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Effects of wildfire on stream temperatures in the Bitterroot River Basin, Montana

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Abstract. Wildfire is a common natural disturbance that can influence stream ecosystems. Of particular concern are increases in water temperature during and following fires, but studies of these phenomena are uncommon. We examined effects of wildfires in 2000 on maximum water temperature for a suite of second- to fourth-order streams with a range of burn severities in the Bitterroot River basin, Montana. Despite many sites burning at high severity, there were no apparent increases in maximum water temperature during the fires. One month after fire and in the subsequent year, increases in maximum water temperatures at sites within burns were 1.4–2.2°C greater than those at reference sites, with the greatest differences in July and August. Maximum temperature changes at sites >1.7 km downstream from burns did not differ from those at reference sites. Seven years after the fires, there was no evidence that maximum stream temperatures were returning to pre-fire norms. Temperature increases in these relatively large streams are likely to be long-lasting and exacerbated by climate change. These combined effects may alter the distribution of thermally sensitive aquatic species.

Additional keywords: aquatic ecosystems, disturbance, recovery, watershed.

Introduction

Many anthropogenic and natural disturbances influence stream characteristics (Kauffman and Krueger 1984; Johnson and Jones 2000; Dunham *et al*. 2007). Fire is a frequent natural disturbance that has both immediate and long-term consequences for stream ecosystems because it affects water temperature (Minshall *et al*. 1997), nutrient dynamics (Spencer *et al*. 2003), channel morphology (Dunham *et al*. 2007), stream biota (Minshall 2003; Pilliod *et al*. 2003; Burton 2005), and habitat complexity and structure (Meyer and Pierce 2003; Wondzell and King 2003). Nevertheless, the magnitude and extent of the initial effects of fires on streams and the rates of recovery to pre-fire conditions are poorly understood (Gresswell 1999; Spencer *et al*. 2003). Because wildfires have become more pervasive in recent years in the western USA and climate change is expected to increase their severity and frequency (Fagre *et al*. 2003; Westerling *et al*. 2006), addressing this knowledge gap is becoming more critical.

Fire-related stream warming has been of particular concern because water temperatures affect primary and secondary productivity, and the vital rates, distribution and diversity of stream biota (Minshall *et al.* 1989; Taniguchi and Nakano 2000; Pilliod *et al*. 2003). In the Rocky Mountains, some cold-adapted taxa such as bull trout (*Salvelinus confluentus* (Suckley); Selong *et al*. 2001), westslope cutthroat trout (*Oncorhynchus clarkii lewisi* (Girard); Bear *et al*. 2007), and tailed frogs (*Ascaphus* spp.; Pilliod *et al*. 2003) are near the southern extremes of their range. Because their distribution in this area is dependent on stream temperature (Hawkins *et al*. 1988; Paul and Post 2001; Dunham *et al*. 2003*a*), fire-related increases may have adverse effects on some populations.

Although fires may alter water temperatures by affecting the magnitude and characteristics of surface and subsurface flows, and rates of evaporation, convection and conduction (Webb *et al*. 2008), the primary driver is usually greater solar radiation following the loss of riparian vegetation, as is the case for streams that have undergone riparian canopy removal during forest harvest (Brown 1969; Johnson and Jones 2000; Johnson 2004; Caissie 2006). In addition, severe wildfires may increase spring runoff and lead to debris flows, resulting in wider, shallower stream channels that absorb more solar radiation (Dunham *et al*. 2007).

Yet pre- and post-fire assessments of stream temperatures, as well as monitoring during fires, are relatively uncommon

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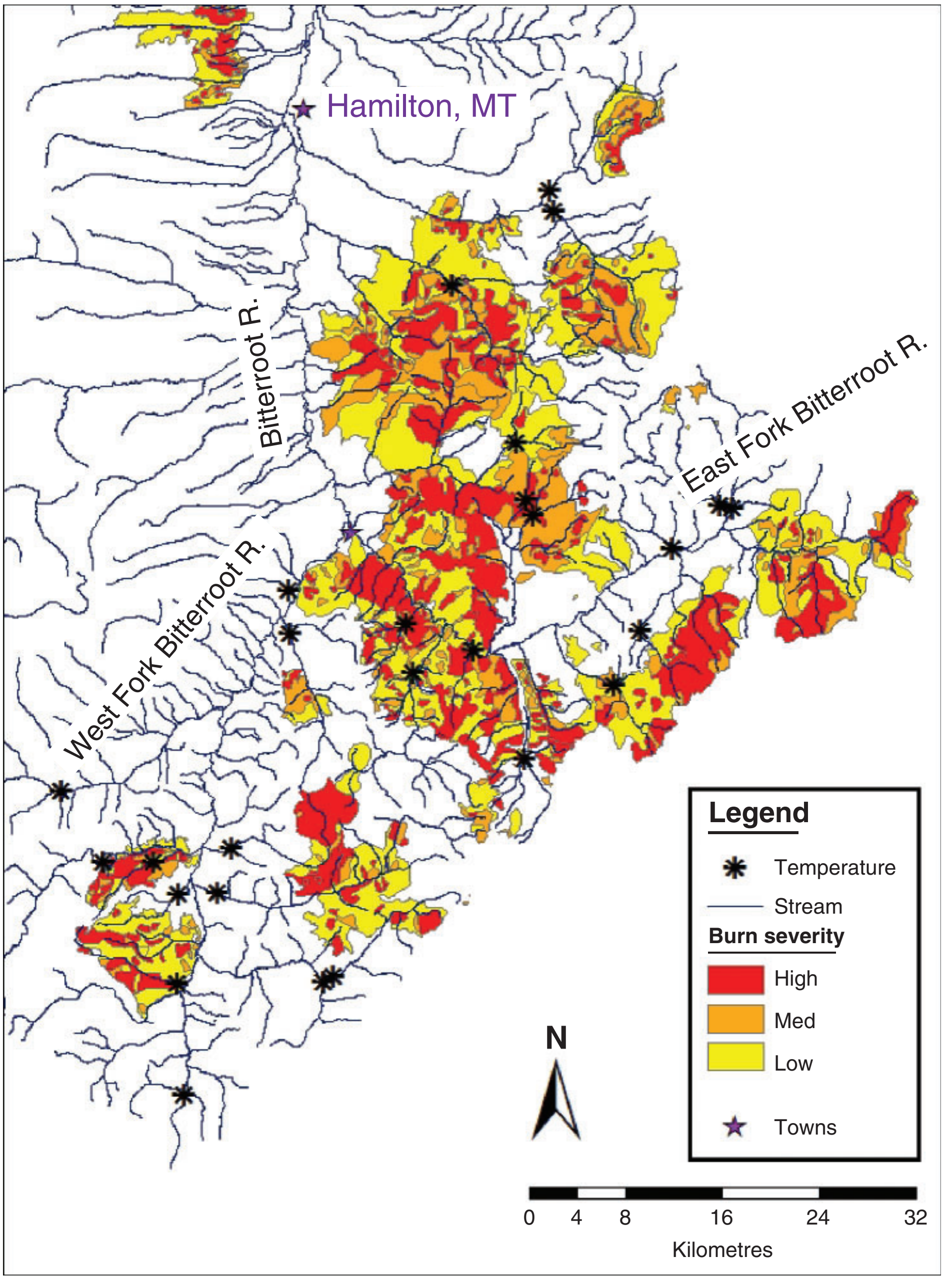

Fig. 1. Temperature logger sites (asterisks) and 2000 burn severity in the southern Bitterroot River basin, Montana. Red, orange and yellow indicate areas of high, medium and low burn severity. White indicates areas that did not burn.

because fires are spatially and temporally unpredictable. Some studies have examined fire effects retrospectively (e.g. Royer and Minshall 1997), but the lack of a pre-fire baseline makes it difficult to quantify temperature changes related to fire. Studies that have reported increased summer maximum water temperatures in the first year following fire have tended to focus on first- or second-order headwater streams (Amaranthus *et al*. 1989; Burton 2005; Dunham *et al*. 2007; but see Minshall *et al*. 1997; Sestrich 2005). In addition, little work has been done on post-fire recovery of stream temperatures (Dunham *et al*. 2007) or the downstream extent of elevated water temperatures associated with wildfire (Amaranthus *et al*. 1989). Manipulative experiments describing stream temperature budgets and dynamics (Story *et al*. 2003; Johnson 2004) and research on the effects of streamside forest harvest (Moore *et al*. 2005) imply that if a wildfire burns in only a portion of a basin, increases in temperature during or after a fire may be confined to stream segments in the vicinity of the burn.

In 2000, several large wildfires burned in the Bitterroot River basin, Montana. Sestrich (2005) documented increased water temperatures in burned watersheds following these fires. We examined these and additional second- to fourth-order streams to address four objectives: (1) whether stream temperatures in burned watersheds increased during the fires; (2) whether and when sites in and downstream from burned areas warmed; (3) at sites adjacent to riparian area burns, whether maximum temperatures declined towards pre-fire levels over time; and (4) whether stream temperatures in unburned watersheds were increasing over time.

Methods

Study area

The study area is located in the Bitterroot River basin in western Montana, USA. The Bitterroot River basin is 7394 km^2 and forests are dominated by stands of Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco)), Engelmann spruce (*Picea engelmannii* (Parry ex Engelm.)) and lodgepole pine (*Pinus contorta* (Douglas ex Louden)). In summer 2000, fires of varying severity burned 1244 km^2 of the southern portion of the basin; most of these fires were ignited on 31 July, burned throughout August, and were largely extinguished by rainfall in early September (Bitterroot National Forest 2000). Immediately after these fires, there were only two watersheds (Little Sleeping Child Creek and Laird Creek) within this study area that experienced large-scale landslides involving channel reorganisation events (Bitterroot National Forest 2000).

The Bitterroot River basin is in a temperate, snowmelt-dominated ecosystem, with streams typically covered with ice or snow from November to March and reaching peak flows in May and June. Summertime air temperature during the post-fire period (2001–07) at a central location in the study area averaged 15.4°C (Sula, MT; http://www.wrcc.dri.edu/index.html, accessed 21 February 2011). Additionally, there were no statistically significant trends in maximum air temperatures in July (average maximum 30°C), August (average maximum 28°C) or September (average maximum 22°C). Data from a gauging station within the study area (West Fork Bitterroot River at Conner, MT; http://waterdata.usgs.gov/mt/nwis/inventory/?site_no=12342500&amp, accessed 4 April 2010) similarly indicated variation but no trends across this time period in mean daily discharge in July, August and September.

Since 1993, biologists with Montana Fish, Wildlife and Parks and the Bitterroot National Forest have deployed temperature loggers at fixed sites in streams throughout the basin (Fig. 1). Onset Hobo temperature loggers (Onset Computer Corporation, see http://www.onsetcomp.com/products/) (accuracy ±0.2°C) were calibrated and placed in pools or runs shielded from direct solar radiation. Stream temperature was recorded every 2 h from 19 July to 30 September between 1993 and 2007, but not all sites were measured every year. There is a higher likelihood of underestimating maximum temperatures in streams with larger daily fluctuations. The maximum daily fluctuations across streams in our dataset varied from 2.1 to 11.4°C, resulting in a potential bias of up to 2% in detecting the maximum daily temperature with a 2-h sampling interval (Dunham *et al*. 2005). Finally, general characteristics, such as wetted widths, have been measured but there are no quantitative measures of historical and current canopy cover at these sites.

Analyses

We retrieved topographic map layers (resolution 1 : 24 000) and stream GIS layers from the Montana State Library, Natural Resource Information System (http://nris.state.mt.us). Fire severity was mapped by the US Forest Service using helicopter reconnaissance and ground visits to validate remote sensing data and was based on agency guidelines (US Forest Service 1995 under burned-area emergency rehabilitation). We obtained these fire severity GIS layers from the Bitterroot National Forest Office, Hamilton, MT. For every stream site, we determined area burned within the watershed and the area of watershed above the site using spatial analysis tools in *ArcGIS* (version 9.1). We used the distance tool in *ArcGIS* to determine descriptive indices associated with the spatial distribution of these fires (e.g. distance from the lower edge of the burn to the nearest downstream temperature site, percentage of channel length disturbed, and severity of burn at and above temperature logger sites). We ran the *TauDEM* terrain analysis program (http://hydrology.neng.usu.edu/taudem/taudem2.0/taudem.html) to create a stream layer with an attribute table to determine temperature logger site characteristics such as slope and elevation.

To gauge the effects of these fires on water temperatures, we divided sites into a reference group, in which 0–6% of the watershed burned but only in upland areas, and two fire-affected groups: sites within a riparian area burn (within-burn) and sites downstream of a riparian area burn (below-burn). At within-burn sites, fires of differing severity affected 34 to 100% of the watershed upstream of the site. Below-burn sites were 1.7–6.9 km downstream of the fire perimeter and 20–95% of the watershed above these sites burned (Table 1). To avoid pseudoreplication, we included only one site per stream. If we included sites on tributaries of a stream with a site already in the analyses, the main-stem site was upstream of the tributaries to avoid their influence. Sites varied in other aspects of their geomorphology (width and slope) and the percentage of the stream length above the logger site where the riparian zone was burned (Table 1).

We used differences in maximum water temperature as a metric because maximum temperature is highly correlated with mean water temperature (Dunham *et al*. 2005), is likely to respond to changes in solar radiation (Moore *et al*. 2005) and is associated with native fish distributions (Dunham *et al*. 2003*a*). In addition, differencing emphasises temporal variation in maximum temperatures at a site rather than absolute differences between sites (Johnson and Jones 2000; Quinn and Wright-Stow 2008). Data for July, August and September were analysed separately to account for seasonal variation. Because not all sites were measured during the same years, sample sizes differed among analyses. For statistical comparisons, we used temperatures from 1999 to represent pre-fire conditions because more sites were monitored in 1999 than in other pre-fire years (1993–98), although findings were similar when based on other years (results not shown).

We used three methods to evaluate stream temperatures during the August 2000 fires. We did not always know when

Table 1. Characteristics of water-temperature monitoring sites in the Bitterroot River basin, Montana

If the proportions of the riparian burn associated with low, medium (med) or high were similar within ~10%, then we described the dominant burn severity as mixed

Stream	Wetted width (m)	Order	Elevation (m)	Channel gradient	Watershed burned (%)	Distance from burn (m)	Dominant burn severity above site (at site)	Percentage of channel length above site that burned
Reference								
Bertie Lord	1.7	4	1530	0.021	6			
West Fork Bitterroot	3.9	3	1792	0.045	0			
Blue Joint	7.4	4	1573	0.033	0			
W. Fk. Camp	2.9	3	1554	0.039	0			
Martin	8.2	4	1342	0.017	3			
Mine	3.0	4	1707	0.021	0			
Moose	6.1	4	1681	0.020	5			
Nez Perce Fork	7.5	4	1548	0.010	0			
Pierce	0.9	3	1305	0.055	0			
Below burn								
Coal	2.3	3	1530	0.032	40	1842	Low (no)	78
Daly	8.0	4	1448	0.017	25	6846	Mixed (no)	53
Hughes	5.5	4	1713	0.022	25	6522	Mixed (no)	28
Meadow	4.7	4	1806	0.023	75	2209	High (no)	73
Overwhich	7.7	4	1329	0.010	37	5100	Mixed (no)	41
Piquett	4.9	4	1509	0.035	20	3677	Mixed (no)	23
Skalkaho	7.8	4	1347	0.019	34	1733	Low (no)	79
Slate	4.5	3	1493	0.020	40	6850	High (no)	57
Within burn								
Cameron	3.5	4	1585	0.022	40	0	Mixed (med)	78
Chicken	3.7	3	1598	0.026	80	0	High (low)	80
Hart	0.6	3	1560	0.039	100	0	High (high)	100
Laird	3.4	4	1347	0.051	80	0	Mixed (high)	77
Little Blue Joint	2.4	3	1573	0.049	80	0	Mixed (high)	78
Maynard	2.0	3	1363	0.104	100	0	Mixed (med)	100
Rye	3.5	4	1646	0.022	61	0	Low (low)	72
Tolan	3.0	4	1449	0.057	86	0	Mixed (med)	100
Sleeping Child	7.1	4	1731	0.027	60	0	Mixed (med)	63
Warm Springs	7.8	4	1381	0.028	40	0	Mixed (low)	34

particular locations within each watershed burned, thus we scanned the data for daily maximum temperature anomalies during August 2000 at within-burn sites. In addition, we used one-way ANOVA and Tukey's pairwise Honestly Significantly Different (HSD) test to determine whether maximum temperature differences between August 1999 and August 2000 significantly differed among reference, below-burn and within-burn sites. We also noted whether water temperature surpassed 20°C at any site, a temperature at which sublethal effects may become evident in some species (e.g. bull trout; Selong *et al*. 2001).

We employed a before–after control–impact (BACI) design to evaluate immediate post-fire effects on maximum water temperature (Smith 2002). To evaluate the immediate effect of fires, we used one-way ANOVA and Tukey's pairwise HSD tests to compare maximum temperature differences between September 1999 and September 2000 (the first post-fire month) and between all 3 months in 1999 and 2001 for reference, below-burn and within-burn sites. We also noted whether maximum temperatures reached or exceeded 20°C at any site before or after the 2000 fires.

To evaluate the longer-term effects of fire on temperature, we determined whether within-burn sites were recovering towards pre-fire norms, using regression to evaluate trends in mean maximum temperature differences between unburned ($n = 10$) and within-burn sites ($n = 6$) from 2001 to 2007. A significant negative regression slope would be interpreted as evidence of a return to pre-fire values, and complete recovery would be indicated by a maximum temperature difference approaching the mean of pre-fire differences in 1999 and 2000 (0.1°C, range 0.0–0.2°C) for these sets of streams. In this analysis, unburned sites were the pooled sample of reference and below-burn sites, because post-fire maximum temperatures differences were not significantly different between these groups. We used ANOVA and Tukey's HSD tests to assess seasonal variation in post-fire maximum temperature differences. Finally, to ascertain whether reference stream temperatures exhibited unidirectional trends that might be attributable to climate, we used regression to assess overall trends in maximum water temperature by month for three unburned sites with data from 1994 to 2007.

Results

We did not detect short-term spikes in water temperature at fire-affected sites in August 2000 during the wildfires (Fig. 2). In addition, the increase in maximum temperature between August

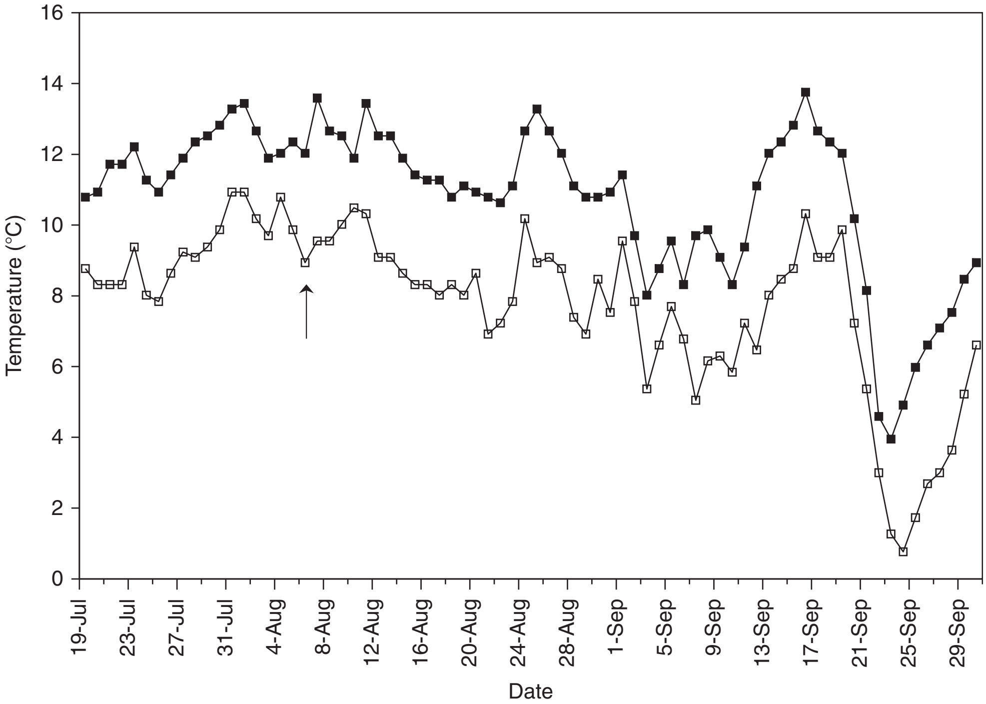

Fig. 2. Daily maximum (filled squares) and minimum (empty squares) temperatures in Laird Creek in 2000. The arrow denotes when the forest adjacent to the temperature logger was burned by a high-severity fire (Bitterroot National Forest 2000).

Table 2. Differences in mean maximum water temperatures (°C) before (1999) and after (September 2000, July–August 2001) the 2000 fires

Standard errors are in parentheses. Positive values denote temperature increases between the earlier and later period. Samples sizes varied among comparisons (September 1999–September 2000: reference, seven sites; below-burn, three sites; within-burn, five sites; remaining comparisons: reference, eight sites; below-burn, five sites; within-burn, four sites) because not all sites were measured in all years. Differences in superscript letters indicate significant differences ($P < 0.05$) between site groups

	Sites		
Period	Reference	Below burn	Within burn
September 1999–September 2000	2.1^{a} (0.1)	2.3^{a} (0.2)	3.5^{b} (0.2)
July 1999–July 2001	-0.1^{a} (0.1)	1.0^{ab} (0.7)	1.9^{b} (0.6)
August 1999–August 2001	1.8^{a} (0.1)	2.1^{a} (0.5)	3.7^{b} (0.4)
September 1999–September 2001	3.2^{a} (0.1)	3.0^{a} (0.2)	5.4^{b} (0.6)

1999 and August 2000 was not significantly different among reference (mean = 2.2°C, s.e. = 0.2, $n = 5$), below-burn (mean = 2.1°C, s.e. = 0.5, $n = 4$) and within-burn (mean = 2.3°C, s.e. = 0.1, $n = 7$) sites. Maximum temperatures never exceeded 18.5°C at any site in August 2000.

Temperature effects were, however, apparent immediately after the fires. Mean maximum temperatures increased more from September 1999 to September 2000 at within-burn sites than at sites not directly influenced by fire, and these increases were also evident when comparing monthly values between 1999 and 2001 (Table 2). There were no significant differences in maximum temperature changes between reference and below-burn sites in any comparison. We observed differences among groups in the number of sites exceeding 20°C before and after the 2000 fires. The highest recorded temperature in any stream from 1993 to 1999 was 18.4°C. The highest recorded temperature at reference sites ($n = 9$) from 2001 to 2007 was 18.6°C, whereas 60% of within-burn sites exceeded 20°C in either July or August (with a maximum temperature of 22.0°C in July and 21.7°C in August), as did 14% of below-burn sites.

There was substantial variation both among streams and years, but post-fire recovery of water temperatures at within-burn sites was not evident (Fig. 3). Regression analysis failed to detect a significant decline in the mean maximum difference between unburned and within-burn sites from 2001 to 2007 in July ($P = 0.88$), August ($P = 0.61$) or September ($P = 0.34$; Fig. 4). Maximum water temperature differences between unburned and within-burn sites were significantly greater in July (2.0°C) and August (1.7°C) than in September (1.3°C; $P < 0.05$ for both comparisons). With respect to unburned sites,

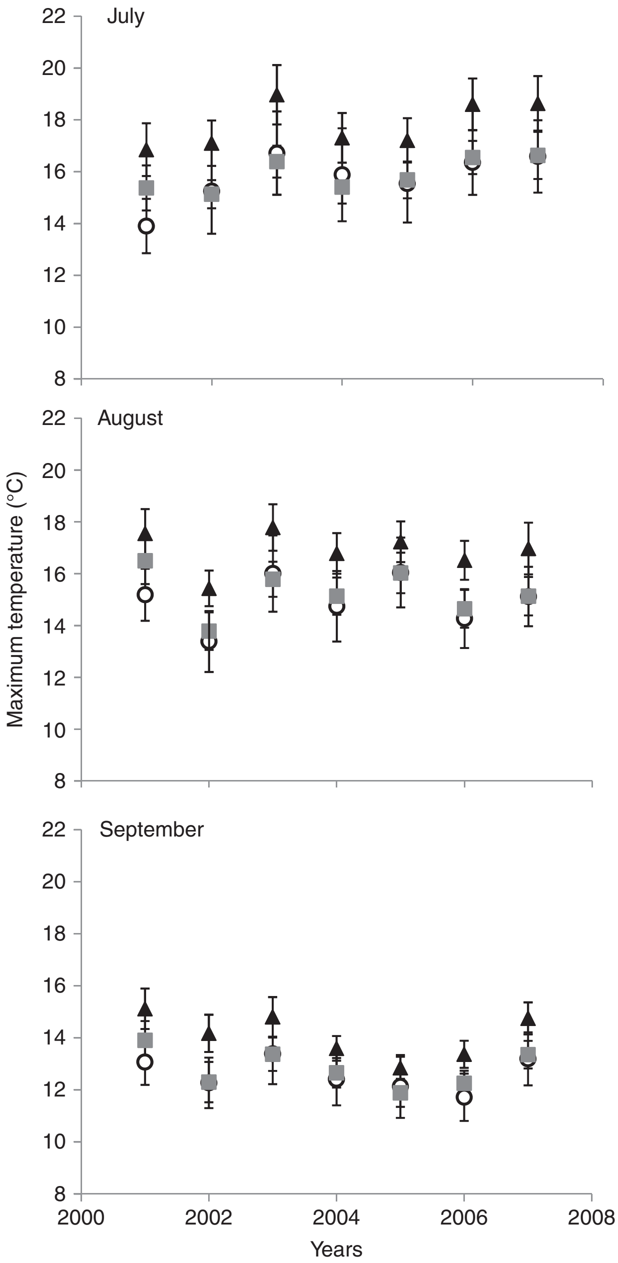

Fig. 3. Maximum summertime temperature unburned (empty circles, $n = 3$), below-burn (grey squares, $n = 7$) and within-burn (filled triangles, $n = 5$) sites for July, August, and September. Error bars are one standard error.

water temperatures significantly increased from 1994 to 2007 at the Meadow Creek site in July (2.6°C, $R^2 = 0.49$, $P = 0.006$) and September (1.9°C, $R^2 = 0.33$, $P = 0.033$) and at the West Fork Bitterroot River site in July (2.1°C, $R^2 = 0.43$, $P = 0.011$), but not for other months at these sites or for any month at the Skalkaho Creek site.

Discussion

We did not detect short-term spikes in stream temperature in August 2000 during the fires, despite that several of the sites

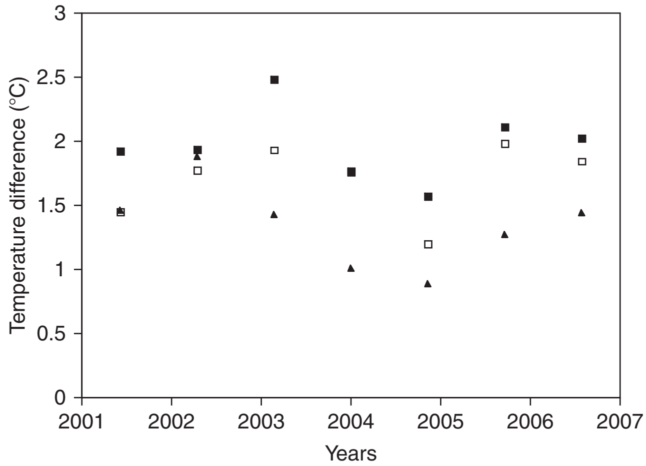

Fig. 4. Post-fire maximum temperature differences (°C) between unburned and burned sites (°C) for July (filled squares), August (empty squares), and September (filled triangles) from 2001 to 2007.

exposed to fire burned at high severity. It is possible that the interval between temperature measurements in this study precluded detection of thermal peaks during the fires, but it seems unlikely that the entire warming event would be missed because conduction from the recently burned stream banks to the water column would have sustained high temperatures for several hours (Feller 1981; Hitt 2003). A more plausible explanation is that watershed size moderated temperature increases because not all portions of a basin were simultaneously burning and because the mostly third- and fourth-order streams in our sample exhibited greater resistance to direct radiative heat transfer (Webb *et al*. 2008) than the mostly smaller streams previously studied (Cushing and Olson 1963; Hall and Lantz 1969; Feller 1981). In addition, smoke during the August 2000 fires may have diffused direct solar radiation and prevented immediate post-fire temperature increases at within-burn sites despite their loss of riparian vegetation.

Nevertheless, by September 2000 and in subsequent years, increases in maximum temperatures were ~1–3°C greater at within-burn sites than at below-burn and reference sites. These temperature increases were comparable to those in other streams affected by fire (Helvey 1972; Amaranthus *et al*. 1989; Hitt 2003; Dunham *et al*. 2007) or riparian timber harvest (Hetrick *et al*. 1998; Macdonald *et al*. 2003; Moore *et al*. 2005). The significantly greater differences in July and August relative to those in September (cf. Johnson and Jones 2000) probably reflect the waning influence of direct solar radiation on air, soil and water temperatures caused by the seasonal decline in sun angle (Johnson and Jones 2000; Danehy *et al*. 2005; Flint and Flint 2008). Although stream size, landscape position and fire characteristics also affect temperature responses (Dunham *et al*. 2007), similar analyses with our study streams were precluded because of small sample sizes.

We observed no differences in post-fire maximum temperature increases between reference and below-burn sites, implying that warming associated with burned areas was fairly localised. All of our below-burn sites were over 1.5 km from riparian burns, and temperature declines of 0.5–2.0°C 100 m^{-1} are common as streams flow from unshaded to shaded reaches (Zwieniecki and Newton 1999; Rutherford *et al*. 2004; Rayne *et al*. 2008). If water temperatures near burn perimeters were on average warmed by 1.5–2.0°C (typical of the within-burn sites), cooling attributable to evaporation, hyporheic exchange and conduction to the substrate (Moore *et al*. 2005) in the unburned downstream reaches could have returned temperatures to pre-fire norms over relatively short distances.

There was no indication that maximum water temperatures at within-burn sites were decreasing 7 years after the fires. This pattern has been observed in the first few years following fire or riparian timber harvest in other watersheds (Feller 1981; Macdonald *et al*. 2003), typically over longer intervals (11 years, Dunham *et al*. 2007; 15 years, Johnson and Jones 2000). In some cases, maximum temperature differences between reference and disturbed sites in these studies lessened or disappeared as deciduous vegetation was re-established. The larger fire-affected streams in the present study may be slower to respond because pre-fire riparian stands consisted largely of conifers, and even where forbs, grasses and shrubs were present, their initial regrowth was probably less effective in reducing solar radiation than in smaller streams (Quinn and Wright-Stow 2008). In addition, because these are relatively high-elevation, snowmelt-dominated systems, recovery of all forms of riparian vegetation will be slower than in lower-elevation, rain-dominated systems (Moore *et al*. 2005). Although additional shade may be provided at within-burn sites when fire-killed trees accumulate as downed wood in and over the channels (McDade *et al*. 1990), it may be several decades before maximum temperature differences between burned and unburned sites return to pre-fire levels.

The biotic consequences of the post-fire maximum temperature increases in this basin are uncertain. Although temperatures occasionally reached levels (>20°C) that may have sublethal effects for some members of the aquatic biota, the warmer temperatures and greater light levels might also be associated with a longer growing season that favours other native species (Dunham *et al*. 2007). These conditions may explain the rapid rebound, after immediate post-fire declines, of westslope cutthroat trout in several of the fire-affected streams (Sestrich 2005). Over the relatively short period covered by the present study, however, maximum temperatures in streams unaffected by fire also appeared to be increasing. Such increases are expected to be sustained or continue under most climate change scenarios (IPCC 2007), and coupled with climate-driven increases in fire effects (Westerling *et al*. 2006), may shrink and fragment suitable thermal environments for the native coldwater fauna such as bull trout, whereas increasingly warm and connected stream networks may serve as conduits for invasions by other native and non-native species with broader temperature tolerances (Dunham *et al*. 2003*b*; Rieman *et al*. 2007).

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