Long-Term Increases in Trout Abundance following Channel Reconstruction, Instream Wood Placement, and Livestock Removal from a Spring Creek in the Blackfoot Basin, Montana

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Published online: 08 Jan 2015.

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Abstract

To restore habitat for wild trout, Kleinschmidt Creek, a low-gradient, groundwater-dominated stream in the Blackfoot Basin, Montana, was reconstructed using natural channel design principles. Reconstruction increased stream sinuosity from a ratio of 1.1 to 1.6, decreased mean channel width from 14.5 to 2.8 m, and increased sediment transport capacity to reduce accumulations of fine instream sediment. To further improve trout habitat, coarse woody debris (CWD) was variably placed within the new channel and livestock were excluded to promote the vegetative recovery of the riparian area. To evaluate the response of wild trout (92% Brown Trout *Salmo trutta*) to channel restoration, the abundance (number of trout per linear meter) and biomass (g/linear m) of age 1+ trout were monitored for 15 years (1998–2012) in a reach with low density CWD (1.3 stems/100 m) and compared with regional (reference) trends. Posttreatment (2002–2012) trout numbers in the low-density CWD reach were also compared with those in a reach with high-density CWD (18.2 stems/100 m). Long-term trends for the reference reaches showed a significant negative trend in trout abundance and no significant trend for biomass. Long-term trends for the low-density CWD reach showed a significant positive trend in abundance, as well as a significant trend in biomass. Trout abundance and biomass increased over the posttreatment period in the low-density CWD reach. However, in the high density CWD reach, while posttreatment abundance increased significantly, there was no significant trend in biomass. These results demonstrated that channel restoration increased wild trout populations in a deep, narrow, vegetated stream and that instream wood provided primarily short-term benefits during the early phase of habitat recovery.

To offset human-induced degradation of river ecosystems, aquatic habitat restoration is expanding across North America. As restoration methods evolve, practitioners are applying natural restoration techniques more frequently to reestablish the ecological integrity and physical habitat necessary for the recovery of sensitive fisheries (Nagle 2007; Baldigo et al. 2008; Ernst et al. 2010; Pierce et al. 2013). Despite broad-scale increases in restoration, few projects document fisheries response to the use of natural channel design (NCD) as an active method to emulate the form and function of geomorphically stable natural streams (Rosgen 1996, 2007, 2011; Klein et al. 2007; Nagle 2007; Baldigo et al. 2008; Ernst et al. 2010). Likewise, few restoration studies investigate overlapping passive methods needed to mediate riparian damage caused by intensive land uses, such as heavy riparian grazing (Meehan 1991; Saunders and Fausch 2007; Pierce et al. 2013).

The paucity of published field studies limits the ability of restoration practitioners to develop and apply informed, effective, and more natural stream restoration techniques (Bernhardt et al. 2005; Baldigo et al. 2008, 2010; Roni et al. 2008).

For over two decades, restoration practitioners have increasingly focused projects on NCD and the recovery fluvial processes to increase channel stability, decrease erosion, restore natural flow, temperature, and sediment regimes, and revitalize sensitive fisheries (Nagle 2007; Baldigo et al. 2008, 2010; Pierce et al. 2013). Central to the NCD concept is the classification of natural rivers into stream types (Rosgen 1994). This classification provides a basis to emulate the dimension (e.g., valley bottom and stream cross-sectional features), pattern (e.g., meander features such as sinuosity), and profile (e.g., valley and channel slopes) of streams that are geomorphically stable (in equilibrium) with their stream valley (Rosgen 1994, 1996, 2007, 2011). Of the stream types described by Rosgen (1994), many are geologically controlled and thus resistant to human alterations. This study focuses on the restoration of a vegetatively controlled stream type that is considered more sensitive to anthropogenic disturbance such as heavy grazing than are streams controlled by geology.

Improper livestock grazing of certain riparian areas can degrade habitat for coldwater salmonids by increasing streambank erosion, reducing riparian vegetation, degrading streambanks, and lowering water tables, all of which cause streams to become wider, shallower, and warmer during summer (Meehan1991; Platts 1991; Pierce et al. 2014a). Though riparian impacts from heavy grazing are widely documented, the efficacy of most riparian grazing strategies to recover salmonid habitat remains uncertain with the exception of exclusion (Platts and Nelson 1985; Platts 1991; Roni et al. 2008; but see Myers and Swanson 1995). The impacts of heavy riparian grazing vary with timing, intensity, and frequency of grazing and type of animal (Platts 1991), as well as site conditions such as riparian plant composition, stream hydrology, and stream type (Hansen et al. 1988; Rosgen 1996; Bengeyfield and Svoboda 1998). Compared with armored channels having coarse (cobble, boulder, and bedrock) substrate, stream types that are controlled vegetatively with noncohesive alluvial soils (sand and gravel) are more sensitive to grazing disturbance (Myers and Swanson 1992; Rosgen 1996). This sensitivity can increase dramatically in areas of groundwater inflow where streambanks are wet during periods of grazing (C. B. Marlow and T. M. Pogacnik [paper presented at the North American Riparian Conference, 1985]).

Similar to the effects of heavy riparian grazing, the anthropogenic loss of instream wood can simplify and degrade salmonid habitat (e.g., Meehan 1991; Gregory et al. 2003; Roni et al. 2008) and ultimately reduce the overall ecological integrity of streams (Schmetterling and Pierce 1999; Bilby 2003; Rosenfeld and Huato 2003). The loss of instream wood is often the result of deforestation, excessive grazing, intentional forest clearing, road construction, and other streamside development pressures (Meehan 1991; Gregory et al. 2003; Jones et al. 2014). Conversely, input of coarse wood can improve the ecological integrity of streams by controlling gradient, increasing pools, and providing essential instream cover for fish (Roni et al. 2008; Whiteway et al. 2010; Jones et al. 2014) and diversifying habitat necessary for spawning and rearing of many salmonids (Schmetterling 2000; Gregory et al. 2003; Jones et al. 2014). Because of its high habitat value, instream wood has been used as a habitat improvement technique for decades (e.g., Tarzwell 1936; Hunt 1976; Binns 2003; Roni et al. 2008).

Despite the biological importance of instream wood, very few long-term (>5 years posttreatment) restoration studies have examined the response of salmonids to the placement of instream wood (Roni 2005; Roni et al. 2008). Exceptions include reach-scale treatments on confined (i.e., V-shaped valleys with narrow floodplains), stable channels and moderate (e.g., 2–4%) stream gradients (Rosgen 1996; Roper et al. 1998; Baldigo et al. 2008; White et al. 2011). The long-term effectiveness of wood placement in low-gradient (e.g., <2%) unconfined channels in alluvial valleys remains uncertain (Frissell and Nawa 1992; Schmetterling and Pierce 1999; Jones et al. 2014), particularly where meandering channel processes and rhizomatous meadow vegetation strongly influence channel morphology (Rosgen 1996).

In the Blackfoot River basin of western Montana, landowners, agencies, and private conservation groups have engaged in riverscape restoration actions for over 20 years (Aitken 1997; Pierce et al. 2013, 2014b). This study expands on a long-term monitoring study that showed restoration-induced reductions in water temperature in Kleinschmidt Creek (Pierce et al. 2014a). The purpose of this study was to describe changes to channel morphology following the conversion of an overwidened and degraded channel to a deep, narrow, meandering, and vegetated stream type, and to specifically examine the fisheries response to this conversion and the variable use of instream coarse woody debris (CWD) within the new channel.

STUDY AREA

Kleinschmidt Creek is a groundwater-dominated tributary to the lower North Fork of the Blackfoot River, located on the floor of the Blackfoot River valley in west-central Montana near the town of Ovando (Figure 1). Kleinschmidt Creek originates along the southern margin of a large glacial outwash plain and flows for approximately 3.4 river kilometers (rkm) within a terraced alluvial and morainal valley before entering the North Fork of the Blackfoot River at rkm 9.9 at an elevation of 1,268 m. The stream gains approximately 90% of its flow from groundwater inflows between rkm 1.6 and 3.2. Stream discharge ranges from 0.26 m^3/s during winter to about 0.42 m^3/s during summer (Pierce and Podner 2006) and the stream has a peak bankfull discharge of 0.71m^3/s (R. Shields, U.S. Geological Survey, retired, unpublished data). Streamside vegetation consists of wetland gramminoids (*Carex* spp., *Juncus* spp., *Phararis* spp.) and shrub
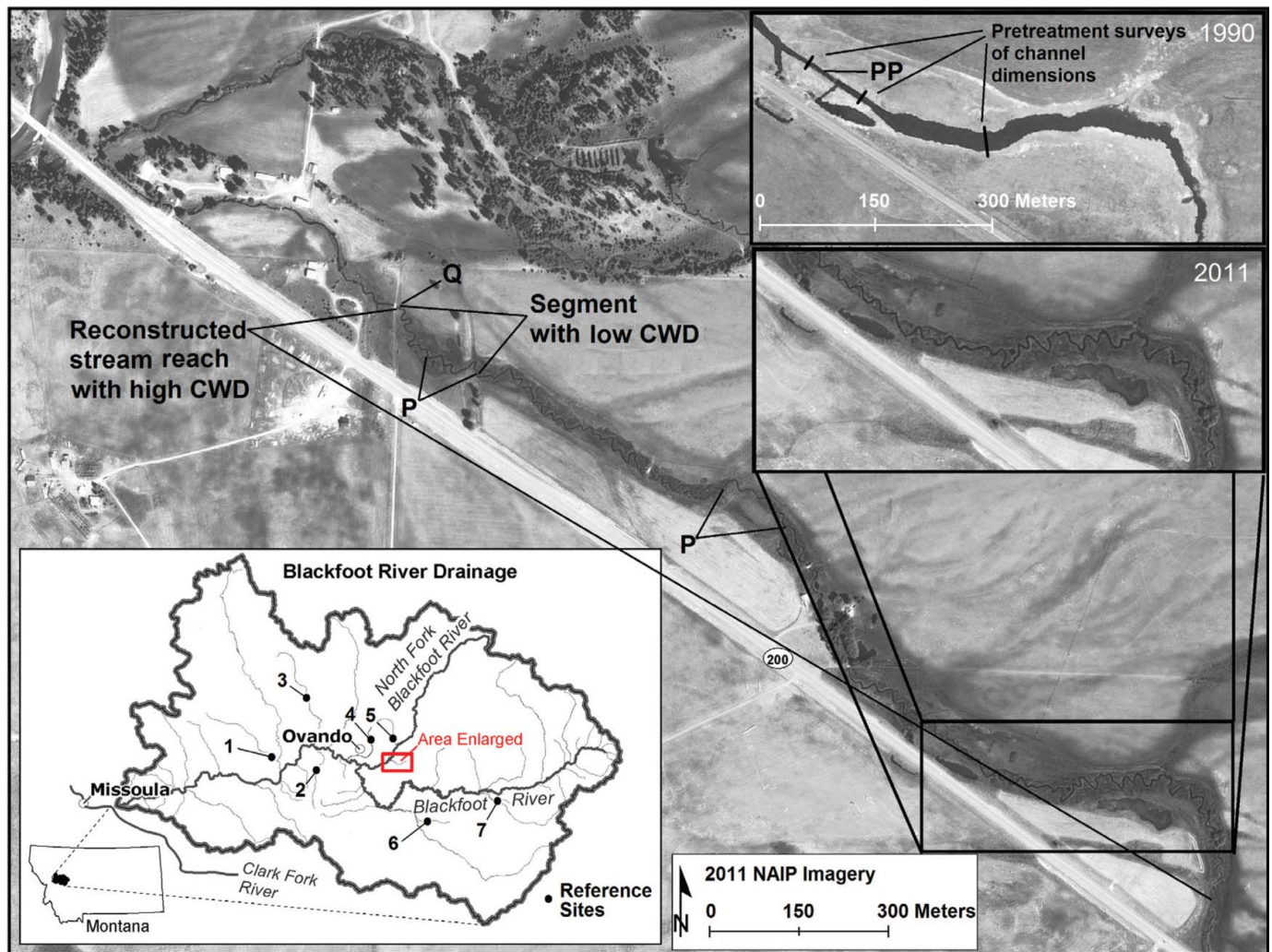
FIGURE 1.   Kleinschmidt Creek and the 2001 restoration project area. High density CWD was placed throughout the reconstruction site with the exception of the stream segment with low density CWD. In addition, two fish population monitoring sites (P) and the bankfull flow measurement site (Q) are identified. The map inset (lower left panel) shows the Blackfoot River basin and the Kleinschmidt Creek study area in Montana, as well as seven reference reach fish population monitoring sites. The two upper right panel insets show pretreatment (1990; aerial photo) and posttreatment (2011; NAIP 2011) images of the upper reaches of the study area and the photopoint (PP) location. [Figure available online in color.]

(*Salix* spp., *Alnus* spp.) cover. Kleinschmidt Creek supports a mixed community of salmonids though Brown Trout *Salmo trutta* comprise >90% of the salmonid community. Other salmonids present in the order of decreasing abundance are Brook Trout *Salvelinus fontinalis*, native Westslope Cutthroat Trout *Oncorhynchus clarkii lewisi*, native Bull Trout *S. confluentus*, and Rainbow Trout *O. mykiss*. The exotic parasite, *Mxyobolus cerebralis*, the cause of whirling disease in salmonids (Bartholomew and Wilson 2002), is also present in Kleinschmidt Creek where infection rates are high (Pierce et al. 2014a). Prior to restoration, Kleinschmidt Creek was subjected to heavy livestock grazing, accelerated streambank erosion and channelization resulting from highway construction, the installation of artificial grade controls that included rock dams and undersized culverts (Pierce 1991; Land and Water Consulting 1999; M. Marler,

1998 unpublished technical report to the U.S. Fish and Wildlife Service, on site assessment and summary of impacts of proposed stream restoration). A combination of these factors caused the channel to become wide, shallow, and straightened with elevated water temperature (Decker-Hess 1986; Pierce et al. 2014a) and high sediment loading (sand and silt), especially upstream from rock dams (Figure 2A). These conditions resulted in a corresponding reduction of instream habitat complexity, degraded wetlands, and the complete loss of woody riparian vegetation (Figure 2A). The pretreatment morphological values are shown in Table 1.

With the ultimate goal of restoring stream habitat for the recovery of wild trout, the restoration of the lower 0.64 rkm of Kleinschmidt Creek was completed in 1998. Then in 2001, restoration on the remaining upper 2.73 rkm of stream
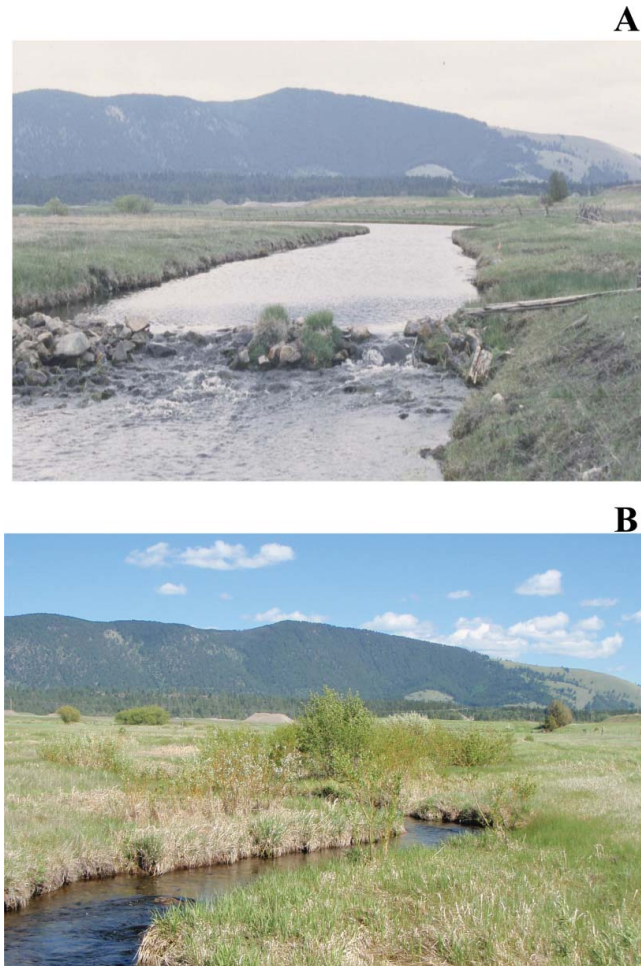
FIGURE 2. Kleinschmidt Creek (**A**) pretreatment, September 2001, and (**B**) posttreatment, June 2014. The upper photo (A) shows a straightened and over-widened section of channel with an example of a channel-altering rock dam that induced upstream deposition of fine sediment within the pretreatment channel. The lower photo (B) shows the restored stream at the same location. [Figure available online in color.]

(the emphasis of this study) was completed to reestablish natural form and function consistent with a vegetative-controlled, more sinuous, deep and narrow channel (Figures 1, 2B). With the use of the Rosgen (1996) stream classification, the 2001 project fit the stream to its valley, reconnected the channel with its original floodplain, removed artificial grade controls (rock dams and culverts), and converted a degraded, wide and shallow stream with fine substrate (i.e., Rosgen impaired C5 stream type; Figure 2A) to form a more natural deep and narrow meandering channel (i.e., Rosgen reference E4 stream type: Figure 2B; Rosgen 1996, 2007, 2011). In the absence of vegetative disturbance, the E4 stream type is a geomorphically stable, hydraulically efficient channel found within alluvial valleys. The stable E4 stream type is specifically characterized by low channel gradients ($<2\%$), low width : depth ($W{:}D$) ratios (<12), high sinuosity (>1.5), gravel substrates, and pool–riffle

bedforms. The degradation of an E4 stream type can convert a stable channel to an impaired C5 stream type through accelerated streambank erosion, channel widening, loss of sinuosity, and the instream accumulation of fine sediment (Rosgen 2007; e.g., Figure 2B). Delineative criteria for the E4 and C5 stream types are shown in Table 2.

During final channel restoration, the upper 2.73 rkm of stream was narrowed to a mean width of 3.0 m, pool–riffle bedforms were reestablished, and 183 coarse woody stems were anchored throughout the streambed and outer banks of most (100 of 108) pool–riffle sequences between rkm 0.95 and rkm 3.40 (Figure 1). This wood was specifically used to offset the anthropogenic loss of instream wood and to increase habitat complexity until riparian vegetation (including woody species) was reestablished. However, during final channel shaping, a 0.31-rkm segment (16 pool–riffle sequences; rkm 0.65–0.95) of new channel was left with a very minimal amount of CWD ($n = 3$ stems total) to explore the fisheries response as described below (Figure 1). Following channel restoration, riparian shrubs were planted, livestock were excluded from the riparian corridor to recover streamside vegetation, and a perpetual conservation easement was placed on a majority of the stream corridor to protect the long-term ecological integrity of wetlands and the riparian area and to prevent grazing-related damage to the new channel (MDT 2001). Although Kleinschmidt Creek had no recent history of dewatering, a large portion of the stream's discharge (0.25 m^3/s) was later dedicated to the maintenance of instream flows to prevent the possibility of future dewatering (MDNRC 2011).

METHODS

Stream channel morphology.—To identify pretreatment and posttreatment project-scale changes (rkm 0.64–3.37) in channel morphology, aerial photo–imagery measurements and field measurements with NCD-related techniques were used (Rosgen 2007, 2008, 2011). Channel lengths and sinuosity were measured from aerial photographs taken prior to restoration (1990) and from National Agricultural Imagery Program (NAIP) high-resolution (1 m) imagery taken 10 years after (2011) full channel reconstruction and livestock exclusion (Figure 1). Aerial photos were georectified and NAIP imagery was classified to contrast surface waters with riparian vegetation. ArcGIS (version 10.1) software was used to measure channel lengths (1:500 scale) along the centerline of the 1990 and 2011 channels. Stream sinuosity was calculated as a ratio of the total centerline channel length divided by the linear distance of the stream valley. In addition to aerial measurements, field surveys of stream slope (using a laser level) and bankfull discharge (measured at bankfull stage with a Marsh–McBirney model 2000 Flo-mate current meter) were measured at the lower portion of the restoration project in June 2011 (Figure 1). Pretreatment channel dimensions were surveyed in the upper treatment area by Land and Water Consulting (1999) (Figure 1). Posttreatment channel

TABLE 1.   Summary of channel morphometrics for Kleinschmidt Creek before restoration (1990) and 10 years after restoration (2011) along with hydraulic relationships of the pretreatment (C5) and posttreatment (E4) stream types.

Variable	Pretreatment (C5)	Posttreatment (E4)
Channel length (km)	1.97	2.73
Sinuosity	1.1	1.6
Stream slope (S; m/m)	0.0058	0.0040
Valley slope (m/m)	0.0064	0.0064
Bankfull discharge (Q_{bkf}; m³/s)	0.71	0.71
Mean bankfull width (W_{bkf}; m)	20.1	3.1
Mean bankfull depth (D_{bkf}; m)	0.13	0.35
Bankfull $W{:}D$ ratio	150	8.8
Mean bankfull cross-sectional area (A_{bkf}; m²)	2.6	1.1
Mean bankfull mean velocity ($\bar{u}_{bkf}$; m/s)	0.27	0.67
Bankfull shear stress (τ; N/m²)	7.4	13.7
Particle entrainment size (mm)	11	21

dimensions were surveyed in 2013 at the low CWD reach (Figure 1). Associated morphological relationships representing the mean values were then calculated to compare the observed pretreatment (C5) and posttreatment (E4) vegetated stream types (Rosgen 2007, 2008, 2011; Table 1).

Hydraulic and sediment transport relationships were then calculated for the pre- and posttreatment channels. Bankfull mean velocity ($\bar{u}_{bkf}$) (m/s) was estimated using the flow continuity equation,

$$\bar{u}_{bkf} = Q_{bkf}/A_{bkf}, \tag{1}$$

where Q_{bkf} = bankfull discharge (m³/s) and A_{bkf} = bankfull cross-sectional area (m²). Pre- and posttreatment bankfull shear stress (τ) (N/m²) was calculated using the relation,

$$\tau = \gamma RS, \tag{2}$$

where γ = specific weight of water (9.81 kN/m³), R = hydraulic radius (substituted by mean bankfull depth), and S = stream slope. Pre- and posttreatment particle entrainment sizes (mm) were derived from the critical bankfull shear stress relation in Leopold et al. (1964):

$$\text{Particle diameter} = 77.966(\tau/47.88)^{1.042}. \tag{3}$$

To obtain specific reach-scale treatment data that may further affect trout populations, stream surveys were undertaken

in each of the two 154-m-long fish population survey sites in 2013 in reaches with low and high density CWD using modified Rosgen level II survey methods (Rosgen 1996). For these surveys, stream surveys were conducted with a laser level and measuring rod to determine wetted widths and wetted depths of all pools ($n = 8$) and riffles ($n = 8$). In addition, bankfull $W{:}D$ ratios at one representative riffle within each section were measured as was substrate composition using Wolman pebble counts (with a minimum of 100 measured pebbles) at each representative riffle. We also counted and measured all instream woody stems (>10 cm diameter and >1 m in length) anchored within each fish population monitoring site and measured the percent shrub cover overhanging the streambanks of each survey reach.

Fisheries data collection and organization.—Three years of pretreatment fish population data (1998–2000) were obtained as a baseline in the impaired (C5) stream reach with minimal wood (Figure 1). Following active channel work in 2001, annual fish population surveys continued over an 11-year posttreatment period (2002–2012) at rkm 0.80–0.95 in the (E4) reach with low CWD and at rkm 1.62–1.77 in the (E4) stream reach with high CWD. Each monitoring site was 154 m in length as described above. Fish populations were not surveyed in Kleinschmidt Creek in 2011 due to sampling difficulties related to high stream flow. Pretreatment and posttreatment fisheries data from the low-density CWD reach were compared with regional trends of abundance and biomass to analyze long-term (15 year) response trends. Reference fisheries data were compiled from seven small, alluvial, low-gradient (<2%) and

TABLE 2.   Delineative criteria for the C5 and E4 stream types (Rosgen 1996).

Stream type	Entrenchment ratio	$W{:}D$ ratio	Sinuosity	Channel slope	Predominate channel materials
C5	>2.2	>12	>1.2	<2%	Sand
E4	>2.2	<12	>1.5	<2%	Gravel

TABLE 3. Site conditions for the seven reference reaches in Kleinschmidt Creek. Stream identification (ID) relates to stream locations on Figure 1.

Stream ID	Stream name	Stream order	Elevation (m)	Stream slope (%)	Sinuosity	Bankfull riffle area (m^2)	W:D ratio	Substrate D50 (mm)	Stream type
1	Blanchard Creek	2	1,167	1.9	1.1	2.2	29	39.0	C4
2	Chamberlain Creek	2	1,306	1.0	1.2	1.8	18	35.0	C4
3	Cottonwood Creek	3	1,324	2.0	1.2	2.6	12	68.0	C4
4	Warren Creek	2	1,316	0.3	1.2	1.1	9	17.0	E4
5	Murphy Spring Creek	1	1,314	0.2	1.5	1.2	16	29.0	C4
6	Wasson Creek	1	1,325	1.2	1.5	0.5	10	12.0	E4
7	Grantier Spring Creek	1	1,379	0.3	1.6	0.8	15	12.0	C4

low-elevation reference reaches (Figure 1; Table 3). Reference reaches were defined as geomorphically and vegetatively stable (Rosgen 1996) with fish populations unaffected by direct human impacts (Pierce et al. 2013). Fish populations were surveyed in reference reaches every year from 1998 through 2012 ($n = 46$ surveys); each discreet reference reach averaged 8 years (range, 3–14 years) of fisheries survey data, and each monitoring year between 1998 and 2012 averaged three surveys (range, 1–6). Reference reaches are listed in Table 3.

All estimates of trout abundance and biomass in this study were derived from of age 1 and older trout. Estimates of abundance (number of trout per linear meter, hereafter trout/m) were conducted between August 7 and October 4 using backpack electrofishing units and depletion (two- and three-pass) estimator methods (Van Deventer and Platts 1985). Estimates of biomass (g/linear m, hereafter g/m) were calculated by multiplying population abundance by mean fish weight. Fish population estimates generated a mean capture probability of 0.75 (range, 0.50–0.94) for reference reaches and 0.61 (range, 0.48–1.0) for surveys in Kleinschmidt Creek. All fish population surveys began at a downstream pool–riffle break (e.g., riffle crest), proceeded upstream, and ended at an upstream pool–riffle break. Block nets were set at the upstream survey boundaries of the Kleinschmidt Creek survey sites in years with high water but not in low-water years. Once captured, all trout were sedated with tricaine methanesulfonate (MS-222; Argent Chemical Laboratories, Redmond, Washington) or clove oil. Individual trout were then identified by species, measured for TL (mm), and weighed (g) and then immediately placed in freshwater to regain their equilibrium before their release within the monitoring section. Because CPUE of age-0 trout did not differ between low-density and high-density CWD reaches in Kleinschmidt Creek during the monitoring period, age-0 trout were excluded from abundance and biomass analyses.

Analyses of trout response to restoration.—To analyze the fisheries response, ANCOVA was used to test for treatment effects between the low-density CWD, high-density CWD, and reference reaches for pre- and posttreatment periods.

Linear regressions were used to test for significant trends (positive or negative slopes) in abundance and biomass before and after treatment and over the entire study period for low-density CWD, high-density CWD, and reference reaches. Increases in trout abundance and biomass were considered statistically significant if the slope of the trend line was significantly different from zero ($P < 0.05$). Prior to statistical analysis, all estimates of abundance were natural log transformed to meet assumptions of normality and homogeneity of variance. Before transformation, a value of 1 was added to each estimate to avoid generating a value of negative infinity when we attempted to transform values of zero. All analyses were performed using SAS version 9.4 statistical software (SAS Institute, Cary, North Carolina).

RESULTS

Stream Channel Morphology

The posttreatment channel was 39% longer, 85% narrower, and 169% deeper than the pretreatment channel, and riparian vegetation was reestablished (Figures 1, 2B; Table 1). With these changes in basic channel morphology, posttreatment hydraulic conditions of bankfull mean velocity, shear stress, and sediment entrainment all increased (Table 1).

Posttreatment habitat surveys of the two 154-m fish population survey sites (i.e., high and low CWD reaches) showed broad similarity between sites, including having low W:D ratios (range, 8.8–9.1) and a dominant gravel substrate (D50 range, 14–21) in riffles (Table 3), which further characterized both monitoring sites as the same E4 stream type (Rosgen 1996). Consistent with uniform features of this deep and narrow, meandering, vegetated stream type, habitat features showed comparable wetted widths and wetted depths for the eight pool–riffle sequences in both reaches (Table 4), although pools averaged 0.3 m deeper in the reach with low CWD. Concentrations of CWD were, however, 14 times higher in the reach with high density CWD compared with the reach with high density CWD, and shrub cover was 9% in the reach with high CWD and <1% in the low-density CWD reach.

TABLE 4. Summary of channel bedform features for low- and high-density CWD reaches in Kleinschmidt Creek including pool and riffle width and depth measurements, substrate, *W:D* ratios, and summaries of instream wood concentrations for the two fish population monitoring sites.

	Pools		Riffles					Instream CWD	
Reach	Mean (range) wetted depth (m)	Mean (range) wetted width (m)	Mean (range) wetted depth (m)	Mean (range) wetted width (m)	Substrate D50 (mm)	*W:D* ratio	Number of stems	Mean (range) diameter (cm)	Mean (range) length (m)
Low CWD	1.1 (0.8–1.2)	3.4 (3.2–3.8)	0.4 (0.3–0.5)	2.8 (2.4–3.6)	14	8.8	2	28 (25–30)	1.2 (1.0–1.5)
High CWD	0.8 (0.7–1.1)	3.5 (3.1–4.0)	0.3 (0.2–0.4)	3.1 (2.7–3.5)	21	9.1	28	25 (18–30)	2.3 (1.2–4.6)

Trout Response to Restoration

Composition of trout species for the two Kleinschmidt Creek fish population survey sections showed Brown Trout comprised 92% of the total catch followed by 7% Brook Trout, 0.5% Westslope Cutthroat Trout, 0.3% Bull Trout, and 0.2% Rainbow Trout (Figure 3). Pre- and posttreatment comparisons showed that pretreatment abundance was significantly higher in the reference reaches than in the low-density CWD reach ($F_{1, 10} = 14.64$, $P = 0.003$) and remained higher posttreatment ($F_{1, 44} = 15.39$, $P < 0.001$). Conversely, there was no significant difference is pretreatment biomass ($F_{1, 10} = 2.08$, $P = 0.18$) or posttreatment biomass ($F_{1, 44} = 0.61$, $P = 0.439$) between the low-density CWD and reference reaches. Further analysis showed a significant treatment effect between the low-density CWD and high-density CWD reaches for trout abundance ($F_{1, 18} = 7.73$, $P = 0.012$) and no significant difference for trout biomass ($F_{1, 18} = 4.11$, $P = 0.06$). Long-term trends for the reference reaches showed a significant negative trend in abundance ($F_{1, 43} = 4.75$, $P = 0.03$, slope $= -0.014$; Figure 4A) and no significant trend for biomass ($F_{1, 43} = 0.74$, $P = 0.395$, slope $= -0.691$; Figure 4B). Long-term trends for the low-density CWD reach showed a significant positive trend in abundance ($F_{1, 11} = 19.26$, $P = 0.001$, slope $= 0.025$) and a significant positive trend in biomass ($F_{1, 11} = 39.89$, $P < 0.001$, slope $= 3.75$).

During the 15-year study period, total trout abundance in the reach with low density CWD increased from a pretreatment average of 0.06 trout/m to a posttreatment average of 0.25 trout/m, compared with a 15-year average of 0.47 trout/m for the reference reaches (Figure 5A). Likewise, biomass in
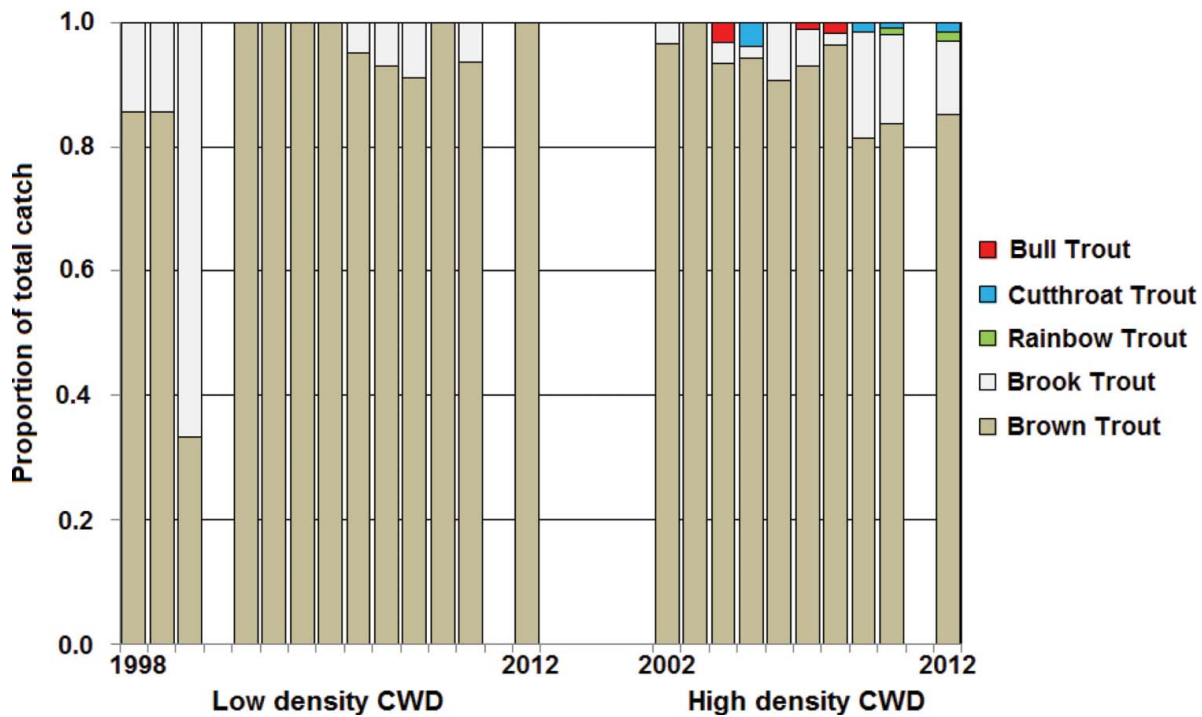

FIGURE 3. Species composition for age-1 and older trout for the low- and high-density CWD reaches. [Figure available online in color.]
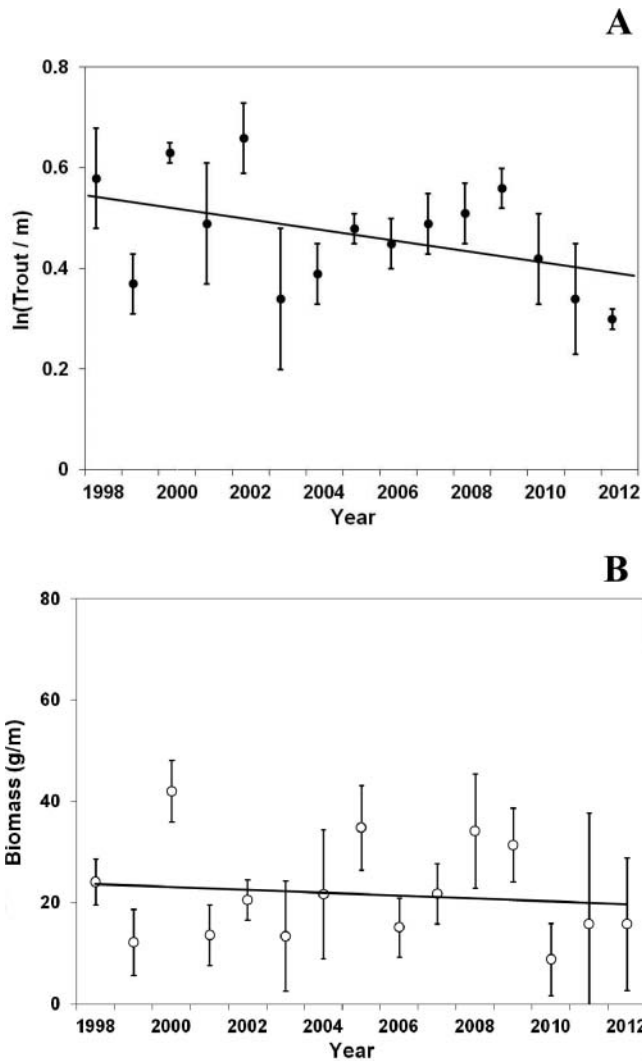
FIGURE 4. Comparisons by year of (**A**) abundance and (**B**) biomass and across-year comparisons of slopes for seven reference reaches.

FIGURE 5. Estimates of (**A**) total trout abundance and (**B**) total trout biomass for the low- and high-density CWD reaches of Kleinschmidt Creek. The solid black trend lines in both panels show the positive increase in the posttreatment high-density CWD reach, and the dashed lines show the positive population increase in the low-density CWD reach over the entire monitoring period.

the low-density CWD reach increased from a pretreatment average of 3.1 g/m to a posttreatment average of 29.1 g/m, compared with 44.2 g/m in the reach with high CWD and the long-term reference reach average of 21.7 g/m. No significant trends in abundance or biomass were found posttreatment for the reference reaches (abundance: $F_{1, 9} = 1.84$, $P = 0.21$; biomass: $F_{1, 9} = 0.18$, $P = 0.68$). Whereas in the low-density CWD reach, abundance and biomass increased significantly over the posttreatment period (abundance: $F_{1, 8} = 11.97$, $P = 0.009$; biomass: $F_{1, 8} = 18.0$, $P = 0.003$; Figure 5A, B). In the high-density CWD reach, posttreatment abundance increased significantly ($F_{1, 8} = 7.45$, $P = 0.03$; Figure 5A) and biomass had no significant trend ($F_{1, 8} = 0.91$, $P = 0.37$; Figure 5B). Posttreatment rates of increase in trout abundance and biomass were found to be highest in the low-density reach (abundance: slope = 0.03; biomass: slope = 4.34).
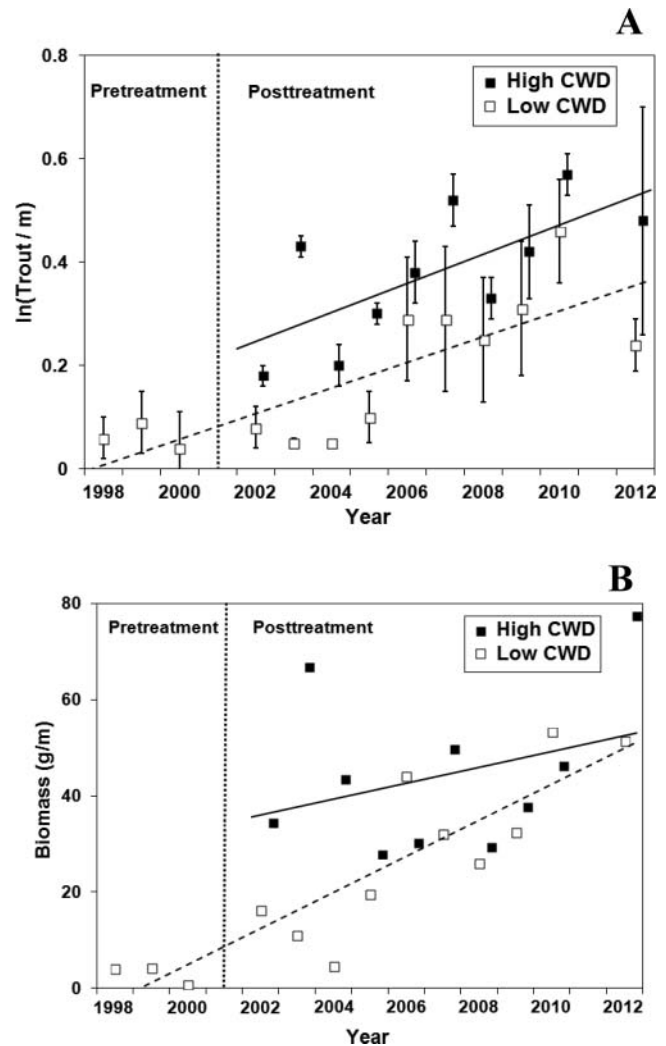
DISCUSSION

This study had several limitations: (1) the inability to separate trout response from the effects of channel reconstruction, CWD, and livestock exclusion in the low-density CWD reach, (2) pretreatment data limitations, especially the lack of fisheries data in the high-density CWD reach, (3) the possible influence of prior restoration on trout response, and (4) the high severity of whirling disease for salmonids other than Brown Trout (Pierce et al. 2014a). Despite these limitations, this study presents one of the few case studies that describe the long-term response of wild trout to comprehensive stream restoration. This study specifically clarifies biological responses to the conversion of an overwidened and degraded stream to a

deep, narrow, more natural channel, as well as changes of fluvial processes as a function of channel shape. This study further provides qualified support that certain low-gradient, meandering, meadow streams (i.e., E4 stream type) may require the addition of a minimal instream woody structure as a long-term (>5 years) wild trout habitat improvement technique.

Stream Channel Morphology

To reestablish natural form and function of Kleinschmidt Creek, restoration began with the removal of instream dams, full channel reconstruction, and the conversion of an overwidened and heavily degraded (C5) stream type to a more natural, vegetated, deep, narrow (E4) stream type. As described by Rosgen (1996), the reconstructed E4 stream type represents the evolutionary end point (i.e., reference condition) for geomorphically stable, self-maintaining channels in gently sloping alluvial valleys where densely rooted vegetation is firmly established. This conversion to a deep and narrow vegetated channel not only created a shift in stream type (Table 1), but also increased transport capacity of fine sediment (Table 1). Because this conversion reduces the potential for aggradation, it helps maintain the stable form and function of gravel-bed stream that has stable vegetated streambanks and a low $W{:}D$ ratio. This increased transport capacity may be especially important in low-gradient, groundwater-dominated streams where high flow (i.e., flushing) events are infrequent. Groundwater-dominated streams are thereby more prone to accumulations of fine sediment from anthropogenic erosion and other forms of channel degradation than are basin-fed streams. Without this conversion to a more hydraulically efficient vegetated stream type, the placement of instream CWD alone would do little to offset basic limiting factors (e.g., low pool frequency, low bank cover, loss of spawning areas) or otherwise mediate sediment and temperature impacts associated with the pretreatment channel (Pierce et al. 2014a).

Though instream wood is widely used to improve salmonid habitat, the use of CWD may be especially effective for habitat improvement in low-gradient (<2%) alluvial stream types with high $W{:}D$ ratios (>12; i.e., morphologically stable C stream types: Rosgen 1996), as well as confined stream types with moderate gradient (>2%) and step-pool morphology (Rosgen 1996; Binns 2003; Gregory et al. 2003; White et al. 2011). In our study area, the placement of CWD was used to improve and diversify trout habitat and provide interim bank cover during the period of vegetative recovery. Though placement of instream wood can clearly increase carrying capacity for trout where bank cover, pool quality, and/or habitat complexity may be limiting (Hunt 1976; Binns 2003; White et al. 2011), the application and efficacy of instream wood for habitat improvement varies widely depending on geomorphic conditions (e.g., stream types: Rosgen 1996; Schmetterling and Pierce 1999; White et al. 2011); although, few fisheries studies

explicitly link wood to geomorphic condition stream types (e.g., stream types: Rosgen 1996; Schmetterling and Pierce 1999; Baldigo et al. 2008). Indeed, we are unaware of any prior published long-term (>5 years) study that links the response of wild trout to the use of instream wood in reconstructed, low-gradient, meandering meadow streams with strong groundwater influence and the prevailing influence of rhizomatous riparian vegetation.

Following channel restoration and grazing changes, perpetual conservation easements were used to ensure long-term vegetative stability and habitat protection given the high sensitivity of the E4 stream type to vegetative disturbance (Rosgen 1996; Figure 2B). For Kleinschmidt Creek, this sensitivity relates to spring seeps, wet stream banks, and high summer flows that significantly elevate the potential for mechanical (e.g., hoof-shear) streambank damage during the summer livestock grazing season. Because of these site conditions, grazing changes and easement protection were considered essential to reestablish a fully vegetated, more natural, self-maintaining stream with sustainable fish habitat.

Trout Response to Restoration

During the study, total trout abundance in the reach with low density CWD increased compared to the reference reaches. Likewise, biomass in the low density CWD reach increased. Elevated biomass, especially in the high-density CWD reach, speaks to the productive nature of spring creeks in general (Decker-Hess 1986, 1987) and the specific presence of large (>500 g) resident Brown Trout in the case of Kleinschmidt Creek. Ironically, the high severity of whirling disease infection in Kleinschmidt Creek may actually favor elevated biomass by favoring resident Brown Trout (Pierce et al. 2014a), a species with natural resistance to *M. cerebralis* (Bartholomew and Wilson 2002), over other salmonids in Kleinschmidt Creek with less resistance and/or greater migratory behavior (Pierce et al. 2014a).

During the initial 4-year recovery period, monitoring identified notable differences in population size between the two fish monitoring sites. Though a lack of pretreatment data limited the strength of between-site comparisons, posttreatment differences were large and biologically revealing. Large differences initially included an 11-times higher abundance and six-times higher biomass in the high-density CWD treatment reach 2 years after restoration was completed (Figure 4A, B). After a 4-year recovery period, differences decreased with estimates of abundance to 1.6 times (range, 1.3–2.2) higher in the high-density CWD reach and only 1.1 times (range, 0.7–1.5) greater biomass in the high-density CWD reach (Figure 4A). These short-term differences indicate movement into and aggregation within pools with complex woody structure (e.g., Hunt 1976; Gowan and Fausch 1996; Dolloff and Warren 2003), which may be especially relevant to Brown Trout given their innate preference for low-gradient streams with undercut

streambanks, abundant cover, and dim light (Wesche 1980; Bachman 1984; Larscheid and Hubert 1992). More incremental long-term (≥5year) changes may reflect compensatory mechanisms driven by density dependence, as indicated by a delayed response in the low-density CWD reach. After this delay, annual estimates of abundance and biomass followed similar trajectories at both monitoring sites (Figure 4A, B).

Though long-term (≥5 year) population trajectories were ultimately similar (Figure 4A, B), we continued to identify biological differences between the two fish population monitoring sites when considering the full monitoring data set. As one example, mean trout abundance was 63% higher (i.e., 0.57 trout/m) in the high-density CWD reach at >5 years posttreatment compared with that (0.36 trout/m) in the low-density CWD reach. In comparison, biomass was only 11% higher (i.e., 45 g/m) in the high-density CWD reach compared with the low-density CWD reach (40 g/m). These differences reflect an increasing percentage of smaller fish in the section with high CWD density (e.g., mean TL = 168 mm, mean weight = 79 g) versus larger fish in the section with low CWD density (e.g., mean TL = 195 mm, mean weight = 122 g). Likewise, the posttreatment CPUE of age-0 trout averaged 26% higher in the high CWD reach although these differences were not significant. Interestingly, all age-1 and older Rainbow Trout, Westslope Cutthroat Trout, and Bull Trout sampled in this study ($n = 10$) were captured in the high-density CWD reach where physical habitat was more diverse, although the proportion of these species within the trout assemblage was very small (Figure 3). Though other factors (e.g., variation in pool depth, shrub cover, or forage availability) could explain some of these differences, such differences were expected and formed the basis of decisions to add instream wood over most of the project area.

Rosgen (1996) suggested the placement of wood within the E stream type provides limited habitat benefit given the low W:D ratios of the stable channel form, lateral habitat forming processes, and increased vegetative bank cover typical of deeper and narrower (E) channels. However, W:D ratios are highly variable (range, 2–12) under the Rosgen geomorphic stream classification for the E stream type (Rosgen 1996). Our stream surveys found W:D ratios within the upper range (i.e., 8.8–9.1) of E stream type classification at the two fish population monitoring sites. Though long-term trout response trends in the low-density CWD reach suggest that wood is a minor habitat feature in the E stream type (Rosgen 1996), higher trout abundance, more diverse population structure, and more species diversity indicate that the addition of wood could be an effective fish habitat improvement. Based on these results, the use of instream wood should be considered when developing restoration goals or species targets, especially at the higher range of W:D ratio values for the E stream types.

After 11 years of posttreatment fisheries monitoring, estimates of abundance and biomass showed no clear indication that wild trout populations have reached equilibrium at either treatment site (Figure 4A, B). This indicates long-term recovery periods (>10 year) may be required in fully reconstructed streams where recovery of biotic communities, including maturation of riparian vegetation, occur over many years before upper trophic aquatic species (e.g., age-1 and older wild trout) reach equilibrium with fully restored streams. In addition to the essential role of vegetation in habitat creation and maintenance for the E stream type, trout population recovery rates in vegetated streams further relate to food-web pathways, including increased aquatic and terrestrial prey (e.g, macroinvertebrates) used by trout. In the case of Kleinschmidt Creek, terrestrial prey may be especially relevant because terrestrial prey can significantly increase following reductions in grazing pressure and corresponding increases in riparian vegetation (Saunders and Fausch 2007).

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