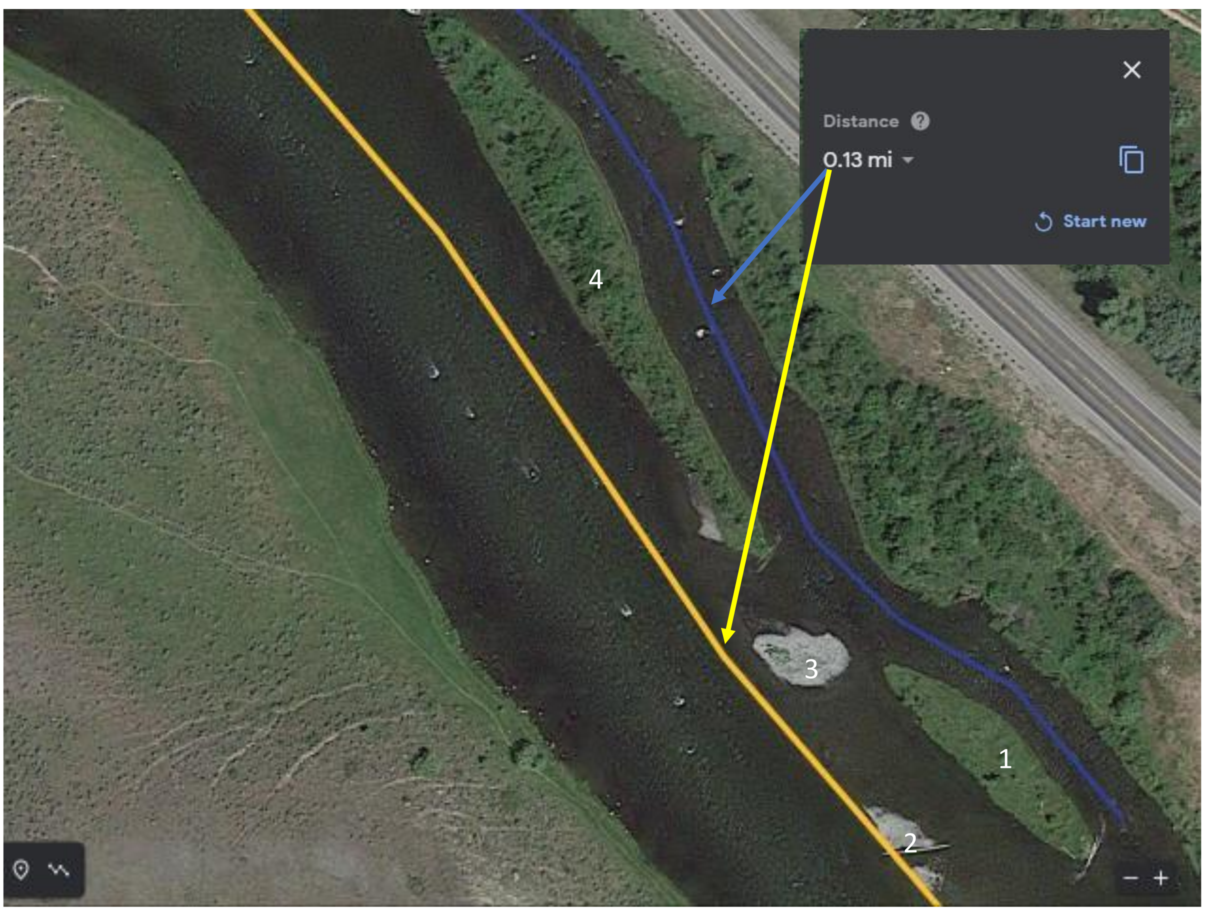

Figure 20. Satellite imagery from Google Earth used to determine channel lengths and covariate densities within sub sections of long-term Madison monitoring reaches. The blue line represents a side channel

length measurement, the yellow line a main channel measurement, and the numbers are identified islands within the reach.

Our modeling approach was focused on assessing the influence of stream characteristics (boulder, side-channel, and island density) on fish abundance while allowing for extra variation from random year effects and a robust negative binomial model for fish abundance. The complex model structure we used (a count-based model estimated from mark-recapture data for multiple sections within stream reaches) had variation at multiple levels, and we would like to highlight the inference available from our model as a series of questions:

1) How much do trout abundances vary through time within sections and within reaches?
2) What is the relationship between stream characteristics and fish abundance, and how does this relationship change between stream reaches?
3) After accounting for variation in abundance due to variation in stream characteristics, how much do fish abundances vary among years within reaches, and among years between reaches (i.e., otherwise unexplained variation)?

Moreover, we ran separate models for two length groups (> 10 inches and > 16 inches), which allowed us to add an additional question:

4) How do the above relationships change between length groups?

The model had two key components: a model to estimate fish abundances using the mark-recapture data, and a model for the estimated fish abundance as a function of stream characteristics (boulder density, islands, and side channels). The mark-recapture data were based on single-pass electrofishing sampling: data were collected by making marking and recapture runs on the right bank, left bank, and center of the channel, respectively and analyzed by treating them as single mark and recapture runs. We treated the marking and recapture runs as two independent sampling events with an identical probability of detection, which we estimated using a binomial model for the number of marked fish captured during the second sampling event (i.e., recapture run). We then estimated fish abundance by modeling it as a binomial random variable assuming the number of fish on the marking run and total fish on the recapture run as replicated observations, i.e., we inflated the number of fish caught on the marking and recapture runs by the estimated detection probability. We used a simple structure for the probabilities of detection: each reach (i.e., Varney, Pine Butte, and Norris) had an independent overall mean probability of detection, and yearly variation in detection probabilities was incorporated using random-effects unique to each reach (i.e., year random effects were not shared between reaches).

We used a negative binomial model for the model for fish abundance as a function of environmental covariates, a flexible count-based model that was able to accommodate more variation in abundance than a simple Poisson model. Using a log-link, we modeled the expected number of fish in each section within each reach using a section-specific overall intercept (corresponding to the estimated number of fish with all covariates held to their mean value), a random year effect specific to each reach (i.e., a random year effect applied to all sections within each reach), reach-specific regression coefficients (i.e., all sections within a reach responded similarly to variation in the covariate), and an offset for the length of the section within the reach.

Initial data exploration and model results indicated that the correlation between side channel and island density was substantial enough to affect inference on regression coefficients. Therefore, we constructed two separate models: a model of fish abundance as a function of boulder density and island density, and a model of fish abundance as a function of boulder density and side-channel density.

Notably, our model had several substantial limitations due to the relationship between the available data and the inference required. First, the relationship(s) between stream covariates and fish abundance (accounting for differences in section length) was assumed to be the same for every section within a reach, i.e., the proportional impact of variation in stream covariates was the same for each section, a simplification required because stream covariates did not vary through time. Second, the random effects of year on fish abundance were shared across sections within reaches (i.e., different sections did not have unique yearly random effects), which assumed that year effects applied equally to all sections within reaches. Finally, the probability of detection for the mark-recapture component of the model was assumed to be constant within reaches and years (i.e., no among-section variation in detection probability, only variation among years and reaches), which we justified after initial modeling efforts suggested very little among-section variation in detection within years.

The complex hierarchical structure of our model, combined with our intent to readily produce figures of the predicted relationships among stream covariates, fish abundance, reaches and sections using derived covariates, necessitated a Bayesian approach to estimation. We used the runjags package as an interface to the JAGS probabilistic programming language in the R environment. Each model was run for 250,000 iterations with 4 chains, with the first 50,000 samples discarded as the adaptation and burn-in phase, and the resulting chain thinned by a factor of 40 (due to memory constraints and autocorrelation issues), resulting in 20,000 iterations for inference. We used the medians and 95% highest posterior density intervals (a credible interval, or Bayesian version of a confidence interval) to summarize the posterior distributions of estimated parameters. All covariates were centered and scaled.

Boulder density was highest in Pine Butte (400 boulders/mile) followed by Norris (248 boulders/mile) and Varney (16 boulders/mile). Overall, the Varney Section had the greatest densities of islands and side channels with 10 islands/mile and 4 side channels/mile. Norris had the lowest island density among all sections with 4 islands/mile and similar side channel density to Pine Butte (Table 2).

Between sub-stops within the Pine Butte section, sub-stop C had the highest density of boulders (715 boulders/mile), sub-stop A had the highest density of islands (10 islands/mile), and sub-stop F had the highest density of side-channels (3 side channels/mile). In the Varney section, sub-stop

A-D had the greatest density of boulders (18 boulders/mile), and sub-stop G-I had the greatest density of islands and side channels at 19 islands/mile and 4 side channels/mile. Boulder densities were greatest among Norris sub-stops in sub-stop D (813 boulders/mile). Island densities were the highest in sub-stop E (8 islands/mile) and side channel densities greatest in sub-stop G (4 side channels/mile; Table 2).

Table 2. Stream habitat covariate densities in the Pine Butte, Varney, and Norris sections.

Section	Sub-stop	Stream length (miles)	#Boulders	Boulder density (#/mile)	#Islands	Island density (#/mile)	#Side channels	Side channel density (#/mile)
Pine Butte	A	1.2	281	244	12	10	2	2.0
Pine Butte	B	0.6	394	657	1	2.0	0	0.0
Pine Butte	C	0.4	293	715	2	5.0	0	0.0
Pine Butte	D	1.2	647	530	5	4.0	3	3.0
Pine Butte	E	0.5	123	256	1	2.0	1	2.0
Pine Butte	F	0.8	120	154	3	4.0	2	3.0
Totals		**4.6**	**1858**	**400**	**24**	**5.0**	**8**	**2.0**
Varney	A-D	4.8	85	18	30	6.0	18	4.0
Varney	E-F	2.7	36	13	16	6.0	7	3.0
Varney	G-I	3.1	50	16	58	19.0	12	4.0
Totals		**10.5**	**171**	**16**	**104**	**10.0**	**37**	**4.0**
Norris	A	0.6	40	67	3	5.0	1	2.0
Norris	B	0.6	148	269	1	2.0	0	0.0
Norris	C	0.4	153	373	1	2.0	0	0.0
Norris	D	0.5	374	813	0	0.0	0	0.0
Norris	E	0.7	184	252	6	8.0	1	1.0
Norris	F	0.4	102	237	0	0.0	0	0.0
Norris	G	0.5	118	227	4	8.0	2	4.0
Norris	H	0.7	97	139	4	6.0	2	3.0
Norris	I	0.9	96	108	4	5.0	3	3.0
Totals		**5.3**	**1312**	**248**	**23**	**4.0**	**9**	**2.0**

The abundance of trout showed considerable variation among length groups, among section sub-stops, within sub-stops, and among years (Table 3; Figures 21 and 22). Within section variation in abundance of > 10” and > 16” trout across sub-stops and years were lowest in Pine Butte and highest in the Varney section; however, sub-stop abundances differed among years in each section. For the Pine Butte section, across all section sub-stops and years the abundance of trout > 10” varied from a minimum of 501 [414, 651] in 2004 to 1526 [1193, 1904] in 2009. Among section sub-stops, the standardized ranges (the difference between the maximum estimated abundance and the minimum estimated abundance divided by the mean estimated abundance across years; higher values indicate more substantial swings in abundance around the long-term average) across years had a minimum of 0.66 and a maximum of 0.81. For trout > 16” the abundances across all section sub-stops and years varied from a minimum of 104 [69, 160] in 2010 to a maximum of 495 [353, 649] in 2013, and standardized ranges had a minimum of 0.63 and a maximum of 1.04. For the Varney section, across all section sub-stops and years the abundance of trout > 10” varied from a minimum of 623 [524, 765] in 2017 to 3440 [2999, 3908] in 2007, and the standardized ranges had a minimum of 0.97 and a maximum of 1.00. For trout > 16” the abundances across all section sub-stops and years varied from a minimum of 256 [185, 312] in 2017 to a maximum of 871 [716, 1089] in 2002, and standardized ranges had a minimum of 0.61 and a maximum of 0.77. It is noteworthy that the Varney Section required considerably more consolidation of sub-stops than other sections to make comparisons among years. For the Norris Section, abundance of trout > 10” varied from a minimum of 498 (95% credible interval = [391, 623]) in 2000 to 2177 [1826, 2557] in 2001, and standardized ranges had a minimum of 0.47 and a maximum of 1.01. For trout > 16”, the abundances across all section sub-stops and years varied from a minimum of 74 [47, 137] in 2002 to a maximum of 359 [264, 495] in 2003, and standardized ranges had a minimum of 0.49 and a maximum of 0.89.

Table 3. Estimated trout abundances by length group, year, section, and sub-stop.

Length group (inches)	Year	Section	Sub-stop	Estimated abundance	2.5%	97.5%
> 10	2000	Norris	A	759	612	927
> 10	2000	Norris	B	618	497	766
> 10	2000	Norris	C	552	434	678
> 10	2000	Norris	D	498	391	623
> 10	2000	Norris	E	638	515	796
> 10	2000	Norris	F	834	671	1014
> 10	2000	Norris	G	545	436	679
> 10	2000	Norris	H	1000	821	1225
> 10	2000	Norris	I	871	724	1087
> 10	2001	Norris	A	1026	829	1213
> 10	2001	Norris	B	1011	847	1238
> 10	2001	Norris	C	1094	931	1354
> 10	2001	Norris	D	1250	1057	1521
> 10	2001	Norris	E	1209	1046	1511
> 10	2001	Norris	F	2177	1826	2557
> 10	2001	Norris	G	1119	919	1331
> 10	2001	Norris	H	2150	1837	2574
> 10	2001	Norris	I	1686	1421	2012
> 10	2002	Norris	A	763	628	968
> 10	2002	Norris	B	1017	823	1245

> 10	2002	Norris	C	766	626	960
> 10	2002	Norris	D	1091	903	1356
> 10	2002	Norris	E	1217	989	1481
> 10	2002	Norris	F	1193	946	1427
> 10	2002	Norris	G	659	531	826
> 10	2002	Norris	H	1316	1109	1653
> 10	2002	Norris	I	842	662	1021
> 10	2003	Norris	A	1123	940	1326
>10	2003	Norris	B	1080	906	1278
> 10	2003	Norris	C	827	692	992
> 10	2003	Norris	D	1002	822	1162
> 10	2003	Norris	E	1250	1052	1473
> 10	2003	Norris	F	1123	959	1351
> 10	2003	Norris	G	803	682	978
> 10	2003	Norris	H	1880	1581	2172
> 10	2003	Norris	I	1203	986	1387
> 10	2004	Norris	A	860	708	1028
> 10	2004	Norris	B	682	551	818
> 10	2004	Norris	C	645	512	762
> 10	2004	Norris	D	841	653	955
> 10	2004	Norris	E	848	676	982
> 10	2004	Norris	F	962	798	1154
> 10	2004	Norris	G	793	645	938
> 10	2004	Norris	H	1427	1175	1662
> 10	2004	Norris	I	740	612	905
> 10	2007	Norris	A	919	762	1076
> 10	2007	Norris	B	1001	826	1163
> 10	2007	Norris	C	696	593	856
> 10	2007	Norris	D	1069	915	1280
> 10	2007	Norris	E	1146	976	1359
> 10	2007	Norris	F	1545	1336	1828
> 10	2007	Norris	G	709	603	866
> 10	2007	Norris	H	1432	1197	1656
> 10	2007	Norris	I	1076	889	1251
> 10	2008	Norris	A	710	621	863
> 10	2008	Norris	B	712	595	832
> 10	2008	Norris	C	664	564	790
> 10	2008	Norris	D	850	700	966
> 10	2008	Norris	E	946	772	1062
> 10	2008	Norris	F	1272	1103	1490
> 10	2008	Norris	G	741	635	881
> 10	2008	Norris	H	986	845	1157
> 10	2008	Norris	I	1025	895	1220
> 10	2010	Norris	A	903	762	1069
> 10	2010	Norris	B	858	707	1001

> 10	2010	Norris	C	699	587	839
> 10	2010	Norris	D	830	711	1004
> 10	2010	Norris	E	1022	863	1203
> 10	2010	Norris	F	1499	1295	1762
> 10	2010	Norris	G	673	567	813
> 10	2010	Norris	H	1216	1039	1431
> 10	2010	Norris	I	1414	1177	1612
> 10	2000	Pine Butte	A	846	668	1062
> 10	2000	Pine Butte	B	949	737	1161
> 10	2000	Pine Butte	C	1038	810	1272
> 10	2000	Pine Butte	D	1058	855	1343
> 10	2000	Pine Butte	E	919	722	1142
> 10	2000	Pine Butte	F	737	570	918
> 10	2001	Pine Butte	A	1064	818	1344
> 10	2001	Pine Butte	B	847	681	1140
> 10	2001	Pine Butte	C	1033	799	1323
> 10	2001	Pine Butte	D	1257	975	1594
> 10	2001	Pine Butte	E	838	658	1101
> 10	2001	Pine Butte	F	762	594	1001
> 10	2002	Pine Butte	A	948	759	1185
> 10	2002	Pine Butte	B	758	637	1003
> 10	2002	Pine Butte	C	761	594	939
> 10	2002	Pine Butte	D	1106	890	1377
> 10	2002	Pine Butte	E	672	534	850
> 10	2002	Pine Butte	F	759	621	981
> 10	2003	Pine Butte	A	557	436	734
> 10	2003	Pine Butte	B	682	524	873
> 10	2003	Pine Butte	C	579	427	726
> 10	2003	Pine Butte	D	784	604	1008
> 10	2003	Pine Butte	E	570	445	748
> 10	2003	Pine Butte	F	534	411	701
> 10	2004	Pine Butte	A	645	506	786
> 10	2004	Pine Butte	B	781	635	973
> 10	2004	Pine Butte	C	563	454	713
> 10	2004	Pine Butte	D	657	550	850
> 10	2004	Pine Butte	E	650	513	797
> 10	2004	Pine Butte	F	501	414	651
> 10	2005	Pine Butte	A	901	645	1208
> 10	2005	Pine Butte	B	841	610	1143
> 10	2005	Pine Butte	C	805	620	1163
> 10	2005	Pine Butte	D	1012	780	1449
> 10	2005	Pine Butte	E	808	572	1079
> 10	2005	Pine Butte	F	861	584	1087
> 10	2006	Pine Butte	A	906	741	1117
> 10	2006	Pine Butte	B	926	780	1170

> 10	2006	Pine Butte	C	924	727	1095
> 10	2006	Pine Butte	D	1183	972	1441
> 10	2006	Pine Butte	E	912	764	1144
> 10	2006	Pine Butte	F	735	604	924
> 10	2007	Pine Butte	A	1042	851	1260
> 10	2007	Pine Butte	B	1056	868	1280
> 10	2007	Pine Butte	C	1093	914	1343
> 10	2007	Pine Butte	D	1399	1155	1682
> 10	2007	Pine Butte	E	1075	891	1314
> 10	2007	Pine Butte	F	903	752	1109
> 10	2008	Pine Butte	A	908	738	1058
> 10	2008	Pine Butte	B	904	743	1060
> 10	2008	Pine Butte	C	691	554	811
> 10	2008	Pine Butte	D	1184	1023	1434
> 10	2008	Pine Butte	E	1134	935	1315
> 10	2008	Pine Butte	F	868	713	1023
> 10	2009	Pine Butte	A	860	675	1107
> 10	2009	Pine Butte	B	901	714	1165
> 10	2009	Pine Butte	C	877	677	1115
> 10	2009	Pine Butte	D	1526	1193	1904
> 10	2009	Pine Butte	E	871	676	1108
> 10	2009	Pine Butte	F	906	705	1147
> 10	2010	Pine Butte	A	998	831	1221
> 10	2010	Pine Butte	B	980	805	1181
> 10	2010	Pine Butte	C	1139	934	1358
> 10	2010	Pine Butte	D	946	781	1157
> 10	2010	Pine Butte	E	713	588	883
> 10	2010	Pine Butte	F	770	619	922
> 10	2011	Pine Butte	A	1006	800	1298
> 10	2011	Pine Butte	B	795	627	1039
> 10	2011	Pine Butte	C	886	699	1142
> 10	2011	Pine Butte	D	936	757	1240
> 10	2011	Pine Butte	E	674	535	901
> 10	2011	Pine Butte	F	754	609	1004
> 10	2012	Pine Butte	A	1116	888	1375
> 10	2012	Pine Butte	B	1050	829	1289
> 10	2012	Pine Butte	C	1346	1031	1590
> 10	2012	Pine Butte	D	1259	987	1521
> 10	2012	Pine Butte	E	1091	851	1327
> 10	2012	Pine Butte	F	1179	953	1468
> 10	2013	Pine Butte	A	1314	1097	1589
> 10	2013	Pine Butte	B	1167	977	1422
> 10	2013	Pine Butte	C	1278	1046	1514
> 10	2013	Pine Butte	D	1315	1107	1592
> 10	2013	Pine Butte	E	1172	983	1420

> 10	2013	Pine Butte	F	1176	929	1349
> 10	2014	Pine Butte	A	1225	1029	1502
> 10	2014	Pine Butte	B	1303	1095	1587
> 10	2014	Pine Butte	C	1246	1017	1476
> 10	2014	Pine Butte	D	1409	1188	1717
> 10	2014	Pine Butte	E	1066	891	1308
> 10	2014	Pine Butte	F	1153	934	1360
> 10	2015	Pine Butte	A	1127	918	1334
> 10	2015	Pine Butte	B	1211	1014	1468
> 10	2015	Pine Butte	C	1271	1064	1525
> 10	2015	Pine Butte	D	1116	912	1328
> 10	2015	Pine Butte	E	1302	1131	1619
> 10	2015	Pine Butte	F	890	742	1088
> 10	2017	Pine Butte	A	792	634	975
> 10	2017	Pine Butte	B	760	630	966
> 10	2017	Pine Butte	C	867	715	1087
> 10	2017	Pine Butte	D	1065	869	1313
> 10	2017	Pine Butte	E	1160	918	1372
> 10	2017	Pine Butte	F	888	731	1107
> 10	2000	Varney	A-D	2711	2370	3178
> 10	2000	Varney	E-F	1157	982	1357
> 10	2000	Varney	G-I	1736	1477	2001
> 10	2001	Varney	A-D	2500	2186	2950
> 10	2001	Varney	E-F	978	819	1148
> 10	2001	Varney	G-I	1276	1079	1494
> 10	2002	Varney	A-D	2519	2157	2977
> 10	2002	Varney	E-F	1160	980	1386
> 10	2002	Varney	G-I	1735	1490	2067
> 10	2003	Varney	A-D	2633	2295	3120
> 10	2003	Varney	E-F	1070	922	1292
> 10	2003	Varney	G-I	1466	1261	1745
> 10	2004	Varney	A-D	2460	2092	2990
> 10	2004	Varney	E-F	1042	853	1260
> 10	2004	Varney	G-I	1390	1193	1729
> 10	2005	Varney	A-D	2242	1878	2671
> 10	2005	Varney	E-F	803	650	970
> 10	2005	Varney	G-I	1230	1047	1526
> 10	2006	Varney	A-D	3309	2852	3823
> 10	2006	Varney	E-F	1589	1383	1889
> 10	2006	Varney	G-I	1964	1702	2312
> 10	2007	Varney	A-D	3440	2999	3908
> 10	2007	Varney	E-F	1579	1363	1814
> 10	2007	Varney	G-I	2068	1814	2397
> 10	2009	Varney	A-D	1451	1272	1753
> 10	2009	Varney	E-F	755	619	869

> 10	2009	Varney	G-I	1118	948	1309
> 10	2010	Varney	A-D	1461	1221	1704
> 10	2010	Varney	E-F	651	529	765
> 10	2010	Varney	G-I	943	826	1160
> 10	2011	Varney	A-D	1814	1470	2169
> 10	2011	Varney	E-F	681	559	858
> 10	2011	Varney	G-I	779	620	961
> 10	2012	Varney	A-D	2589	2208	3162
> 10	2012	Varney	E-F	1182	963	1413
> 10	2012	Varney	G-I	1517	1265	1845
> 10	2013	Varney	A-D	2357	1991	2835
> 10	2013	Varney	E-F	995	848	1240
> 10	2013	Varney	G-I	1515	1287	1850
> 10	2014	Varney	A-D	2868	2531	3411
> 10	2014	Varney	E-F	1081	920	1285
> 10	2014	Varney	G-I	1585	1344	1844
> 10	2015	Varney	A-D	1624	1418	1935
> 10	2015	Varney	E-F	781	664	931
> 10	2015	Varney	G-I	1217	1033	1420
> 10	2017	Varney	A-D	1209	988	1423
> 10	2017	Varney	E-F	623	524	765
> 10	2017	Varney	G-I	884	753	1087
> 10	2018	Varney	A-D	1815	1535	2130
> 10	2018	Varney	E-F	737	608	876
> 10	2018	Varney	G-I	1058	913	1292
> 10	2019	Varney	A-D	1597	1292	1845
> 10	2019	Varney	E-F	694	586	861
> 10	2019	Varney	G-I	993	828	1204
> 10	2020	Varney	A-D	1747	1474	2072
> 10	2020	Varney	E-F	835	709	1018
> 10	2020	Varney	G-I	910	749	1086
> 16	2000	Norris	A	151	105	215
> 16	2000	Norris	B	84	49	125
> 16	2000	Norris	C	106	67	153
> 16	2000	Norris	D	122	75	172
> 16	2000	Norris	E	191	125	258
> 16	2000	Norris	F	223	156	309
> 16	2000	Norris	G	227	164	325
> 16	2000	Norris	H	298	214	412
> 16	2000	Norris	I	204	146	293
> 16	2001	Norris	A	106	69	160
> 16	2001	Norris	B	81	52	129
> 16	2001	Norris	C	108	70	159
> 16	2001	Norris	D	131	92	201
> 16	2001	Norris	E	134	84	197

> 16	2001	Norris	F	230	159	320
> 16	2001	Norris	G	192	125	262
> 16	2001	Norris	H	327	239	466
> 16	2001	Norris	I	198	133	278
> 16	2002	Norris	A	80	46	140
> 16	2002	Norris	B	74	47	137
> 16	2002	Norris	C	85	59	159
> 16	2002	Norris	D	132	82	220
> 16	2002	Norris	E	154	99	256
> 16	2002	Norris	F	122	75	214
> 16	2002	Norris	G	134	89	232
> 16	2002	Norris	H	179	114	298
> 16	2002	Norris	I	131	79	222
> 16	2003	Norris	A	163	109	224
> 16	2003	Norris	B	139	89	191
> 16	2003	Norris	C	104	71	163
> 16	2003	Norris	D	181	127	257
> 16	2003	Norris	E	232	169	329
> 16	2003	Norris	F	189	131	273
> 16	2003	Norris	G	198	139	282
> 16	2003	Norris	H	359	264	495
> 16	2003	Norris	I	240	161	322
> 16	2004	Norris	A	149	97	207
> 16	2004	Norris	B	102	67	152
> 16	2004	Norris	C	113	75	169
> 16	2004	Norris	D	116	77	172
> 16	2004	Norris	E	134	91	205
> 16	2004	Norris	F	169	108	234
> 16	2004	Norris	G	209	145	294
> 16	2004	Norris	H	280	207	403
> 16	2004	Norris	I	154	100	221
> 16	2007	Norris	A	133	88	199
> 16	2007	Norris	B	168	110	235
> 16	2007	Norris	C	123	81	187
> 16	2007	Norris	D	195	132	278
> 16	2007	Norris	E	212	156	323
> 16	2007	Norris	F	272	187	376
> 16	2007	Norris	G	183	129	280
> 16	2007	Norris	H	244	172	362
> 16	2007	Norris	I	246	166	342
> 16	2008	Norris	A	85	54	134
> 16	2008	Norris	B	82	52	127
> 16	2008	Norris	C	106	71	158
> 16	2008	Norris	D	149	94	203
> 16	2008	Norris	E	194	127	263

> 16	2008	Norris	F	189	131	270
> 16	2008	Norris	G	193	143	288
> 16	2008	Norris	H	183	124	263
> 16	2008	Norris	I	193	131	272
> 16	2010	Norris	A	127	94	198
> 16	2010	Norris	B	111	78	170
> 16	2010	Norris	C	142	98	205
> 16	2010	Norris	D	220	147	292
> 16	2010	Norris	E	270	184	353
> 16	2010	Norris	F	277	202	385
> 16	2010	Norris	G	175	122	251
> 16	2010	Norris	H	245	165	331
> 16	2010	Norris	I	264	180	347
> 16	2000	Pine Butte	A	162	102	296
> 16	2000	Pine Butte	B	196	121	328
> 16	2000	Pine Butte	C	139	89	246
> 16	2000	Pine Butte	D	228	149	386
> 16	2000	Pine Butte	E	161	104	272
> 16	2000	Pine Butte	F	110	74	208
> 16	2001	Pine Butte	A	273	166	401
> 16	2001	Pine Butte	B	236	154	375
> 16	2001	Pine Butte	C	191	114	284
> 16	2001	Pine Butte	D	261	162	388
> 16	2001	Pine Butte	E	152	100	252
> 16	2001	Pine Butte	F	160	103	261
> 16	2002	Pine Butte	A	298	215	464
> 16	2002	Pine Butte	B	248	156	350
> 16	2002	Pine Butte	C	146	86	213
> 16	2002	Pine Butte	D	223	161	358
> 16	2002	Pine Butte	E	178	107	250
> 16	2002	Pine Butte	F	138	85	204
> 16	2003	Pine Butte	A	189	126	300
> 16	2003	Pine Butte	B	263	173	387
> 16	2003	Pine Butte	C	159	103	247
> 16	2003	Pine Butte	D	229	142	326
> 16	2003	Pine Butte	E	142	93	222
> 16	2003	Pine Butte	F	131	82	203
> 16	2004	Pine Butte	A	291	204	403
> 16	2004	Pine Butte	B	374	255	490
> 16	2004	Pine Butte	C	134	94	208
> 16	2004	Pine Butte	D	228	156	322
> 16	2004	Pine Butte	E	160	116	245
> 16	2004	Pine Butte	F	184	115	242
> 16	2005	Pine Butte	A	309	203	477
> 16	2005	Pine Butte	B	220	137	341

> 16	2005	Pine Butte	C	146	94	247
> 16	2005	Pine Butte	D	241	154	371
> 16	2005	Pine Butte	E	143	89	231
> 16	2005	Pine Butte	F	177	116	278
> 16	2006	Pine Butte	A	303	198	418
> 16	2006	Pine Butte	B	213	152	328
> 16	2006	Pine Butte	C	142	100	228
> 16	2006	Pine Butte	D	252	187	392
> 16	2006	Pine Butte	E	207	141	296
> 16	2006	Pine Butte	F	159	97	220
> 16	2007	Pine Butte	A	287	198	422
> 16	2007	Pine Butte	B	196	137	309
> 16	2007	Pine Butte	C	159	106	247
> 16	2007	Pine Butte	D	229	154	341
> 16	2007	Pine Butte	E	148	98	226
> 16	2007	Pine Butte	F	157	99	225
> 16	2008	Pine Butte	A	346	241	459
> 16	2008	Pine Butte	B	249	178	351
> 16	2008	Pine Butte	C	179	118	246
> 16	2008	Pine Butte	D	234	165	331
> 16	2008	Pine Butte	E	180	123	253
> 16	2008	Pine Butte	F	136	91	197
> 16	2009	Pine Butte	A	254	156	387
> 16	2009	Pine Butte	B	241	159	389
> 16	2009	Pine Butte	C	185	121	297
> 16	2009	Pine Butte	D	257	181	432
> 16	2009	Pine Butte	E	122	73	202
> 16	2009	Pine Butte	F	140	83	220
> 16	2010	Pine Butte	A	321	240	449
> 16	2010	Pine Butte	B	277	205	388
> 16	2010	Pine Butte	C	258	185	350
> 16	2010	Pine Butte	D	227	169	326
> 16	2010	Pine Butte	E	104	69	160
> 16	2010	Pine Butte	F	159	105	214
> 16	2011	Pine Butte	A	365	265	502
> 16	2011	Pine Butte	B	266	193	377
> 16	2011	Pine Butte	C	192	137	278
> 16	2011	Pine Butte	D	222	155	309
> 16	2011	Pine Butte	E	139	93	199
> 16	2011	Pine Butte	F	146	97	208
> 16	2012	Pine Butte	A	351	266	559
> 16	2012	Pine Butte	B	328	226	482
> 16	2012	Pine Butte	C	278	197	422
> 16	2012	Pine Butte	D	293	203	442
> 16	2012	Pine Butte	E	259	168	368

> 16	2012	Pine Butte	F	235	159	346
> 16	2013	Pine Butte	A	495	353	649
> 16	2013	Pine Butte	B	359	268	504
> 16	2013	Pine Butte	C	318	227	429
> 16	2013	Pine Butte	D	297	210	405
> 16	2013	Pine Butte	E	240	167	327
> 16	2013	Pine Butte	F	229	159	313
> 16	2014	Pine Butte	A	467	359	647
> 16	2014	Pine Butte	B	363	287	524
> 16	2014	Pine Butte	C	292	213	400
> 16	2014	Pine Butte	D	386	290	532
> 16	2014	Pine Butte	E	246	169	327
> 16	2014	Pine Butte	F	177	128	260
> 16	2015	Pine Butte	A	412	296	553
> 16	2015	Pine Butte	B	381	276	513
> 16	2015	Pine Butte	C	329	232	438
> 16	2015	Pine Butte	D	262	187	369
> 16	2015	Pine Butte	E	292	214	405
> 16	2015	Pine Butte	F	228	158	311
> 16	2017	Pine Butte	A	330	237	444
> 16	2017	Pine Butte	B	257	171	335
> 16	2017	Pine Butte	C	238	180	339
> 16	2017	Pine Butte	D	344	262	481
> 16	2017	Pine Butte	E	242	178	338
> 16	2017	Pine Butte	F	217	163	313
> 16	2000	Varney	A-D	713	576	856
> 16	2000	Varney	E-F	343	262	411
> 16	2000	Varney	G-I	546	443	661
> 16	2001	Varney	A-D	737	579	858
> 16	2001	Varney	E-F	360	296	456
> 16	2001	Varney	G-I	541	428	647
> 16	2002	Varney	A-D	871	716	1089
> 16	2002	Varney	E-F	486	406	632
> 16	2002	Varney	G-I	712	600	912
> 16	2003	Varney	A-D	708	593	879
> 16	2003	Varney	E-F	422	335	512
> 16	2003	Varney	G-I	577	472	708
> 16	2004	Varney	A-D	824	670	1012
> 16	2004	Varney	E-F	406	319	502
> 16	2004	Varney	G-I	488	414	643
> 16	2005	Varney	A-D	730	585	898
> 16	2005	Varney	E-F	340	271	435
> 16	2005	Varney	G-I	519	407	639
> 16	2006	Varney	A-D	857	702	1019
> 16	2006	Varney	E-F	447	365	547

> 16	2006	Varney	G-I	598	489	721
> 16	2007	Varney	A-D	792	652	960
> 16	2007	Varney	E-F	440	363	547
> 16	2007	Varney	G-I	616	500	743
> 16	2009	Varney	A-D	740	604	892
> 16	2009	Varney	E-F	399	307	473
> 16	2009	Varney	G-I	547	446	669
> 16	2010	Varney	A-D	661	529	789
> 16	2010	Varney	E-F	345	277	431
> 16	2010	Varney	G-I	636	520	770
> 16	2011	Varney	A-D	506	403	732
> 16	2011	Varney	E-F	284	208	394
> 16	2011	Varney	G-I	321	241	459
> 16	2012	Varney	A-D	652	536	886
> 16	2012	Varney	E-F	358	266	459
> 16	2012	Varney	G-I	471	367	626
> 16	2013	Varney	A-D	494	395	655
> 16	2013	Varney	E-F	290	224	381
> 16	2013	Varney	G-I	426	330	551
> 16	2014	Varney	A-D	810	659	973
> 16	2014	Varney	E-F	366	298	462
> 16	2014	Varney	G-I	562	460	689
> 16	2015	Varney	A-D	633	486	752
> 16	2015	Varney	E-F	328	242	390
> 16	2015	Varney	G-I	456	364	569
> 16	2017	Varney	A-D	447	347	561
> 16	2017	Varney	E-F	256	185	312
> 16	2017	Varney	G-I	329	245	406
> 16	2018	Varney	A-D	650	518	788
> 16	2018	Varney	E-F	299	234	375
> 16	2018	Varney	G-I	443	360	558
> 16	2019	Varney	A-D	570	449	692
> 16	2019	Varney	E-F	288	230	369
> 16	2019	Varney	G-I	429	351	545
> 16	2020	Varney	A-D	777	644	953
> 16	2020	Varney	E-F	439	343	522
> 16	2020	Varney	G-I	455	362	563

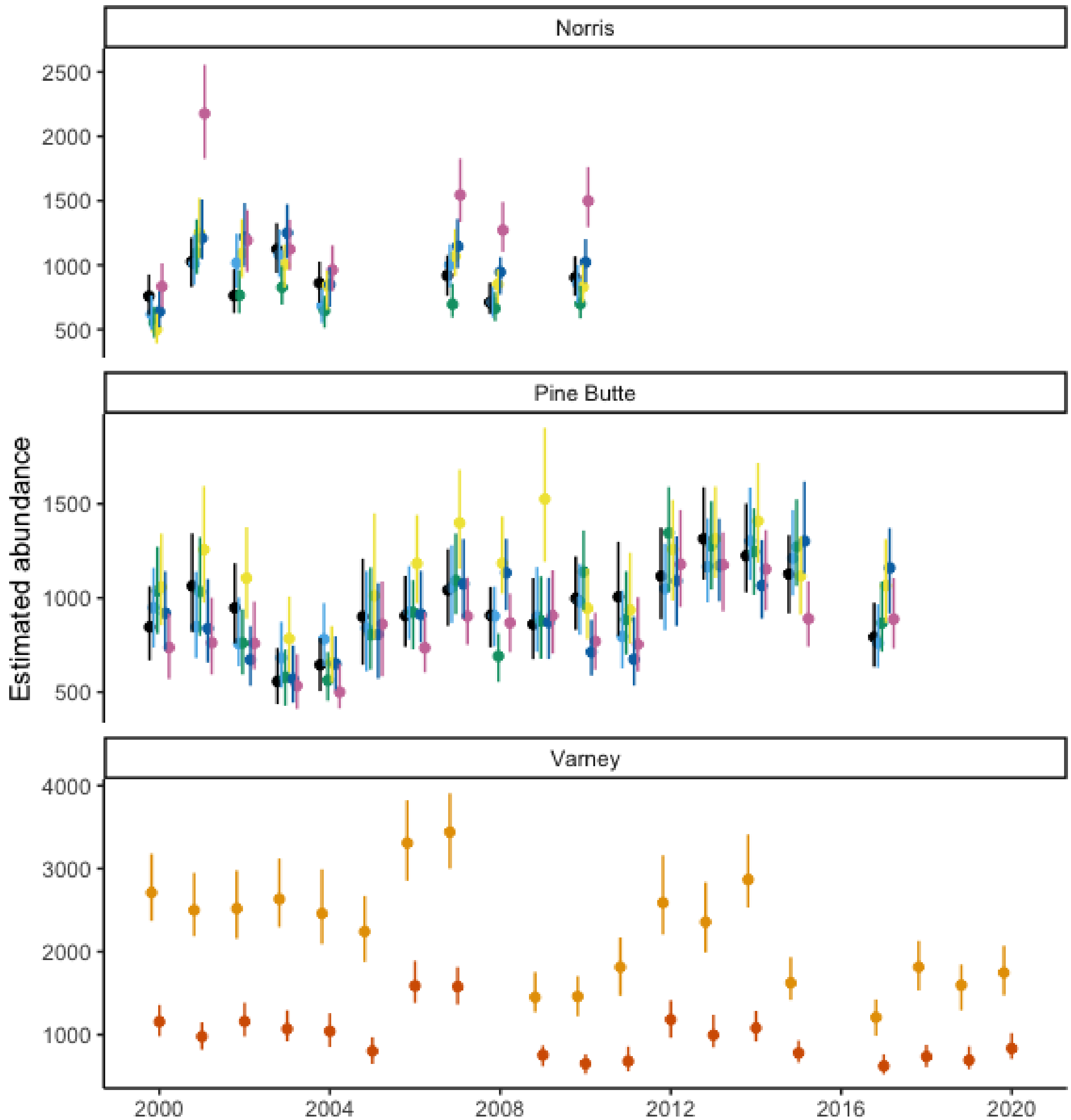

Figure 21. Estimated abundances of trout > 10” (brown trout and rainbow trout) in the Norris, Pine Butte, and Varney sections. Circles indicate the medians and lines indicate 95% credible intervals. Colors represent the different sub-stops in each section. Note the different scales on the y-axes in each panel.

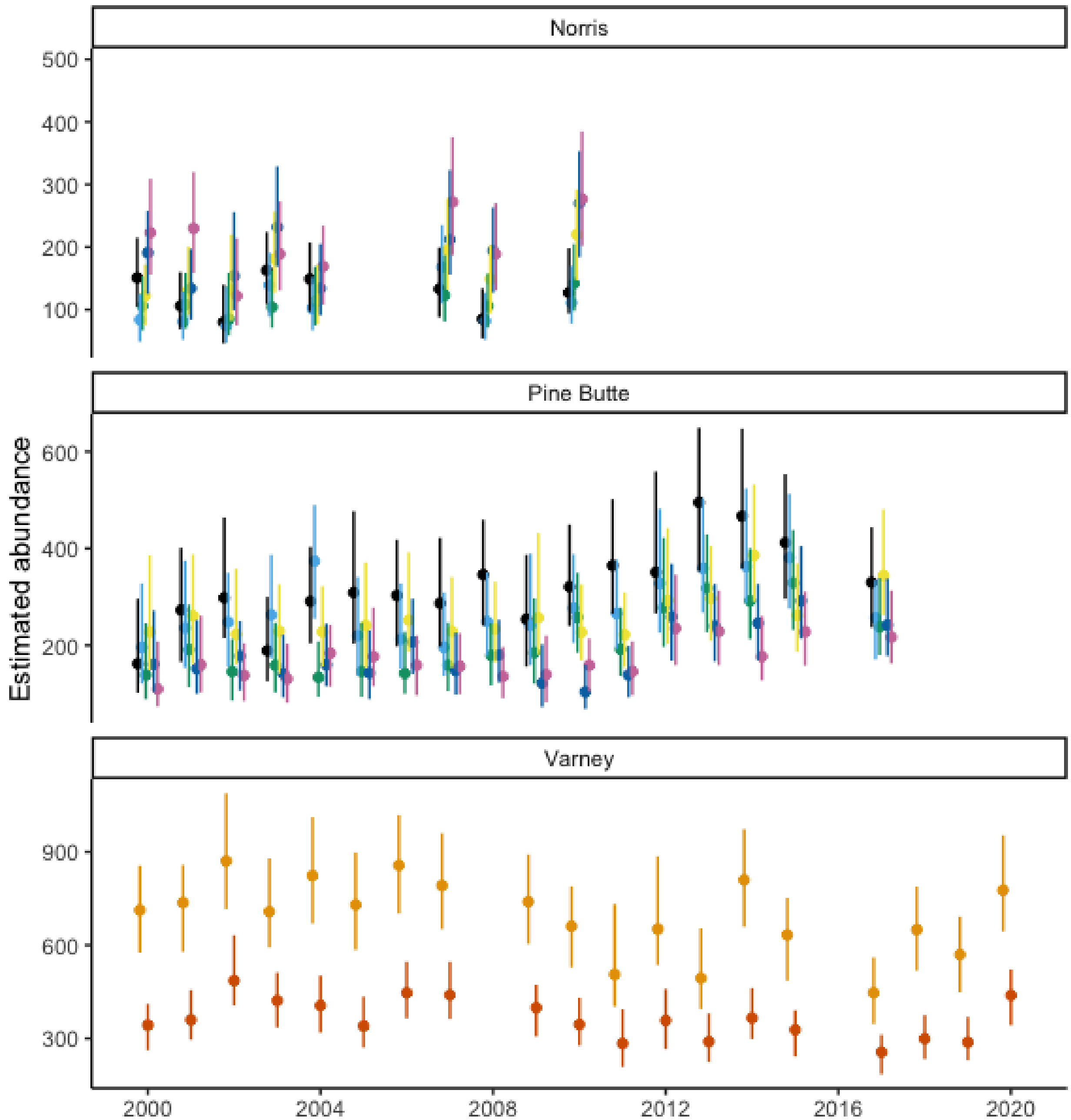

Figure 22. Estimated abundances of trout > 16" (Brown Trout and Rainbow Trout) in the Norris, Pine Butte, and Varney sections. Circles indicate the medians and lines indicate 95% credible intervals. The colors represent the different sub-stops in each section. Note the different scales on the y-axes in each panel.

Variation in trout abundances were not related to boulder densities; however, a suggestive positive relationship existed between abundance of trout > 16" and island and side channel densities. No evidence for an association between the abundance of trout > 10" and stream characteristics existed (Figure 23 and Figure 24). However, the credible intervals for the estimated coefficients for island density (-1.16 [-0.07, 0.04]) and side channel density (-0.004 [-

0.18, 0.18]) overlapped zero and prevented strong inference. We wanted to provide an easily-interpreted explanation for the effect of these covariates (log-scale estimates are hard to interpret), so we used the approximate posterior distributions to create predictions of how much trout abundance would vary in response to variation in stream characteristics, assuming average conditions. The lack of an estimated relationship translated into weak predictions of how much trout abundance would vary over the ranges of stream characteristics (Figure 23). In contrast, we found suggestive but inconclusive evidence (i.e., point estimates different than zero, but with credible intervals whose tails overlapped zero) that the abundance of trout > 16” was positively related to island density (0.02 [-0.02, 0.06]) and side channel density (0.11 [-0.07, 0.24]; Figure 23). These estimated relationships translated into predictions that suggested for the average sub-stop in the average year, predicted abundances could vary from 252 [121, 471] to 412 [180, 717] over the range of side channel density, and from 300 [134, 508] to 402 [153, 718] over the range of island density (Figure 25). However, the uncertainty in the point estimates translated into considerable uncertainty in these projections, and strong inference was not possible.

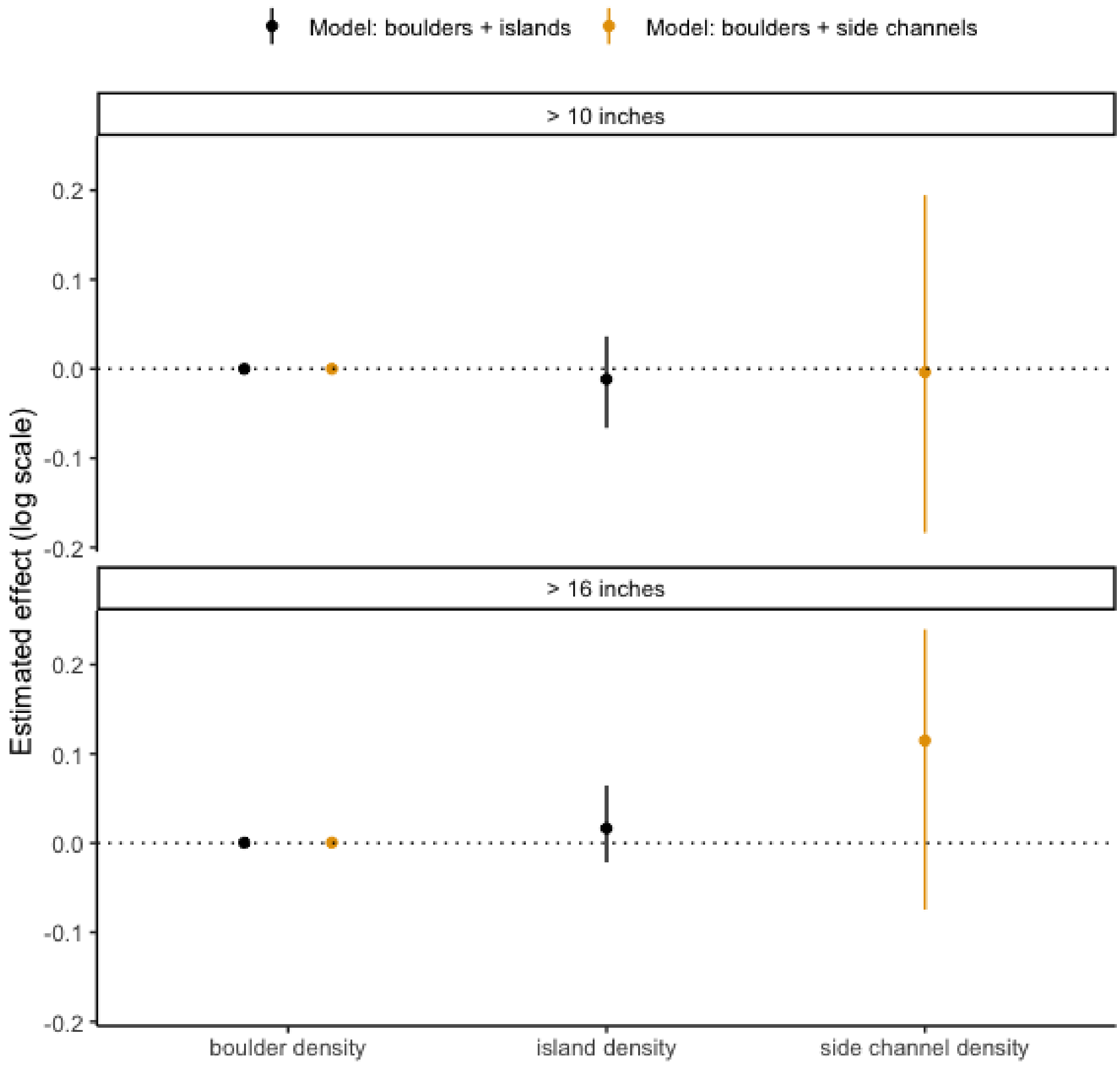

Figure 23. Estimated effects of physical characteristics of the waterbody on trout abundance (on the log scale) for the two length groups (> 10” and > 16”). Circles indicate medians and lines indicate the 95% credible intervals. Estimates greater than zero suggest abundances increase as the physical characteristics increase, estimates less than zero indicate abundances decreased as the physical covariates increase. Credible intervals that overlap zero indicate we cannot confidently claim relationships exist.

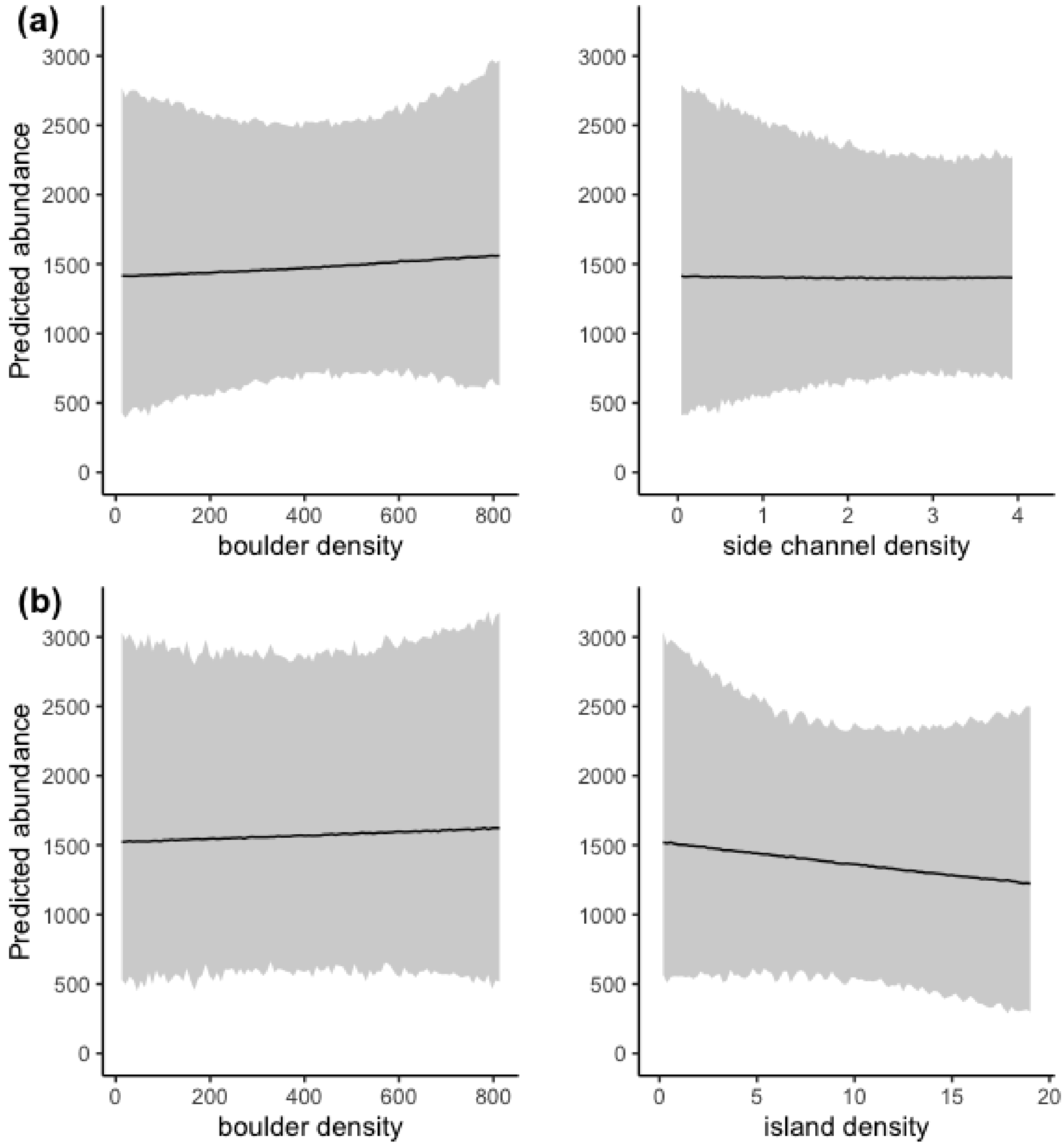

Figure 24. Estimated relationships between physical characteristics of the waterbody and abundances of trout > 10”. Lines indicate medians and ribbons indicate 95% credible intervals. Note the different scales on the y-axes in each panel. These predictions were made for an average year and represent an average across all sections and sub-stops.

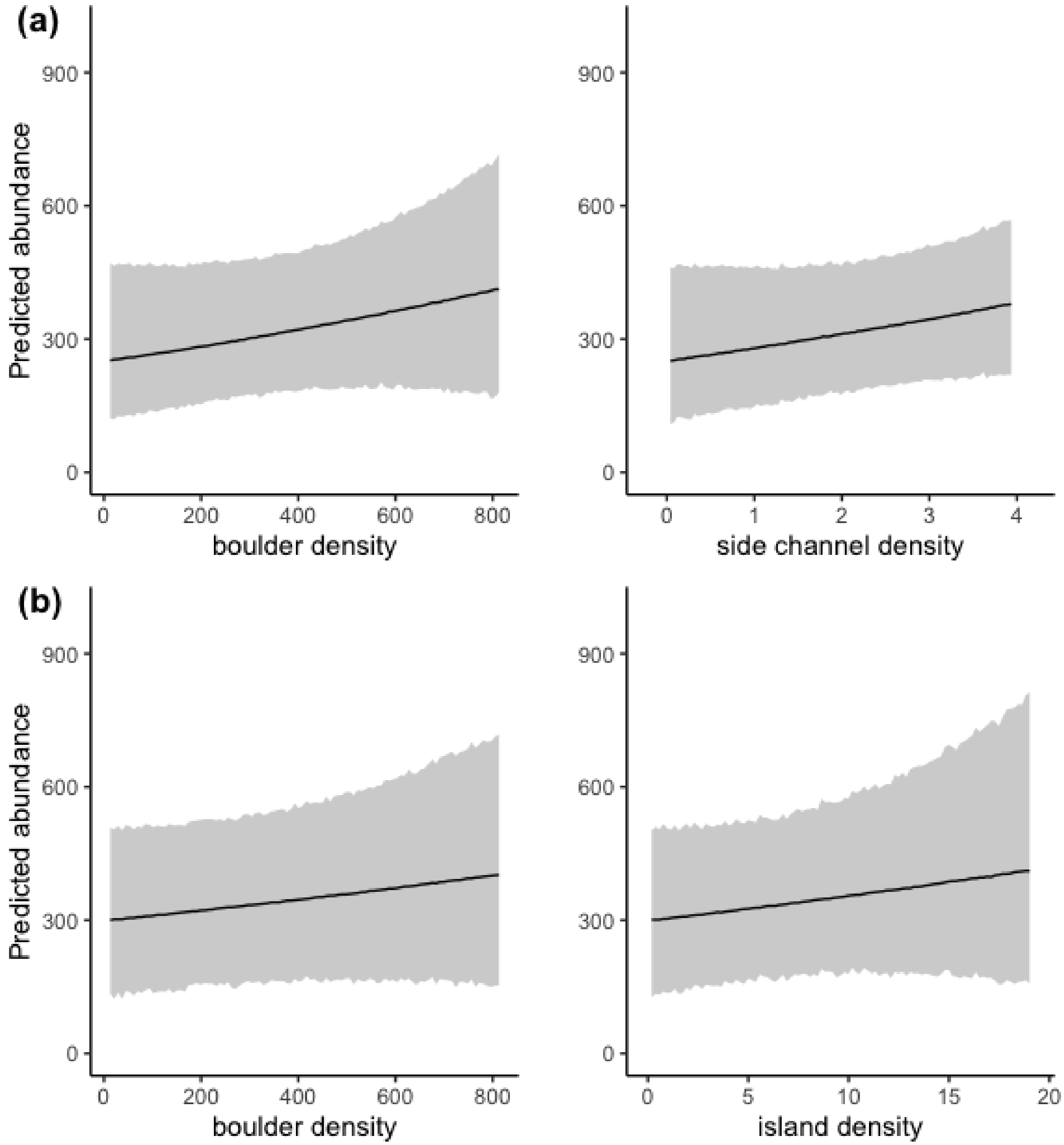

Figure 25. Estimated relationships between the physical characteristics of the waterbody and the abundances of trout > 16”. Lines indicate medians and ribbons indicate 95% credible intervals. Note the different scales on the y-axes in each panel. These predictions were made for an average year and represent an average across all sections and sub-stops.

Overall abundances of trout > 10” are influenced by unexplained among year variation; however, annual variation in factors other than physical features had a similar level of effect on abundance of trout > 16” as physical features. We incorporated a random effect of year on trout abundance for each section to estimate the significance of unmodeled variation in trout abundance (e.g., stream discharge and hydrograph, temperature, population structure, crew efficiency). We then took our approximate posterior distributions and created predictions of how those random effects translated into variation in abundance by assuming the average within-section among-sub-stop abundance. Although substantial uncertainty existed in the estimates (note the wide credible intervals; Figures 26 and 27), the results indicate that otherwise-unexplained variation captured as a yearly random effect is influencing trout abundances significantly for both length groups. For trout > 10”, predicted abundances among years varied from 1482 [876, 2220] to 2867 [1694, 4194] for the Norris Section, 551 [0, 1680] to 1348 [0, 4057] for the Pine Butte Section, and 902 [0, 2745] to 2243 [1382, 3355] for the Varney Section. For trout > 16”, predicted abundances among years varied from 285 [126, 495] to 413 [206, 683] for the Norris Section, 210 [86, 330] to 414 [196, 664] for the Pine Butte Section, and 313 [148, 492] to 576 [293, 934] for the Varney Section. Finally, we wanted to compare the variation in predicted abundances in response to stream characteristics for trout > 16” (recall we found no evidence for a relationship for trout > 10”) and otherwise-unexplained yearly variation to get a rough feel for the relative importance of the two model results (Figure 28). Results indicate nearly commensurate variation (i.e., yearly variation and stream characteristic variation are about several hundred fish across sections).

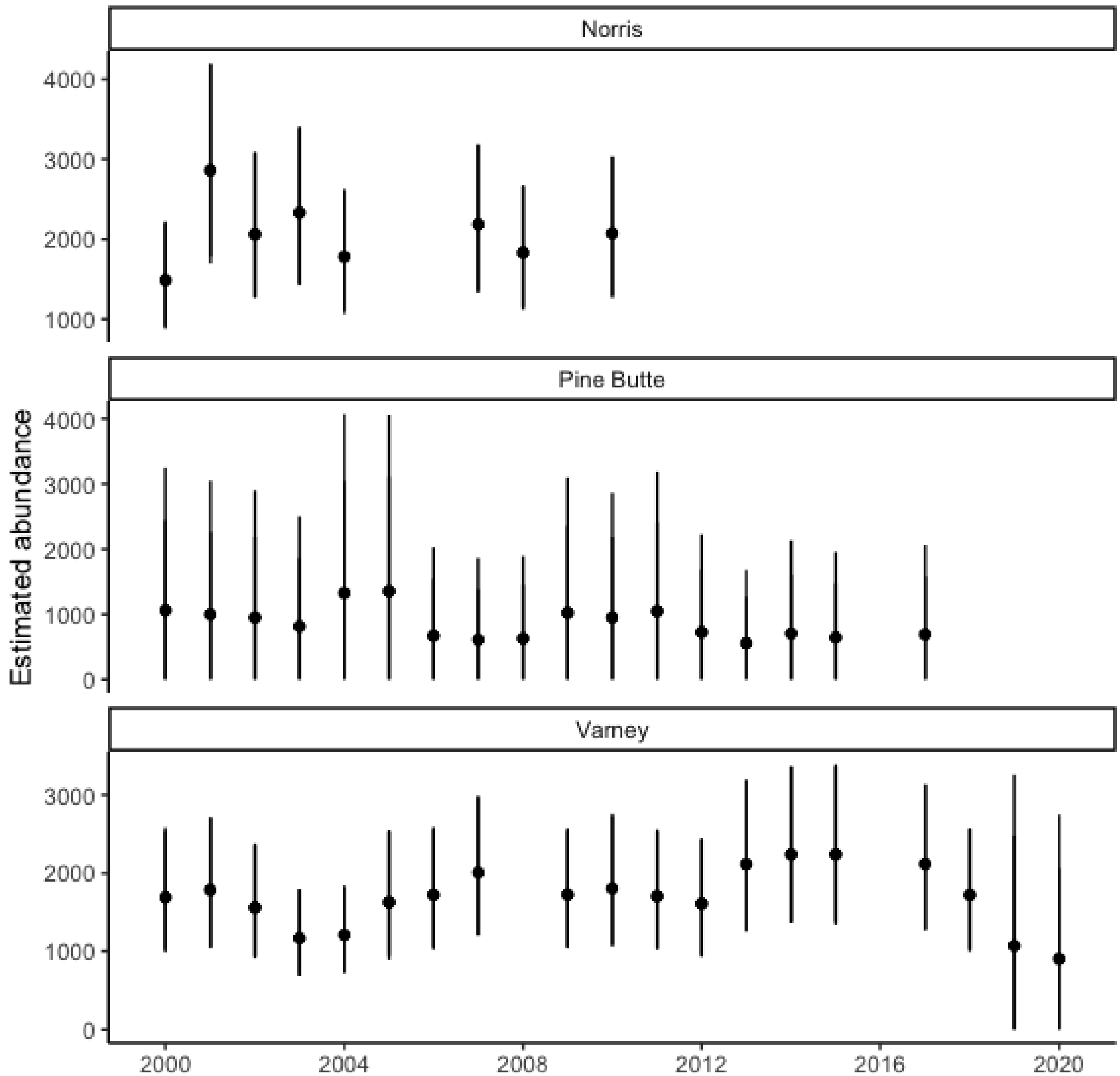

Figure 26. Predicted among-year variation in the abundance of trout > 10” for each section (predictions were made for the mean covariate values in each section). Circles are medians and lines are 95% credible intervals.

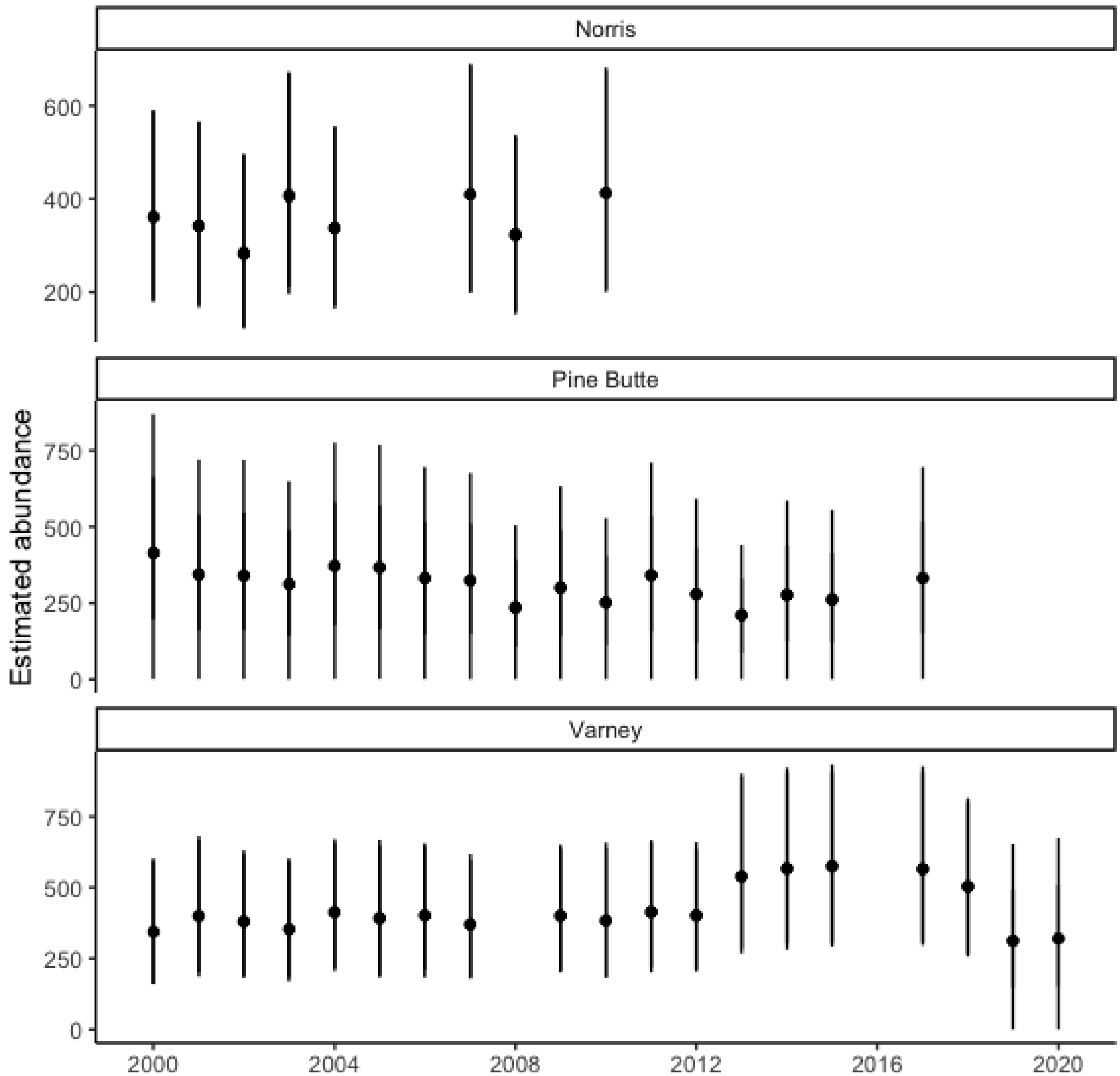

Figure 27. Predicted among-year variation in the abundance of trout > 16” for each section (predictions were made for the mean covariate values in each section). Circles are medians and lines are 95% credible intervals.

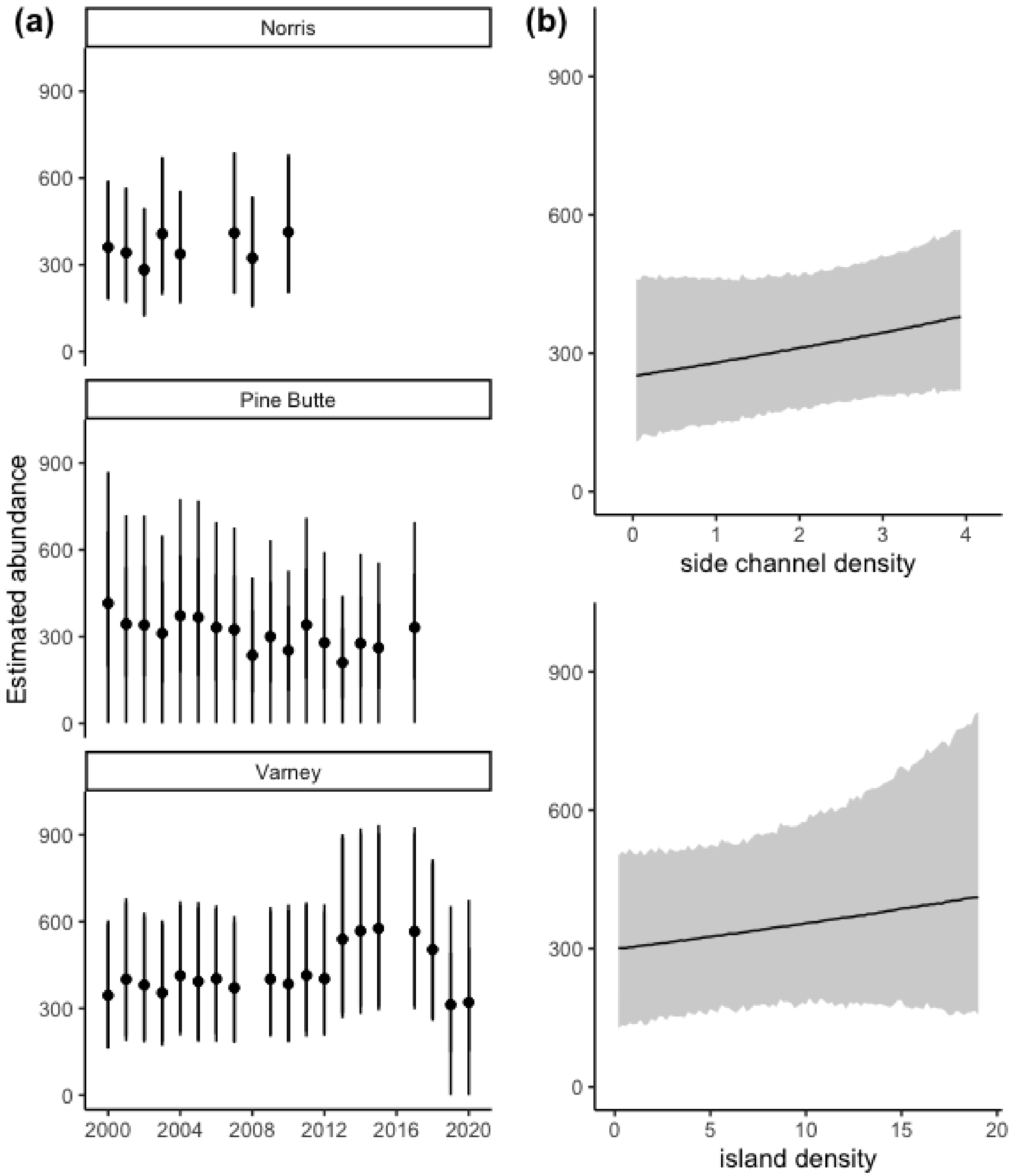

Figure 28. Observed (a) trout abundances in each Madison River sampling section and expected (b) trout abundances based on habitat characteristics combined across sections.

We found no evidence that addition of boulder and side channels will influence overall abundances of Madison River trout > 10”; however, increasing side channel or island density may increase abundances of large trout > 16”. Consistent within section variation among sub-stops across years suggests that factors other than year affects influence abundances; if abundances were related solely to year-specific effects standardized abundances would be similar within sections. However, the physical features we investigated had either no or an unclear effect on trout abundances. Boulder density, which has been suggested as a possible mitigative action to improve the fishery, had no effect on overall trout abundances or the abundance of large trout. Island and side channel density also had no effect on overall trout abundance, but there was a suggestive positive relationship with the abundance of large trout > 16”. Although islands and side channels most influenced large trout abundances, it is important to note that we did not investigate the effect of these features on trout < 10”; relative abundances of young-of-year and age-1 trout are commonly linked to complex habitats like side channels and high island density.

Inference about the effect of physical features on trout abundances was limited by the resolution of historic electrofishing data and the scale and observational nature of this assessment, limitations that precluded clear inference regarding the effect of these features. Abundances derived from electrofishing data are assumed to be homogeneous within a sub-stop and cannot account for the spatial distribution (and re-distribution among years) of fish within sub-stop and sections in response to physical drivers such as stream characteristics. Moreover, abundances estimated from small sample sizes are notoriously imprecise: this has the practical effect of conflating sampling variation (variation originating from the sampling process) with process variation (actual variation in the abundances of fish among sub-stops, sections and years) into a “noisy” representation of population dynamics. This problem of “noisy” abundance is amplified due to population size being the result of a complex interplay of biological mechanisms (i.e., the process variation component of variation in abundance results from variation in both reproduction and survival). These vital rates do not necessarily respond to environmental and intrinsic drivers of population demography in the same way; the classic example of which is the negative relationship between survival and reproduction when populations near carrying capacity in a density-dependent model. We suggest a clearer picture of the influence of extrinsic drivers on fish populations requires a better understanding of the vital rates that underly population dynamics, rather than the aggregated result of vital rates that is abundance. Our work indicates a multi-year monitoring program designed to estimate key reproduction and survival rates is required to improve our understanding. Moreover, we could improve our inference by incorporating experimental manipulations into the monitoring program; the inference available from the current observational study hinders our better understanding by conflating a wide series of unmodeled parameters into a very simple model structure to account for among-sub-stop variation. Experimental manipulation of stream characteristics would dramatically improve our inference by creating variation in stream characteristics within sub-stops, a notable characteristic lacking in the current study. Specifically, if the goal is to better understand the relative costs and benefits of island or side channel construction to the Madison River trout population we recommend side channel and/or island creation in a relatively simple reach (or set of reaches) and monitoring of vital rates of all age classes relative to one or more control sections for multiple years.

South Meadow Creek riparian enhancement: In 2011, FWP cosponsored a riparian fencing and off channel water development project with the Madison Conservation District to address water quality and degraded riparian and in-stream habitat conditions on South Meadow Creek, a tributary to Ennis Reservoir. Past grazing practices had largely eliminated the presence of riparian vegetation leaving streambanks unstable and in a highly erosive condition. Riparian vegetation has started to become re-established, and streambanks stabilized within much of the treatment area. However, 1500 ft of channel within the 2011 treatment reach was not exhibiting the rate of recovery observed in the downstream portion of the treatment reach. The 1500 ft section had previously been straightened for water delivery purposes. The straightening of the channel resulted in abandonment of the historic floodplain, channel widening, and loss of instream habitat. In 2019, FWP implemented restoration activities that re-established floodplain connectivity, appropriate channel dimensions, and in-stream habitat (Figure 29 and Figure 30). New willow and riparian vegetation within the project reach colonized the reach. FWP anticipates this will provide shade and moderate summer water temperatures during reduced flows associated with irrigation withdrawals. FWP did not sample the South Meadow Creek fish assemblage through the project reach in 2021.

Figure 29. South Meadow Creek Fall 2021. Circles and ellipses are new willow growth.

Figure 30. New Willow growth along South Meadow Creek restoration in July 2021.

O'Dell Creek habitat enhancement: O'Dell Creek is a spring-fed tributary of the Madison River that originates southeast of Ennis. The stream flows north for about 13 miles to its confluence with the mainstem Madison about 1.5 miles downstream of Ennis and 5.0 miles above Ennis Reservoir (Figure 31). From 2005 to 2009, stream restoration efforts on O'Dell Creek narrowed stream channels, increased stream sinuosity, lowered streambank elevation, and increased stream channel water surface elevations. FWP monitored responses in Brown Trout abundance and size structure, as Brown Trout are the predominant game fish species in the restoration area. Additional restoration work has occurred downstream of the monitoring area annually. Monitoring occurred in the headwater reaches of O'Dell Creek in April 2021.

Six monitoring sections were established throughout the restoration area (Figure 31; Table 4). In 2021, fish were collected by a crew of three to four individuals using a mobile anode electro-fisher in all sections except the O'Dell Spring North section where a backpack electro-fisher was used. C/f was used in all sampling sections to determine relative abundance and was calculated as the number of fish per mile. In 2021, FWP completed three mark-recapture abundance estimates among sections to assess catchability and determine whether comparisons of C/f among years and reaches were valid. Most fish were weighed (g) and measured (mm). However, some fish in the Old Middle Channel and O'Dell Spring North sections were released after recording species. Biomass per mile was calculated by multiplying the mean weight observed by the calculated C/f for each individual section where weights were taken. Age was assigned as age-0: 0-150 mm, age-1: 151-277 mm, age-2: 278-404 mm, age-3 or older: ≥ 404 mm based on Inter-Fluve Inc (1989).

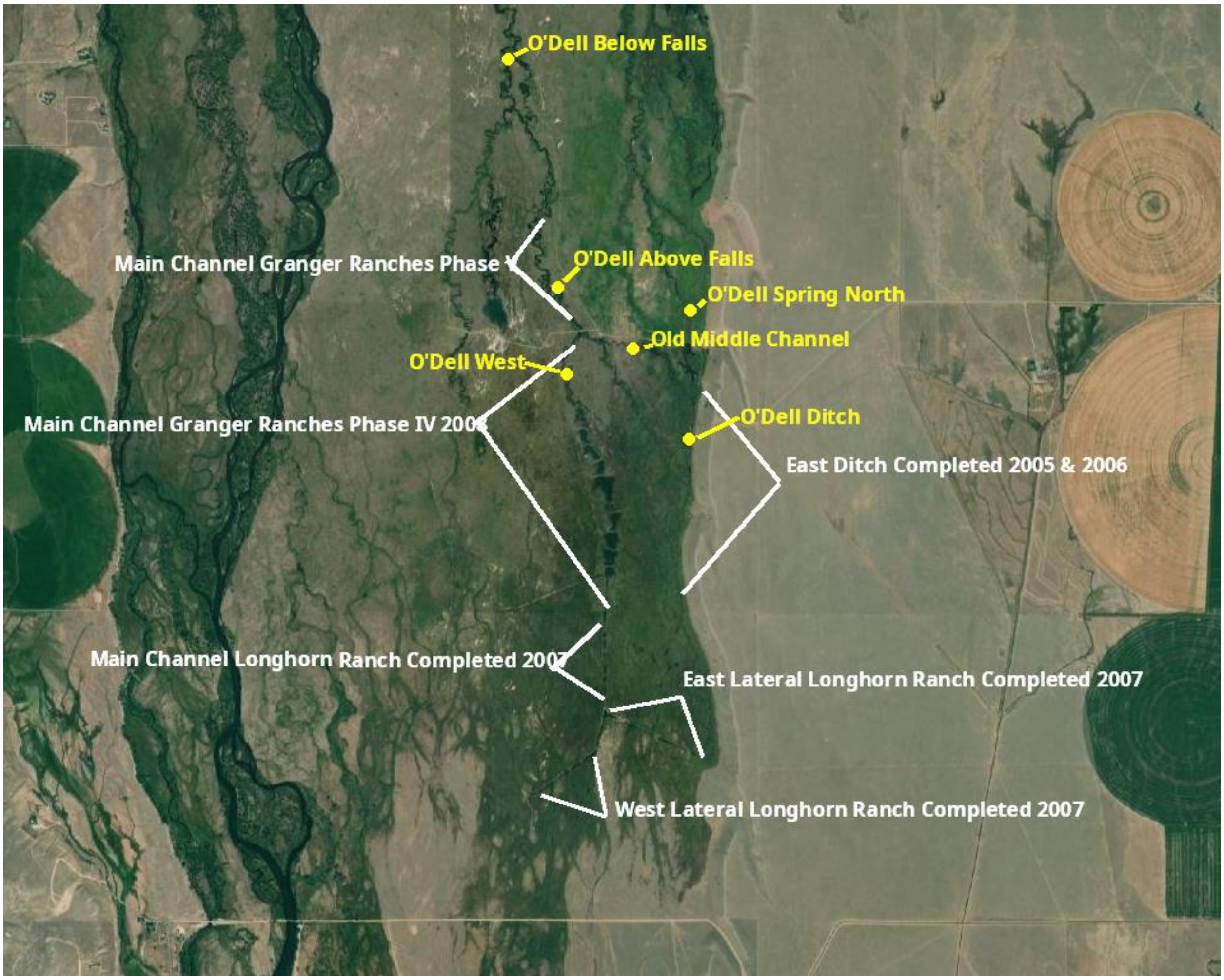

Figure 31. Aerial view of O'Dell Spring Creek Restoration sites (white) and FWP sampling sites (yellow).

Table 4. Stream restoration actions on fish monitoring sites at O'Dell Creek, 2005 - 2012.

Site	Stream channel modification	Section length/ft	Years
O'Dell Ditch	Backfilled	500	2005
O'Dell Spring North	Increase in stream discharge, no physical modifications	500	2005-2010
Old Middle	Historic channel reconnected and reconstructed	500	2005-2012
O'Dell West	Channel narrowed & deepened, increase in stream discharge	500	2005
Above Falls	Increase in stream discharge, stream channel restoration	1000	2005-2010
Below Falls	Increase in stream discharge, no physical modifications	1000	2005-2008

Fewer and larger fish were captured in 2021 but limited inferences can be made about relative abundance. Catchability ranged 33% to 68%, so comparisons of relative abundance among sections and years should be made cautiously. For example, a C/f of 1000 fish/mile could describe a point estimate of abundance between 1470 and 3030 fish/mile. Unless there was at least a two-fold difference in C/f among years, inference is speculative. However, relative abundances (Tables 5, 6, 7, and 8) in 2021 were lower than those in previous years. Overall, the reduction can largely be attributed to a decline in juvenile trout (Figures 32, 33, and 34). Relative abundance of age-2 and older fish was greater than that observed in all previous sampling events in the Above Falls and Old Middle Channel sections and was similar to abundances observed prior to restoration in the Below Falls section (Tables 5, 6, and 7).

Median lengths and weights were statistically significantly different among years in all sections, although some differences may not be biologically significant. In general, the Above Falls and Below Falls sections (Tables 5 and 6; Figures 35 and 36) had larger fish by length (mm) and weight (g) in 2021, and a reduction in median fish length was observed in the Old Middle (Table 7; Figure 36) section, while fish size in O'Dell Spring North (Table 8) section showed no significant change.

In summary, it appears that restoration activities, such as deepening and narrowing the channel as well as increasing discharge, have ultimately enhanced conditions for and increased abundance of large adult fish after initially improving abundances of younger fish.

Table 5. Median lengths and weights (interquartile range), biomass, and relative abundances for Above Falls Section. Asterisks denote pre-restoration monitoring. Median lengths and weights with different superscripts are significantly different among years (α =0.05).

Year	Median length (mm)	Median weight (g)	C/f (fish/mile)	C/*f* (fish/mile) by age group 0+	1+	2+	> 2+	Biomass (kg/mile)
2005*	180^{a} (109)	73^{a} (170)	1063	374	389	274	26	181
2006*	174^{a} (71)	77^{a} (130)	1916	316	1258	300	42	291
2007	178^{a} (79)	54^{a} (100)	543	137	374	32	0	54
2008	264^{b} (157)	213^{b} (290)	837	174	316	321	26	202
2010	173^{a} (110)	59^{a} (33)	1137	268	658	200	11	133
2021	266^{b} (210)	215^{b} (470)	316	63	111	68	74	110
	178 (104)	82(186)	969 ± 228	222 ± 48	517 ± 164	199 ± 50	30 ± 11	162 ± 34

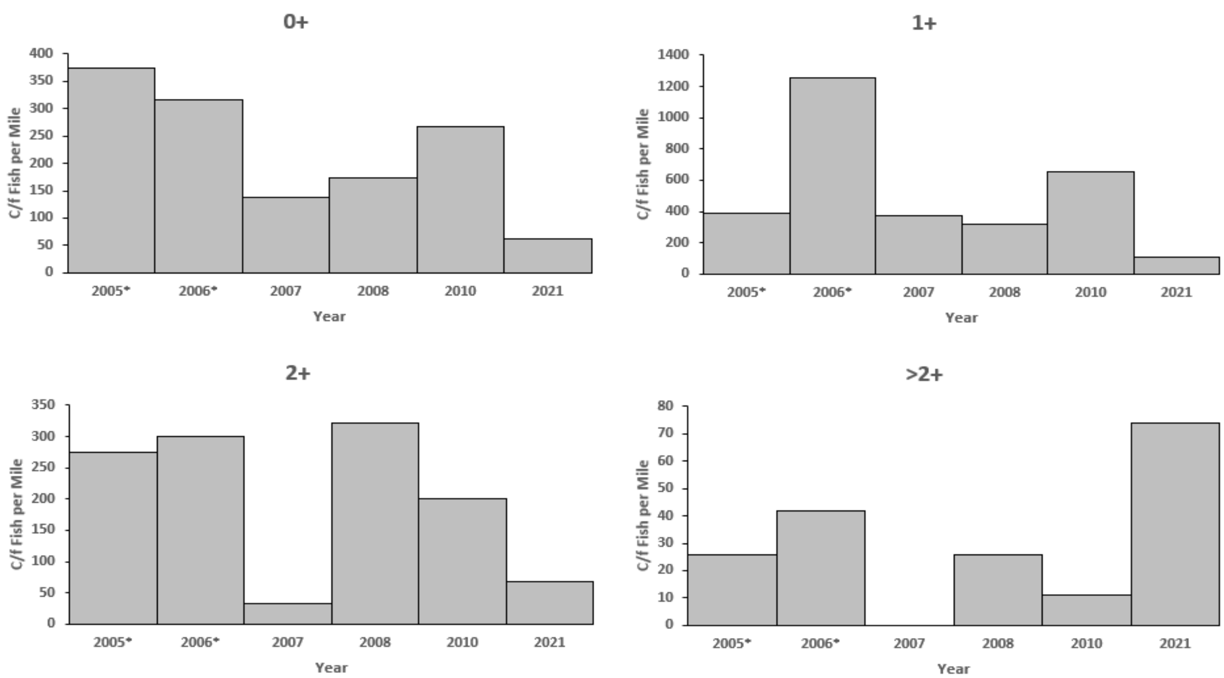

Figure 32. Relative abundance histograms of age groups for the Above Falls Section. Pre-restoration years are denoted with asterisks. Note that the y-axes are not the same scale.

Table 6. Median lengths and weights (interquartile range), biomass, and relative abundances for Below Falls Section. Asterisks denote pre-restoration monitoring. Median lengths and weights with different superscripts are significantly different among years (α =0.05).

				C/*f* (fish/mile) by age group				
Year	Median length (mm)	Median weight (g)	C/f (fish/mile)	0+	1+	2+	> 2+	Biomass (kg/mile)
1989*	161	145	1121	705	195	121	100	163
2005*	206[a] (145)	91[a] (227)	721	90	389	168	74	167
2006*	221[a] (150)	127[a] (254)	763	121	411	163	68	183
2007	188[a] (121)	82[a] (204)	537	53	358	105	21	99
2008	319[b] (97)	358[b] (324)	221	21	32	142	26	89
2021	283[b] (223)	240[b] (520)	326	63	89	100	74	122
	243 (148)	150 (290)	614 ± 133	176 ± 107	246 ± 67	133 ± 12	61 ± 13	137 ± 16

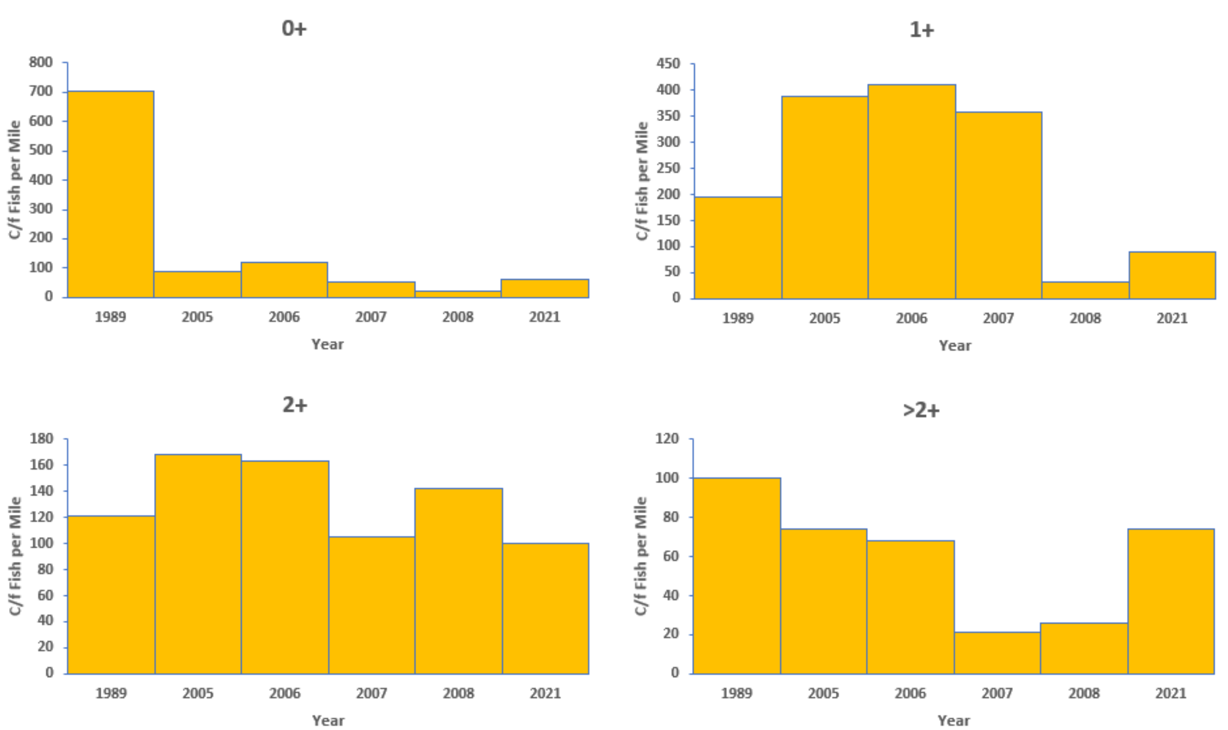

Figure 33. Relative abundance histograms of age groups for the Below Falls Section. Pre-restoration years are denoted with asterisks. Note that the y-axes are not the same scale.

Table 7. Median lengths and weights (interquartile range), biomass, and relative abundances for Old Middle Channel Section. Asterisks denote pre-restoration monitoring. Median lengths and weights with different superscripts are significantly different among years (α = 0.05).

Year	Median length (mm)	Median weight (g)	*C/f* mile (fish/mile)	C/f (fish/mile) by age group 0+	1+	2+	> 2+	Biomass (kg/mile)
2005*	123[a] (25)	-	2211	1989	222	0	0	-
2006*	147[b] (62)	-	1289	712	522	33	22	-
2007	163[bc] (53)	54[a] (64)	1056	279	733	44	0.0	81
2008	168[c] (102)	41[a] (109)	2422	900	1366	156	0.0	203
2010	221[d] (138)	154[b] (218)	1922	511	878	522	11	332
2012	216[d] (127)	127[b] (213)	1367	289	700	367	11	234
2021	176[bcd] (121)	52[a] (172)	667	211	300	122	33	102
	157 (104)	91 (166)	1,557 ± 226	695 ± 219	675 ± 135	175 ± 69	11 ± 4	189 ± 42

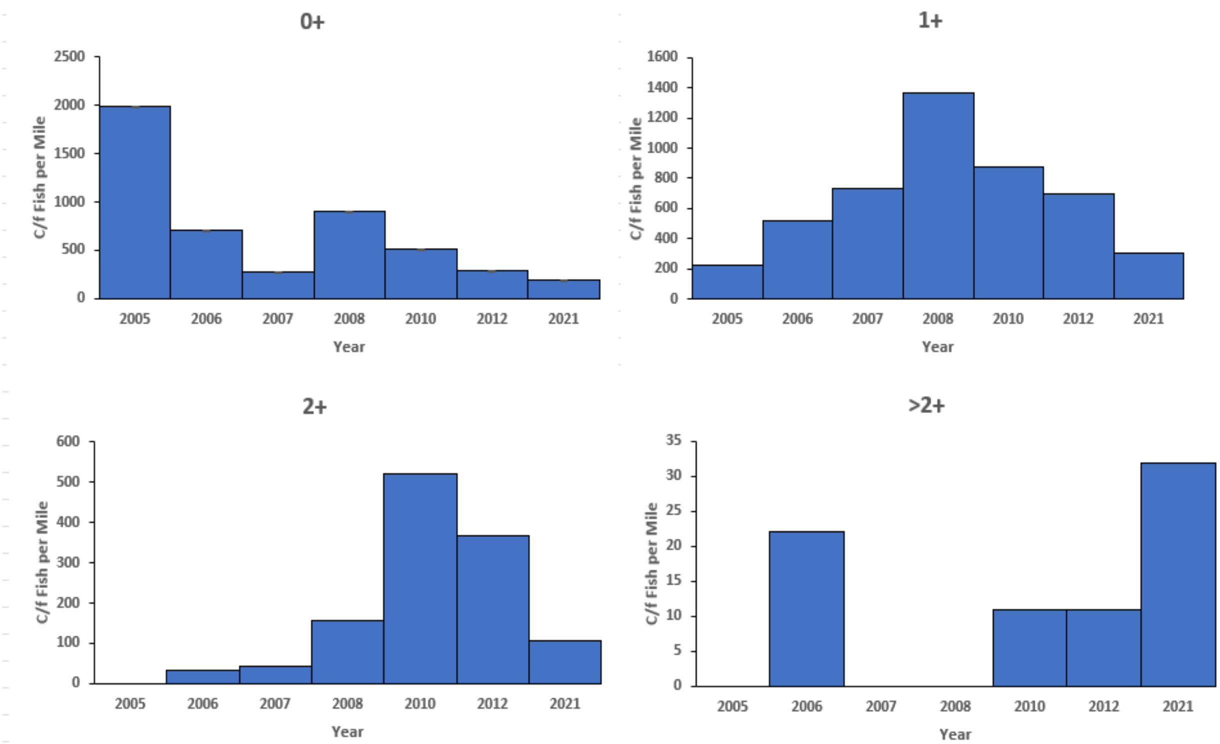

Figure 34. Relative abundance histograms of age groups for the Old Middle Section. Pre-restoration years are denoted with asterisks. Note that the y-axes are not the same scale.

Table 8. Median lengths (interquartile range) and relative abundances for O'Dell Spring North Section. Asterisks denote pre-restoration monitoring. Median lengths and weights with different superscripts are significantly different among years (α = 0.05).

Year	Median length (mm)	C/*f* (fish/mile)	C/*f* (fish/mile) by age group 0+	1+	2+	> 2+
2005*	156[a] (81)	1,367	289	700	0	0
2006	117[ab] (25)	2,044	1,789	256	0	0
2007	114[abc] (25)	1,033	956	78	0	0
2008	124[abcd] (28)	1,144	1,011	133	0	0
2010	132[ad] (33)	811	622	189	0	0
2012	144[a] (26)	867	500	356	11	0
2021	130[ad] (32)	466	322	144	0	0
	127 (41)	1,104 ± 189	784 ± 198	265 ± 80	11 ± 0	0

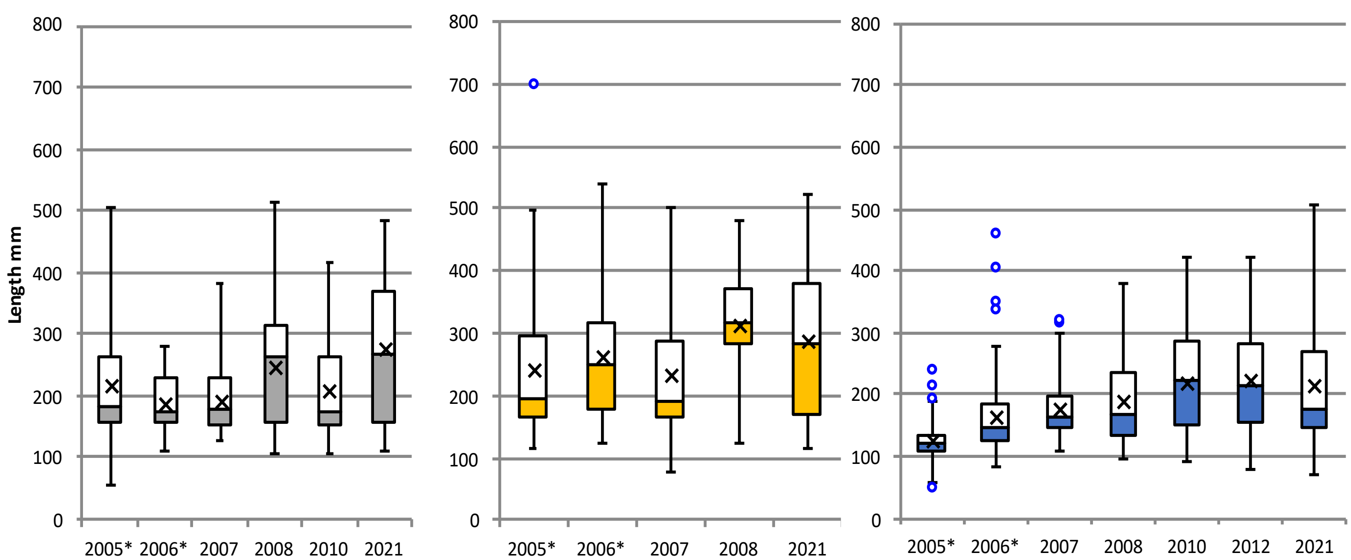

Figure 35. Median and mean lengths for Above Falls (gray), Below Falls (yellow), and O'Dell Old Middle (blue) sections. Asterisks denote pre-restoration monitoring years. Xs denote mean values, horizontal lines are medians, bars are the 5th and 95th percentiles, and circles are outliers.

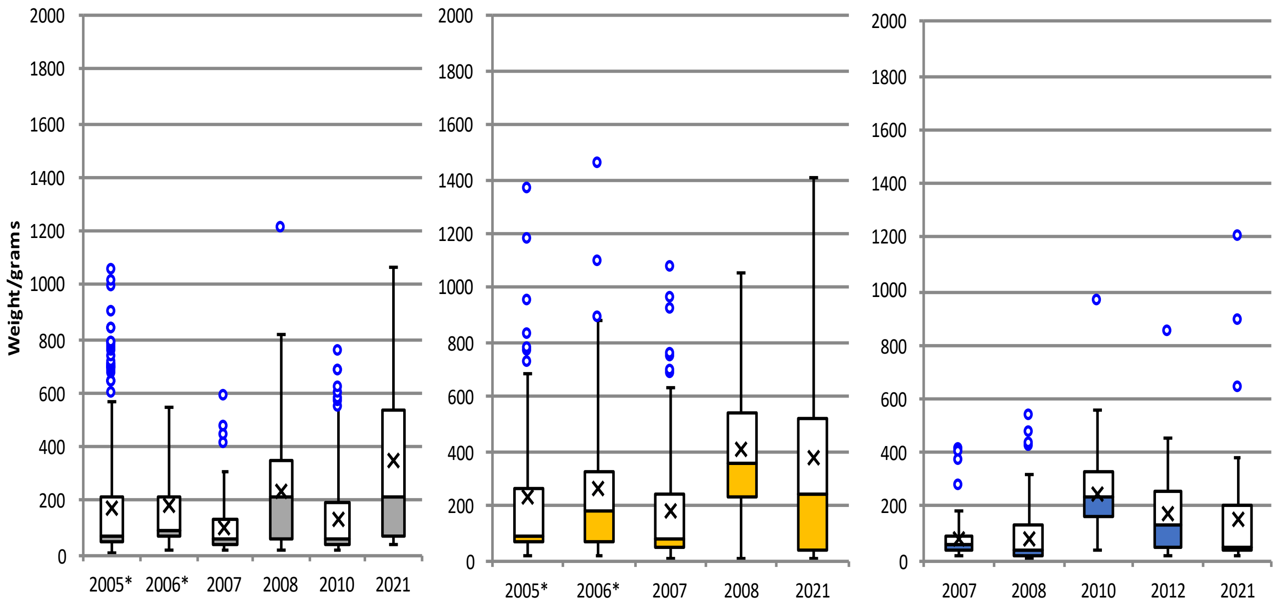

Figure 36. Median and mean weights for Above Falls (gray), Below Falls (yellow), and O'Dell Old Middle (blue) sections. Asterisks denote pre-restoration monitoring years. Xs denote mean values, horizontal lines are medians, bars are the 5th and 95th percentiles, and circles are outliers.

Article 413-Pulsed Flows: Temperature affects all living organisms and fish species have specific thermal ranges that are optimal for their persistence. Exposure to extreme temperatures for extended durations can be lethal to fish. In 1988, a fish kill occurred in the Lower Madison River when temperatures reached 82.5ºF. FWP and NWE have since implemented monitoring programs to mitigate the effects of high-water temperatures on fish. FWP has monitored water and air temperatures throughout the Madison River basin from upstream of Hebgen Reservoir to the mouth of the Madison River at Headwaters State Park since 1993 (Figure 37). Temperature data has been used by FWP as criteria for implementing angling restrictions to reduce mortality of adult trout during periods of thermally induced stress. Angling restrictions are implemented when daily maximum water temperature ≥ 73ºF for three consecutive days. Additionally, to mitigate high water temperatures and reduce the risk of a thermally induced fish kill in the Lower Madison River, NWE implemented the Madison Decision Support System (DSS) program. The Madison DSS program is designed to predict a pulse volume of water that will limit thermal heating sufficiently to keep maximum daily water temperatures ≤ 80ºF at Sloan and avoid the 82.5ºF lethal thermal limit of resident fish in the Lower Madison River. The Madison DSS is comprised of two methods to determine a pulse volume to the delivered to the Lower Madison River: a thermo-dynamic physics model (physics model) and a manual protocol. Pulsed flows are triggered when water temperature at the Madison (Ennis) Powerhouse is 68ºF or higher and the predicted air temperature at the Sloan Station (River Mile 17) near Three Forks, MT for the following day is 80º F or higher. NWE enters the maximum water temperature recorded at the McAllister USGS gage and the next days forecasted maximum air temperature at Three Forks to the manual protocol and the physics model to derive the volume of pulse needed for the following day (Table 9). NWE determines the larger derived pulse of the two methods and directs the operations to release that volume the following day from 6:00 am to noon. Timing of the release is designed to allow for travel time of the water to arrive in the lower Madison River near Sloan during the late afternoon when daily solar radiation is greatest.

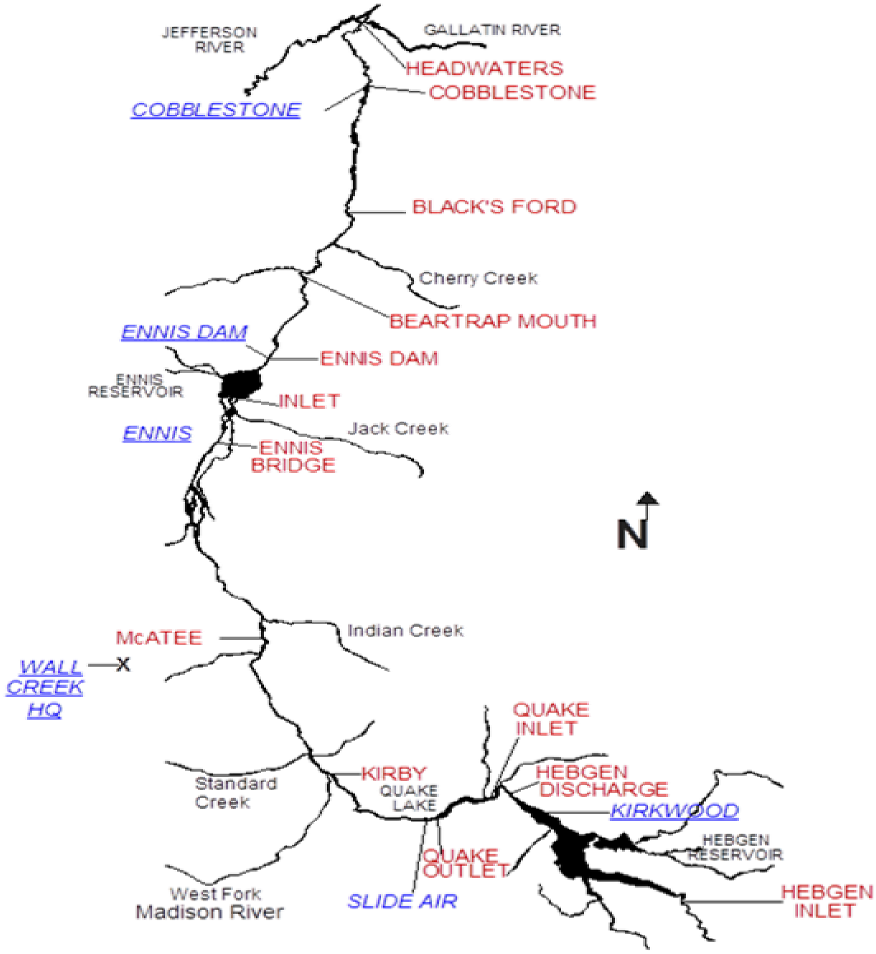

Figure 37. FWP temperature monitoring sites. Air temperature monitoring sites are blue and underlined; water temperature monitoring sites are red.

Table 9. Madison DSS Manual Protocol (Northwestern Energy 2020).

Maximum powerhouse release temperature (◦F) at the Madison DSS website or USGS McAllister gage on or after 8:30 p.m.	Predicted maximum air temperature (◦F) at Sloan Gage the following day and corresponding pulse flows (cfs).		
	75.0—84.9	85.0—94.9	≥ 95.0
68.0—68.9	1150	1150	1400
69.0—69.9	1150	1400	1600
70.0—70.9	1150	1600	2000
71.0—71.9	1400	1600	2100
72.0—72.9	1450	1800	2400
73.0—73.9	1600	2100	2800
74.0—74.9	1800	2600	3000
≥ 75.0	2600	3200	3200

Daily maximum water temperatures recorded in the upper river were ≥ 73^{o}F on 29 occasions, (once at the Ennis Bridge and 28 times at Ennis Reservoir inlet; Table 10); maximum daily temperatures at the Ennis Reservoir inlet met or exceeded the ≥ 73^{o}F on June 28 -July 4, July 28-July 30, and again Aug 11-Aug 14, for periods of 7, 3, and 5 successive days, respectively. Daily maximum temperatures were ≥ 73^{o}F at the lower river monitoring sites, Bear Trap Mouth and Black's Ford for 58 and 63 days, respectively (Table 10). Since 2000, maximum daily water temperatures at the Black's Ford monitoring site have been ≥ 73^{o}F an average of 45 times a year causing FWP to regularly implement restrictions that prohibited angling from 2 p.m. to 12 a.m. during summer months.

In 2021, there were 64 calls for a pulse flow, but only 51 of those resulted operational changes to accommodate a pulse flow. This was the highest number of days where pulsing occurred since the program's inception. Pulse flows kept maximum daily water temperatures from reaching 80^{o} F at Sloan; however, we were not able to ascertain if values for maximum daily water temperatures reached or exceed 80^{o} F below the Sloan site, because the temperature loggers at the Cobblestone and Headwaters sites were not recovered (Table 10). Pulse flows have been implemented an average of 21 days since 2000 and have been effective at moderating maximum daily water temperatures and preventing the occurrence of a thermally induced fish kill in the lower river (Table 11). FWP recommends continued monitoring of Madison River temperatures and that NWE continue to adjust the pulse flow program as needed.

Table 10. Maximum and minimum temperatures (oF) recorded at monitoring sites in the Madison River Drainage, 2021. Mean temperature is mean daily temperate ± 95% confidence intervals (CI). Days ≥ 73oF the number of days daily maximum temperatures were at or excee ded 73°F, and days ≥ 80°F are the number of days daily maximum temperatures were at or exceeded 80°F. NA denotes temperature data was unable to be recovered.

Site	Max°F	Min°F	Mean daily temperature ± 95% CI	Days ≥ 73°F	Days ≥ 80°F
Hebgen inlet	NA	NA	NA	NA	NA
Hebgen discharge	56.8	38.4	56.8 ± 0.1	0	0
Quake Lake inlet	65.7	35.7	56.5 ± 1.2	0	0
Quake Lake outlet	67.2	38.9	55.5 ± 1.2	0	0
Kirby Bridge	68.7	36.4	55.9 ± 1.0	0	0
McAtee Bridge	70.9	34.3	56.9 ± 1.0	0	0
Ennis Bridge	73.1	34.2	59.0 ± 1.0	1	0
Ennis Reservoir Inlet	76.5	34.2	59.7 ± 1.0	28	0
Madison Dam	74.6	41.2	63.6 ± 1.2	15	0
Bear Trap Mouth	78.1	41.2	63.7 ± 1.1	58	0
Blacks Ford	79.2	40.7	63.1 ± 1.1	63	0
Cobblestone	NA	NA	NA	NA	NA
Headwaters S.P. (Madison mouth)	NA	NA	NA	NA	NA

Table 11. The number of days that maximum daily water temperatures at Sloan ≥ 73°F and ≥ 80°F.

Year	≥ 73°F	≥ 80.0°F	Number of days pulsing occurred
2009	34	0	2
2010	29	0	1
2011	27	0	0
2012	50	0	0
2013	69	1	22
2014	42	0	7
2015	50	7	15
2016	51	0	21
2017	57	0	34
2018	38	0	25
2019	40	0	10
2020	50	0	26
2021	59	0	51

Article 419-Coordinate and Monitor Flushing Flows: Article 419 of the 2188 FERC license requires that NWE develop and implement a plan to coordinate and monitor flushing flows in the Madison River downstream of Hebgen Dam. A flushing flow should be large enough to mobilize substrates and produce scour in some locations and deposition in other locations. This is a natural occurrence in unregulated streams and rivers that maintains and creates spawning, rearing, and foraging habitats for fish as well as providing fresh mineral and organic soil for terrestrial vegetation and other wildlife needs. Impoundments such as dams interrupt the natural hydrograph of rivers and high flow events that are responsible for the replenishment and cleaning of spawning gravels are often reduced in magnitude and duration. These effects may be exacerbated by operational parameters the owner or operators of the dam prefer or must comply with. Streambed embeddedness and excessive amounts of fines (particles ≤ 0.8 mm) in spawning gravels can adversely affect the survival of embryos and emergence of fry by inhibiting the delivery of oxygenated water and reducing the amount of interstitial space required for development (McNeil and Ahneil 1964; Kondolof 2000). Accordingly, a goal to maintain ≤ 10% fines in the upper Madison River and ≤ 15% in the lower Madison River were established with the understanding that release of a flushing flow from Hebgen Dam has limited influence on sediment mobility in the lower Madison River. This goal was selected because these targets are known to provide suitable conditions for salmonid spawning.

Operational constraints for Hebgen Reservoir outflow and reservoir elevation limit implementation, magnitude, and duration of a flushing flow. These constraints 1) limit discharge at USGS gage # 6-388 (Kirby gage) to no more than 3500 cubic feet per second (cfs) to limit erosion of the Quake Lake outlet, 2) limit changes in outflow from Hebgen Dam to no more than 10% per day for the entire year, and 3) require that snowpack and runoff forecasts allow for the filling of Hebgen to a minimum elevation of 6,532.26 msl by June 20. Several approaches have been implemented to evaluate the efficacy of flushing flows to recruit and rejuvenate spawning gravels and maintain fine sediment thresholds under current operational constraints, including redd counts, core sampling, and scour chains.

A redd is a nest constructed in the streambed by salmonids where fertilized eggs are deposited and develop until fry emerge from the gravel. Gravels selected for redd construction typically have a median diameter ≤ 10% of the female's body size, can be easily excavated, and contain minimal amounts of fine sediment and organic debris (Chambers et. al 1955; Kondolf and Wolman 1993). Sediment core sampling at the Kirby, Ennis, Norris, and Greycliff sections has occurred annually since 2002. These sites were selected to represent conditions in the upper (Kirby & Ennis) and lower (Norris & Greycliff) Madison River sediment core data provides an index of relative spawning habitat suitability during years with and without flushing flows. Redd counts were initiated in 2012 to ensure complementary substrate sampling (e.g., core samples, scour chains) occurs in actual spawning habitats.

Redd counts are completed by walking upstream and identifying streambed disturbances consistent with redd morphology. A typical redd consists of a pit where gravel was excavated with a mound of gravel (tail spill) immediately downstream of the pit (Figure 38). The number, physical dimensions, and location of individual redds within each monitoring section were recorded. Core samples were collected with a 12-inch McNeil core sampler in substrate previously identified as spawning habitat during redd counts. The core sampler was manually drilled into the substrate to a depth of 8". Substrate from within the 12" x 8" area was removed, dried, and sorted using a sieve method. The percent composition of the sample was calculated according to particle size.

Figure 38. Redd (nest) at the Norris redd counting site. Pit is denoted with the X and black arrow shows the direction of stream flow over tail spill.

Two sample *t*-tests were conducted at $\alpha = 0.05$ to test whether the mean number of redds differed in years with and without flushing flows and 95% CIs were calculated for the mean percent fines ≤ 0.84 mm in core samples from the upper river monitoring sites (Kirby, Ennis) and the lower river monitoring sites (Norris and Greycliff). No significant difference between the number of redds for years with and without flushing flows existed; however, sparse redd data and few flushing flows precluded meaningful statistical inference at any of the sites (Table 12). Inconsistencies in the timing and frequency of counts likely influenced the number of redds observed between years (Table 12). Additionally, flushing flows have had no observed effect on the percent fines present in spawning habitat. Median values for percent fines ≤ 0.8 mm in the upper river ranged from 3.7% (2002) to 10.7% (2020) and from 8.5% (2007) to 22.9% (2014) in the lower river (Table 12). There have been no statistical differences in the percent fines ≤ 0.8 mm observed between years with and without a flushing flow (Figure 39). The flushing flow program and its utility is being evaluated. Discussions about continuing the flushing flow program between NWE and FWP will continue.

In 2021 the number of Fall Brown Trout redds recorded in the lower river were the highest observed since redd counts were implemented. Simple linear regression was used to test if the mean discharge for the month of October affected the ability of observers to identify redds. A negative relationship existed between river discharge in the month of October and the number of Brown Trout redds with 45% of the variation in the number of redds observed explained by the magnitude of the October discharge ($P = 0.05$; $R^2 = 0.45$). The high number of Brown Trout redds observed in 2021 could be due in part to increased visibility of redds at lower flows and or spawning fish being concentrated into limited habitats of suitable depth. The number of Brown Trout redds in the lower Madison River were lowest from 2018-2020. However, mean discharge in October during those years was on average 258 cfs greater than that observed in 2021. Because redd count data is not focused river wide, no inference can be made as to the number of adult spawning fish or their success in a given year. Additionally, observations are potentially skewed by river conditions and other factors. Therefore, FWP recommends discontinuing redd counts as a primary tool to evaluate flushing flow performance.

Table 12. Median % fines ≤ 0.84mm ± standard deviation (SD) and Brown (LL) and Rainbow (RB) Trout redds in the Upper and Lower Madison River, incidence of a NWE flushing flow event, and peak flow in cubic feet per second (CFS) at USGS gage 06041000.

	Upper Madison River			Lower Madison River				
Year	% fines < 0.84 mm median ± SD	LL Redds	RB Redds	% fines < 0.84 mm median ± SD	LL Redds	RB Redds	NWE flushing flow	Peak Flow CFS USGS gage 0604100
1995	6.6 ± 4.4			15.9 ± 5.4				7360
1996	5.8 ± 1.2			8.3 ± 4.5				7980
1997	7.4 ± 3.9			9.8 ± 4.5				7910
1998								6820
1999								5500
2000								4450
2001								2460
2002	3.7 ± 1.5			9.6 ± 4.1			No	5180
2003	8.6 ± 3.2			10.0 ± 5.7			No	4670
2004	7.6 ± 2.7			10.7 ± 5.2			No	3440
2005	6.9 ± 4.1			13.5 ± 8.0			No	4470
2006	9.7 ± 3.7			13.5 ± 5.0			Yes	5390
2007	5.1 ± 2.5			8.5 ± 4.0			No	3400
2008	5.4 ± 2.9			9.7 ± 4.8			Yes	5390
2009	9.3 ± 3.2			12.4 ± 11.7			No	4050
2010	7.0 ± 5.3			11.9 ± 5.7			No	5540
2011	10.1 ± 3.4			13.8 ± 8.2			Yes	7100
2012	6.8 ± 7.2			15.9 ± 5.4			No	4810
2013	5.8 ± 2.1	8	39	18.8 ± 18.7	36	26	No	2850
2014	8.4 ± 3.4	39		22.9 ± 13.7	21		No	5560
2015	8.3 ± 6.1	39	42	12.6 ± 8.3	29	34	No	4490
2016	7.1 ± 4.0	17	78	14.7 ± 10.2	40	48	No	3180
2017	7.9 ± 2.4	14	54	11.7 ± 5.7	46	56	No	4520
2018	8.7 ± 2.6	6		11.4 ± 4.8	20		Yes	6510
2019	7.2 ± 4.5	5	16	10.3 ± 11.3	14	1	No	4670
2020	10.5 ± 4.5	23	22	19.2 ± 6.5	16	59	Yes	6180
2021	9.9 ± 3.5	52	28	14.7 ± 11.5	64	16	No	3260

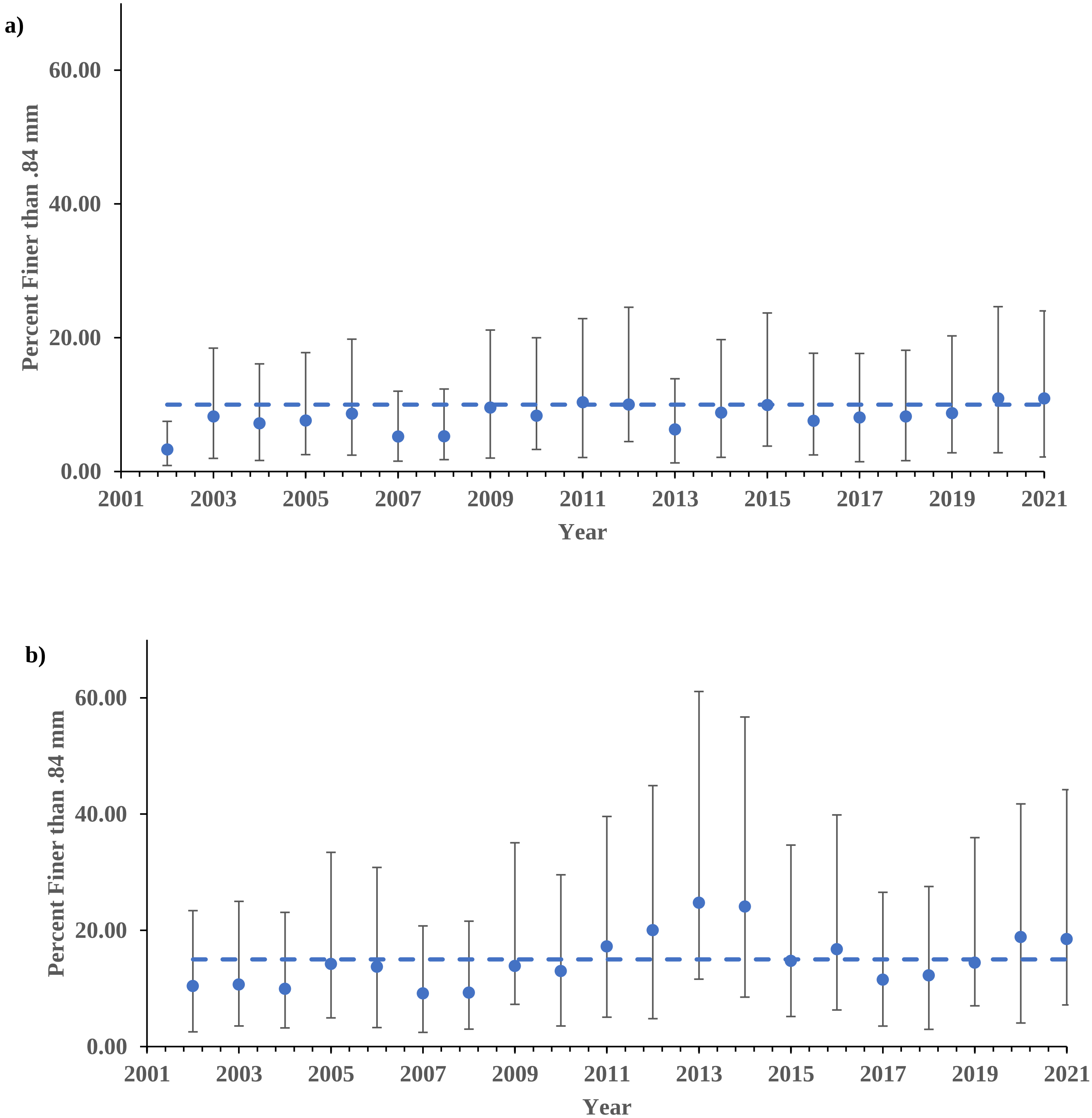

Figure 39 Mean percent fines and 95% CI's of < 0.84 mm in core samples from the Madison River in the (a) Upper River where the blue dashed line is the 10% threshold for fines and (b) Lower River where the blue dashed line is the 15% threshold for fines.

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