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Stream water responses to timber harvest: Riparian buffer width effectiveness

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A B S T R A C T

Vegetated riparian buffers are critical for protecting aquatic and terrestrial processes and habitats in southern Appalachian ecosystems. In this case study, we examined the effect of riparian buffer width on stream water quality following upland forest management activities in four headwater catchments. Three riparian buffer widths were delineated prior to cutting; 0 m (no-buffer), 10 m, and 30 m, and one reference site (REF). A two-age prescription timber harvest was conducted on all cut sites with a target residual basal area of approximately 4.0 m^2 ha^{-1}. Harvesting occurred from October 2005 through February 2007. Stream sampling was conducted weekly from January 2004 through December 2008. Stream water chemistry, temperature, and total suspended solids (TSS) were used as metrics of water quality. Analyses were conducted on weekly grab samples. Pre-treatment concentrations of all solutes were similar to conditions found in other headwater streams at similar topographic positions around the region. The greatest responses to cutting occurred on the no-buffer site. Compared with pre-harvest levels on the no-buffer site, stream nitrate concentration ([NO_3–N]) increased 2-fold during both base and stormflow following harvest, and all base cations increased in concentration. [NO_3–N] on the no-buffer site showed steady decline with time following the initial post-harvest increase. The other sites did not show increases in [NO_3–N] and very small or no responses in other stream chemistry parameters. There was no TSS response at stormflow on any site, and during baseflow, TSS decreased on all but the no-buffer site. Stream water temperature increased during the summer on the no-buffer site. Although alternative land uses may have different requirements, these results suggest that for riparian buffer widths of 10 m and wider, the forest harvest activities implemented in this study did not substantially impact stream water quality. Hence, 10 m wide buffers in these ecosystems may provide effective protection with respect to stream water chemistry, TSS, and temperature.

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1. Introduction

Forest management operations have long been considered non-point sources of pollution potentially regulated by a myriad of local, state, and federal controls (Brown et al., 1993; Nesbit, 2001). One of the key factors influencing water quality responses to forest management is the degree to which structural and functional attributes of the riparian zone are impacted. In a synthesis of studies, Binkley and Brown (1993) found that stream water responses to forest management activities were strongly influenced by the implementation of best management practices (BMPs) such as riparian buffers. Riparian buffers moderate the effects of upland land management activities through nutrient sequestration (Hill, 1996), maintenance of local microclimates (Rykken et al., 2007), and filtering of sediment and other materials (Neary et al., 1993). The regulation of nutrient export is considered one of the most fundamental and important benefits of undisturbed riparian areas, particularly in agricultural systems where nutrient fluxes are often well in excess of natural conditions (Lowrance et al., 1984).

Many studies have quantified the impacts of riparian zone management on aquatic resources in the southern Appalachians (Greene, 1950; Swift and Messer, 1971; Webster et al., 1992; Jones et al., 1999) and most have focused on the effects of removing riparian zone vegetation. Results from these studies varied, but in general, indicated that manipulation of vegetation in steep mountain watersheds can alter thermal, sediment, and discharge regimes of the affected stream through reduced shading, soil disturbance, and decreased water uptake.

Prescribing a consistent riparian buffer width has been difficult because of considerable variation in slope, vegetation composition, soils, etc., as well as the level of acceptable risk. Hence, the importance of maintaining critical riparian zone structural or functional attributes depends on the intensity of management activities and the resource(s) needing protection. For example, Kreutzweiser and Capell (2001) examined the influence of timber harvest activities on fine sediment deposition in streams in areas without riparian

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buffers. They concluded that in the absence of road construction or road improvement activities, timber removals of less than 50% of canopy area (approx. 40% basal area removal) may not require riparian buffers to prevent fine sediment deposition. Others have reported the need for buffer widths greater than 30 m to protect specific aquatic habitats and species (Jones et al., 1999; Peterman and Semlitsch, 2009). Many BMPs (best management practices) restrict activities within or near the riparian zone to minimize risks of resource degradation. Requirements for buffer widths vary from state to state and are usually conditional based on side slopes, soil erosivity, or sensitivity of flora and fauna to disturbance. For example, BMP requirements for riparian zones in the state of North Carolina, USA, range from 5 m to 90 m depending upon the resource in question, i.e., flora and fauna, sediment and nutrient retention, and its condition and sensitivity (North Carolina Division of Forest Resources, 2006). Hence, for effective protection or enhancement of all critical resources, riparian zone definition requires a careful, site specific and complete examination of terrestrial and aquatic resource needs.

In the southern Appalachian region there has been little effort to characterize what structural and functional attributes define the riparian zone, particularly in headwater catchments. Considerable interest and debate surrounds the use of structural and functional attributes of riparian ecosystems as metrics to assign appropriate riparian buffer widths to protect aquatic and terrestrial resources and habitats from upland disturbance (Castelle et al., 1994; Grubbs and Cummins, 1996; Knoepp and Clinton, 2009; Clinton et al., 2010). While scientists and policy makers often contend that forest ecosystems should be managed, protected, or restored based on our understanding of structure, function, and their interactions (Franklin, 1999; National Research Council, 2002), in many cases not enough is known about these characteristics of specific ecosystems (or their components) to make sound, science-based management decisions that protect water quality.

As a result of inadequate or incomplete knowledge of riparian zone width, land managers are often faced with the challenge of making decisions about appropriate riparian zone protection without a science-based understanding of the structural and functional characteristics that define the riparian area and serve to protect stream water quality (see Clinton et al., 2010). To address this lack of understanding, our study goals were 2-fold. During base and stormflow conditions we; (1) examined stream water chemistry, TSS, and temperature responses to timber harvest in headwater catchments, and (2) determined the effectiveness of varying riparian buffer widths at protecting stream water quality following timber harvest.

2. Methods

2.1. Study catchments

Study sites were located on the Nantahala Ranger District of the Nantahala National Forest in the Blue Ridge Physiographic Province of western North Carolina (35°6′N, 83°6′W). This region of the southern Appalachians receives abundant rainfall (approx. 1800 mm year^{-1}) distributed more or less evenly throughout the year (Swift et al., 1988). Less than 5% of total annual precipitation falls as snow or ice. Mean annual air temperature is 12.6 °C, and ranges from 3.3 °C to 21.6 °C in January and July, respectively. Four headwater catchments containing first order perennial streams, having similar vegetation, topography, and soils were selected for the study. Among our study sites, elevations range from 850 to 950 m, were east-facing, ranged in size from 6 to 10 ha, and had stream gradients ranging from 7 to 23%. Catchment side-slopes ranged from 25 to 75% among catchments, but side-slopes were generally consistent within catchments. Overstory on all sites consisted of mixed hardwoods (*Quercus* spp., *Acer* spp., *Carya* spp., *Liriodendron tulipifera* L.), with a lesser component of conifers (*Pinus strobus* L., *Tsuga canadensis* (L.) Carr.) (Clinton et al., 2010), and logging had not occurred for more than 60 years on any of the sites. All sites contained existing forest access roads and on two sites (later referred to as no-buffer and 10 m buffer sites), there was a single stream crossing with culverts at the upper end of both study reaches. Sites have similar soils that are generally loamy to coarse loamy and derived from material weathered from high grade metamorphosed rock or from colluvium. Side slope soils are mapped in the Evard–Cowee complex (fine-loamy, mixed, mesic Typic Hapludults) which contains about 20% inclusion of the Trimont series (fine-loamy, mixed, mesic Humic Hapludults). These Ultisols are very deep moderately well-drained to well-drained, with solum thickness ∼1 m, and are greater than 1.5 m to bedrock. The saprolite layer beneath the solum may be up to 6 m deep (Thomas, 1996). Cove or stream side soils were formed in colluvium, 15–50% slope, and are mapped in the Cullasaja series. These soils are loamy-skeletal, mixed, mesic Typic Haplumbrepts and very deep, well drained soils, with solum thickness is <1.5 m, and are greater than 1.8 m to bedrock (Thomas, 1996).

2.2. Treatments

Three of the four catchments (sites) selected for this study were designated by the Nantahala National Forest to receive a two-age silvicultural prescription using primarily cable-yarding technology. Cable skidder technology was used on one side of the stream on the 10 m buffer site. With the exception of the area logged with the cable skidder, very little forest floor was disturbed during harvest. Where disturbance due to skid trails did occur, disturbed areas were seeded and mulched immediately after logging ceased. Prior to harvest, existing roads were reshaped and where available, brush was applied to road shoulders particularly around stream crossings to neutralize pre-existing sources of sediment. Pre-harvest stem density and basal area across all sites, including within the proposed buffer, ranged from approximately 350 to 400 stems ha^{-1} and 32 to 41 m^2 ha^{-1}, respectively. Overstory tree DBH (diameter at 1.4 m) averaged approximately 55 cm. Tree height on all sites frequently exceeded 30 m. Understory vegetation consisted of advanced regeneration in species of *Quercus* primarily, with scattered *Pinus strobus* L. and *Tsuga canadensis* (L.) Carr. Although usually abundant in southern Appalachian riparian areas, very few evergreen shrubs (*Rhododendron maximum* L. and *Kalmia latifolia* L.) were present on our study sites (see Clinton et al. (2010) for more a complete account of pretreatment conditions). Target residual basal area and stem density outside the buffers following harvest ranged from 3.4 to 4.6 m^2 ha^{-1} and 20 to 30 stems ha^{-1} of 30 to 40 cm diameter at breast height overstory vegetation. Each treatment site was leave-tree marked and target residual conditions were achieved through the felling of all remaining unmarked standing timber outside the buffer. Site harvest dates are presented in Table 1. The fourth site was not harvested and served as a reference (REF). Each harvested site was assigned one of the following riparian buffer widths: 0 m, 10 m, and 30 m. Harvesting began in October 2005 on the 10 m site and concluded in February 2007 on the 30 m site, and no timber was removed from within the designated buffer zones. The site with the smallest area harvested was the 10 m site (6 ha). The 30 m and no-buffer site had harvest areas of 8.5 and 9.7 ha, respectively (Table 1). The percent of the total watershed harvested for each site was: 10 m, 1.2% of 495 ha; 30 m, 2.0% of 425 ha; and no-buffer, 4.4% of 220 ha (Table 1) (Joan Brown, Nantahala Ranger District Silviculturist; personal communication). After logging, roads were reshaped and a light application of grass seed and fertilizer was applied to the disturbed road beds. Subsequent

Table 1
Site harvest dates for the three harvested study sites, area harvested and percent of total watershed area harvested.

Site	Harvest initiated	Harvest completed	Harvested area (ha)	Percent of watershed
Reference	n/a	n/a	n/a	n/a
30 m	July 2006	February 2007	8.5	2.0
10 m	October 2005	December 2005	6.0	1.2
0 m	January 2006	July 2006	9.7	4.4

road use has been access to the experimental areas by light-weight vehicles.

2.3. Stream water sample collection and analysis

Grab samples were collected weekly and consistently at the same location at the most down-stream point within the treated area of each study catchment beginning January 2004 and continuing through December 2008. Sampling took place during both baseflow and stormflow conditions. Stream gauging did not take place on any of the study reaches to determine stream discharge rates. The rationale for not measuring discharge was based on two main factors: (1) very low discharge rates in the streams would make continuous flow measures problematic and most likely unreliable, and (2) conductivity measurements taken longitudinally along the study reach indicated that no additional measureable sources of surface or subsurface water contributed to discharge. Stormflow was determined based on the occurrence of recent or ongoing rain events within the study catchments that resulted in obvious and substantial increases in discharge. Because of the remote nature of the study sites, it was not always known at what point during stormflow (i.e., increasing or decreasing discharge) samples were actually retrieved, and only one sample was taken at each site for each storm event sampled. Previous studies in the region suggest that TSS can be higher on the rising limb of the hydrograph versus the recession limb (Riedel et al., 2004). Our sampling took most of one day to complete so it is likely that stormflow sampling sometimes occurred on the rising limb of the hydrograph and sometimes occurred on the falling limb of the hydrograph (Riedel et al., 2004). We believe that while this introduces variability into the stormflow TSS and chemistry data, it does not bias the sampling results because we averaged TSS and stream chemistry concentrations across sample collections, thereby including samples collected on both rising and falling limbs of the hydrograph (see Section 3 below). Estimates of stream discharge were not made on any site due to the short length of the study reach (200 m) and the absence of surface or subsurface tributary flows along the reaches; a conclusion based on conductivity sampling along the reach (F. Benfield unpublished data); hence, only minimal increases in discharge were expected to have occurred along our study reaches.

Water samples were analyzed for NO_3–N, NH_4–N, HCO_3, K, Na, Ca, Mg, SO_4, SiO_2, conductivity, total suspended solids (TSS), and pH. Stream water temperature was measured using submersible data-loggers ([1]Hobo Data Loggers, Onset Computer Corp., Pocasset, MA, USA) deployed in 2004 and recorded stream water temperature every 2 h. Pre-treatment concentrations of all solutes were similar to conditions found in other headwater streams at similar topographic positions around the region. We determined SO_4 using ion chromatography; Ca, Mg, Na, and K using atomic adsorption spectrophotometry; and NH_4–N using an autoanalyzer alkaline phenol method on all water samples (USEPA, 1983a,b).

[1] The use of trade or firm names in this publication is for reader information and does not imply endorsement by the U.S. Department of Agriculture of any product or service.

For analysis of TSS, all weekly sample collections were refrigerated until analyzed. Filtration of stream samples for estimating concentrations of TSS (mg L^{-1}) was conducted at the analytical lab of the Coweeta Hydrologic Laboratory in accordance with established Coweeta Q_A/Q_C Laboratory protocols (Coweeta Q_A/Q_C Manual, Analytical Laboratory Publication).

3. Data analysis

The general experimental design was based on four catchments (fixed factor), before and after timber harvest (fixed factor), and month (random factor). Stream water chemistry and total suspended solids data were analyzed on the basis of discrete weekly concentration values. Weekly values for stream water temperature were calculated as averages of continuous hourly readings. Comparisons were made among study sites between pre- and post-harvest conditions using a repeated measures analysis of variance given AR(1) covariance structure (PROC MIXED SAS Institute Inc., 2004). Comparisons of monthly mean stream water chemistry and TSS were made for treatment effects for each study site during both growing (May through October) and dormant (November through April) seasons, and for base and stormflow conditions. For [NO_3–N], TSS, and temperature, tests for treatment effects were made across seasons (winter, spring, summer, fall) among study sites for pre- versus post-harvest periods using the General Linear Model procedure (PROC GLM; SAS Institute Inc., 2004). To normalize the data to the REF site as a way to adjust for variation unrelated to treatment effects, we subtracted stream water chemistry and TSS concentrations on the REF site from those on the treatment sites (differencing) to examine treatment effects with respect to the REF site. Means were compared between pre- and post-harvest periods and base versus stormflow using PROC GLM (SAS Institute Inc., 2004). When main effects or interactions were significant, least square means were computed and Tukey's pairwise comparisons were performed using an experimentwise alpha of 0.05 and Tukey's adjustment.

4. Results

4.1. Stream water responses to treatment

Under baseflow conditions most cations (K, Ca, Na, Mg) increased in concentration following harvest on all sites including the REF site (Table 2). During stormflow (Table 3) cation responses were mixed but in general increased in concentration (Table 3). Conductivity increased significantly following harvest on all sites during baseflow (Table 2) and on the 10 m buffer site during stormflow (Table 3), likely in response to increases in base cation concentrations.

Seasonal patterns of [NO_3] indicated no response to the timber harvest treatment on all but the no-buffer site (Fig. 1), where concentrations increased during and immediately following harvest. Nitrate concentrations declined following the initial post-harvest increase on that site (Fig. 1). Compared with pre-harvest levels, mean monthly stream [NO_3] increased on the no-buffer site during baseflow following harvest from 0.04 to 0.12 mg L^{-1} (F = 64.51, P < 0.0001) (Table 2 and Fig. 2a), and during stormflow from 0.05 to 0.12 mg L^{-1} (F = 6.43, P = 0.042) (Table 3 and Fig. 2b). Following har-

Table 2
Mean monthly values for stream water chemistry comparing pre-harvest with post-harvest for baseflow conditions. Values in parentheses are one standard error. Minimum sample size of $N = 20$. Means in bold type are significantly difference between pre and post harvest at $\alpha = 0.05$.

	Reference		No buffer		10 m buffer		30 m buffer	
Const.	Pre	Post	Pre	Post	Pre	Post	Pre	Post
NO_3 (mg L^{-1})	**0.082**	**0.053**	**0.041**	**0.120**	**0.049**	**0.119**	0.019	0.028
	(0.004)	**(0.004)**	**(0.002)**	**(0.005)**	**(0.004)**	**(0.039)**	(0.002)	(0.002)
NH_4 (mg L^{-1})	**0.007**	**0.004**	0.007	0.006	**0.007**	**0.005**	**0.007**	**0.004**
	(0.000)	**(0.000)**	(0.000)	(0.001)	**(0.001)**	**(0.001)**	**(0.000)**	**(0.000)**
K (mg L^{-1})	**0.36**	**0.43**	**0.50**	**0.68**	**0.57**	**0.69**	**0.32**	**0.39**
	(0.006)	**(0.009)**	**(0.009)**	**(0.014)**	**(0.017)**	**(0.068)**	**(0.006)**	**(0.007)**
Na (mg L^{-1})	**1.62**	**1.71**	**1.13**	**1.28**	**1.23**	**1.33**	**1.25**	**1.38**
	(0.016)	**(0.020)**	**(0.010)**	**(0.017)**	**(0.026)**	**(0.040)**	**(0.014)**	**(0.018)**
Ca (mg L^{-1})	**1.07**	**1.19**	**1.04**	**1.28**	1.33	1.25	**0.82**	**0.99**
	(0.017)	**(0.022)**	**(0.019)**	**(0.029)**	(0.064)	(0.064)	**(0.017)**	**(0.020)**
Mg (mg L^{-1})	**0.98**	**1.07**	**0.76**	**0.89**	0.89	0.87	**0.76**	**0.87**
	(0.010)	**(0.012)**	**(0.011)**	**(0.013)**	(0.027)	(0.055)	**(0.011)**	**(0.013)**
SO_4 (mg L^{-1})	0.51	0.48	0.88	0.88	0.81	0.98	0.55	0.52
	(0.010)	(0.013)	(0.014)	(0.013)	(0.027)	(0.055)	(0.009)	(0.008)
SiO_2 (mg L^{-1})	14.51	14.60	11.88	11.57	12.59	11.17	11.75	12.05
	(0.159)	(0.154)	(0.161)	(0.120)	(0.320)	(0.455)	(0.143)	(0.140)
HCO_3(mg L^{-1})	8.22	8.43	6.39	6.94	7.47	6.85	**6.29**	**7.04**
	(0.137)	(0.154)	(0.091)	(0.138)	(0.277)	(0.484)	**(0.114)**	**(0.136)**
Cl (mg L^{-1})	0.55	0.56	**0.55**	**0.59**	0.55	0.67	0.49	0.56
	(0.008)	(0.009)	**(0.005)**	**(0.006)**	(0.014)	(0.051)	(0.004)	(0.045)
pH	7.03	6.93	6.94	6.93	7.03	6.96	6.91	6.92
	(0.018)	(0.016)	(0.014)	(0.017)	(0.037)	(0.050)	(0.019)	(0.017)
Cond. (μS)	**20.63**	**22.30**	**17.83**	**20.63**	19.93	20.60	**16.47**	**18.41**
	(0.223)	**(0.209)**	**(0.206)**	**(0.228)**	(0.456)	(0.992)	**(0.199)**	**(0.224)**
TSS (mg L^{-1})	**7.16**	**4.20**	8.23	8.38	**12.05**	**9.09**	**4.55**	**4.04**
	(0.440)	**(0.393)**	(0.652)	(0.559)	**(0.906)**	**(0.525)**	**(0.298)**	**(0.288)**

vest there was a significant decrease in [NO_3] on the 10 m buffer site ($F = 4.72$, $P = 0.035$) and on the REF site ($F = 11.90$, $P = 0.001$) during baseflow conditions. Overall [NO_3] during baseflow was greater in the growing season on the 10 m buffer site ($F = 23.84$, $P = 0.001$) and on the 30 m buffer site at both base ($F = 10.02$, $P = 0.010$) and stormflow ($F = 6.27$, $P = 0.049$). Although seasonal variation did occur on some sites, no interactions were detected between season of year and pre- and post-harvest periods. Overall patterns for mean stream water [NO_3] were similar between base and stormflow conditions on each site (Fig. 2a and b); concentrations increased following harvest on the no-buffer site during base and stormflow, and decreased on the 10 m and REF site during baseflow and on the 10 m buffer site during stormflow (Fig. 2b). When comparing treatment sites to the REF site [NO_3] on the no-buffer site increased substantially following harvest during both base and stormflow conditions ($F = 94.05$, $P < 0.0001$; $F = 9.80$, $P = 0.004$, respectively) (Fig. 3a and b), but no significant responses were found on either the 10 m or 30 m buffer sites. Maximum values for [NO_3] for pre-

Table 3
Mean monthly values for stream water chemistry comparing pre-harvest with post-harvest for stormflow conditions. Values in parentheses are one standard error. Minimum sample size of $N = 20$. Means in bold type are significantly difference between pre and post harvest at $\alpha = 0.05$.

	Reference		No buffer		10 m buffer		30 m buffer	
Const.	Pre	Post	Pre	Post	Pre	Post	Pre	Post
NO_3 (mg L^{-1})	0.069	0.044	**0.049**	**0.119**	0.074	0.053	0.026	0.019
	(0.009)	(0.028)	**(0.004)**	**(0.039)**	(0.007)	(0.003)	(0.003)	(0.007)
NH_4 (mg L^{-1})	**0.007**	**0.003**	0.005	0.004	0.006	0.005	**0.006**	**0.004**
	(0.001)	**(0.001)**	(0.001)	(0.001)	(0.001)	(0.001)	**(0.001)**	**(0.001)**
K (mg L^{-1})	0.40	0.57	**0.57**	**0.69**	**0.51**	**0.56**	**0.37**	**0.46**
	(0.013)	(0.133)	**(0.017)**	**(0.068)**	**(0.024)**	**(0.033)**	**(0.015)**	**(0.091)**
Na (mg L^{-1})	1.70	1.86	**1.23**	**1.33**	1.71	1.83	1.35	1.40
	(0.025)	(0.183)	**(0.026)**	**(0.040)**	(0.033)	(0.117)	(0.029)	(0.103)
Ca (mg L^{-1})	1.21	1.25	1.33	1.25	1.38	1.42	**0.93**	**1.04**
	(0.036)	(0.121)	(0.064)	(0.064)	(0.031)	(0.073)	**(0.029)**	**(0.100)**
Mg (mg L^{-1})	1.07	1.10	0.89	0.87	**0.72**	**0.78**	0.85	0.89
	(0.017)	(0.084)	(0.027)	(0.055)	**(0.015)**	**(0.028)**	(0.021)	(0.072)
SO_4 (mg L^{-1})	0.48	0.59	0.81	0.98	**0.47**	**0.53**	0.51	0.61
	(0.046)	(0.092)	(0.027)	(0.055)	**(0.015)**	**(0.024)**	(0.014)	(0.051)
SiO_2 (mg L^{-1})	15.28	14.25	12.59	11.17	13.56	13.85	12.33	11.37
	(0.301)	(0.839)	(0.320)	(0.455)	(0.404)	(0.488)	(0.225)	(0.604)
HCO_3(mg L^{-1})	8.71	7.76	7.47	6.85	8.64	7.41	6.97	6.72
	(0.395)	(0.731)	(0.277)	(0.484)	(0.216)	(0.643)	(0.302)	(0.611)
Cl (mg L^{-1})	0.52	0.63	0.55	0.67	0.43	0.47	0.49	0.58
	(0.010)	(0.084)	(0.014)	(0.051)	(0.018)	(0.028)	(0.009)	(0.053)
pH	**7.02**	**6.91**	7.03	6.96	**7.10**	**6.81**	6.93	6.89
	(0.063)	**(0.060)**	(0.037)	(0.050)	**(0.021)**	**(0.086)**	(0.052)	(0.049)
Cond. (μS)	21.96	23.50	19.93	20.60	**20.58**	**21.58**	18.16	18.67
	(0.285)	(1.647)	(0.456)	(0.992)	**(0.380)**	**(0.786)**	(0.439)	(1.274)
TSS (mg L^{-1})	23.07	15.74	29.29	24.16	37.00	31.95	11.29	8.18
	(3.213)	(15.74)	(6.700)	(6.145)	(5.162)	(10.23)	(1.647)	(2.625)

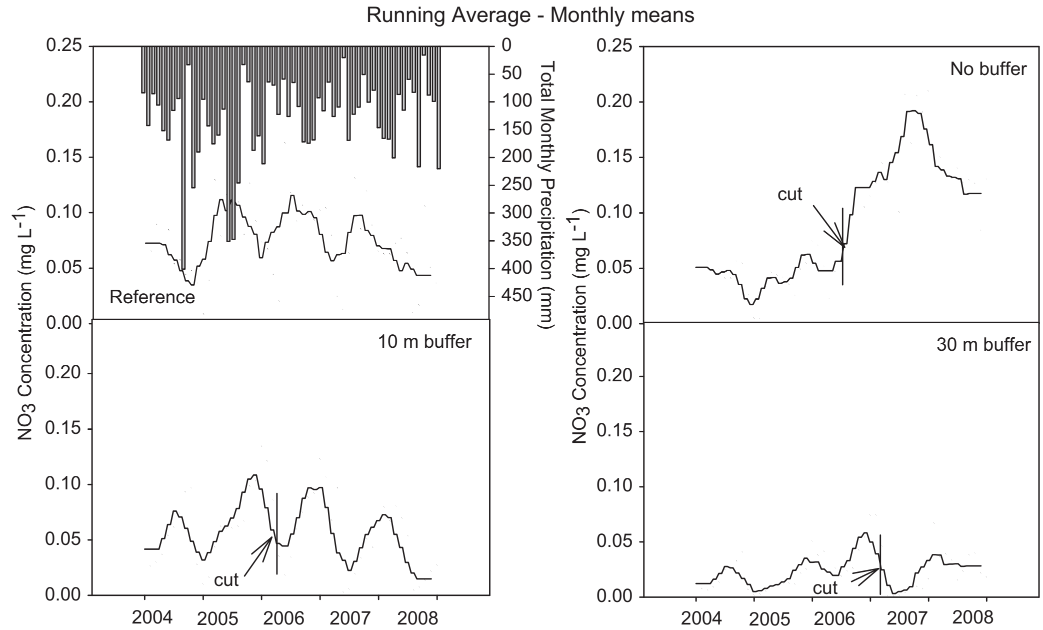

Fig. 1. Monthly means and three month running averages for NO_3–N concentration (mg L^{-1}) for each study site during baseflow. Beginning of harvest for the three treatment sites is indicated on each panel. Total monthly precipitation (mm) for the study period is shown in the reference panel.

and post-harvest, respectively, were: no-buffer, 0.09–0.247: 10 m, 0.156–0.367: 30 m, 0.094–0.162 mg L^{-1}. We observed decreases in [NH_4] during base and stormflow on the REF and 30 m buffer sites (REF: $F = 28.55$, $P < 0.0001$; $F = 7.12$, $P = 0.032$, respectively; 30 m site: $F = 20.21$, $P < 0.0001$; $F = 4.55$, $P = 0.049$, respectively), and during baseflow on the 10 m buffer site ($F = 4.80$, $P = 0.033$). No overall seasonal responses in [NH_4] were detected.

4.2. Stream water suspended solids

During baseflow, clear seasonal patterns in TSS were observed; with higher values during growing season compared with dormant season, but there were no discernable responses to the harvest treatment on any of our study sites (Fig. 4). Although small in magnitude, significant decreases in TSS were observed on all but the

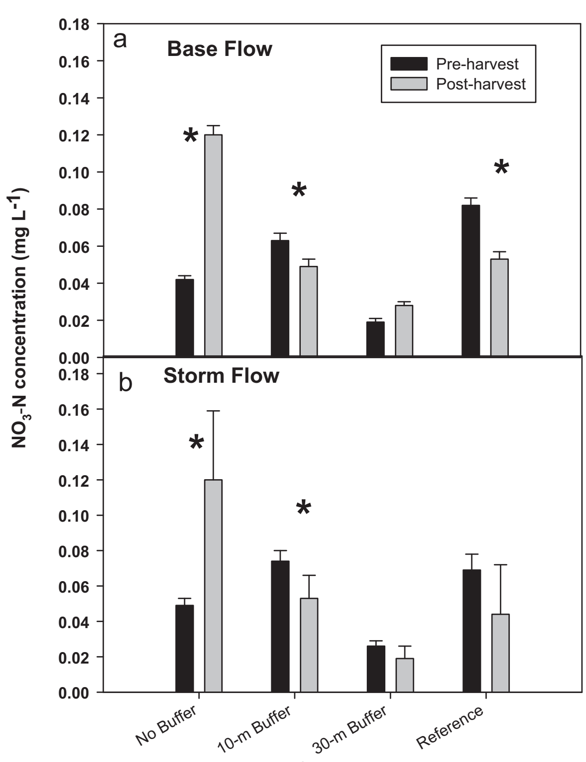

Fig. 2. Overall means for NO_3–N concentration (mg L^{-1}) for pre- and post-harvest periods for each site. An '*' above pre- and post-harvest means for each site indicates significant difference between the means at the $\alpha = 0.05$ level.

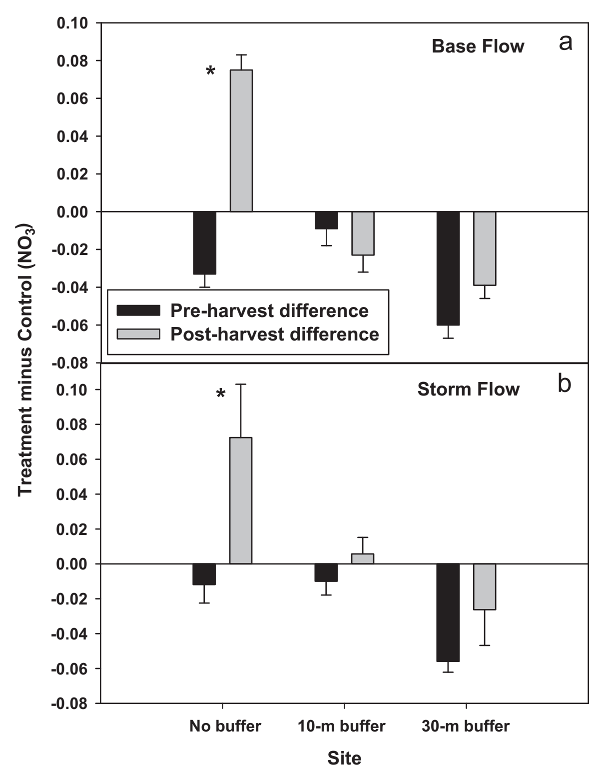

Fig. 3. Pre- and post-harvest differences in NO_3–N concentration between the treated sites and the REF site. An '*' above pre- and post-harvest differences in means for each site indicates significant difference between pre- and post-harvest nitrate levels relative to the REF site. Differences were evaluated at the $\alpha = 0.05$ level.

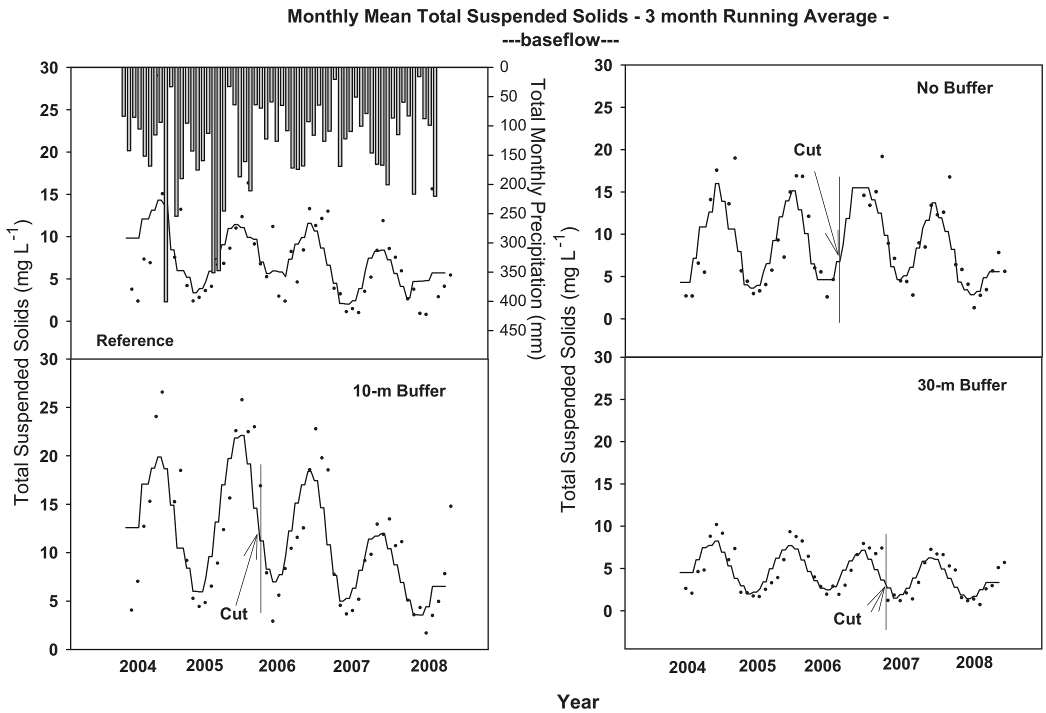

Fig. 4. Monthly means and three month running averages for total suspended solids (TSS; mg L^{-1}) for each study site during baseflow. Beginning of harvest for the three treatment sites is indicated on the each panel. Total monthly precipitation (mm) for the study period is shown in the reference panel.

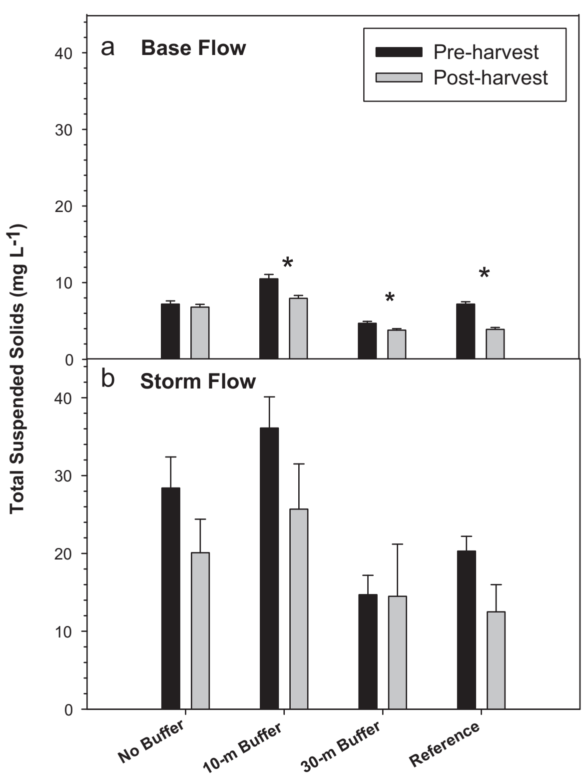

Fig. 5. Overall means for total suspended solids (TSS; mg L^{-1}) for pre- and post-harvest period for each site at (a) baseflow and (b) stormflow. An '*' above pre- and post-harvest means for each site indicates significant difference between the means at the $\alpha = 0.05$ level.

no-buffer site at baseflow (Fig. 5a). TSS averaged less than or equal to 10 mg L^{-1} on all sites during baseflow. Although concentrations were generally greater, no TSS responses to the harvest treatment were detected on any site at stormflow (Fig. 5b). When comparing treatment watersheds to the REF site, TSS increased during baseflow on the no-buffer site ($F = 4.52$, $P = 0.039$), and decreased on the 10 m buffer site ($F = 4.04$, $P = 0.049$) (Fig. 6a). Using this analysis, there were no significant responses to treatment during stormflow (Fig. 6b). During stormflow, the highest mean concentration was 35 mg L^{-1} and that occurred on the 10 m site prior to the beginning of timber harvesting activities. Prior to harvest, stormflow TSS values ranged from approximately 18 mg L^{-1} on the 30 m buffer site to 35 mg L^{-1} on the 10 m site, and after harvest ranged from 12 mg L^{-1} on the 30 m buffer site to 25 mg L^{-1} on the 10 m buffer site. Maximum concentrations on the no-buffer site were intermediate in value to the above.

4.3. Stream water temperature

There were no overall differences in stream water temperature on any of our study sites. However, on a seasonal basis, differences in stream water temperature were significant only on the no-buffer site and only during the summer ($F = 22.90$, $P < 0.0001$; Table 4), when mean daily stream water temperature averaged 16.1 °C before harvest and 18.5 °C after harvest. When comparing treatment watersheds to the REF site, stream water temperature increased significantly following harvest on the 30 m buffer site, although actual average differences were < 1 °C (Fig. 7). Maximum daily summertime water temperatures for pre- and post-harvest periods were: no-buffer, 18.5–22.7; 10 m buffer, 18.4–18.8; 30 m buffer, 19.4 – 19.8; REF, 16.9 – 18.0, respectively. On the no-buffer site, sustained temperatures of greater than 20 °C occurred on several consecutive days during the summer of 2006 and lasted for several hours on each occurrence.

5. Discussion

Water quality responses to timber harvest were generally confined to the no-buffer site. Observed increases in all cation concentrations, particularly K (Tables 2 and 3), illustrate the mobility

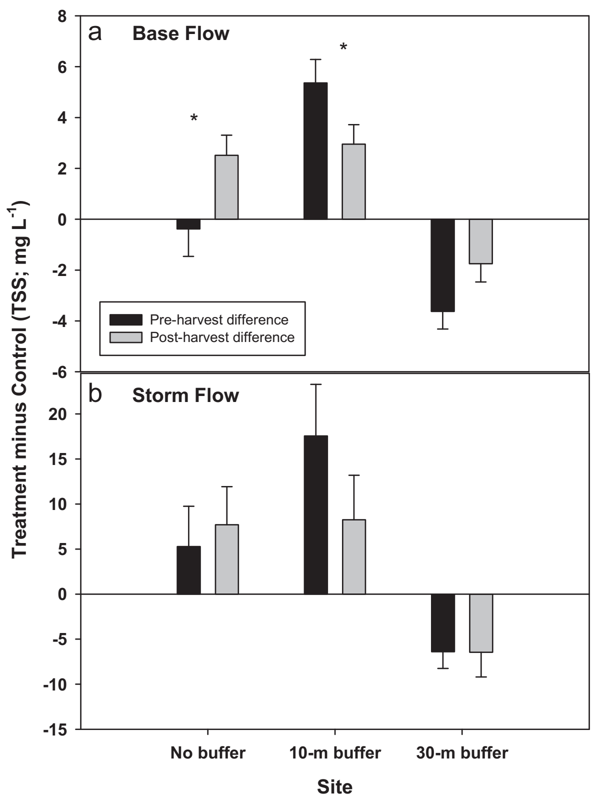

Fig. 6. Pre- and post-harvest differences in total suspended solids (TSS) between the treated and the REF sites during (a) baseflow and (b) stormflow. An '*' above pre- and post-harvest differences in means for each site indicates significant difference between pre- and post-harvest TSS concentrations relative to the REF site. Differences were evaluated at the $\alpha = 0.05$ level.

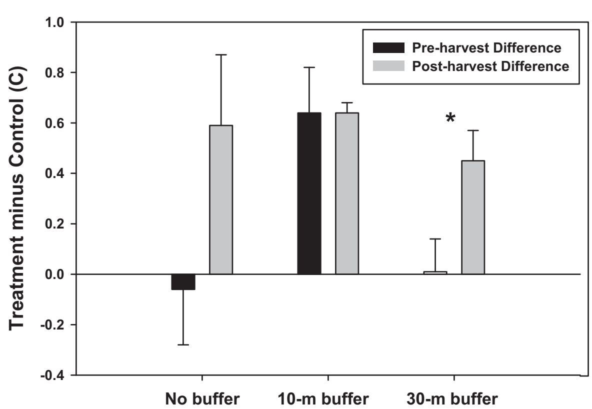

Fig. 7. Pre- and post-harvest differences in stream water temperature (C) between the treated and the REF sites. An '*' above pre- and post-harvest differences in means for each site indicates significant difference between pre- and post-harvest temperature relative to the REF site. Differences were evaluated at the $\alpha = 0.05$ level.

of these nutrients following upland disturbance. Mechanisms for cation leaching have long been understood. McColl and Cole (1968) demonstrated how the leaching of cations is linked to the formation of bicarbonate in soil solution where the hydrogen ions associated with bicarbonate replace exchangeable cations on the soil matrix. Bicarbonate is formed from dissolved carbon dioxide derived in large part from microbial and root activity in the soil and thus any increases in bicarbonate may be indirectly linked to increases in production of carbon dioxide (Remezov et al., 1964). The only increases in bicarbonate in this study occurred during baseflow on the 10 m and 30 m buffer sites (12% on both sites). The result was concentration increases for all base cations (K, Na, Ca, Mg) of from 8 to 22% on the 30 m site and from 8 to 21% for Na and K, respectively, on the 10 m site. Although bicarbonate did not increase significantly on the no-buffer site, cations increased in concentration by 13–36%. On the 10 m and 30 m sites, the increase in base cation concentration may be due to increased fluxes of mobilized cations from the upslope harvested areas. On the no-buffer site, this response may be due to both increased fluxes of mobilized cations from the upslope harvested areas and the lack of uptake in the riparian area due to vegetation removal. Stormflow responses were more variable when K increased in concentration on all cut sites following harvest, likely due to greater mobility in the soil compared with other base cations.

With the exception of increases in [NO_3], increases in other solutes were small by comparison with pre-harvest levels, and represented no threat to water quality. The large increase in [NO_3], particularly during stormflow when stream solute concentrations tend to decrease due to dilution (Fig. 2), may have been the result of the flushing of accumulated nitrate in the riparian zone. Typical changes in [NO_3] from base to stormflow in headwater systems in the region are decreases by as much as two-thirds under undisturbed conditions (Coweeta Hydrologic Laboratory, unpublished data). The steady decline in [NO_3] with time on the no-buffer site, illustrated in Fig. 1, may be the result of increased uptake by recovering vegetation on that site. Natural regeneration occurs rapidly in this region following timber harvest. Boring et al. (1981) reported leaf area recovery on a clearcut south-facing southern Appalachian watershed had reached greater than 25% of pre-harvest levels within the first year following harvest, and 68% by the third year (Boring et al., 1988). Nitrate is a very mobile soil nutrient that forms during the decomposition of organic material, and during dry periods in the absence of plant uptake, can accumulate in the upper soil layers but becomes mobile during rain events that flush the accumulated N into the stream (Creed et al., 1996). Most of the post-harvest period and part the pre-harvest period during this study can be characterized by prolonged periods of no or very small amounts of precipitation punctuated by infrequent and intense storms. This pattern of precipitation may explain the strong response in nitrate levels on the no-buffer site during stormflow, where stream side vegetation that would have normally sequestered mobilized nitrate was removed during harvest. This phenomenon has been reported by other investigators and is referred to as the 'Flushing Hypothesis' (Hornberger et al., 1994; Creed et al., 1996). As described in Creed et al. (1996), when the soil saturation deficit is high, N accumulates in the upper layers of the soil, and as the soil saturation deficit decreases (following storms), the formation of a saturated subsurface layer flushes N from the upper layers of the soil into the stream. Ocampo et al. (2006) drew similar conclusions about N flushing in a project in a western Australian catchment. They concluded that low antecedent moisture conditions at the time of precipitation events resulted in the largest amount of N flushing. In our study, the exaggerated increase in nitrate concentration during stormflow shown in Fig. 2, we hypothesize to be the result of the combination of two mechanisms; (1) no or very little uptake by riparian vegetation following harvest on the no buffer site, and (2) flushing of accumulated nitrate from the terrestrial system into the stream following rain events. The lack of a stream nitrate response on the 10 m and 30 m sites may be due primarily to vegetation uptake within the uncut riparian area, and the steady decrease in NO_3 concentration on the no-buffer site following its peak is likely due to rapid recovery of vegetation and consequent uptake of mobilized N on that site.

TSS decreased or remained unchanged following harvest on all cut sites during baseflow (Fig. 5 and Table 2), and it is believed to be the result of overall site improvements, primarily the reshaping of existing roads and the addition of brush-barriers, during preparation for harvest that eliminated pre-existing sources of sediment.

Table 4
Mean seasonal stream water temperature (°C) ($N = 12$) by site for pre- and post-harvest periods. Statistics are derived from General Linear Model procedure (SAS Institute Inc., 2004) and differences between pre- and post-harvest were evaluated at $\alpha = 0.05$.

Site	Season	Pre-harvest	Post-harvest	*F*-Value	*P*-Value
Reference	Winter	6.6 (0.25)	6.2 (0.55)		n.s.
	Spring	10.6 (0.61)	10.6 (0.70)		n.s.
	Summer	15.1 (0.20)	16.0 (0.33)		n.s.
	Fall	12.9 (0.62)	14.8 (0.48)		n.s.
No-buffer	Winter	6.0 (0.34)	4.1 (0.88)		n.s.
	Spring	10.4 (0.72)	11.2 (0.93)		n.s.
	Summer	16.1 (0.29)	18.5 (0.31)	22.9	<0.0001
	Fall	13.5 (0.95)	11.6 (1.67)		n.s.
10 m buffer	Winter	6.3 (0.36)	6.5 (0.27)		n.s.
	Spring	11.1 (0.72)	11.8 (0.67)		n.s
	Summer	16.5 (0.21)	16.9 (0.29)		n.s.
	Fall	13.6 (0.72)	15.4 (0.54)		n.s.
30 m buffer	Winter	6.5 (0.31)	5.9 (0.55)		n.s.
	Spring	10.5 (0.65)	10.9 (0.77)		n.s.
	Summer	16.4 (0.40)	17.3 (0.37)		n.s.
	Fall	14.3 (0.69)	15.6 (0.70)		n.s.

The two dominant mechanisms for reductions in sediment delivery to streams are (1) filtering through riparian vegetation (Cooper and Gilliam, 1987; Lakel et al., 2006), and (2) particle size sorting as the overland flow velocity decreases as it passes through riparian vegetation (Cooper et al., 1987). Research has demonstrated that it is common for harvesting activities to increase surface water runoff for up to five years or until vegetation advances sufficiently to reduce overland flow (Kochenderfer and Edwards, 1990; Aust and Blinn, 2004). Neary et al. (1993) reported that the effectiveness of these processes is a measure of the adequacy of the riparian buffer for protecting water quality. In this study, mitigation in the form of brush barriers along the roads may have served as filters for sediment leaving the road surface, and the addition of slash to the buffer zone itself on all sites as a result of the harvest operation, may have functioned in a similar manner. When examining overall means, there was no difference in TSS on the no-buffer site between pre- and post-harvest periods; although, relative to the control, the observed increase in TSS on the no-buffer site was statistically significant, the magnitude of the increase was small. The lack of a TSS response during stormflow may be due to legacy sediment migration in the stream channels overwhelming any additional sediment due to the harvest operation. In either case, average concentrations are at least equal to or below published water quality standards (USEPA, 1999).

The use of cable-yarding technology provides additional explanation for the small or no TSS response following harvest. Cable-yarding technology was developed to facilitate access to steep landscapes deemed at risk to erosion in the event of extensive disturbance associated with ground-based yarding systems. The benefit of its use is minimal forest floor disturbance and exposed mineral soil. In addition, fewer access roads are required using cable-yarding and on our study sites, no additional roads were constructed. On the area where skidder logging occurred, exposed surfaces were immediately stabilized. Hence, the combined benefits of minimal soil disturbance, lack of the need for additional road construction, and prompt attention given to disturbed areas; best explain the lack of a TSS response following timber harvest in this study.

Stream water temperature, an important water quality parameter for trout reproduction and survival and overall stream productivity, did not change appreciably following harvest except on the no-buffer site (Table 4). Although maximum water temperature increased on that site by 4.2 °C following harvest to an absolute maximum of 22.7 °C, average daily temperature during the summer increased by 2.4 °C from 16.1 to 18.5 °C, which was below the temperature threshold for reproduction and establishment of young for all species of trout (i.e., 22.3 °C for brook trout [*Salvelinus frontinalis* Mitchell], 24.0 °C for rainbow trout [*Oncorhynchus mykiss* Walbaum], and 24.1 °C for brown trout [*Salma truta* L.]) found in the region (Eaton et al., 1995). Maximum temperature remained below 20 °C on all other study sites. Ringler and Hall (1975) reported increases in water temperature after logging in an Oregon watershed and attributed this increase to reduced forest cover over the stream surface. Similarly, Johnson and Jones (2000) reported increases in stream maximum summer time temperature that remained elevated for 15 years following harvest. They attributed the increase to removal of riparian vegetation. Swift (1983) reported that daily maximum summer time temperatures increased by 3.3 °C for the first two years following harvest, including riparian vegetation, on a south facing watershed. Daily maximum temperature increases were reduced the following three years to 1.2 °C on average. In that study, over 950 m^2 of stream were exposed on a south facing slope following logging. In contrast, exposed stream area on the no-buffer site was considerably less, approximately 300 m^2, and the aspect was more easterly. The observed increase in daily maximum temperatures in our study (4.2 °C) immediately after harvest is consistent with the pattern reported by Swift (1983), but the magnitude of increase was greater on the no-buffer site in our study, even though there was less exposed stream surface area on that study site. The explanation may lie in the fact that because of the unusually low baseflows due to below average precipitation, soil warming in the harvested riparian zone likely contributed to increases in stream water temperature more so than when discharge is at normal levels. This postulate may also hold for the observed increase relative to the REF on the 30 m buffer site.

6. Conclusions

Because of considerable variation among first-order catchments with respect to above and below ground conditions in this region, it is recommended that when attempted, extrapolation of these results beyond this case study should be viewed with caution. However, the results of this study do provide important insight into the degree to which headwater streams respond to upland disturbance with and without vegetated riparian buffers.

It is important to keep in mind that although concentrations did increase for many stream water constituents, levels were not high enough to pose a threat to water quality as defined by U.S. Environmental Protection Agency (USEPA) and North Carolina State published standards. Although these streams are not designated as sources for drinking water, the highest observed NO_3–N concentrations that occurred during stormflow after harvest (0.37 mg L^{-1}) were a fraction of the USEPA standard for drinking water of 10 mg L^{-1}. Similarly, TSS concentrations during baseflow were at or below USEPA standards for trout water (10 mg L^{-1}). Further, increases in stream water NO_3 concentration were confined to the no-buffer site, and TSS remained the same or decreased after harvest on all sites. Compared to the REF, TSS increased on the no-buffer site but differences were minimal.

Although alternative land uses, or terrestrial and aquatic resources, may have different requirements, these results suggest that for riparian buffer widths of 10 m and wider, forest harvesting activities like the ones implemented in this study do not degrade stream water quality to any great degree. Hence, where cable-yarding technology is employed, 10 m wide buffers in these ecosystems may provide effective protection from timber harvesting activities with respect to stream water chemistry, sediment, and temperature.

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