Prescribed Fire Effects on an Aquatic Community of a Southwest Montane Grassland System

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Abstract

Use of prescription fire has long been recognized as a reliable management tool to suppress vegetative succession processes and to reduce fuel loading to prevent catastrophic wildfires, but very little attention has been paid to the effects on aquatic systems. A late fall prescribed burn was implemented to characterize effects on an aquatic community within a montane grassland system in northcentral New Mexico, USA. The fire treatment was consistent with protocols of a managed burn except that the fire was allowed to burn through the riparian area to the treatment stream to replicate natural fire behavior. In addition to summer and fall pre-burn assessment of the treatment and a reference stream, we characterized immediate post-fire effects (within a week for macroinvertebrates and within 6 months for fish), and seasonal effects over a 2 year period. Responses within the treatment stream were compared to an unburned reference stream adjacent to the prescription burn. During the burn, the diel range in air temperature increased by 5°C while diel range in water temperature did not change. Carbon-nitrogen ratios did not differ between treatment and reference streams, indicating the contribution of ash from the surrounding grassland was negligible. Although total taxa and species richness of aquatic macroinvertebrates were not altered, qualitative indices revealed departure from pre-burn condition due to loss of sensitive taxa (mayflies and stoneflies) and an increase in tolerant taxa (midges) following the burn. Within one year of the burn, these attributes returned to pre-burn conditions. Adult Brown Trout Salmo trutta density and recruitment did not differ between pre- and post-burn collections, nor did fish condition differ. Fire is rarely truly replicated within a given study. Although our study represents one replication, the results will inform managers
about the importance in timing (seasonality) of prescription burn and anticipated effects on aquatic communities.

Forested ecosystems in western North America have evolved with fire and rely on its influence to increase ecological productivity through mobilization of nutrients to increase energy flow. The timing and severity of fire remain causes of concern. Heavy fuel loads and low fuel moisture attributable to periods of drought or seasonal ebbs in precipitation exacerbate fire conditions. The impacts of fire on aquatic systems are complex and not entirely dependent on fire severity, but on the fundamental characteristics of the watershed (Bêche et al. 2005), stream channel morphology (Benda et al. 2003), riparian vegetation (Rieman et al. 2003; Pettit and Naiman 2007), and the frequency of post–fire disturbances such as flooding (Vieira et al. 2004). Persistence of aquatic communities is closely related to periodic disturbances which shape habitat quantity (Rieman and McIntyre 1995), quality (Minshall et al. 2001), and heterogeneity (Resh et al. 1988). Direct effects of wildfire include increases in stream temperature (Hitt 2003) and changes in aquatic chemistry which can approach lethal toxicity thresholds for fishes (Spencer and Hauer 1991). Indirect include the loss of streamside vegetation, resultant increases in stream temperature from increased solar radiation, and increased sedimentation (Gresswell 1999).

Fire effects can vary in severity from acute (between onset of fire and initial runoff post–fire) to chronic (extending for years), and influence both biotic communities and abiotic components of aquatic systems (Minshall 2003). Acute effects result in localized changes to the structure and composition of the riparian area that include...
accumulation of downed woody debris, shifts in composition of aquatic
macroinvertebrates and fish densities due to mortality or emigration (Minshall et al. 1997; Hitt 2003; Burton 2005; Bêche et al. 2005; Howell 2006), decreases in stream
channel stability with variable discharge (Rieman and Clayton 1997; Dunham et al. 2007), elevated sediment transport (Benda et al. 2003), and elevated water temperatures
(Dunham et al. 2007). In sharp contrast, chronic effects from a severe burn due to
vegetative re-growth and in-stream woody debris are not often realized for decades.
Long term effects also will likely occur if stream channels undergo geomorphic
reorganization, if the variability in timing and amount of runoff exceeds normal annual
and inter-annual patterns that disrupt hydrologic patterns, and if riparian areas are unable
to stabilize soil erosion and ameliorate heat insolation (Dunham et al. 2003).

Land managers have employed controlled burning or prescription fire to
proactively manage potential hazardous fuel loading while maintaining ecological
integrity of forested and grassland systems. If carefully planned, prescriptions for
wildland fire should protect against erosion and sedimentation to aquatic systems and
have little to no effect on water quality (USEPA 2005). Fires that burn down through the
litter to the mineral soil will adversely affect water quality through erosion. The amount
of erosion will depend on fire severity, amount of ground cover remaining on the soil,
grade of slope (the steeper, the more likely soils will move), erodibility of the soil,
removal of riparian vegetation, time and amount of rainfall post-burn, and how rapidly
the site re-vegetates. While the effects of prescription fire on water quality has received
some attention (Richter et al. 1982; Stephens et al. 2004; Elliott and Vose 2005; Bêche et
al. 2005), less attention has been paid to short- and long-term effects on fish and
macroinvertebrate communities. Bêche et al. (2005) related low to moderate burn severity from a prescription burn to only minor effects in the benthic macroinvertebrate community despite altered periphyton levels. Population responses of fishes to prescription burn have been less documented and would presumably be related to severity, size, and patchiness of the burn. Arkle and Pilloid (2010) found no immediate or delayed effects to fish populations in Idaho streams within watersheds subjected to prescribed burn of low to moderate severity. While the authors did not observe deleterious effects, this is not sufficient evidence to assume prescribed burns, if conducted according to protocol, would not affect fish populations and the invertebrate community they rely upon.

The purpose of our research was to assess impacts of a prescribed burn during the cool, autumn season on an aquatic community within a grassland system on the Valles Caldera National Preserve, New Mexico. An important management goal of the Valles Caldera National Preserve, hereafter referred to as the Preserve, is to utilize prescription burn throughout its grasslands and Ponderosa pine forests as a reliable management tool to return pre-settlement fire regimes, and reduce fuel loading to prevent uncharacteristic wildfires. Extensive logging operations and wildfire suppression during the 20th Century throughout the Preserve resulted in increased secondary succession and encroachment of shrubs and trees on grasslands that provide recreational fishing opportunities. To evaluate the ecological impacts of managed low-intensity fire on the Preserve, we designed and implemented an experimental prescribed burn in one of the Preserve’s upland watersheds (Valle Toledo). A prescribed burn treatment consistent with protocols for a managed burn in the fall season (November) was applied to low gradient open
meadow grassland and adjacent Ponderosa pine forest, except that the fire was allowed to
burn through the riparian area to the stream bank to replicate natural fire behavior. Our
study objectives were to characterize immediate (within a week), seasonal and longer-
term (two years) effects of this prescription burn on water quality, benthic
macroinvertebrate communities, and fish populations. Thus, our null hypothesis was that
no detectable effects of selected variables will be observed between burned and unburned
treatments.

Methods

Study Site Description.—Valles Caldera National Preserve encompasses
approximately 36,000 ha of the Jemez Mountains in northcentral New Mexico (Sandoval
County). The Preserve is a caldera of a 1.25 million year-old super-volcano,
characterized by multiple volcanic domes and intervening grassland valles (Spanish term
for treeless valleys). Under private ownership from 1860 through 2000, the area was
managed for livestock (sheep and cattle) grazing, timber harvest, hunting and fishing,
mining, and geothermal energy exploration (Martin 2003). After federal acquisition in
2000, the Preserve was mandated to maintain a multiple land-use policy balancing
sustainable harvest of natural resources (i.e., grazing for livestock, timber production,
hunting and fishing) with recreation and ecosystem health. Approximately 9,000 ha of
the Preserve are highly productive grassland valles, surrounded by upland montane
coniferous forests. Within the watershed are a series of first- and second-order streams
with riparian areas composed of sedges Carex spp., bunchgrasses Danthonia, Festuca,
Phleum spp., sod-forming grasses Poa spp., and forbs. The dominant soils in the valles
are Cosey-Jarmillo association, which are deep, well drained, and moderately permeable (USDA 2008). The remaining areas throughout the Preserve and adjacent federal lands consist of montane forests, dominated by ponderosa pine *Pinus ponderosa*, white fir *Abies concolor*, Engelmann spruce *Picea engelmannii*, Douglas-fir *Pseudotsuga menziesii*, Colorado blue spruce *Picea pungens*, aspen *Populus tremuloides*, and juniper *Juniperus* spp. Average pre-1900 fire return intervals in forested areas throughout the Preserve varied from 5 to 20 years, while grassland fires in the open meadows of the valles occurred on average every 1.6 years (Allen 2002; Touchan et al. 1996; Allen et al. 2008; Dewar 2011).

**Prescribed fire treatment.**—The prescribed burn was undertaken in Valle Toledo (35°57’50” N, 106°29’31” W, elevation 2,607 m) 1–2 November 2005. A late-season burn was selected because of higher fuel moisture reducing chances of a burn escaping the targeted area. Valle Toledo is a 1,040 ha grassland containing a second order stream (Rio San Antonio) that runs east to west and is bordered on the western edge by a first order stream (Rito de los Indios; Figure 1). A graded dirt road surrounds the burn area, and was used as a firebreak during the burn. Interior areas of the burn were lit by aerial ignition devices from a helicopter. The burn covered 728 ha or approximately 70% of the target area. Areas that did not burn either had insufficient fuel loads or were too moist and replete with green vegetation. The majority of riparian areas within the study reaches of the stream were burned to the stream edge resulting in successful attainment of the targeted goal which was to replicate natural fire behavior. Grassland fuel loads (dry mass) averaged 1,740 kg/ha. Flame lengths in the grassland ranged from 0.3 m to 3.0 m high, with rate of spread averaging 0.4 m/s. In the forest, the fire burned less intensively;
sub-canopy herbaceous fuel loads averaged 1,205 kg/ha, with flame lengths between 0.1 and 0.3 m and spread-rate of 0.04 m/s. Few trees were killed, although needle and branch loss in the lower canopy were extensive. In the forest, standing grasses and forbs were burned and consumption of the pine needle litter layer was limited to the uppermost 2–3 cm.

<Study stream.– Rio San Antonio is a second order stream draining approximately 120 km² flowing east to west across the Preserve’s northern portion, with headwaters in the Sierra de los Valles near the Preserve’s eastern boundary. The portion of the stream flowing through the Valle Toledo is a C-4 type stream (i.e., slightly entrenched, gravel dominated, with a well-developed flood plain) according to the Rosgen (1994) scale. Rito de los Indios is a spring-fed first order stream with its headwaters in the Sierra de los Valles on the Preserve’s northern boundary that drains approximately 19 km². The portion of the Rito de los Indios that flows through Valle Toledo is characterized as an E-4 type stream (i.e., slightly entrenched, gravel and cobble dominated, well developed flood plain; Rosgen 1994). Although Rito de los Indios exhibited lower width to depth ratios, the stream was selected to represent reference conditions due to its close proximity to the burn area and comparable physical features such as entrenchment, substrata, and area of flood plain. Rito de los Indios borders the western edge of the Valle Toledo and eventually joins Rio San Antonio downstream of the Valle Toledo burn. Hereafter, the Rio San Antonio and Rito de los Indios will be referred to as the treatment and reference streams, respectively.
Water quality and C:N ratios in sediment. – Water quality was assessed by placing two data sondes (Model 6920, Yellow Springs Instruments, Yellow Spring, OH) within the reference and treatment streams upstream from the confluence (Figure 1). Water temperature (°C), dissolved oxygen (mg/L), pH, and conductivity (µS/cm) were monitored at 15 min intervals from June to November 2005 and from April to November 2006 and 2007. Air temperature (°C) was obtained from an on-site Remote Automated Weather Station within the Valle Toledo. Monthly cumulative precipitation (mm) throughout the study (June 2005–September 2007) and over 31 years (1981–2011) was obtained from the Natural Resources Conservation Service’s Quemazon snow telemetry station (SNOTEL) located near the eastern boundary of the Rio San Antonio watershed (Site 708; 35°55′ N, 106°24′ W, Elevation 3,167 m, Los Alamos County, New Mexico).

In each of the treatment and reference streams, three study reaches were arbitrarily selected to collect sediments (n = 3) pre-burn (fall 2005), immediately post-burn (late-fall 2005), following snow-melt but before monsoonal summer rains (six months post-burn, spring 2006), and one year post-burn (fall 2006). To assess the contribution of carbon from ash, sediments were collected to assess carbon and nitrogen ratios (C:N) using a Hess–style plastic sediment sampler (500 mm [height] by 200 mm [dia]). Sediments were processed through a series of U.S. Standard stainless steel sieves that ranged from 0.25 mm to 16 mm and then oven-dried (60°C). From the smallest sized fractions (0.25–0.5 mm), a sub–sample was weighed to the nearest 0.001 g and analyzed for carbon and nitrogen by high temperature combustion on a ThermoQuest Elemental Analyzer (Model NC2100, CE Instruments, Wigan, UK). Percentages of carbon and nitrogen gases were eluted on a gas chromatography column, detected by thermal
conductivity and integrated to yield carbon and nitrogen content (Pella 1990a, 1990b). Carbon and nitrogen were confirmed by mass percent loss on ignition by combustion at 500°C for 2 h. Pre- and post-combustion weights of each sample were determined as mass percent loss of organic matter (Ben-Dor and Banin 1988; Nelson and Sommers 1996).

**Macroinvertebrates.**– Three study reaches in each stream (treatment and reference) were selected to collect benthic macroinvertebrates, pre-burn (spring and fall 2005), immediately after the burn (late fall 2005), and four times post-burn (spring and fall 2006, and spring and fall 2007; Figure 1). These study reaches spanned the area to be burned in the Valle Toledo, were adjacent to the burned area in the reference stream (Rito de los Indios), and were 5 to 10 m upstream to where sediments were collected. Two habitats were sampled each season for benthic macroinvertebrates: Riffles of comparable substrates (i.e., pebble and cobble ranging from 50 to 150 mm); and pools (< 0.5 m depth). Macroinvertebrates were sampled within a few weeks of the pre-burn sample date to minimize seasonal variation and to avoid peak flows. Within each reach, benthic aquatic macroinvertebrates were collected from 3 pools and 3 riffles using a modified Hess–type circular sampler (0.06 m²; Jacobi 1978). This was repeated across all three reaches within treatment and reference streams. Collections at all locations were usually completed within two consecutive days to minimize spurious effects of weather. Samples were preserved in 99% ethanol in the field and later identified to lowest taxonomic level possible.
Ecological metrics of Plafkin et al. (1989), Barbour et al. (1992, 1999) were used to assess effects of the burn on community structure, community balance, and functional feeding components of the aquatic macroinvertebrate community compared to pre-burn conditions through time and across habitat (pools and riffles). The Hilsenhoff Biotic Index (Hilsenhoff 1987) assessed sensitivity of the aquatic macroinvertebrate community to sediment and nutrient perturbation. Additional indices included mayflies (Ephemeroptera), stoneflies (Plecoptera), and caddisflies (Tricoptera), collectively referred to as the EPT index. Additionally, when chironomids (i.e., tolerant organisms) are included with EPT assemblage (EPT/EPT + Chironomidae), an even distribution among all four taxonomic groups indicates optimal biotic conditions; whereas, a shift to a lower ratio indicates a disproportionately greater number of tolerant chironomids.

**Fish surveys.**– Preliminary surveys in both treatment and reference streams revealed a fish community dominated by Brown Trout *Salmo trutta*. Longnose Dace *Rhinichthys cataractae* were incidental throughout the study and not considered in the analysis. Three transects (50 m each) were established in close proximity to the macroinvertebrate sample sites within treatment and reference streams to obtain a representative selection of runs, riffles, and pools (Figure 1). Baseline data for the Brown Trout population within both streams was collected spring and fall 2005 to assess adult (≥ 170 mm) and juvenile (< 170 mm) densities (#/100 m of stream) and relative weight ($W_r$) prior to the prescribed burn. Treatment and reference streams were surveyed spring and fall of 2006 and 2007 to evaluate delayed effects from the prescribed burn as well as annual recruitment to the Brown Trout population. Within each sample reach, Brown Trout were sampled using
direct current in a three-pass depletion with a backpack electrofishing unit (Model LR–24, Smith–Root Inc., Vancouver, WA). Block nets (6 mm mesh) were set above and below the sample reach to prevent emigration and immigration of fish during electrofishing. Upon capture, total length (TL; mm) and weight (g) were recorded for all captured fish. These fish were subsequently placed in a live box within the stream until completion of the final electrofishing pass and then returned to the sample reach.

Abundance estimates for each sampled reach were calculated based on maximum likelihood removal methods (Zippen 1958) and corrected to relative density (# fish/100 m²) by measuring stream wetted width every 10 m throughout the 50 m electrofishing reach. Adult and juvenile population estimates were calculated separately to minimize potential bias associated with differences in capture efficiency using electrofishing among size classes. Juvenile fish were not captured during spring electrofishing occasions. Thus, density of juvenile fish was estimated for only fall surveys and used to evaluate recruitment. Adult fish were assigned to age classes from length frequencies and assigned to age classes and Wr indices for ages 2–5; age estimates from length frequency histograms were verified from a subsample of scales from the population. Fish body condition was evaluated prior to and following the prescribed fire using Wr for Lotic Brown Trout (Milewski and Brown 1994).

Data analyses.—The purpose of the study was to assess impacts of a prescription burn on the aquatic community within a grassland system with the null hypothesis of no detectable effects of selected variables between burned and unburned treatments. As this study was based on only one prescription burn in one watershed, the “treatment” was not
replicated; however, fire studies are often opportunistic and lack replication (Mantgem et al. 2001). Our replications were spatial units within burned and reference areas. Therefore, we restricted our conclusions to differences between pre- and post-burn collections at our treatment (burned) and reference (unburned) sites.

Models with equal and unequal variances were fit as well as different covariance structures over time using PROC MIXED in SAS (SAS Institute 2009). The model with the lowest Akaike’s Information Criterion value corrected for small samples (AICc; Akaike 1973) was selected to make inferences. Residual analysis was conducted to identify outliers to visually assess variance structures incrementally. Unless otherwise noted, model assumptions of equal variance and normality were met. Differences were detectably different when $P \leq 0.05$.

Differences in C:N ratios were assessed using a repeated measures analysis of variance (ANOVA) comparing sites (treatment and reference) during 2005 (pre-burn spring and post-burn fall) and post-burn spring and fall 2006. Site*time comparisons were conducted within seasons (e.g., spring versus spring and fall versus fall). To identify differences between the treatment and reference areas, the treatment*time interaction was examined because a main effect for treatment without an interaction would reflect differences between treatment and reference areas prior to the burn that were maintained throughout the study period regardless of the burn.

Summary statistics were calculated for the treatment and reference areas using raw abundance for each habitat (pool, riffle) across seasons (spring, fall) to obtain total standing crop (total number of organisms/m²) and taxa richness (total number of detected taxa). Differences in total standing crop and species richness of benthic
macroinvertebrates were assessed using repeated measures ANOVA with a 2 x 2 factorial
treatment structure using sites (treatment and reference) by habitat (pools and riffles). To
identify differences in total standing crop and species richness between treatment and
reference areas, treatment*time interactions were examined for each habitat using pre-
planned contrasts to identify an immediate burn effect (pre-burn fall 2005 versus late-fall
2005, and longer term effects by comparing pre-burn spring 2005 versus post-burn spring

Differences in fish density and \( W_r \) by age classes (2, 3) were assessed using
repeated measures ANOVA comparing sites (treatment and reference) over time (pre-
burn, post-burn 2005, post-burn 2006, post-burn 2007). Site*time comparisons were
conducted within seasons (e.g., spring versus spring and fall versus fall). To identify
differences between the treatment and reference areas, the treatment*time interaction was
examined because a main effect for treatment without an interaction would reflect
differences between treatment and reference areas prior to the burn that were maintained
throughout the study period regardless of the burn. Few age 4 and 5 fish were captured
throughout the study and thus were not analyzed, but summary statistics are reported.

Results

Water quality and C:N ratios in sediment.— Stream and air temperature profiles prior
to the prescribed burn (27–31 October) revealed consistent diel patterns (Figure 2). Prior
to the burn, the diel range in air temperature was 21°C (−8.0 –13.0°C) while the diel
range in stream temperature was 6°C (7.4–13.4°C). During the burn (1–2 November), the
diel range in air temperature increased to 26°C (−8.0 –18°C) in contrast to the diel range
for stream temperature which remained 6°C (7.1–13.1°C). We noted an immediate
decline following the burn (3–5 November) in which minimum air temperature increased
to 3.2°C (Figure 2). Despite the rise in air temperature, stream temperatures were within
the diel range prior to the burn. Between pre- and immediate post-burn, daily maximum
temperatures did not exceed criteria for impaired cool water streams in New Mexico
(20°C; New Mexico Environment Department, NMED 2006). Water temperature,
dissolved oxygen concentrations, conductivity, and pH varied little prior to, during, and
immediately after the burn within the treatment stream and were deemed acceptable when
compared to water quality criteria of coldwater fishes (NMED 2006). Mean daily water
temperature (MDWT) before the burn in 2005 was 15.1°C and did not vary after the burn
in 2006 (15.1°C) or 2007 (15.0°C; Table 1). The highest MDWT recorded in one day
was prior to the burn in 2005 (17.9°C) compared to subsequent years (16.4°C in 2006 and
2007). Conductivity and pH in the treatment stream throughout summer months (1 June–
31 August) did not vary pre- and post-burn (Table 1). Monthly precipitation was below
the long-term average during spring 2006 (post-burn) which was reflected by severe
drought (USDM 2011). Precipitation patterns deemed “normal” returned the summer of
2006.

An increase in C:N was observed over time ($F_{2,18} = 5.76; \ P = 0.012$) for both
treatment and reference sites, but there was no detectable treatment*time interaction ($F_{4,18} = 1.49; \ P = 0.247$) indicating changes were not attributable to the burn (Figure 3).
Average C:N varied between pre-burn spring (11.5 ± 0.42; mean ± standard error) and
after snowmelt post-burn spring 2006 (13.2 ± 0.39) similarly to the reference site (pre-
burn spring 13.4 ± 0.47 versus post-burn spring 16.2 ± 0.37) indicating contribution of a
source unrelated to the fire from within the watershed. Pre-planned contrasts of C:N changes between the two time frames (pre-burn spring 2005 and post-burn spring 2006) revealed no detectable difference ($T_{18} = -0.78; P = 0.448$) between treatment and reference sites, further indicating no fire effect.

Macroinvertebrates—A diverse assemblage of aquatic benthic macroinvertebrates totaling 110 taxa was collected throughout the study (Appendix). Of this total, 88% were insects and included stoneflies (Plecoptera – five taxa), mayflies (Ephemeroptera – seven taxa), caddisflies (Trichoptera – 18 taxa), true flies (Diptera – 52 taxa), damselflies and dragonflies (Odonata – two taxa), true bugs (Hemiptera – three taxa), beetles (Coleoptera – eight taxa), and an aquatic moth (Lepidoptera – one taxon; see Appendix). The most dominant group belonged to the true fly family Chironomidae (non-biting midges) which consisted of 34 genera or 31% of the total taxa collected throughout the study. All dominant taxa collected pre-burn were collected post-burn throughout sites in the treatment stream while one dominant taxon (caddisfly Limnephilus sp.) appeared for the first time in the reference stream one year and two years post-burn.

The top ranked model allowing for unequal variances over time for total standing crop in fall collections revealed no detectable treatment by time interaction prior to the burn (2005) and immediately following the burn (2005; $F_{3,11} = 2.01; P = 0.171$). Thus no immediate burn effects on total standing crop of benthic macroinvertebrates in either pools or riffles were observed between treatment and reference sites (Figure 4). This conclusion was reaffirmed in the pre-planned contrasts that indicated little change for total standing crop in pools ($T_{11} = 0.31; P = 0.760$) and riffles ($T_{11} = -1.95; P = 0.078$).
When we compared total standing crop in the fall prior to the burn (2005) with fall collections one year (2006) and two years (2007) post-burn, the top ranked model allowing for unequal variances over time also revealed no differences in the treatment by time interaction ($F_{6,14.8} = 1.73; P = 0.183$). Pre-planned contrasts of changes in total standing crop revealed no detectable differences in pools ($T_{11} = -0.20; P = 0.843$) or riffles ($T_{11} = -2.0; P = 0.066$). In spring collections, the top ranked model allowing for unequal variances among treatments revealed no treatment*time interaction prior to the burn (2005), one year (2006), and two years (2007) post-burn ($F_{6,12.3} = 2.10; P = 0.127$). Pre-planned contrasts of changes in total standing crop revealed no detectable differences in either habitat ($P > 0.30$).

Changes in species richness in the treatment stream prior to the burn (2005) and immediately following the burn (2005) were not affected by fire as indicated by no detectable treatment* time interaction ($F_{3,16} = 0.96; P = 0.436$). There were no immediate effects of the burn on species richness in either pools or riffles between treatment and reference sites (Figure 5). Pre-planned contrasts of changes in species richness indicated differences were negligible in pools ($T_{16} = 0.00; P = 1.0$) and riffles ($T_{16} = -1.67; P = 0.115$). When we compared species richness in the fall prior to the burn (2005) with fall collections one year (2006) and two years (2007) post-burn, the top ranked model revealed no treatment*time interaction ($F_{6,24} = 2.10; P = 0.090$). The pre-planned contrasts of changes in species richness throughout the fall collections revealed no detectable differences in pools ($T_{24} = -0.17; P = 0.869$), but differences were observed in riffles ($T_{24} = -3.34; P = 0.003$). These differences were due to a decrease in species richness in riffles throughout the treatment site while species richness increased.
in riffles throughout the reference site (Figure 5). The top ranked model for species richness revealed no treatment*time interaction prior to the burn in spring (2005), and in the following springs post-burn ($F_{6,24} = 1.03; P = 0.428$). Pre-planned contrasts of changes in species richness revealed no detectable differences in pools ($T_{24} = -1.55; P = 0.135$) or riffles ($T_{24} = -1.08; P = 0.289$).

Small sample size ($n = 3$), considerable variation within pools and riffles, and seasonal response of taxa obscured detectable differences between pre- and post-burn sample collections. However, changes in sensitive and tolerant taxa across time and microhabitat in the treatment stream are worth noting. Departure of the EPT index in both riffles and pools immediately post-burn were due to a decline in mayfly *Tricorythodes* sp. and the predacious stonefly *Isoperla* sp. (Figure 6). *Tricorythodes* sp. decreased from $128.0/m^2 (± 26.82)$ prior to the burn to $74.0/m^2 (± 18.66)$ by the first spring. By the second spring, mayfly had rebounded throughout the pools ($219.0/m^2 ± 72.20$). *Isoperla* sp. decreased in riffles from $81.3/m^2 (± 13.58)$ prior to the burn to $3.0/m^2 (± 3.00)$ spring 2006 and began to return to pre–burn levels by spring 2007 ($49.0/m^2 ± 13.58$).

Departure from the EPT/EPT + Chironomindae index in pools (Figure 6) occurred immediately post-burn as a result of an increase in the tolerant chironomid *Microtendepes* sp. throughout the treatment sites. Prior to the burn, *Microtendepes* sp. increased throughout the treatment pools from $4.3/m^2 (± 1.33)$ to $56.0/m^2 (± 18.21)$ immediately following the burn returning to pre–burn levels ($2.0/m^2 ± 1.15$) by fall of 2007. Two years post-burn, departure from pre-burn conditions throughout pools in the treatment stream as reflected by the suite of ecological indices was negligible.
Regardless of season, we did not detect an effect of the burn treatment on adult salmonid density (Figure 7). Changes in population densities of adult Brown Trout in the fall prior to the burn and one year post-burn (fall 2006) in treatment and reference streams did not differ as revealed by no detectable treatment*time interaction ($F_{2,12} = 0.15; P = 0.861$). Changes in population densities between pre-burn spring (2005) and post-burn spring (2006) in treatment and reference streams did not differ ($F_{2,5.26} = 1.74; P = 0.262$) according to the top ranked model that allowed for heterogeneous variances over time. Densities of adult Brown Trout decreased from 79.6 fish/100 m$^2$ (± 14.48) prior to the burn (fall 2005) to 56.3 fish/100 m$^2$ (± 1.03) one-year post-burn (fall 2006). A similar response was also observed during the same time frame in the reference sites where densities decreased from 105.0 fish/100 m$^2$ (± 29.3) to 67 fish/100 m$^2$ (± 15.5; Figure 7). The top ranked model allowing for unequal variances over time for recruitment revealed no detectable treatment*time interaction ($F_{2,7.2} = 0.05; P = 0.952$). Recruitment in juvenile fish was greater in the treatment stream (range 20.3 – 20.7 fish/100 m$^2$) compared to the reference stream (3.3 – 6.7 fish/100 m$^2$) throughout the study (Figure 7). The top ranked model allowing for unequal variances by treatment for $W_r$ in age 2 fish revealed no detectable treatment*time interaction for fall ($F_{2,7.93} = 0.51; P = 0.616$). Spring collections did not demonstrate any treatment* time interaction either ($F_{2,12} = 0.61; P = 0.558$), but the top ranked model assumed equal variances over time and treatments. There was, however, a detectable difference in $W_r$ of age 3 fish as seen in the treatment*time interaction for fall collections ($F_{2,12} = 9.84; P = 0.003; Table 2$). Pre-
planned contrasts revealed detectable differences ($T_{12} = -4.41; P = 0.001$) in changes of $W_r$ between treatment and reference sites for age 3 Brown Trout between pre-burn (2005) and post-burn (2006). Age 3 Brown Trout in the treatment site exhibited a decrease in $W_r$ while the same age group experienced an increase in $W_r$ at the reference site (Table 2). Differences in $W_r$ changes for age 3 fish between the two streams were no longer evident by 2007 ($T_{12} = -1.79; P = 0.098$). Although spring collections exhibited a similar pattern of decreasing condition at the treatment site and increasing fish condition at the reference site, these differences were not statistically different ($F_{2,12} = 1.71, P = 0.222$; Table 2).

Relative weight was greater in adult Brown Trout (ages 4 and 5) in the treatment stream (range 78 – 98 g) compared to the reference stream (range 70 – 88 g) throughout the study.

Discussion

As global temperatures increase over the next century (IPCC 2007), projected aridity throughout the Southwest will be intensified by La Niña events beyond recent human record (Seager et al. 2007). Grasslands as well as low- to mid-elevation forests comprise the largest land cover in the southwestern U.S. and are prone to frequent fires (Litschert et al. 2012). Severity and wildfire intervals throughout these systems are expected to increase due to precipitation patterns that increase fuel loads during wet El Niño followed by drier La Niña cycles. As such, prescribed burns by land managers during cool seasons should become more commonplace as a safe, effective approach to returning fire to the landscape. Despite extensive burn throughout the riparian corridor, the cool-season prescription burn exerted negligible effects on the aquatic community.
Detrimental effects of fire often occur when a burn is followed by precipitation events resulting in increased stream flow and elevated sediment transport (Benda et al. 2003; Wondzell and King 2003). While the timing of the burn in late-fall was planned to maximize soil moisture, the Southwest was also experiencing drought conditions which minimized chances for heavy rainfall and effects that an inflow of ash would have had on the aquatic community. During the first eight months of the study (November 2005 – June 2006), the Preserve experienced lower than normal precipitation, reducing spring snowmelt runoff and the contribution of ash from surrounding grasslands and riparian area. By the time precipitation returned to the Preserve with the onset of summer monsoon rains July 2006, re-growth of vegetation was returning to the grasslands and riparian areas, minimizing erosion processes and ash input. Within a southwestern grassland, Gori and Backer (2005) demonstrated prescription fire increased ground cover of perennial grasses throughout burned plots compared to un-burned plots thereby increasing the watershed’s capacity to capture runoff and reduce soil erosion. Others have attributed the importance of riparian vegetation as a sediment trap on post-fire sediment deposition to pools (Spina and Tormey 2000).

Fire mobilizes organic matter increasing the C:N signature within the top soil layer (Ojima et al. 1994; Monleon et al. 1997; Rhoades et al. 2004) where precipitation and snowmelt eventually carry these to stream sediments. The hydrology throughout the Preserve is snowmelt-dominated and the grasslands had not experienced an extensive burn since Federal acquisition in 2000. Thus, we anticipated an increase in C:N within the treatment stream (Rio San Antonio) following the first spring snowmelt. However, C:N ratios varied little, indicating that fire effects and subsequent ash input following the...
first snowmelt were negligible. The low to moderate burn severity, combined with lower
than average precipitation within the first year of the burn, resulted in reduced ash and
particle deposition to Rio San Antonio. Of note, C:N ratios were slightly higher (15–
20%) throughout Rito de los Indios compared to Rio San Antonio, reflecting the
contribution of organic matter from the close proximity of the forested watershed.
Conifers are the dominant vegetation within the watershed of the Rito de los Indios and
may have contributed to a higher seasonal pulse of particulate C relative to N (McGroddy
et al. 2004; Starry et al. 2005). While this may have affected the C:N ratio of the
reference stream, C:N ratio in the treatment stream varied only slightly throughout the
study indicating negligible ash and particle deposition from the burn.
Aquatic macroinvertebrate communities have been widely accepted as useful
indicators of fire effects to lotic systems (Minshall et al. 1997; Barbour et al. 1999).
These communities are relatively sensitive and tend to respond rapidly to changes in
temperature, dissolved oxygen, and shifts of instream substrate (Vannote and Sweeney
Taxonomic richness and community composition were related to hydrological
disturbances that affected habitat stability post-fire (Vieira et al. 2004) and varied
according to the time between disturbances rather than the total number of disturbances
(Death and Winterbourn 1995). While the prescribed burn had no immediate or longer-
term effects on the benthic macroinvertebrate community, we noted shifts in ecological
indices reflected in the reduction of a sensitive stonefly *Isoperla* sp. in riffle habitat,
reduction of a mayfly *Tricorythodes* sp. in pool habitats, and an increase in moderately
tolerant chironomid midges in both habitats. In circumstances involving high-intensity
forest fires in northcentral New Mexico, Pippin and Pippin (1981) and Vieira (2003) reported that while densities of benthic macroinvertebrates in wildfire-affected streams recovered within one year, taxa richness was less resilient with the loss of several species of less tolerant stoneflies for up to three years post-fire.

The absence of fire effects on Brown Trout is clearly related to the prescribed burn’s minimal impact on water quality, stream habitat, and food resources (macroinvertebrates). In addition, the fish population’s lack of response may be due in part to the presence of small springs within both streams but upstream of the burn area which may have ameliorated changes in water quality from fire. Dunham et al. (2007) attributed ground water inflow to streams having less pronounced effects on water temperature after severe wildfires in an Idaho watershed. The authors point out that despite the loss of vegetation throughout the riparian areas and the potential for long term effects on the hydrology, ground water inflow would ameliorate water temperature fluctuations and fish would be less likely to suffer exposure to daily peak temperatures from solar insulation of suspended sediments. The overall health and condition of the Brown Trout population in this study revealed a population skewed toward younger healthier fish with overall decreases in $W_r$ as fish age. While there appeared to be a treatment effect on older fish, as seen in diminished $W_r$ of age 3 fish in the Rio San Antonio, we were unable to tease apart the combined effects of fire with overwintering and lower than normal rain patterns.

Fire has long been recognized as one of the most influential of natural disturbances affecting aquatic systems in the western United States (see review by Gresswell 1999). Factors related to a changing climate (e.g., earlier snowmelt, higher
summer temperatures) have been positively associated with increased wildfire activity in
recent decades (Westerling et al. 2006; Littell et al. 2009). In the Southwest, average
annual air temperatures have increased 0.29°C per decade with a reduction in summer
flows (2% per decade; Zeigler et al. 2012). No doubt aquatic communities throughout
the region will be at greater risk to wildfire. We demonstrated that a cool season
prescription burn through the riparian area within a Southwest montane grassland system
can be successfully conducted, when fire risk conditions are low, and with minimal
impact on the aquatic community. Within montane landscapes, prescription burns offer a
reliable tool to reduce the severity of fire effects while enhancing resiliency of aquatic
communities. Fire behavior is complex and because each fire will be different, we
encourage monitoring prior to and after prescribed burn to assess the effects that timing
(seasonality) and magnitude of fire have on aquatic communities relative to management
goals.

Acknowledgments

Financial support for this study was provided by a grant from the U.S. Department of Agriculture Forest Service, Rocky Mountain Experimental Research Station, Albuquerque, New Mexico, and the Valles Caldera National Preserve. Additional support was provided by New Mexico State University, Agriculture Experiment Station, Department of Fish, Wildlife and Conservation Ecology, and U.S. Geological Survey, New Mexico Cooperative Fish and Wildlife Research Unit. D. McGuire (McGuire Consulting, Española, New Mexico) was consulted on identification of Chironomidae (non–biting midges) and M. Myers (U.S. Fish and Wildlife Service,
Albuquerque, New Mexico) was consulted on identification of Trichoptera (caddisflies).

D. Klemm (U.S. Environmental Protection Agency, Cincinnati, Ohio) identified the Hirudinea (leeches). K. Montgomery and J. Gulbransen of the Valles Caldera National Preserve assisted with collection, some sample processing, and sample site verifications.

The project was conducted under New Mexico State University Animal Care and Use Committee procedures under Project 2007-014. Mention of trade names or commercial products does not constitute endorsement or recommendations for use by the U.S. Government.

References


Howell, P. J. 2006. Effects of wildfire and subsequent hydrologic events on fish
distribution and abundance in tributaries of North Fork John Day River. North

Intergovernmental Panel on Climate Change (IPCC). 2007. Climate Change 2007:
Synthesis Report. Contribution of Working Groups I, II and III to the Fourth
Assessment Report of the Intergovernmental Panel on Climate Change.

Jacobi, G. Z. 1978. An inexpensive circular sampler for collecting benthic

Litschert, S. E., T. C. Brown, and D. M. Theobald. 2012. Historic and future extent of
wildfires in the southern Rockies ecoregion, USA. Forest Ecology and
Management 269:124–133.

Littell, J. S., D. McKenzie, D. L. Peterson, and A. L. Westerling. 2009. Climate and
Applications 19:1003–1021.

burns and wildfires: coming to terms with pseudoreplication. Natural Areas

Publishing, Los Alamos, New Mexico.

in forests worldwide: Implications of terrestrial Redfield–type ratios. Ecology
85:2390–2401.


New Mexico Environment Department (NMED). 2006. Standards for interstate and intrastate surface waters. 20.6.4 NMAC.


Table 1. Mean daily water temperature (MDWT) (°C), and average dissolved oxygen (mg/L), conductivity (µS/cm), and pH for summer months (1 June – 31 August) for 2005 (pre–burn), 2006 and 2007 (post–burn) in the treatment (Rio San Antonio) and reference (Rito de los Indios) streams of the Valles Caldera National Preserve, New Mexico. Standard deviation in parentheses.

<table>
<thead>
<tr>
<th>Stream</th>
<th>Water Temperature (°C)</th>
<th>Dissolved Oxygen (mg/L)</th>
<th>Conductivity (µS/cm)</th>
<th>pH</th>
</tr>
</thead>
<tbody>
<tr>
<td>Treatment</td>
<td>15.1</td>
<td>15.1</td>
<td>15.0</td>
<td>7.2</td>
</tr>
<tr>
<td></td>
<td>(2.22)</td>
<td>(2.11)</td>
<td>(2.24)</td>
<td>(0.87)</td>
</tr>
<tr>
<td>Reference</td>
<td>13.1</td>
<td>12.7</td>
<td>12.2</td>
<td>6.9</td>
</tr>
<tr>
<td></td>
<td>(3.77)</td>
<td>(3.11)</td>
<td>(2.87)</td>
<td>(0.83)</td>
</tr>
</tbody>
</table>
Table 2. Average relative weight ($W_r$) (± standard error) among Age–2, Age–3, and Age–4 and Age–5 adult brown trout in the treatment (Rio San Antonio) and reference (Rito de los Indios) streams of the Valles Caldera National Preserve, New Mexico. Fish were collected prior to the prescribed burn (spring and fall 2005) and post–burn (spring and fall 2006 and 2007). “–“ indicates Age–4 and Age–5 fish were not observed during the sample collection.

<table>
<thead>
<tr>
<th>Year</th>
<th>Season</th>
<th>Age–2 Treatment</th>
<th>Age–2 Reference</th>
<th>Age–3 Treatment</th>
<th>Age–3 Reference</th>
<th>Age–4 and 5 Treatment</th>
<th>Age–4 and 5 Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>2005</td>
<td>Spring</td>
<td>128 (10.6)</td>
<td>89 (5.0)</td>
<td>108 (1.4)</td>
<td>84 (2.5)</td>
<td>78 (4.8)</td>
<td>71 (4.5)</td>
</tr>
<tr>
<td></td>
<td>Fall</td>
<td>130 (4.9)</td>
<td>91 (0.7)</td>
<td>105 (1.2)</td>
<td>85 (2.3)</td>
<td>87 (7.5)</td>
<td>76 (4.5)</td>
</tr>
<tr>
<td>2006</td>
<td>Spring</td>
<td>128 (3.8)</td>
<td>94 (3.6)</td>
<td>100 (3.7)</td>
<td>87 (3.7)</td>
<td>84 (8.0)</td>
<td>78 (3.0)</td>
</tr>
<tr>
<td></td>
<td>Fall</td>
<td>129 (2.9)</td>
<td>98 (0.02)</td>
<td>99 (1.8)</td>
<td>94 (0.1)</td>
<td>83 (2.4)</td>
<td>–</td>
</tr>
<tr>
<td>2007</td>
<td>Spring</td>
<td>122 (6.2)</td>
<td>96 (1.1)</td>
<td>99 (3.1)</td>
<td>80 (2.1)</td>
<td>90 (4.2)</td>
<td>70 (0.6)</td>
</tr>
<tr>
<td></td>
<td>Fall</td>
<td>136 (8.6)</td>
<td>96 (0.3)</td>
<td>109 (1.0)</td>
<td>95 (2.3)</td>
<td>98 (5.6)</td>
<td>88 (4.4)</td>
</tr>
</tbody>
</table>
FIGURE LEGENDS

Figure 1. Location of the Valles Caldera National Preserve in northcentral New Mexico, USA.

Figure 2. Stream and air temperature (°C) prior to (27–31 October), during (1–2 November), and following the prescribed burn (3–7 November) within the treatment stream (Rio San Antonio) of the Valles Caldera National Preserve, New Mexico, USA. Dashed vertical lines indicate prescribed burn.

Figure 3. Ratio of carbon:nitrogen (C:N) from sediment (0.25–0.5 mm) collected prior to the prescribed burn (spring 2005), immediately following the burn (late fall 2005) and after snow melt (spring 2006) within the Rio San Antonio (Treatment), Rito de los Indios (Reference) of the Valles Caldera National Preserve, New Mexico. Additional samples were collected twelve months post-burn (fall 2006), 18 months post-burn (spring 2007) and 24 months post-burn (fall 2007). Error bars represent standard error of the mean of three samples (n = 3) for each site after which one qualitative sample was collected from each site.

Figure 4. Total standing crop (No./m²) of benthic macroinvertebrates collected prior to the prescribed fire (spring 2005 and fall 2005), immediately following the fire (late fall 2005) and post-burn (spring 2006, fall 2006, spring 2007, fall 2007) within the Rio San Antonio (Treatment), Rito de los Indios (Reference) of the Valles Caldera National Preserve, New Mexico. Error bars represent standard error of the mean of three samples (n = 3) within habitats (Riffles and Pools).

Figure 5. Taxa richness of benthic macroinvertebrates collected prior to the prescribed fire (spring 2005 and fall 2005), immediately following the fire (late fall 2005) and post-
burn (spring 2006, fall 2006, spring 2007, fall 2007) within the treatment (Rio San Antonio) and reference (Rito de los Indios) streams of the Valles Caldera National Preserve, New Mexico. Error bars represent standard error of the mean of three samples ($n = 3$) within microhabitats (Riffles and Pools).

Figure 6. Ecological indices (%) of the benthic community in pool and riffle microhabitat immediately after the burn throughout the treatment (Rio San Antonio) and reference (Rito de los Indios) streams of the Valles Caldera National Preserve, New Mexico. The reference value of each ecological index was obtained prior to the burn in the spring and fall of 2005. Thus, each index in the fall was compared to its respective fall reference value and each index in the spring was compared to its respective spring reference value. For example, 100% reflects no change from the respective reference value obtained prior to the burn. EPT = Ephemeroptera + Plecoptera + Trichoptera.

Figure 7. Mean density (fish/100 m$^2$) and standard error of adult Brown Trout (top) and juvenile Brown Trout (bottom) prior to the prescribed burn (pre-burn) spring and fall 2005, post-burn spring and fall 2006 and 2007 in the treatment (Rio San Antonio) and reference (Rito de los Indios) streams of the Valles Caldera National Preserve, New Mexico.
Stream Temperature

DATE vs AIR

Prescribed fire
1-2 November 2005
Density (fish/100 m²)

Pre-burn 2005

Spring Fall Spring Fall Spring Fall

Post-burn 2007

Fire

Treatment Reference

Density (fish/100 m²)

Pre-burn 2005

Post-burn 2006

Post-burn 2007

Fire

Treatment Reference

Pre-burn 2005

Post-burn 2006

Post-burn 2007
Appendix. Taxa list of aquatic macroinvertebrates collected prior to the prescribed in 2005 through 2007 post-burn among riffles and pools of the treatment (Rio San Antonio) and reference (Rito de los Indios) streams on the Valles Caldera National Preserve, New Mexico.

<table>
<thead>
<tr>
<th>Taxon</th>
<th>Family, Genus, Species</th>
<th>Taxon</th>
<th>Family, Genus, Species</th>
</tr>
</thead>
<tbody>
<tr>
<td>Plecoptera - stoneflies</td>
<td><em>Amphinemura</em> sp.</td>
<td>Diptera – true flies</td>
<td></td>
</tr>
<tr>
<td></td>
<td><em>Pteronarcella</em> <em>badia</em></td>
<td><em>Chronomidae</em> – midges</td>
<td></td>
</tr>
<tr>
<td></td>
<td><em>Isoperla</em> sp.</td>
<td><em>Radotanypus</em> sp.</td>
<td></td>
</tr>
<tr>
<td></td>
<td><em>Swelsa</em> sp.</td>
<td><em>Procladius</em> sp.</td>
<td></td>
</tr>
<tr>
<td></td>
<td><em>Hesperoperla</em> <em>pacific</em></td>
<td><em>Thienemannimyia</em> sp.</td>
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<tr>
<td></td>
<td></td>
<td><em>Diamesa</em> sp.</td>
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<tr>
<td></td>
<td></td>
<td><em>Pagastia</em> sp.</td>
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<td></td>
<td></td>
<td><em>Pothastia</em> sp.</td>
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<td></td>
<td></td>
<td><em>Odontomesa</em> sp.</td>
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<td></td>
<td></td>
<td><em>Prodiamesa</em> sp.</td>
<td></td>
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<tr>
<td></td>
<td></td>
<td><em>Thienemanniella</em> sp.</td>
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<td></td>
<td></td>
<td><em>Brillia</em> sp.</td>
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<tr>
<td></td>
<td></td>
<td><em>Cricotopus</em> spp.</td>
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<td></td>
<td></td>
<td><em>C. nostococladius</em></td>
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<tr>
<td></td>
<td></td>
<td><em>Eukiefferiella</em> sp.</td>
<td></td>
</tr>
<tr>
<td>Ephemeroptera - mayflies</td>
<td><em>Baetis</em> <em>tricaudatus</em></td>
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<tr>
<td></td>
<td><em>Acentrella</em> <em>insignificans</em></td>
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<tr>
<td></td>
<td><em>Ecdyonurus</em> sp.</td>
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<tr>
<td></td>
<td><em>Cinygmula</em> sp.</td>
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<tr>
<td></td>
<td><em>Paraleptophlebia</em> sp.</td>
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<tr>
<td></td>
<td><em>Tricorythodes</em> sp.</td>
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<tr>
<td></td>
<td><em>Drunella</em> <em>grandis</em></td>
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<td></td>
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<td></td>
<td></td>
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<tr>
<td>Tricoptera - caddisflies</td>
<td><em>Rhyacophila</em> <em>brunea</em> cpx.</td>
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<tr>
<td></td>
<td><em>Glossosoma</em> sp.</td>
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<tr>
<td></td>
<td><em>Protoptila</em> sp.</td>
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<td></td>
<td><em>Hydropsyche</em> sp.</td>
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<tr>
<td></td>
<td><em>Cheumatopsyche</em> sp.</td>
<td></td>
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<tr>
<td></td>
<td><em>Arctopsyche</em> <em>grandis</em></td>
<td></td>
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</tr>
<tr>
<td></td>
<td><em>Hydroptila</em> sp.</td>
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<tr>
<td></td>
<td><em>Ithyrichia</em> sp.</td>
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<tr>
<td></td>
<td><em>Oxyethira</em> sp.</td>
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<tr>
<td></td>
<td><em>Limnephilus</em> sp.</td>
<td></td>
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<tr>
<td></td>
<td><em>Hesperophylax</em> sp.</td>
<td></td>
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<tr>
<td></td>
<td><em>Oecetis</em> sp.</td>
<td></td>
<td></td>
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<tr>
<td></td>
<td><em>Brachycentrus</em> sp.</td>
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<tr>
<td></td>
<td><em>Lepidostoma</em> sp.</td>
<td></td>
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<tr>
<td></td>
<td><em>Micrasema</em> sp.</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td><em>Helicopsyche</em> sp.</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td><em>Cernotina</em> sp.</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td><em>Oligophlebodes</em> sp.</td>
<td></td>
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## Appendix. (Continued)

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<th>Family, Genus, Species</th>
<th>Taxon</th>
<th>Family, Genus, Species</th>
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<tbody>
<tr>
<td>Other Diptera</td>
<td><em>Antocha</em> sp.</td>
<td>Odonata – damsel/dragonflies</td>
<td><em>Ophiogomphus</em> sp.</td>
</tr>
<tr>
<td></td>
<td><em>Dicranota</em> sp.</td>
<td></td>
<td><em>Argia</em> sp.</td>
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<tr>
<td></td>
<td><em>Hexatoma</em> sp.</td>
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<tr>
<td></td>
<td><em>Limonia</em> sp.</td>
<td>Hemiptera – bugs</td>
<td><em>Gerris</em> sp.</td>
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<td></td>
<td><em>Tipula</em> sp. 1</td>
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<td><em>Ambrysus</em> sp.</td>
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<td><em>Tipula</em> sp. 2</td>
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<td>Cicadellidae</td>
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<tr>
<td></td>
<td><em>Dixa</em> sp.</td>
<td>Lepidoptera – moths and butterflies</td>
<td><em>Petrophyla</em> sp.</td>
</tr>
<tr>
<td></td>
<td><em>Simulium</em> sp.</td>
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<tr>
<td>Ceratopogonidae</td>
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<td>Amphipoda - scuds</td>
<td><em>Hyalella</em> sp.</td>
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<tr>
<td>Tabanus sp.</td>
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</tr>
<tr>
<td></td>
<td><em>Chrysops</em> sp.</td>
<td>Arachnida – spiders and mites</td>
<td>Hydrachnidia</td>
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<tr>
<td></td>
<td><em>Chelifera</em> sp.</td>
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<tr>
<td></td>
<td><em>Oreogeton</em> sp.</td>
<td>Mollusca – clams and snails</td>
<td><em>Physella</em> sp.</td>
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<tr>
<td></td>
<td><em>Hemerodromia</em> sp.</td>
<td></td>
<td><em>Gyraulus</em> sp.</td>
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<tr>
<td>Ephyrididae</td>
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<td><em>Pisidium</em> sp.</td>
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<tr>
<td>Limnophora sp.</td>
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<tr>
<td>Ptychopteridae</td>
<td></td>
<td>Annelida – segmented worms</td>
<td>Helobdella stagnalis</td>
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<td>Motobdella sedonensis</td>
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<tr>
<td>Coleoptera – beetles</td>
<td>Dytiscidae</td>
<td></td>
<td>Tubificidae</td>
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<tr>
<td></td>
<td><em>Hydroporus</em> sp.</td>
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<td>Naididae</td>
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<td></td>
<td><em>Helichus</em> sp.</td>
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<td>Megadrilli</td>
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<td></td>
<td><em>Zaitzevia</em> sp.</td>
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<td><em>Dubiraphia</em> sp.</td>
<td>Platyhelminthes - flatworms</td>
<td>Tricladiida</td>
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<td><em>Optioservus</em> sp.</td>
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<td></td>
<td><em>Heterlimnus</em> sp.</td>
<td>Nematomorpha – Gordian worm</td>
<td><em>Gordius</em> sp.</td>
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<td></td>
<td><em>Microcylloepus</em> sp.</td>
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<td>Curculionidae</td>
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