

soils & hydrology

Influence of Variable Streamside Management Zone Configurations on Water Quality after Forest Harvest

Emma L. Witt, Christopher D. Barton, Jeffrey W. Stringer, Randall K. Kolka, and Mac A. Cherry

Streamside management zones (SMZs) are a common best management practice (BMP) used to reduce water quality impacts from logging. The objective of this research was to evaluate the impact of varying SMZ configurations on water quality. Treatments (T1, T2, and T3) that varied in SMZ width, canopy retention within the SMZ, and BMP utilization were applied at the watershed scale to two watersheds each. Watersheds with wider SMZs (T3: 110 ft, 100% canopy retention) and improved crossings were not significantly different from unharvested control (C) watersheds for all parameters except nitrate and diurnal stream temperatures. Changes in total suspended solids, turbidity, nitrate, dissolved oxygen, and maximum stream temperature were detected for watersheds treated with the current recommended minimum SMZ configurations (T1: 55 ft, 50% canopy retention) and watersheds with unharvested SMZs (T2: 55 ft, 100% canopy retention) and improved crossings.

Keywords: headwater streams, total suspended solids, best management practices, forest harvest

Forestry best management practices (BMP) were developed to minimize nonpoint source pollution from silvicultural activities (Aust and Blinn 2004). Water quality degradation due to changes in sediment, nutrients, dissolved oxygen (DO), and stream water temperatures has been associated with forestry activities (Binkley and Brown 1993). Streamside management zones (SMZs) are designated areas where management recommendations are provided to minimize sediment delivery,

regulate temperature fluctuations, and filter surface runoff (Stringer and Perkins 2001). These management recommendations can include equipment limitations, canopy and basal area retention limits, and chemical use guidelines (Aust and Blinn 2004).

Forests have several characteristics that promote good surface water quality, including high infiltration rates, litter layers that minimize exposed mineral soil, efficient nutrient cycling, stream temperatures moderated by canopy shading, and high DO con-

centrations (Brown 1973, Binkley and Brown 1993, Richardson and Danehy 2007). Forest harvesting operations can result in negative impacts to water quality, including increased total suspended solid (TSS) concentrations resulting from soil compaction, litter disturbance, larger peak flows and elevated flow durations resulting from reductions in evapotranspiration and interception, and increased connectivity of the hydrologic and sediment delivery systems via skid trails and harvest roads and stream crossings (Troendle and Olsen 1993, Wemple et al. 1996, Lacey 2000, Wynn et al. 2000, Croke et al. 2001, Bracken and Croke 2007, Litschert and MacDonald 2009, Witt et al. 2013). In addition, nitrate export may increase because of decreased uptake by plants, increases in mineralization and nitrification rates, and decay of residues remaining from harvesting activities (Likens et al. 1970, Wynn et al. 2000, Prescott 2002, Mayer et al. 2007). Riparian canopy removal can cause increased water tempera-

Received August 29, 2014; accepted August 27, 2015; published online October 22, 2015.

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Acknowledgments: We express gratitude to those at Robinson Forest who contributed to the project including Matt Strong, Aaron Klee, Daniel Bowker, David Collett, Ted Sizemore, Chris Reeves, Will Marshall, and Jim Marshall. This work was supported in part by the Kentucky Agricultural Experiment Station.

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tures, larger diel temperature fluctuations, and decreased minimum temperatures (Brown 1973, Lynch et al. 1984, Beschta et al. 1987, Richardson and Danehy 2007). DO concentrations may also be negatively affected by harvesting as a result of decreased solubility that accompanies increased water temperature and increased fine organic matter inputs from logging debris, which can stimulate biologic oxygen demand (Binkley and Brown 1993, Ice 2008).

SMZs have been shown to be effective in mitigating the impacts of harvesting on TSS, nitrate, DO, and temperature (Binkley and Brown 1993). Previous studies have shown the effectiveness of BMP in mitigating harvest-induced changes in sediment and nitrate (Arthur et al. 1998, Lakel et al. 2010) as well as temperature and water quality impacts (Swift and Baker 1973, Aubertin and Patric 1974, Clinton 2011). In each of these cases, reduced impacts to surface water were found in watersheds harvested with BMP that included SMZs relative to watersheds harvested without BMP.

Most states in the Appalachian Region currently have two specifications associated with SMZs: one related to the width of the SMZ and a second relating to the allowable harvest within the SMZ (Stringer and Thompson 2000). For perennial streams, the minimum permitted distance to severe disturbance increases as upland slope increases due to the higher potential of surface runoff impacts with higher upland grades (Trimble and Sartz 1957, Stringer et al. 1998). Most states allow some amount of overstory removal within the SMZ. For example, Kentucky allows 50% overstory removal within their SMZs. The zone width is also dependent on upland slope. For intermittent streams, the distance to the nearest severe disturbance (generally a road or landing) is also related to upland slope, but the distance is shorter relative to perennial streams and 100% harvest is allowed within the SMZ. Finally, no SMZ is required for ephemeral streams in Kentucky (Stringer et al. 1998). Logging debris must be removed from all channels so that flow is not impeded. Elevated crossings (e.g., bridge or culvert) must be used where feasible to cross all stream types. Other riparian zone requirements vary considerably among states (Stringer and Thompson 2000). For example, North Carolina requires that 75% of the trees remain in the perennial and intermittent riparian zone after harvest. West Virginia and Pennsylvania allow 100% harvest

within the riparian zone on both perennial and intermittent streams. Ohio allows 100% harvest up to the stream bank but requires that a stringer of shade trees be left along all stream types. The differences in SMZs among states do not necessarily reflect the best available knowledge but are the culmination of compromises among forestry groups, environmental groups, and policymakers within each state (Stringer and Thompson 2000).

Although BMP use has been generally adopted across the United States, information on SMZ effectiveness and watershed and riparian processes is still needed (Anderson and Lockaby 2011). Recommendations on whether SMZs should further address ephemeral and intermittent streams are also needed. Many state BMP programs emphasize the necessity for best professional judgment that may result in going beyond mandated recommendations in the planning and implementing of silvicultural practices to optimize the efficacy of forestry BMP, but specific details on how that is performed are lacking. Given these conditions, a study was initiated with the objective of quantifying the impact of forest harvesting on water quality under three SMZ configurations applied at the watershed scale. Configurations include the minimal state-mandated BMP and two others with increasingly more restrictions and potential protection for water resources.

Methods

Site Description

This study was conducted at the University of Kentucky's Robinson Forest (37°27' N latitude and 83°08' W longi-

tude), located in the Cumberland Plateau physiographic region of southeastern Kentucky. The forest is approximately 15,000 acres and was harvested by the Mowbray-Robinson Lumber Company between 1890 and 1920 (Overstreet 1984). The regenerated forest is categorized as mixed mesophytic forest dominated by oak (*Quercus* spp.), hickory (*Carya* spp.), maple (*Acer* spp.), and yellow-poplar (*Liriodendron tulipifera*). The topography is highly dissected, with narrow stream bottoms, elevation ranges of 850 to 1,510 ft, and steep side slopes ranging from 35 to 90% (Coltharp and Springer 1980).

The climate of Robinson Forest is temperate-humid-continental with warm summers and cool winters. The average annual precipitation for southeastern Kentucky is 45.8 in., whereas the 26-year average for three precipitation collectors at Robinson Forest was 46.3 in. (Cherry 2006). Average monthly precipitation is 3.85 in. March tends to be wetter than average and October tends to be drier.

Eight first-order perennial watersheds were included in this study (Figure 1). Each was located in the 3,800-acre Clemons Fork watershed and equipped with either a 3:1 broad-crested weir or H-flume at the perennial outlet and a trapezoidal or cut-throat flume at the perennial-intermittent transition. Active channel widths range from 10 to 20 ft in perennial channels and <10 ft in intermittent channels. Ephemeral streams had defined channels along at least part of their course, if not the entirety, and were not old gullies. Channel substrates in ephemeral streams were bedrock with large cobbles and gravel common. Watersheds ranged in area

Management and Policy Implications

Use of streamside management zones (SMZs) to protect water quality from logging is a common practice, but information on their effectiveness is limited. Moreover, strategies to strengthen existing SMZ guidelines should be explored in light of uncertainty associated with future resource use and potential alterations in storm frequency and intensity associated with climate change. In this study, we found that SMZ recommendations should focus on minimizing the connectivity of sediment sources to the hydrologic system during storm events. Further, we found that our treatment T1 (55-ft width and 50% overstory retention) resulted in limited alterations to many stream parameters. However, a combination of increasing SMZ width and SMZ canopy retention and use of elevated crossing of ephemeral streams (T3) exhibited sediment levels and many water quality parameters that were similar to those of unharvested control watersheds in both baseflow and storm flow conditions. These results suggest that attention to overstory retention and setbacks are warranted for harvest operations around sensitive, high ecological values, or pristine streams. Although this study was conducted in the steep-sloped forested lands of eastern Kentucky, these results are applicable throughout the Cumberland Plateau and to similar physiographic regions with moderate to steeply sloping topography.

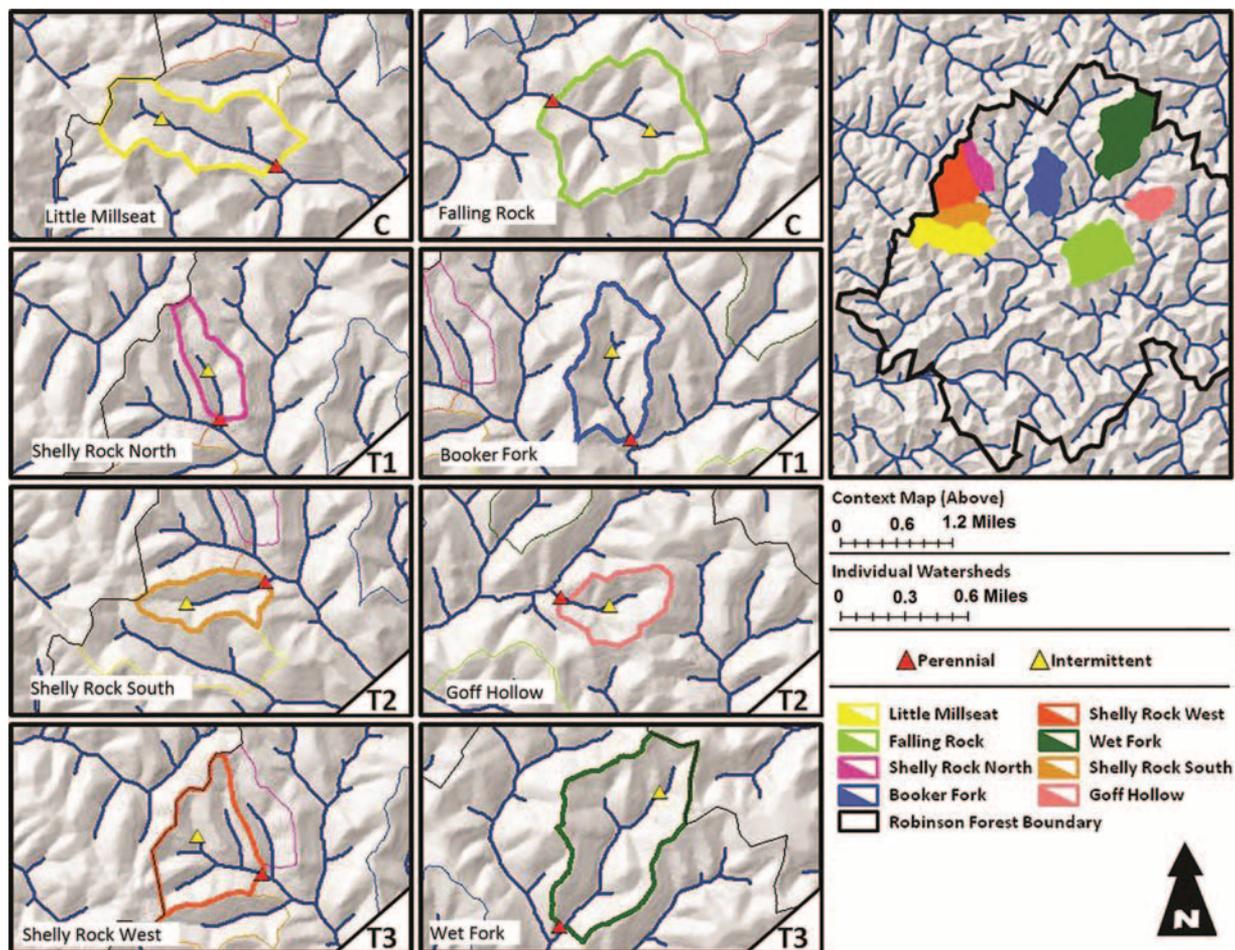


Figure 1. Robinson Forest is an approximately 15,000-acre experimental forest located in portions of Breathitt, Perry, and Knott counties in Kentucky (37°27' N latitude and 83°08' west longitude). This study was conducted on the main block (outlined in black) in the 3,800-acre Clemons Fork watershed of Robinson Forest.

from 67 to 277 acres. Six watersheds were harvested from June 2008 to October 2009. The remaining two watersheds remained unharvested to serve as controls. Treatment watersheds were harvested using a shelterwood with reserves or a two-aged deferment harvest (Smith et al. 1989, Miller et al. 2006) with a target postharvest basal area of approximately 15 ft²/ac. After harvest, the sites were allowed to regenerate naturally. Fertilizers and herbicides were not used. Harvesting equipment included wheeled cable and grapple skidders (John Deere models 540 and 648 and Caterpillar models 525 and 545), tracked dozers (John Deere models 650, 700, or 800), and tracked feller-bunchers (Timbco swing-armed feller-bunchers model 445 or 445EXL).

The interior rugged section of the Cumberland Plateau physiographic region is a highly dissected landscape with steep but relatively short slopes compared with the adjacent Appalachian Mountains. These conditions make cable yarding difficult, and

ground skidding with wheeled skidders is a common practice in the region. To facilitate skidding, a set of parallel bladed skid trails were constructed along slope contours generally parallel to the perennial and intermittent streams and perpendicular to ephemeral streams. Skid trails were constructed at intervals that allowed felling and bunching between the skid trails (Figure 2). Skid trail construction across these steep slopes requires considerable cut and fill particularly when the often deeply entrenched ephemeral channels are crossed. The skid trail system comprised 6–12% of the watershed area (Table 1). After harvest, skid trails were retired using BMP recommended for Kentucky (Stringer and Perkins 2001). Retirement practices included construction of water bars and other cross-drained structures as well as revegetation of the trail system.

Treatments

This study included three treatment configurations applied to a total of six wa-



Figure 2. Photograph of a portion of two of the treated watersheds clearly showing the parallel bladed skid trails. Note the SMZ along the perennial stream associated with the T3 treatment in the bottom right corner.

tersheds ($n = 2$) (Figure 3). Two other watersheds were unharvested and served as controls (C). Each treatment included prescriptions for perennial, intermittent, and ephemeral streams (Table 2).

Treatment 1 (T1) was based on the current Kentucky BMP and included a 55-ft perennial SMZ with 50% overstory reten-

Table 1. Characteristics of monitored watersheds at the perennial and intermittent monitoring locations.

Watershed name	Monitoring location	Treatment	Area (acre)	Drainage density (ft/acre)	Aspect	Watershed occupied by skid trails (%)	Ephemeral crossings	Crossings per acre
Little Millseat	Intermittent	C	67	66	Southeast			
	Perennial		195	64				
Falling Rock	Intermittent	C	62	94	Northeast			
	Perennial		240	50				
Booker Hollow	Intermittent	T1	67	48	Northeast	6.8	8	0.05
	Perennial		146	62				
North Shelly Rock	Intermittent	T1	40	81	South	9.4	5	0.07
	Perennial		67	68				
South Shelly Rock	Intermittent	T2	47	53	East	11	7	0.08
	Perennial		82	60				
Goff Hollow	Intermittent	T2	76	31	Northeast	8.8	3	0.03
	Perennial		94	77				
Wet Fork	Intermittent	T3	79	81	Southwest	7.6	12	0.04
	Perennial		277	61				
West Shelly Rock	Intermittent	T3	44	141	Southeast	12	12	0.06
	Perennial		178	76				

tion and a 25-ft intermittent SMZ with no overstory retention requirement. SMZ overstory buffers were not used along ephemeral streams. Nonelevated stream crossings (fords) were used to cross ephemeral streams in T1. Treatment 2 (T2) maintained the 55-ft perennial SMZ but required 100% canopy retention and 25% canopy retention in the 25-ft intermittent SMZ. In addition, elevated crossings (temporary skidder bridges and culverts) were used to cross ephemeral streams and the nearest channel bank tree was retained. Treatment 3 (T3) increased the perennial SMZ width to 110 ft with 100% canopy retention. T3 also increased the intermittent SMZ width to 55 ft with 25% canopy retention and included a 25-ft SMZ around ephemeral streams that limited harvesting equipment to the crossings only. Elevated crossings were used to cross ephemeral streams in T3, and the nearest tree to the channel was retained along the length of the stream.

Sampling and Statistical Methodologies

Sampling at perennial and intermittent monitoring locations began in 2003 and concluded in October 2010. Nonstorm (grab) samples were collected at both perennial and intermittent monitoring locations, monthly before harvesting and weekly after the onset of harvesting in each watershed. Analyses of nonstorm samples included TSS, turbidity, DO, nitrate (as nitrate-N) and ammonium (as ammonium-N). Storm samples were taken only at perennial monitoring locations using automated water samplers (Teledyne ISCO, Lincoln, NE)

equipped with liquid level actuators positioned just above base flow levels.

Depending on the duration of the event, samples were taken at 10- to 30-minute intervals from the start of the storm, and sampling concluded 8–24 hours afterward. Up to 47 samples were taken per event and composited, resulting in one sample per storm event. A total of 162 storm samples were collected from 76 precipitation events. The primary purpose of the storm sampling was to detect sediment changes resulting from harvesting; thus, only TSS and turbidity were analyzed.

Total suspended solids were determined gravimetrically using a 59- μ m filter (Whatman 934-AH glass microfiber filters; GE Biosciences Corp., Piscataway NJ) (American Public Health Association 1992). Turbidity was analyzed using a portable turbidity meter and recorded as formazin turbidity units (FTU) (model HI 93703; Hanna Instruments, Woonsocket, RI). For each storm sample, TSS and turbidity were measured twice and reported as the mean of the two samples.

Nitrate and ammonium samples were refrigerated within an hour of collection and analyzed within 72 hours. Concentrations were determined colorimetrically using an AutoAnalyzer 3 (Bran Luebbe, Norderstedt, Germany). The DO concentration was measured with a Yellow Springs Instruments 556 multiparameter probe (Yellow Springs Instruments Incorporated, Yellow Springs, OH).

Stream temperature was recorded every 15 minutes using a data logging mini-TROLL instrument (In-Situ, Inc., Fort Collins, CO). The loggers used to measure temperature also recorded water table height for

determining flow. The flow sensor could be damaged if frozen, so sensors were deployed from mid-February to mid-December each year for the duration of the study. Temperature-related parameters included daily mean temperature, daily maximum temperature, daily minimum temperature, and diurnal fluctuation. Diurnal fluctuation was calculated as the difference between daily maximum and minimum temperatures.

A preharvest calibration study was conducted on the eight watersheds included in this study from 2003 to 2006 and did not detect significant differences among watersheds at either the perennial or intermittent monitoring locations for nonstorm DO, TSS, turbidity, nitrate, or ammonium (Cherry 2006). Because pre- and postharvest sampling intensities differed and storm sampling was not conducted preharvest, the paired watershed approach for evaluating water quality treatment effects was not used. Instead, a protected least significant differences approach was used and included data from the harvest and postharvest period only.

Data for each watershed were grouped by treatment. Differences among treatments were evaluated using the Kruskal-Wallis method. Where differences among treatments were observed using Kruskal-Wallis, pairwise comparisons of the treatments were performed using Dunn's method. All statistical analyses were performed with Sigma Plot version 13.

Results

Storm Samples

TSS. Significant differences in storm TSS were found among treatments ($P <$

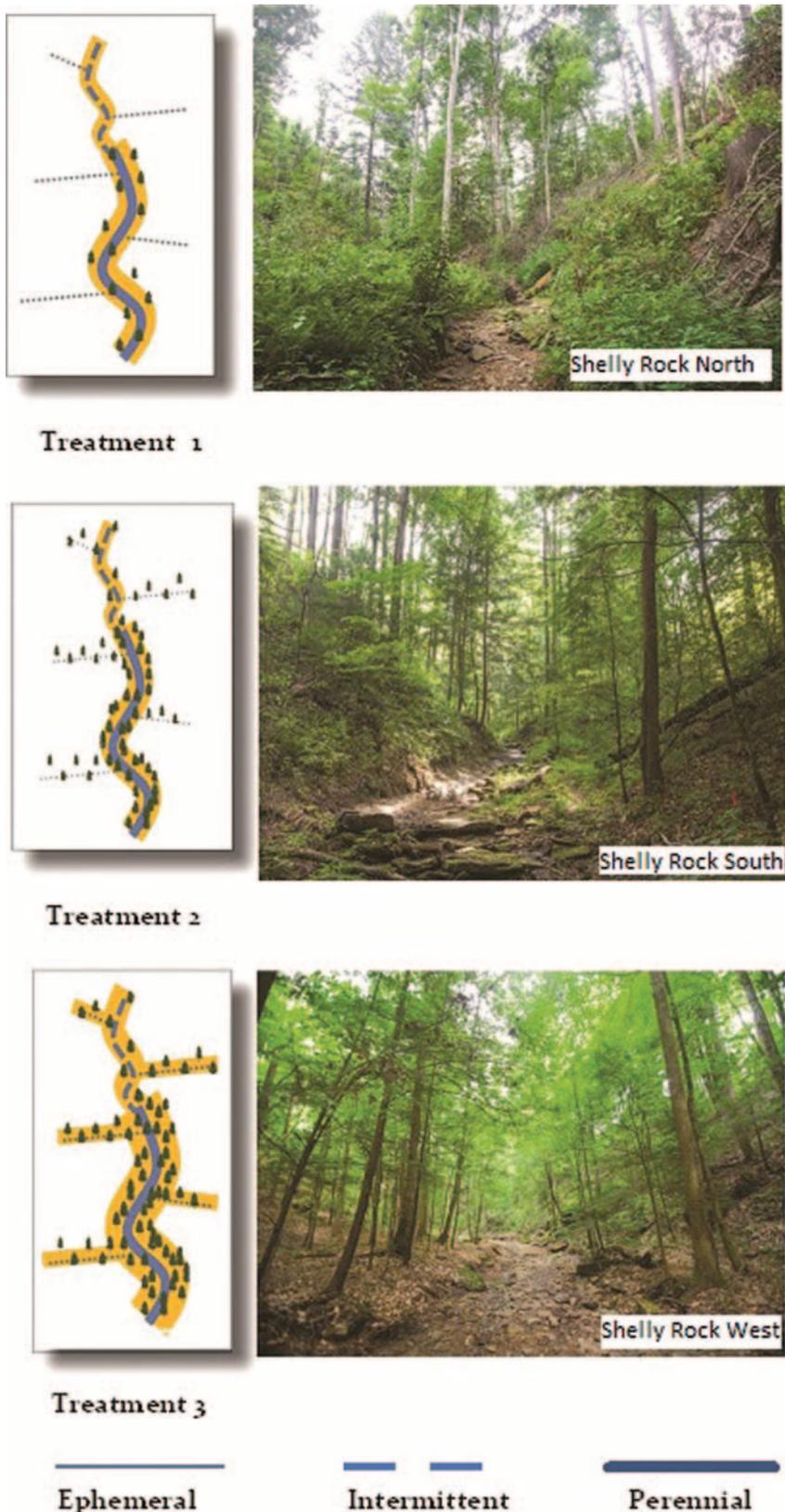


Figure 3. Treatments used in the study varied by stream type. Treatment 1 was established following requirements outlined in Kentucky's Forestry Best Management Practices, requiring a 55-ft perennial SMZ with 50% overstory retention and a 25-ft intermittent SMZ with no overstory retention requirement. Note the open canopy and light infiltration. Treatments 2 and 3 require 100% canopy retention in 55- and 110-ft SMZ, respectively, for perennial streams. For intermittent streams, treatments 2 and 3 require 25% canopy retention in 25- and 55-ft SMZs, respectively.

0.0001, $df = 3$) (Figure 4A). Storm TSS was significantly higher in T1 streams than in T3 streams (difference of ranks: 50.4, $P = 0.002$) and C streams (difference of ranks: 47.0, $P \leq 0.001$). Storm TSS in T2 streams was not different from that in T1 streams (difference of ranks: 26.0, $P = 0.19$), T3 streams (difference of ranks: 24.4, $P = 0.24$), or C streams (difference of ranks: 21.1, $P = 0.10$). No difference in storm TSS was detected between C streams and T3 streams (difference of ranks: 3.4, $P = 1.00$).

Turbidity. Turbidity at the perennial outlet in storm samples was significantly different among the three treatment groups and the unharvested control ($P < 0.0001$, $df = 3$) (Figure 4B). Turbidity from storm events was significantly higher in T1 streams than in T3 streams (difference of ranks: 49.6, $P = 0.002$) and C streams (difference of ranks: 65.3, $P < 0.0001$). Storm sample turbidity was not different between T1 and T2 streams (difference of ranks: 32.4, $P = 0.06$). Turbidity in storm samples from T2 streams was significantly higher than in C streams (difference of ranks: 32.8, $P = 0.004$) but was statistically similar to turbidity in T3 streams (difference of ranks: 17.1, $P = 0.89$). Turbidity was statistically similar between T3 and C streams (difference of ranks: 15.7, $P = 0.94$).

Nonstorm Samples

TSS. Analysis of nonstorm TSS concentrations at perennial monitoring sites did not result in significant differences among the treatments ($P = 0.09$, $df = 3$). Nonstorm TSS at intermittent monitoring locations did result in differences among treatments ($P = 0.048$, $df = 3$) (Figure 5A). Intermittent TSS concentrations from T1 streams were higher than concentrations in T3 streams (difference of ranks: 39.96, $P = 0.047$) but were not different from those in T2 streams (difference of ranks: 30.6, $P = 0.18$) or C streams (difference of ranks: 30.1, $P = 0.38$) (Figure 5B). Differences were not observed between T3 and C streams (difference of ranks: 5.8, $P = 1.00$) or T2 streams (difference of ranks: 5.4, $P = 1.00$).

Turbidity. Significant differences in nonstorm sample turbidity from perennial monitoring points were found among the treatment types ($P < 0.0001$, $df = 3$) (Figure 6A). Nonstorm sample turbidity was higher in T1 streams than in T2 streams (difference of ranks: 113, $P < 0.001$), T3 streams (difference of ranks: 170, $P < 0.0001$), and C streams (difference of ranks:

Table 2. Whole watershed treatment combinations utilized in the study.

Treatment	Perennial		Intermittent		Ephemeral		Improved crossings (Yes/No)
	SMZ width (ft)	Canopy retention (%)	SMZ width (ft)	Canopy retention (%)	SMZ width (ft)	Canopy retention (%)	
T1*	55	50	25	0	0	0	No
T2	55	100	25	25	0	Stringer†	Yes
T3	110	100	55	25	25	Stringer	Yes

Each treatment was applied to two watersheds.

* T1 was based on the current Kentucky best management practice regulations.

† A stringer refers to the retention of the overstory tree nearest the channel bank on either side of the stream.

138, $P < 0.0001$). Nonstorm sample turbidity in T2 streams was higher than that in T3 stream samples (difference of ranks: 57.3, $P = 0.01$) but was similar to turbidity in C streams (difference of ranks: 25.5, $P = 1.00$). Turbidity concentrations in T3 streams were similar to those in C streams (difference of ranks: 31.8, $P = 0.76$).

Turbidity at the intermittent monitoring locations was significantly different among treatments and the unharvested control ($P < 0.001$, $df = 3$) (Figure 6B). Similar to nonstorm perennial outlets, mean turbid-

ity at intermittent monitoring locations was significantly higher in T1 streams than in T2 streams (difference of ranks: 82.4, $P < 0.001$), T3 streams (difference of ranks: 130, $P < 0.001$), and C streams (difference of ranks: 106, $P < 0.001$). Turbidity in T2 streams was significantly higher than that in T3 streams (difference of ranks: 47.5, $P = 0.003$) but was statistically similar to turbidity in C streams (difference of ranks: 23.2, $P = 0.94$). Turbidity in T3 and C streams was statistically similar (difference of ranks: 24.3, $P = 0.79$).

Nitrate and Ammonium Concentrations. Significant differences in nitrate concentrations in nonstorm samples from perennial monitoring locations were observed among treatments ($P < 0.001$, $df = 3$) (Figure 7). Nitrate concentrations at perennial monitoring locations in C streams were lower than concentrations at perennial locations in T1 streams (difference of ranks: 90.6, $P < 0.001$), T2 streams (difference of ranks: 202, $P < 0.001$), or T3 streams (difference of ranks: 188, $P < 0.0001$). Nitrate concentrations in T1 streams were lower than concentrations in T2 streams (difference of ranks: 111, $P < 0.001$) and T3 streams (difference of ranks: 97.3, $P < 0.001$). Nitrate concentrations in T2 streams were similar to concentrations in T3 streams (difference of ranks: 13.7, $P = 1.00$).

Differences by treatment in nitrate concentration at intermittent monitoring locations were also observed ($P < 0.001$, $df = 3$). Nitrate concentrations in C streams were lower than concentrations in T1 streams (difference of ranks: 53.1, $P = 0.012$), T2 streams (difference of ranks: 123, $P < 0.001$), and T3 streams (difference of ranks: 114, $P < 0.001$). Intermittent nitrate concentrations were also lower in T1 than in T2 streams (difference of ranks: 70.1, $P < 0.001$) and T3 streams (difference of ranks: 60.7, $P < 0.001$). Similar nitrate concentrations were found at intermittent monitoring locations in T2 and T3 streams (difference of ranks: 9.4, $P = 1.00$).

Comparisons of ammonium nitrate concentrations from treatment watersheds and unharvested control watersheds did not result in statistical differences for either perennial ($P = 0.25$) or intermittent ($P = 0.78$) samples (Figure 7).

DO Concentrations. Significant differences in perennial nonstorm DO concentrations were observed ($P = 0.003$). No differences in DO were detected between C and T1 streams (difference of ranks: 2.4,

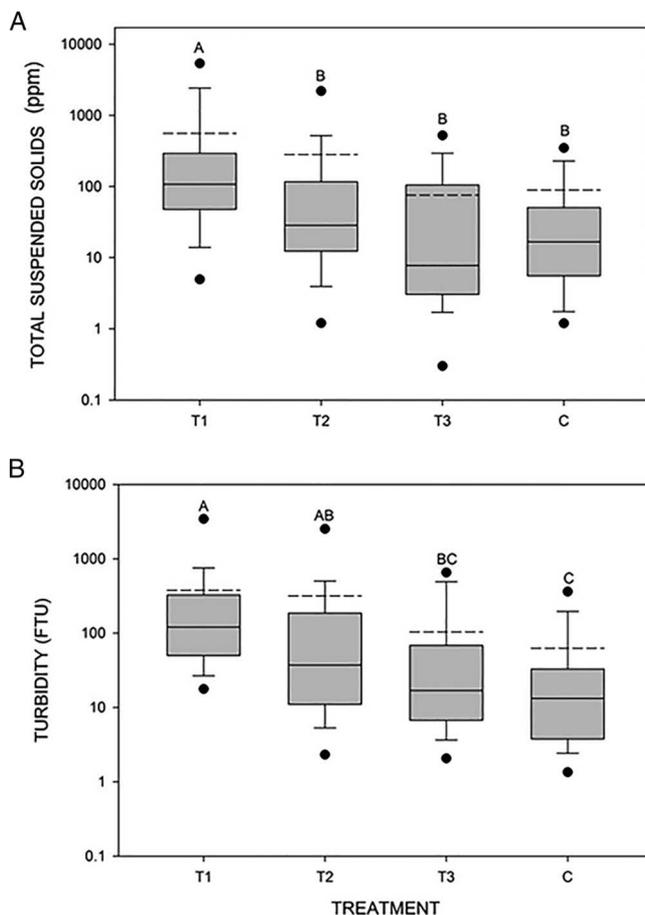


Figure 4. Box and whisker plot of storm TSS (A) and turbidity (B) concentrations by treatment at monitoring locations in perennial streams after forest harvest in Robinson Forest, Kentucky. Letters denote significant differences at the $P = 0.05$ level.

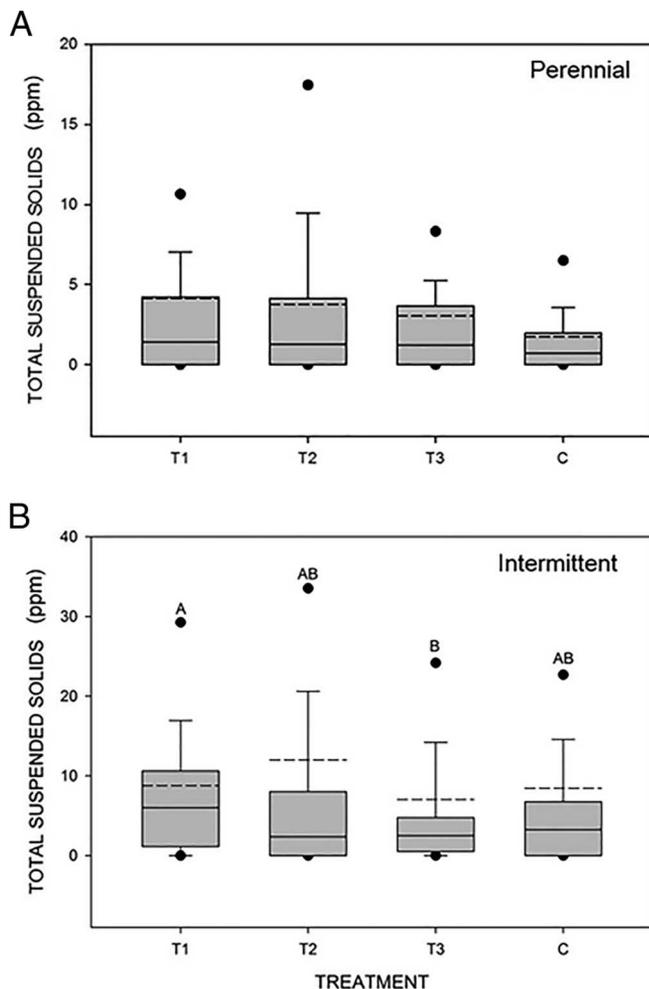


Figure 5. Box and whisker plot of nonstorm TSS concentrations by treatment at perennial stream (A) and intermittent stream (B) monitoring locations after forest harvest in Robinson Forest, Kentucky. Letters denote significant differences at the $P = 0.05$ level.

$P = 1.00$) or T2 streams (difference of ranks: 21.9, $P = 1.00$). DO concentrations were higher in T3 streams than in T2 streams (difference of ranks: 61.5, $P = 0.002$). No differences were detected between T1 and T2 streams (difference of ranks: 19.6, $P = 1.00$) (Figure 8).

Significant differences in DO concentrations were not observed at intermittent monitoring locations ($P = 0.16$).

Temperature. No difference in mean daily temperature was observed when watersheds were grouped by treatment ($P = 0.48$) (Table 3). Significant differences were detected in maximum daily temperature when watersheds were grouped by treatment ($P < 0.001$). Mean maximum daily temperature was higher in T1 streams ($66.7 \pm 1.1^\circ \text{F}$) than in T2 streams ($64.4 \pm 1.2^\circ \text{F}$) ($P = 0.005$), T3 streams ($64.4 \pm 1.2^\circ \text{F}$) ($P = 0.002$), and C streams ($63.3 \pm 1.1^\circ \text{F}$) ($P < 0.0001$). Differences in maximum daily temperature were not found between any of

the other watersheds. Differences in minimum daily temperature did not occur among treatments ($P = 0.22$) (Table 3). Comparison of diurnal fluctuation among watersheds was significant ($P < 0.0001$). Mean fluctuation for C streams ($3.7 \pm 0.2^\circ \text{F}$) was lower than that for T1 streams ($8.6 \pm 0.4^\circ \text{F}$) ($P < 0.0001$), T2 streams ($6.7 \pm 0.3^\circ \text{F}$) ($P < 0.0001$), and T3 streams ($5.9 \pm 0.3^\circ \text{F}$) ($P < 0.0001$). Mean diurnal fluctuation for T1 streams was also significantly different from that for T2 streams ($P = 0.0004$) and T3 streams ($P < 0.0001$). The diurnal fluctuation in T2 streams was higher than the fluctuation in T3 streams ($P = 0.02$).

Discussion

Storm Samples

Mean storm TSS concentrations of 76–555 ppm for harvested watersheds in this study were similar to those reported

by Keim and Schoenholtz (1999) in forests in Mississippi bluffs with SMZ configurations that included treatments with no buffer, a partially harvested 100-ft SMZ, an unharvested 100-ft SMZ, and an unharvested control (197–664 ppm). Storm TSS concentrations from this study were also similar to median storm event TSS concentrations observed after harvest by Wynn et al. (2000) in Virginia, which included a no-BMP treatment and a treatment with a 50-ft SMZ with partial harvesting, as well as an unharvested control (control: 166 ppm, BMP: 99 ppm, and no BMP: 3,299 ppm). BMP were effective in storm sediment reduction regardless of the geologic setting (loessial bluffs, coastal plains, or Cumberland Plateau) and various practices used among the studies.

Although storm TSS concentrations were similar to those of other studies that evaluated similar BMP, T1 storm TSS levels were much higher than those for the other treatments examined. We did not implement a no-BMP treatment in this study, so it is difficult to evaluate the effectiveness of this particular treatment. However, the change in T1 storm TSS over that for C represents an increase of 525%, which is concerning. Storm TSS concentrations and turbidity readings in T2 and T3 watersheds were statistically similar to those in C watersheds. Because the BMP prescriptions varied by stream type within a watershed, it is difficult to discern whether an individual recommendation was responsible for the observed reduction or whether it was the result of combined practices. For example, it was noted in a separate study examining ephemeral stream crossings in these watersheds that TSS concentrations from two types of improved crossings (skidder bridges and culverts) used in T2 and T3 watersheds were statistically similar to concentrations in C watersheds (Witt et al. 2013). Differentiating the impact of improved crossings from SMZ width was not possible when the whole watershed impact of these BMP was considered.

It appears that limiting the connectivity of the skid trail system to the hydrologic system, as well as providing a large undisturbed SMZ, is adequate to minimize sediment concentration increases after forest harvests. Precipitation events effectively link the exposed soils of the skid trail system to the hydrologic system via either ephemeral streams or concentrated flow through the SMZ (Bowker 2013,

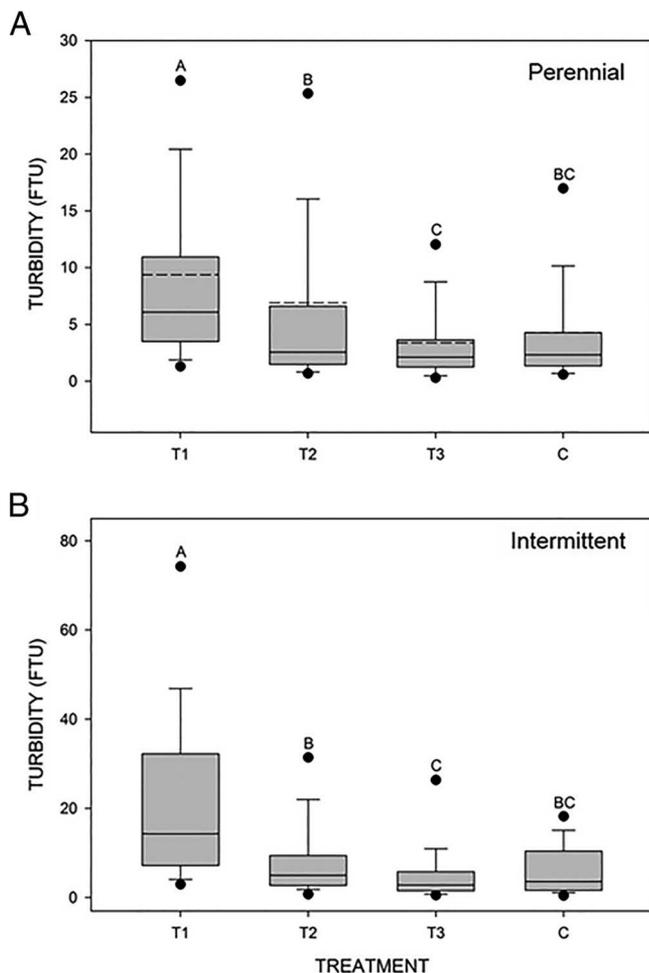


Figure 6. Box and whisker plot of nonstorm turbidity concentrations by treatment at perennial stream (A) and intermittent stream (B) monitoring locations after forest harvest in Robinson Forest, Kentucky. Letters denote significant differences at the $P = 0.05$ level.

Witt et al. 2013). The skid trail system concentrates runoff because of reductions in infiltration relative to the intact forest floor and through interception of lateral

subsurface flow at trail cuts into the hill-slope (Moore and Wondzell 2005). The contribution of skid trails to increased sediment after harvesting activities has

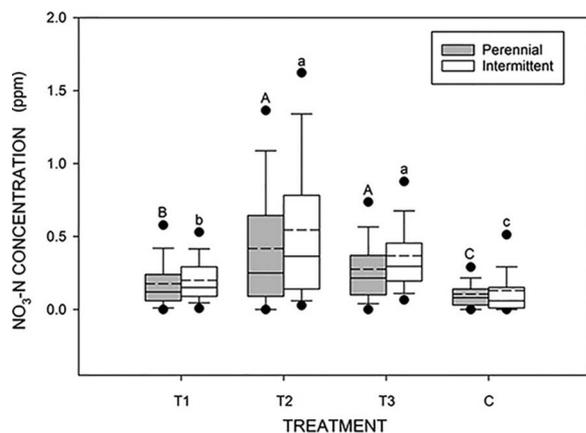


Figure 7. Box and whisker plot of nonstorm nitrate concentrations by treatment at perennial stream (gray) and intermittent stream (white) monitoring locations after forest harvest in Robinson Forest, Kentucky. Letters denote significant differences at the $P = 0.05$ level. Uppercase letters show significance for perennial streams, whereas lowercase letters are used for intermittent streams.

been previously observed by Gomi et al. (2005), Kreutzweiser and Capell (2001), Lacey (2000), and Litschert and MacDonald (2009), among others.

Nonstorm Samples

TSS and Turbidity. Differences in TSS concentrations from unharvested control watersheds were not observed at either the perennial or intermittent monitoring locations under base flow conditions for any of the treatment watersheds. In contrast to the TSS, differences in turbidity were apparent among treatments from both perennial and intermittent nonstorm samples. The combination of an intact SMZ and elevated crossings in ephemeral streams significantly reduced turbidity by 16% relative to that of treatment watersheds that did not incorporate improved crossings and included 50% basal area removal in perennial SMZs.

The overall impact of the harvest on TSS concentration and turbidity was low. Negative impacts on aquatic biota from increased TSS and turbidity may include reductions in primary productivity by periphyton, increased macroinvertebrate drift and decreased diversity, and negative impacts on fish reproduction and feeding behaviors. Many of these impacts were observed in studies where TSS concentrations were greater than 100 ppm, and turbidity was greater than 1,000 nephelometric turbidity units (NTU) (Bilotta and Brazier 2008). One NTU is equal to one FTU. However, negative impacts of turbidity have been observed for some species, particularly cold water fish species, at much lower levels (10–30 NTU) (Barrett et al. 1992, Zamor and Grossman 2007). In our study, mean nonstorm TSS concentrations in perennial streams were less than 5 ppm for all treatments, with mean nonstorm TSS concentrations in intermittent streams of less than 10 ppm, which is typical of unharvested forested streams (Binkley and Brown 1993, Lebo and Herrmann 1998, Clinton 2011). Mean nonstorm TSS concentrations from intermittent and perennial streams were approximately half of mean annual concentrations found in predominately agricultural (13.3 ppm), urban (20.8 ppm), and an urban-agricultural mixed watershed (23.1 ppm) in central Kentucky, approximately 100 miles from our study site (Coulter et al. 2004).

Nitrate and Ammonium. Nitrate concentrations were higher in harvested watersheds at both the perennial and intermittent

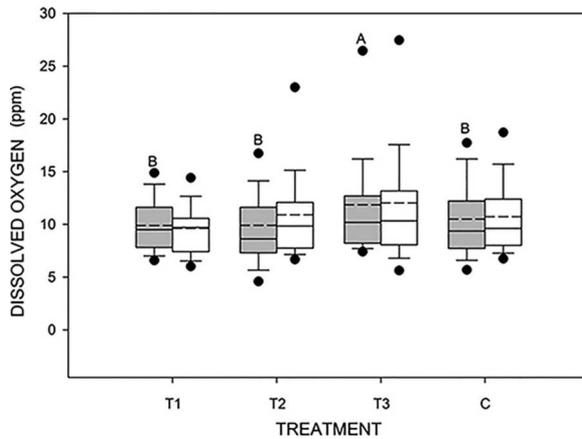


Figure 8. Box and whisker plot of nonstorm DO concentrations by treatment at perennial stream (gray) and intermittent stream (white) monitoring locations after forest harvest in Robinson Forest, Kentucky. Treatments followed by the same letter are not significantly different ($P > 0.05$). Absence of letters indicates no statistical difference between treatments.

monitoring locations. These results are similar to those reported previously by Blackburn and Wood (1990), Martin et al. (1986), and Wynn et al. (2000). Ammonium concentrations did not vary by treatment. Wider perennial SMZs did not influence nitrate concentrations, as there was no difference between T2 and T3. The retention of 100% of canopy trees in perennial SMZs resulted in higher nitrate concentrations in T2 and T3 relative to those in T1. Canopy removal in the perennial SMZ in T1 may have increased light and temperature conditions of the stream bed, which increased in-stream productivity by periphyton and assimilation of nitrate. Andrews et al. (2011) noted a similar response on a restored stream segment with open light conditions and no canopy cover. In addition, the 50% canopy removal in the perennial SMZ in T1 resulted in increased light penetration to the ground surface and a flush of herbaceous understory vegetation observed during sampling, which may have also contributed to rapid nitrate uptake in T1 water-

sheds. Nitrate concentrations in these watersheds postharvest were still low, with averages near the 0.5 ppm threshold that approximately 70% of studies reviewed by Binkley and Brown (1993) reported and were close to the preharvest ranges observed in this study (intermittent means = 0.26–0.58 ppm; perennial means = 0.34–0.57 ppm). Stream concentrations in this study were also similar to the mean stream water concentrations of nitrate for the United States, which were reported as 0.31 ppm, and to mean concentrations for all hardwood forests, which were 0.46 ppm (National Council for Air and Stream Improvement 2001). Concentrations were also similar to those found in Noland Creek in the Great Smoky Mountain National Park (annual mean 0.62 ppm) and were higher than those observed in Walker Branch at Oak Ridge National Laboratory (0.02 ppm) (Martin et al. 2001).

The impact of elevated nutrient concentrations, including nitrate, on forested streams is difficult to determine after harvest

because of the many complexities associated with light, temperature, and carbon that control nutrient cycling and warm water stream food webs (Miltner and Rankin 1998, Smith et al. 1999). Mulholland et al. (2008) demonstrated the importance of small streams such as these in processing elevated nitrate across many land uses, and Adams et al. (2014) demonstrated how disturbance can alter the inorganic nitrogen dynamics of forested watersheds. Changes in the fish community structure have been observed when total inorganic nitrogen concentrations exceeded 3.61 ppm (Miltner and Rankin 1998). Mean nitrate plus ammonium concentrations in streams draining harvested watershed in this study ranged from 0.24 to 0.48 ppm and are not expected to negatively affect aquatic biota but are important given the complexity of nitrogen cycling in headwater streams and the role of the headwaters in exporting nitrogen downstream.

DO and Stream Temperature. Differences in DO concentrations due to the harvest were observed among the three treatments, but none were different from the unharvested control watersheds. Postharvest DO concentrations were at the high end of the normal range of 5–10 ppm reported by Binkley and Brown (1993). These differences in DO relative to those for unharvested control watersheds should not have a detrimental impact on aquatic biota, and the range of the mean preharvest DO concentrations (10.32–11.51 ppm) was close to the mean postharvest range (8.22–12.17 ppm).

Differences in maximum daily temperature and diurnal fluctuation were detected among the treatments. The magnitude of differences relative to C watersheds for T1 was +3.4° F, which was greater than the changes observed by Studinski et al. (2012) when 50% of the basal area was removed within a 100-ft riparian zone in West Virginia (+1.8° F) and greater than changes reported in western Oregon by Groom et al. (2011) when a 50- to 70-ft riparian area was partially harvested. Differences in daily maximum temperature were similar to those observed by Cole and Newton (2013) in Oregon for streams with upland clearcuts and a variety of riparian management including areas of no harvest, partial harvest, and overstory harvest. Relative to the unharvested control, differences in maximum daily temperature were not observed in T2 and T3, similar to the report of Fraser et al. (2012), who found no change in maximum temper-

Table 3. Summary of postharvest temperature data at perennial monitoring locations.

Treatment	Mean daily temperature	Maximum daily temperature	Minimum daily temperature	Mean diurnal fluctuation
.....(° F).....				
T1	62.2 (1.1)	66.7 (1.1)a	58.2 (1.0)	8.6 (0.4)a
T2	61.0 (1.1)	64.4 (1.2)b	57.7 (1.1)	6.7 (0.3)b
T3	61.4 (1.1)	64.4 (1.2)b	58.6 (1.0)	5.9 (0.3)c
C	61.4 (1.1)	63.3 (1.1)b	59.6 (1.1)	3.7 (0.2)d

Data are presented as means (SE). Letters denote significant differences at the $P = 0.05$ level tested separately for each parameter using nonparametric least significant difference tests. Parameters without letters exhibited no significant differences among treatments (mean daily temperature $P = 0.62$; minimum daily temperature $P = 0.22$). Nonparametric least significant difference tests were used for each parameter.

ature after an upland clearcut when a 40- to 65-ft SMZ remained unharvested. Differences in diurnal fluctuation in this study of +3.7–8.6° F were similar to those observed by Johnson and Jones (2000) in an unharvested watershed and two differentially harvested watersheds in Oregon.

Stream temperatures may be influenced by a number of processes, primarily solar inputs, but also evaporation, convection, and advection from upstream and groundwater inputs (Brown and Krygier 1970, Johnson and Jones 2000, Webb et al. 2008). Although changes to solar inputs via overstory removal may be assumed to be the primary influence on harvested versus nonharvested stream temperature, other factors (e.g., substrate type) can influence the degree to which changes in solar inputs influence stream temperatures (Johnson 2004). Results from Kibler et al. (2013) and Jackson et al. (2001) showed that maintaining shaded channels using logging debris was also an effective strategy to minimize temperature changes.

Many aquatic species are adapted to narrow temperature ranges (Vannote and Sweeney 1980, Ward and Stanford 1982, Moore et al. 2005). Both T2 and T3 were able to maintain stream temperatures within 3° F of the unharvested control watersheds for the four temperature parameters measured, which should result in limited impact to aquatic biota (Binkley and Brown 1993).

Conclusion

In summary, the treatments examined in this study varied by the use of elevated crossings at ephemeral streams and increased canopy retention in perennial and intermittent streams and retention of canopy trees along ephemeral stream channels. Although the exact contribution of improved crossings versus increased canopy retention to sediment concentration reduction at the perennial outlet may not be determined from these data, the combination of BMP used did minimize sediment input in streams draining harvested watersheds. However, increased TSS and turbidity concentrations in T1 over those for the other treatments and control watersheds clearly implies that the effectiveness of state-mandated BMP can be improved on.

Operationally, improvements over state mandated BMP were accomplished by increasing the amount of residual overstory trees left next to stream channels, by restricting the operation of equipment next to

channels, and/or by using elevated stream crossings. In addition to T2 and T3 being more effective for minimizing sediment input, these treatments also provided canopy retention around stream channels that offered thermal protection, maintained coarse woody debris inputs, influenced carbon and nitrogen dynamics, and changed habitat characteristics. Many state BMP programs emphasize the necessity for best professional judgment that may result in going beyond mandated recommendations when planning and implementing silvicultural practices to optimize the efficacy of forestry BMP. In Kentucky and throughout the Cumberland Plateau and similar physiographic regions with moderate to steeply sloping topography, T2 and T3 provide a suitable alternative to state mandated BMP. Use of alternatives such as additional canopy retention, equipment limiting zone recommendations and use of elevated stream crossings could prove valuable in watersheds where sediment generation would be particularly harmful, as would be the case with streams containing flora or fauna that are sensitive to sedimentation.

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