

1 **Prescribed Fire Effects on an Aquatic Community of a Southwest Montane**

2 **Grassland System**

3

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30

31 Abstract

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

54 about the importance in timing (seasonality) of prescription burn and anticipated effects
55 on aquatic communities.

56

57 Forested ecosystems in western North America have evolved with fire and rely on
58 its influence to increase ecological productivity through mobilization of nutrients to
59 increase energy flow. The timing and severity of fire remain causes of concern. Heavy
60 fuel loads and low fuel moisture attributable to periods of drought or seasonal ebbs in
61 precipitation exacerbate fire conditions. The impacts of fire on aquatic systems are
62 complex and not entirely dependent on fire severity, but on the fundamental
63 characteristics of the watershed (Bêche et al. 2005), stream channel morphology (Benda
64 et al. 2003), riparian vegetation (Rieman et al. 2003; Pettit and Naiman 2007), and the
65 frequency of post-fire disturbances such as flooding (Vieira et al. 2004). Persistence of
66 aquatic communities is closely related to periodic disturbances which shape habitat
67 quantity (Rieman and McIntyre 1995), quality (Minshall et al. 2001), and heterogeneity
68 (Resh et al. 1988). Direct effects of wildfire include increases in stream temperature (Hitt
69 2003) and changes in aquatic chemistry which can approach lethal toxicity thresholds for
70 fishes (Spencer and Hauer 1991). Indirect include the loss of streamside vegetation,
71 resultant increases in stream temperature from increased solar radiation, and increased
72 sedimentation (Gresswell 1999).

73 Fire effects can vary in severity from acute (between onset of fire and initial
74 runoff post-fire) to chronic (extending for years), and influence both biotic communities
75 and abiotic components of aquatic systems (Minshall 2003). Acute effects result in
76 localized changes to the structure and composition of the riparian area that include

77 accumulation of downed woody debris, shifts in composition of aquatic
78 macroinvertebrates and fish densities due to mortality or emigration (Minshall et al.
79 1997; Hitt 2003; Burton 2005; Bêche et al. 2005; Howell 2006), decreases in stream
80 channel stability with variable discharge (Rieman and Clayton 1997; Dunham et al.
81 2007), elevated sediment transport (Benda et al. 2003), and elevated water temperatures
82 (Dunham et al. 2007). In sharp contrast, chronic effects from a severe burn due to
83 vegetative re-growth and in-stream woody debris are not often realized for decades.
84 Long term effects also will likely occur if stream channels undergo geomorphic
85 reorganization, if the variability in timing and amount of runoff exceeds normal annual
86 and inter-annual patterns that disrupt hydrologic patterns, and if riparian areas are unable
87 to stabilize soil erosion and ameliorate heat insolation (Dunham et al. 2003).

88 Land managers have employed controlled burning or prescription fire to
89 proactively manage potential hazardous fuel loading while maintaining ecological
90 integrity of forested and grassland systems. If carefully planned, prescriptions for
91 wildland fire should protect against erosion and sedimentation to aquatic systems and
92 have little to no effect on water quality (USEPA 2005). Fires that burn down through the
93 litter to the mineral soil will adversely affect water quality through erosion. The amount
94 of erosion will depend on fire severity, amount of ground cover remaining on the soil,
95 grade of slope (the steeper, the more likely soils will move), erodibility of the soil,
96 removal of riparian vegetation, time and amount of rainfall post-burn, and how rapidly
97 the site re-vegetates. While the effects of prescription fire on water quality has received
98 some attention (Richter et al. 1982; Stephens et al. 2004; Elliott and Vose 2005; Bêche et
99 al. 2005), less attention has been paid to short- and long-term effects on fish and

100 macroinvertebrate communities. Bêche et al. (2005) related low to moderate burn
101 severity from a prescription burn to only minor effects in the benthic macroinvertebrate
102 community despite altered periphyton levels. Population responses of fishes to
103 prescription burn have been less documented and would presumably be related to
104 severity, size, and patchiness of the burn. Arkle and Pilloid (2010) found no immediate
105 or delayed effects to fish populations in Idaho streams within watersheds subjected to
106 prescribed burn of low to moderate severity. While the authors did not observe
107 deleterious effects, this is not sufficient evidence to assume prescribed burns, if
108 conducted according to protocol, would not affect fish populations and the invertebrate
109 community they rely upon.

110 The purpose of our research was to assess impacts of a prescribed burn during the
111 cool, autumn season on an aquatic community within a grassland system on the Valles
112 Caldera National Preserve, New Mexico. An important management goal of the Valles
113 Caldera National Preserve, hereafter referred to as the Preserve, is to utilize prescription
114 burn throughout its grasslands and Ponderosa pine forests as a reliable management tool
115 to return pre-settlement fire regimes, and reduce fuel loading to prevent uncharacteristic
116 wildfires. Extensive logging operations and wildfire suppression during the 20th Century
117 throughout the Preserve resulted in increased secondary succession and encroachment of
118 shrubs and trees on grasslands that provide recreational fishing opportunities. To
119 evaluate the ecological impacts of managed low-intensity fire on the Preserve, we
120 designed and implemented an experimental prescribed burn in one of the Preserve's
121 upland watersheds (Valle Toledo). A prescribed burn treatment consistent with protocols
122 for a managed burn in the fall season (November) was applied to low gradient open

123 meadow grassland and adjacent Ponderosa pine forest, except that the fire was allowed to
124 burn through the riparian area to the stream bank to replicate natural fire behavior. Our
125 study objectives were to characterize immediate (within a week), seasonal and longer-
126 term (two years) effects of this prescription burn on water quality, benthic
127 macroinvertebrate communities, and fish populations. Thus, our null hypothesis was that
128 no detectable effects of selected variables will be observed between burned and unburned
129 treatments.

130 <A>Methods

131 *Study Site Description.*– Valles Caldera National Preserve encompasses
132 approximately 36,000 ha of the Jemez Mountains in northcentral New Mexico (Sandoval
133 County). The Preserve is a caldera of a 1.25 million year-old super-volcano,
134 characterized by multiple volcanic domes and intervening grassland *valles* (Spanish term
135 for treeless valleys). Under private ownership from 1860 through 2000, the area was
136 managed for livestock (sheep and cattle) grazing, timber harvest, hunting and fishing,
137 mining, and geothermal energy exploration (Martin 2003). After federal acquisition in
138 2000, the Preserve was mandated to maintain a multiple land-use policy balancing
139 sustainable harvest of natural resources (i.e., grazing for livestock, timber production,
140 hunting and fishing) with recreation and ecosystem health. Approximately 9,000 ha of
141 the Preserve are highly productive grassland valles, surrounded by upland montane
142 coniferous forests. Within the watershed are a series of first- and second-order streams
143 with riparian areas composed of sedges *Carex* spp., bunchgrasses *Danthonia*, *Festuca*,
144 *Phleum* spp., sod-forming grasses *Poa* spp., and forbs. The dominant soils in the valles

145 are Cosey-Jarmillo association, which are deep, well drained, and moderately permeable
146 (USDA 2008). The remaining areas throughout the Preserve and adjacent federal lands
147 consist of montane forests, dominated by ponderosa pine *Pinus ponderosa*, white fir
148 *Abies concolor*, Engelmann spruce *Picea engelmannii*, Douglas-fir *Pseudotsuga*
149 *menziesii*, Colorado blue spruce *Picea pungens*, aspen *Populus tremuloides*, and juniper
150 *Juniperus* spp. Average pre-1900 fire return intervals in forested areas throughout the
151 Preserve varied from 5 to 20 years, while grassland fires in the open meadows of the
152 valleys occurred on average every 1.6 years (Allen 2002; Touchan et al. 1996; Allen et al.
153 2008; Dewar 2011).

154 *Prescribed fire treatment.*– The prescribed burn was undertaken in Valle Toledo
155 (35°57'50" N, 106°29'31" W, elevation 2,607 m) 1–2 November 2005. A late-season
156 burn was selected because of higher fuel moisture reducing chances of a burn escaping
157 the targeted area. Valle Toledo is a 1,040 ha grassland containing a second order stream
158 (Rio San Antonio) that runs east to west and is bordered on the western edge by a first
159 order stream (Rito de los Indios; Figure 1). A graded dirt road surrounds the burn area,
160 and was used as a firebreak during the burn. Interior areas of the burn were lit by aerial
161 ignition devices from a helicopter. The burn covered 728 ha or approximately 70% of the
162 target area. Areas that did not burn either had insufficient fuel loads or were too moist
163 and replete with green vegetation. The majority of riparian areas within the study reaches
164 of the stream were burned to the stream edge resulting in successful attainment of the
165 targeted goal which was to replicate natural fire behavior. Grassland fuel loads (dry
166 mass) averaged 1,740 kg/ha. Flame lengths in the grassland ranged from 0.3 m to 3.0 m
167 high, with rate of spread averaging 0.4 m/s. In the forest, the fire burned less intensively;

168 sub-canopy herbaceous fuel loads averaged 1,205 kg/ha, with flame lengths between 0.1
169 and 0.3 m and spread-rate of 0.04 m/s. Few trees were killed, although needle and branch
170 loss in the lower canopy were extensive. In the forest, standing grasses and forbs were
171 burned and consumption of the pine needle litter layer was limited to the uppermost 2–3
172 cm.

173

174 *Study stream.*– Rio San Antonio is a second order stream draining approximately 120
175 km² flowing east to west across the Preserve’s northern portion, with headwaters in the
176 Sierra de los Valles near the Preserve’s eastern boundary. The portion of the stream
177 flowing through the Valle Toledo is a C-4 type stream (i.e., slightly entrenched, gravel
178 dominated, with a well-developed flood plain) according to the Rosgen (1994) scale.
179 Rito de los Indios is a spring-fed first order stream with its headwaters in the Sierra de los
180 Valles on the Preserve’s northern boundary that drains approximately 19 km². The
181 portion of the Rito de los Indios that flows through Valle Toledo is characterized as an E-
182 4 type stream (i.e., slightly entrenched, gravel and cobble dominated, well developed
183 flood plain; Rosgen 1994). Although Rito de los Indios exhibited lower width to depth
184 ratios, the stream was selected to represent reference conditions due to its close proximity
185 to the burn area and comparable physical features such as entrenchment, substrata, and
186 area of flood plain. Rito de los Indios borders the western edge of the Valle Toledo and
187 eventually joins Rio San Antonio downstream of the Valle Toledo burn. Hereafter, the
188 Rio San Antonio and Rito de los Indios will be referred to as the treatment and reference
189 streams, respectively.

190

191 *Water quality and C:N ratios in sediment.*— Water quality was assessed by placing
192 two data sondes (Model 6920, Yellow Springs Instruments, Yellow Spring, OH) within
193 the reference and treatment streams upstream from the confluence (Figure 1). Water
194 temperature (°C), dissolved oxygen (mg/L), pH, and conductivity (µS/cm) were
195 monitored at 15 min intervals from June to November 2005 and from April to November
196 2006 and 2007. Air temperature (°C) was obtained from an on-site Remote Automated
197 Weather Station within the Valle Toledo. Monthly cumulative precipitation (mm)
198 throughout the study (June 2005–September 2007) and over 31 years (1981–2011) was
199 obtained from the Natural Resources Conservation Service’s Quemazon snow telemetry
200 station (SNOTEL) located near the eastern boundary of the Rio San Antonio watershed
201 (Site 708; 35°55’ N, 106°24’ W, Elevation 3,167 m, Los Alamos County, New Mexico).

202 In each of the treatment and reference streams, three study reaches were
203 arbitrarily selected to collect sediments ($n = 3$) pre-burn (fall 2005), immediately post-
204 burn (late-fall 2005), following snow-melt but before monsoonal summer rains (six
205 months post-burn, spring 2006), and one year post-burn (fall 2006). To assess the
206 contribution of carbon from ash, sediments were collected to assess carbon and nitrogen
207 ratios (C:N) using a Hess–style plastic sediment sampler (500 mm [height] by 200 mm
208 [dia]). Sediments were processed through a series of U.S. Standard stainless steel sieves
209 that ranged from 0.25 mm to 16 mm and then oven-dried (60°C). From the smallest sized
210 fractions (0.25–0.5 mm), a sub–sample was weighed to the nearest 0.001 g and analyzed
211 for carbon and nitrogen by high temperature combustion on a ThermoQuest Elemental
212 Analyzer (Model NC2100, CE Instruments, Wigan, UK). Percentages of carbon and
213 nitrogen gases were eluted on a gas chromatography column, detected by thermal

214 conductivity and integrated to yield carbon and nitrogen content (Pella 1990a, 1990b).
215 Carbon and nitrogen were confirmed by mass percent loss on ignition by combustion at
216 500°C for 2 h. Pre- and post-combustion weights of each sample were determined as
217 mass percent loss of organic matter (Ben-Dor and Banin 1988; Nelson and Sommers
218 1996).

219

220 **Macroinvertebrates.**– Three study reaches in each stream (treatment and reference)
221 were selected to collect benthic macroinvertebrates, pre-burn (spring and fall 2005),
222 immediately after the burn (late fall 2005), and four times post-burn (spring and fall
223 2006, and spring and fall 2007; Figure 1). These study reaches spanned the area to be
224 burned in the Valle Toledo, were adjacent to the burned area in the reference stream (Rito
225 de los Indios), and were 5 to 10 m upstream to where sediments were collected. Two
226 habitats were sampled each season for benthic macroinvertebrates: Riffles of comparable
227 substrates (i.e., pebble and cobble ranging from 50 to 150 mm); and pools (< 0.5 m
228 depth). Macroinvertebrates were sampled within a few weeks of the pre-burn sample
229 date to minimize seasonal variation and to avoid peak flows. Within each reach, benthic
230 aquatic macroinvertebrates were collected from 3 pools and 3 riffles using a modified
231 Hess–type circular sampler (0.06 m²; Jacobi 1978). This was repeated across all three
232 reaches within treatment and reference streams. Collections at all locations were usually
233 completed within two consecutive days to minimize spurious effects of weather.
234 Samples were preserved in 99% ethanol in the field and later identified to lowest
235 taxonomic level possible.

236 Ecological metrics of Plafkin et al. (1989), Barbour et al. (1992, 1999) were used
237 to assess effects of the burn on community structure, community balance, and functional
238 feeding components of the aquatic macroinvertebrate community compared to pre-burn
239 conditions through time and across habitat (pools and riffles). The Hilsenhoff Biotic
240 Index (Hilsenhoff 1987) assessed sensitivity of the aquatic macroinvertebrate community
241 to sediment and nutrient perturbation. Additional indices included mayflies
242 (Ephemeroptera), stoneflies (Plecoptera), and caddisflies (Tricoptera), collectively
243 referred to as the EPT index. Additionally, when chironomids (i.e., tolerant organisms)
244 are included with EPT assemblage (EPT/EPT + Chironomidae), an even distribution
245 among all four taxonomic groups indicates optimal biotic conditions; whereas, a shift to a
246 lower ratio indicates a disproportionately greater number of tolerant chironomids.

247

248 **Fish surveys.**— Preliminary surveys in both treatment and reference streams revealed
249 a fish community dominated by Brown Trout *Salmo trutta*. Longnose Dace *Rhinichthys*
250 *cataractae* were incidental throughout the study and not considered in the analysis.

251 Three transects (50 m each) were established in close proximity to the macroinvertebrate
252 sample sites within treatment and reference streams to obtain a representative selection of
253 runs, riffles, and pools (Figure 1). Baseline data for the Brown Trout population within
254 both streams was collected spring and fall 2005 to assess adult (≥ 170 mm) and juvenile
255 (< 170 mm) densities (#/100 m of stream) and relative weight (W_r) prior to the prescribed
256 burn. Treatment and reference streams were surveyed spring and fall of 2006 and 2007
257 to evaluate delayed effects from the prescribed burn as well as annual recruitment to the
258 Brown Trout population. Within each sample reach, Brown Trout were sampled using

259 direct current in a three-pass depletion with a backpack electrofishing unit (Model LR–
260 24, Smith–Root Inc., Vancouver, WA). Block nets (6 mm mesh) were set above and
261 below the sample reach to prevent emigration and immigration of fish during
262 electrofishing. Upon capture, total length (TL; mm) and weight (g) were recorded for all
263 captured fish. These fish were subsequently placed in a live box within the stream until
264 completion of the final electrofishing pass and then returned to the sample reach.

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

277

278 **Data analyses.**– The purpose of the study was to assess impacts of a prescription
279 burn on the aquatic community within a grassland system with the null hypothesis of no
280 detectable effects of selected variables between burned and unburned treatments. As this
281 study was based on only one prescription burn in one watershed, the “treatment” was not

282 replicated; however, fire studies are often opportunistic and lack replication (Mantgem et
283 al. 2001). Our replications were spatial units within burned and reference areas.
284 Therefore, we restricted our conclusions to differences between pre- and post-burn
285 collections at our treatment (burned) and reference (unburned) sites.

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

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

301 Summary statistics were calculated for the treatment and reference areas using
302 raw abundance for each habitat (pool, riffle) across seasons (spring, fall) to obtain total
303 standing crop (total number of organisms/m²) and taxa richness (total number of detected
304 taxa). Differences in total standing crop and species richness of benthic

305 macroinvertebrates were assessed using repeated measures ANOVA with a 2 x 2 factorial
306 treatment structure using sites (treatment and reference) by habitat (pools and riffles). To
307 identify differences in total standing crop and species richness between treatment and
308 reference areas, treatment*time interactions were examined for each habitat using pre-
309 planned contrasts to identify an immediate burn effect (pre-burn fall 2005 versus late-fall
310 2005, and longer term effects by comparing pre-burn spring 2005 versus post-burn spring
311 2006 and 2007, and pre-burn fall 2005 versus post-burn fall 2006 and 2007).

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

321

322 <A>Results

323 *Water quality and C:N ratios in sediment.*— Stream and air temperature profiles prior
324 to the prescribed burn (27–31 October) revealed consistent diel patterns (Figure 2). Prior
325 to the burn, the diel range in air temperature was 21°C (–8.0–13.0°C) while the diel
326 range in stream temperature was 6°C (7.4–13.4°C). During the burn (1–2 November), the
327 diel range in air temperature increased to 26°C (–8.0–18°C) in contrast to the diel range

328 for stream temperature which remained 6°C (7.1–13.1°C). We noted an immediate
329 change following the burn (3–5 November) in which minimum air temperature increased
330 to 3.2°C (Figure 2). Despite the rise in air temperature, stream temperatures were within
331 the diel range prior to the burn. Between pre- and immediate post-burn, daily maximum
332 temperatures did not exceed criteria for impaired cool water streams in New Mexico
333 (20°C; New Mexico Environment Department, NMED 2006). Water temperature,
334 dissolved oxygen concentrations, conductivity, and pH varied little prior to, during, and
335 immediately after the burn within the treatment stream and were deemed acceptable when
336 compared to water quality criteria of coldwater fishes (NMED 2006). Mean daily water
337 temperature (MDWT) before the burn in 2005 was 15.1°C and did not vary after the burn
338 in 2006 (15.1°C) or 2007 (15.0°C; Table 1). The highest MDWT recorded in one day
339 was prior to the burn in 2005 (17.9°C) compared to subsequent years (16.4°C in 2006 and
340 2007). Conductivity and pH in the treatment stream throughout summer months (1 June–
341 31 August) did not vary pre- and post-burn (Table 1). Monthly precipitation was below
342 the long-term average during spring 2006 (post-burn) which was reflected by severe
343 drought (USDN 2011). Precipitation patterns deemed “normal” returned the summer of
344 2006.

345 An increase in C:N was observed over time ($F_{2,18} = 5.76$; $P = 0.012$) for both
346 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).
347 Average C:N varied between pre-burn spring (11.5 ± 0.42 ; mean \pm standard error) and
348 after snowmelt post-burn spring 2006 (13.2 ± 0.39) similarly to the reference site (pre-
349 burn spring 13.4 ± 0.47 versus post-burn spring 16.2 ± 0.37) indicating contribution of a
350

351 source unrelated to the fire from within the watershed. Pre-planned contrasts of C:N
352 changes between the two time frames (pre-burn spring 2005 and post-burn spring 2006)
353 revealed no detectable difference ($T_{18} = -0.78$; $P = 0.448$) between treatment and
354 reference sites, further indicating no fire effect.

355

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

367 The top ranked model allowing for unequal variances over time for total standing
368 crop in fall collections revealed no detectable treatment by time interaction prior to the
369 burn (2005) and immediately following the burn (2005; $F_{3,11} = 2.01$; $P = 0.171$). Thus
370 no immediate burn effects on total standing crop of benthic macroinvertebrates in either
371 pools or riffles were observed between treatment and reference sites (Figure 4). This
372 conclusion was reaffirmed in the pre-planned contrasts that indicated little change for
373 total standing crop in pools ($T_{11} = 0.31$; $P = 0.760$) and riffles ($T_{11} = -1.95$; $P = 0.078$).

374 When we compared total standing crop in the fall prior to the burn (2005) with fall
375 collections one year (2006) and two years (2007) post-burn, the top ranked model
376 allowing for unequal variances over time also revealed no differences in the treatment by
377 time interaction ($F_{6,14.8} = 1.73$; $P = 0.183$). Pre-planned contrasts of changes in total
378 standing crop revealed no detectable differences in pools ($T_{11} = -0.20$; $P = 0.843$) or
379 riffles ($T_{11} = -2.0$; $P = 0.066$). In spring collections, the top ranked model allowing for
380 unequal variances among treatments revealed no treatment*time interaction prior to the
381 burn (2005), one year (2006), and two years (2007) post-burn ($F_{6,12.3} = 2.10$; $P = 0.127$).
382 Pre-planned contrasts of changes in total standing crop revealed no detectable differences
383 in either habitat ($P > 0.30$).

384 Changes in species richness in the treatment stream prior to the burn (2005) and
385 immediately following the burn (2005) were not affected by fire as indicated by no
386 detectable treatment* time interaction ($F_{3,16} = 0.96$; $P = 0.436$). There were no
387 immediate effects of the burn on species richness in either pools or riffles between
388 treatment and reference sites (Figure 5). Pre-planned contrasts of changes in species
389 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
390 (2005) with fall collections one year (2006) and two years (2007) post-burn, the top
391 ranked model revealed no treatment*time interaction ($F_{6,24} = 2.10$; $P = 0.090$). The pre-
392 planned contrasts of changes in species richness throughout the fall collections revealed
393 no detectable differences in pools ($T_{24} = -0.17$; $P = 0.869$), but differences were
394 observed in riffles ($T_{24} = -3.34$; $P = 0.003$). These differences were due to a decrease in
395 species richness in riffles throughout the treatment site while species richness increased
396

397 in riffles throughout the reference site (Figure 5). The top ranked model for species
398 richness revealed no treatment*time interaction prior to the burn in spring (2005), and in
399 the following springs post-burn ($F_{6,24} = 1.03$; $P = 0.428$). Pre-planned contrasts of
400 changes in species richness revealed no detectable differences in pools ($T_{24} = -1.55$; $P =$
401 0.135) or riffles ($T_{24} = -1.08$; $P = 0.289$).

402 Small sample size ($n = 3$), considerable variation within pools and riffles, and
403 seasonal response of taxa obscured detectable differences between pre- and post-burn
404 sample collections. However, changes in sensitive and tolerant taxa across time and
405 microhabitat in the treatment stream are worth noting. Departure of the EPT index in
406 both riffles and pools immediately post-burn were due to a decline in mayfly
407 *Tricorythodes* sp. and the predacious stonefly *Isoperla* sp. (Figure 6). *Tricorythodes* sp.
408 decreased from $128.0/m^2 (\pm 26.82)$ prior to the burn to $74.0/m^2 (\pm 18.66)$ by the first
409 spring. By the second spring, mayfly had rebounded throughout the pools ($219.0/m^2 \pm$
410 72.20). *Isoperla* sp. decreased in riffles from $81.3/m^2 (\pm 13.58)$ prior to the burn to
411 $3.0/m^2 (\pm 3.00)$ spring 2006 and began to return to pre-burn levels by spring 2007
412 ($49.0/m^2 \pm 13.58$).

413 Departure from the EPT/EPT + Chironomidae index in pools (Figure 6) occurred
414 immediately post-burn as a result of an increase in the tolerant chironomid *Microtendipes*
415 sp. throughout the treatment sites. Prior to the burn, *Microtendipes* sp. increased
416 throughout the treatment pools from $4.3/m^2 (\pm 1.33)$ to $56.0/m^2 (\pm 18.21)$ immediately
417 following the burn returning to pre-burn levels ($2.0/m^2 \pm 1.15$) by fall of 2007. Two
418 years post-burn, departure from pre-burn conditions throughout pools in the treatment
419 stream as reflected by the suite of ecological indices was negligible.

420

421 *Fish*.– Regardless of season, we did not detect an effect of the burn treatment on
422 adult salmonid density (Figure 7). Changes in population densities of adult Brown Trout
423 in the fall prior to the burn and one year post-burn (fall 2006) in treatment and reference
424 streams did not differ as revealed by no detectable treatment*time interaction ($F_{2,12} =$
425 0.15 ; $P = 0.861$). Changes in population densities between pre-burn spring (2005) and
426 post-burn spring (2006) in treatment and reference streams did not differ ($F_{2,5,26} = 1.74$;
427 $P = 0.262$) according to the top ranked model that allowed for heterogeneous variances
428 over time. Densities of adult Brown Trout decreased from $79.6 \text{ fish}/100 \text{ m}^2 (\pm 14.48)$
429 prior to the burn (fall 2005) to $56.3 \text{ fish}/100 \text{ m}^2 (\pm 1.03)$ one-year post-burn (fall 2006).
430 A similar response was also observed during the same time frame in the reference sites
431 where densities decreased from $105.0 \text{ fish}/100 \text{ m}^2 (\pm 29.3)$ to $67 \text{ fish}/100 \text{ m}^2 (\pm 15.5$;
432 Figure 7). The top ranked model allowing for unequal variances over time for
433 recruitment revealed no detectable treatment*time interaction ($F_{2,7,2} = 0.05$; $P = 0.952$).
434 Recruitment in juvenile fish was greater in the treatment stream (range $20.3 - 20.7$
435 $\text{fish}/100 \text{ m}^2$) compared to the reference stream ($3.3 - 6.7 \text{ fish}/100 \text{ m}^2$) throughout the
436 study (Figure 7).

437 The top ranked model allowing for unequal variances by treatment for W_r in age 2
438 fish revealed no detectable treatment*time interaction for fall ($F_{2,7,93} = 0.51$; $P = 0.616$).
439 Spring collections did not demonstrate any treatment* time interaction either ($F_{2,12} =$
440 0.61 ; $P = 0.558$), but the top ranked model assumed equal variances over time and
441 treatments. There was, however, a detectable difference in W_r of age 3 fish as seen in the
442 treatment*time interaction for fall collections ($F_{2,12} = 9.84$; $P = 0.003$; Table 2). Pre-

443 planned contrasts revealed detectable differences ($T_{12} = -4.41$; $P = 0.001$) in changes of
444 W_r between treatment and reference sites for age 3 Brown Trout between pre-burn (2005)
445 and post-burn (2006). Age 3 Brown Trout in the treatment site exhibited a decrease in W_r
446 while the same age group experienced an increase in W_r at the reference site (Table 2).
447 Differences in W_r changes for age 3 fish between the two streams were no longer evident
448 by 2007 ($T_{12} = -1.79$; $P = 0.098$). Although spring collections exhibited a similar pattern
449 of decreasing condition at the treatment site and increasing fish condition at the reference
450 site, these differences were not statistically different ($F_{2,12} = 1.71$, $P = 0.222$; Table 2).
451 Relative weight was greater in adult Brown Trout (ages 4 and 5) in the treatment stream
452 (range 78 – 98 g) compared to the reference stream (range 70 – 88 g) throughout the
453 study.

454

455 <A>Discussion

456 As global temperatures increase over the next century (IPCC 2007), projected
457 aridity throughout the Southwest will be intensified by La Niña events beyond recent
458 human record (Seager et al. 2007). Grasslands as well as low- to mid-elevation forests
459 comprise the largest land cover in the southwestern U.S. and are prone to frequent fires
460 (Litschert et al. 2012). Severity and wildfire intervals throughout these systems are
461 expected to increase due to precipitation patterns that increase fuel loads during wet El
462 Niño followed by drier La Niña cycles. As such, prescribed burns by land managers
463 during cool seasons should become more commonplace as a safe, effective approach to
464 returning fire to the landscape. Despite extensive burn throughout the riparian corridor,
465 the cool-season prescription burn exerted negligible effects on the aquatic community.

466 Detrimental effects of fire often occur when a burn is followed by precipitation
467 events resulting in increased stream flow and elevated sediment transport (Benda et al.
468 2003; Wondzell and King 2003). While the timing of the burn in late-fall was planned to
469 maximize soil moisture, the Southwest was also experiencing drought conditions which
470 minimized chances for heavy rainfall and effects that an inflow of ash would have had on
471 the aquatic community. During the first eight months of the study (November 2005 –
472 June 2006), the Preserve experienced lower than normal precipitation, reducing spring
473 snowmelt runoff and the contribution of ash from surrounding grasslands and riparian
474 area. By the time precipitation returned to the Preserve with the onset of summer
475 monsoon rains July 2006, re-growth of vegetation was returning to the grasslands and
476 riparian areas, minimizing erosion processes and ash input. Within a southwestern
477 grassland, Gori and Backer (2005) demonstrated prescription fire increased ground cover
478 of perennial grasses throughout burned plots compared to un-burned plots thereby
479 increasing the watershed's capacity to capture runoff and reduce soil erosion. Others
480 have attributed the importance of riparian vegetation as a sediment trap on post-fire
481 sediment deposition to pools (Spina and Tormey 2000).

482 Fire mobilizes organic matter increasing the C:N signature within the top soil
483 layer (Ojima et al. 1994; Monleon et al. 1997; Rhoades et al. 2004) where precipitation
484 and snowmelt eventually carry these to stream sediments. The hydrology throughout the
485 Preserve is snowmelt-dominated and the grasslands had not experienced an extensive
486 burn since Federal acquisition in 2000. Thus, we anticipated an increase in C:N within
487 the treatment stream (Rio San Antonio) following the first spring snowmelt. However,
488 C:N ratios varied little, indicating that fire effects and subsequent ash input following the

489 first snowmelt were negligible. The low to moderate burn severity, combined with lower
490 than average precipitation within the first year of the burn, resulted in reduced ash and
491 particle deposition to Rio San Antonio. Of note, C:N ratios were slightly higher (15–
492 20%) throughout Rito de los Indios compared to Rio San Antonio, reflecting the
493 contribution of organic matter from the close proximity of the forested watershed.
494 Conifers are the dominant vegetation within the watershed of the Rito de los Indios and
495 may have contributed to a higher seasonal pulse of particulate C relative to N (McGroddy
496 et al. 2004; Starry et al. 2005). While this may have affected the C:N ratio of the
497 reference stream, C:N ratio in the treatment stream varied only slightly throughout the
498 study indicating negligible ash and particle deposition from the burn.

499 Aquatic macroinvertebrate communities have been widely accepted as useful
500 indicators of fire effects to lotic systems (Minshall et al. 1997; Barbour et al. 1999).
501 These communities are relatively sensitive and tend to respond rapidly to changes in
502 temperature, dissolved oxygen, and shifts of instream substrate (Vannote and Sweeney
503 1980; Ward and Stanford 1982) resulting in insect drift (Earl and Blinn 2003).
504 Taxonomic richness and community composition were related to hydrological
505 disturbances that affected habitat stability post-fire (Vieira et al. 2004) and varied
506 according to the time between disturbances rather than the total number of disturbances
507 (Death and Winterbourn 1995). While the prescribed burn had no immediate or longer-
508 term effects on the benthic macroinvertebrate community, we noted shifts in ecological
509 indices reflected in the reduction of a sensitive stonefly *Isoperla* sp. in riffle habitat,
510 reduction of a mayfly *Tricorythodes* sp. in pool habitats, and an increase in moderately
511 tolerant chironomid midges in both habitats. In circumstances involving high-intensity

512 forest fires in northcentral New Mexico, Pippin and Pippin (1981) and Vieira (2003)
513 reported that while densities of benthic macroinvertebrates in wildfire-affected streams
514 recovered within one year, taxa richness was less resilient with the loss of several species
515 of less tolerant stoneflies for up to three years post-fire.

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

532 Fire has long been recognized as one of the most influential of natural
533 disturbances affecting aquatic systems in the western United States (see review by
534 Gresswell 1999). Factors related to a changing climate (e.g., earlier snowmelt, higher

535 summer temperatures) have been positively associated with increased wildfire activity in
536 recent decades (Westerling et al. 2006; Littell et al. 2009). In the Southwest, average
537 annual air temperatures have increased 0.29°C per decade with a reduction in summer
538 flows (2% per decade; Zeigler et al. 2012). No doubt aquatic communities throughout
539 the region will be at greater risk to wildfire. We demonstrated that a cool season
540 prescription burn through the riparian area within a Southwest montane grassland system
541 can be successfully conducted, when fire risk conditions are low, and with minimal
542 impact on the aquatic community. Within montane landscapes, prescription burns offer a
543 reliable tool to reduce the severity of fire effects while enhancing resiliency of aquatic
544 communities. Fire behavior is complex and because each fire will be different, we
545 encourage monitoring prior to and after prescribed burn to assess the effects that timing
546 (seasonality) and magnitude of fire have on aquatic communities relative to management
547 goals.

548

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556 McGuire (McGuire Consulting, Española, New Mexico) was consulted on identification
557 of Chironomidae (non-biting midges) and M. Myers (U.S. Fish and Wildlife Service,

558 Albuquerque, New Mexico) was consulted on identification of Trichoptera (caddisflies).
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564 products does not constitute endorsement or recommendations for use by the U.S.
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758 Table 1. Mean daily water temperature (MDWT) (°C), and average dissolved oxygen (mg/L), conductivity (µS/cm), and pH for
 759 summer months (1 June – 31 August) for 2005 (pre–burn), 2006 and 2007 (post–burn) in the treatment (Rio San Antonio) and
 760 reference (Rito de los Indios) streams of the Valles Caldera National Preserve, New Mexico. Standard deviation in parentheses.
 761

Stream	Water Temperature (°C)			Dissolved Oxygen (mg/L)			Conductivity (µS/cm)			pH		
	2005	2006	2007	2005	2006	2007	2005	2006	2007	2005	2006	2007
Treatment	15.1 (2.22)	15.1 (2.11)	15.0 (2.24)	7.2 (0.87)	6.7 (2.82)	7.6 (1.12)	80 (1.5)	80 (3.58)	80 (1.39)	8.1 (0.54)	8.1 (0.60)	8.1 (0.63)
Reference	13.1 (3.77)	12.7 (3.11)	12.2 (2.87)	6.9 (0.83)	8.3 (0.98)	7.4 (0.70)	70 (3.9)	76 (0.72)	72 (3.5)	7.6 (0.26)	7.4 (0.20)	7.4 (0.12)

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765 Table 2. Average relative weight (W_r) (\pm standard error) among Age-2, Age-3, and Age-4 and Age-5 adult brown trout in the
 766 treatment (Rio San Antonio) and reference (Rito de los Indios) streams of the Valles Caldera National Preserve, New Mexico. Fish
 767 were collected prior to the prescribed burn (spring and fall 2005) and post-burn (spring and fall 2006 and 2007). “–” indicates Age-4
 768 and Age-5 fish were not observed during the sample collection.
 769

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

770

771 FIGURE LEGENDS

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

774 Figure 2. Stream and air temperature ($^{\circ}\text{C}$) prior to (27–31 October), during (1–2
775 November), and following the prescribed burn (3–7 November) within the treatment
776 stream (Rio San Antonio) of the Valles Caldera National Preserve, New Mexico, USA.
777 Dashed vertical lines indicate prescribed burn.

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

786 Figure 4. Total standing crop (No./m^2) of benthic macroinvertebrates collected prior to
787 the prescribed fire (spring 2005 and fall 2005), immediately following the fire (late fall
788 2005) and post-burn (spring 2006, fall 2006, spring 2007, fall 2007) within the Rio San
789 Antonio (Treatment), Rito de los Indios (Reference) of the Valles Caldera National
790 Preserve, New Mexico. Error bars represent standard error of the mean of three samples
791 ($n = 3$) within habitats (Riffles and Pools).

792 Figure 5. Taxa richness of benthic macroinvertebrates collected prior to the prescribed
793 fire (spring 2005 and fall 2005), immediately following the fire (late fall 2005) and post-

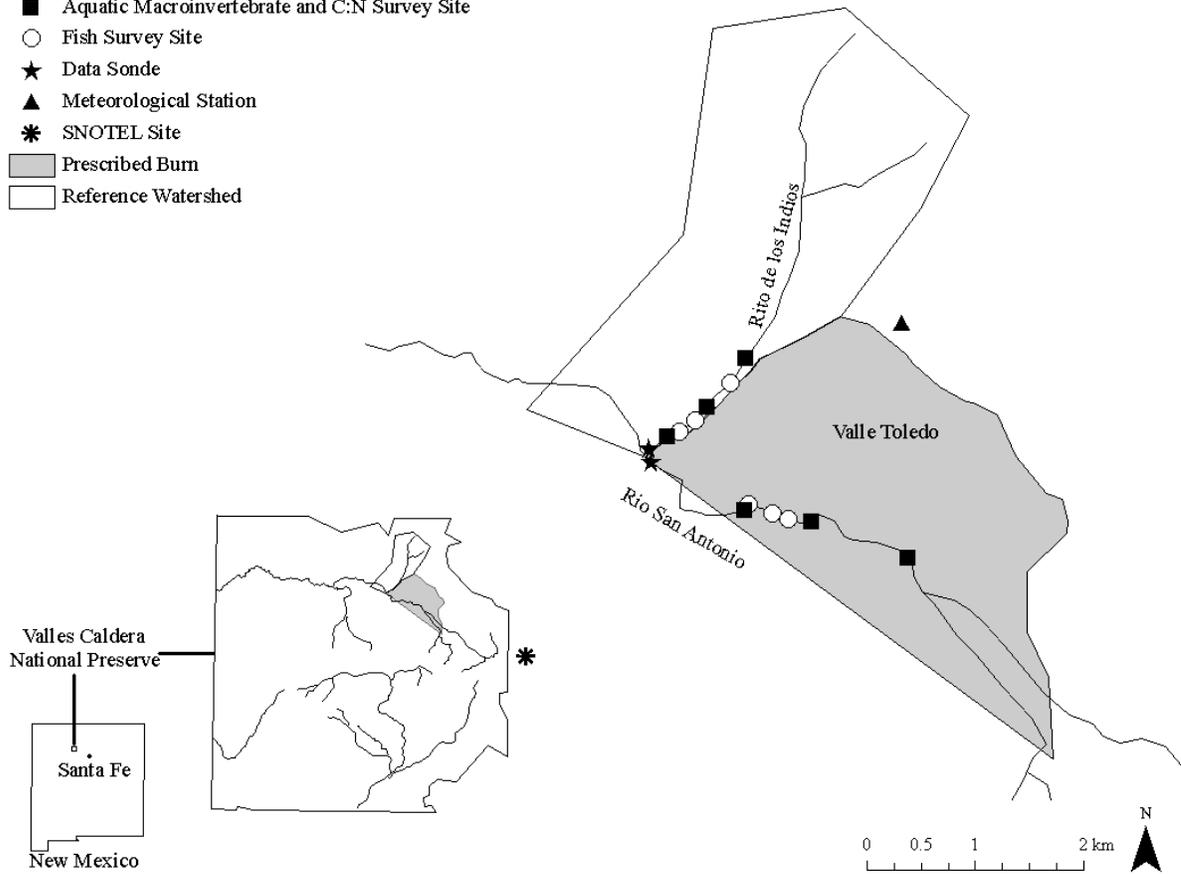
794 burn (spring 2006, fall 2006, spring 2007, fall 2007) within the treatment (Rio San
795 Antonio) and reference (Rito de los Indios) streams of the Valles Caldera National
796 Preserve, New Mexico. Error bars represent standard error of the mean of three samples
797 ($n = 3$) within microhabitats (Riffles and Pools).

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

806 Figure 7. Mean density (fish/100 m²) and standard error of adult Brown Trout (top) and
807 juvenile Brown Trout (bottom) prior to the prescribed burn (pre-burn) spring and fall
808 2005, post-burn spring and fall 2006 and 2007 in the treatment (Rio San Antonio) and
809 reference (Rito de los Indios) streams of the Valles Caldera National Preserve, New
810 Mexico.

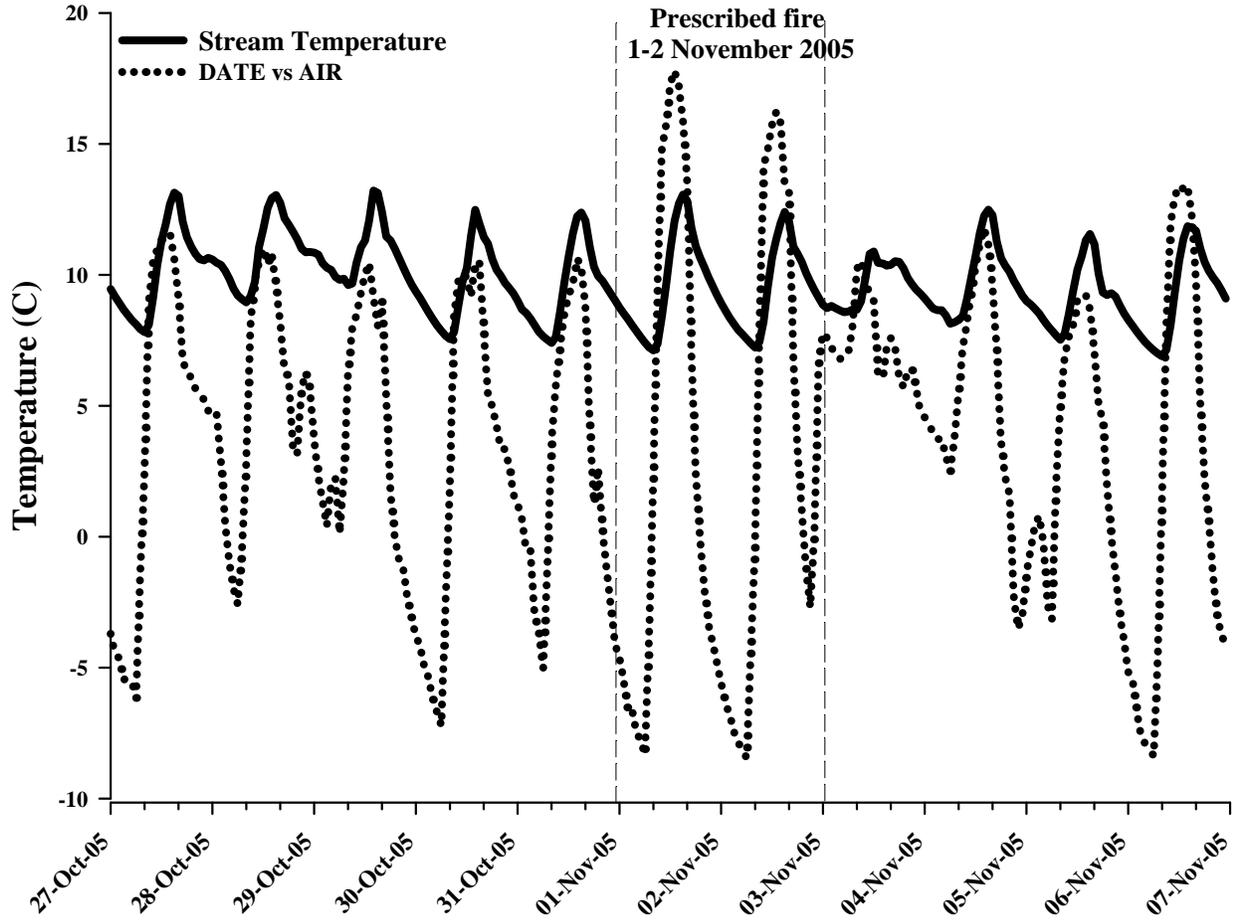
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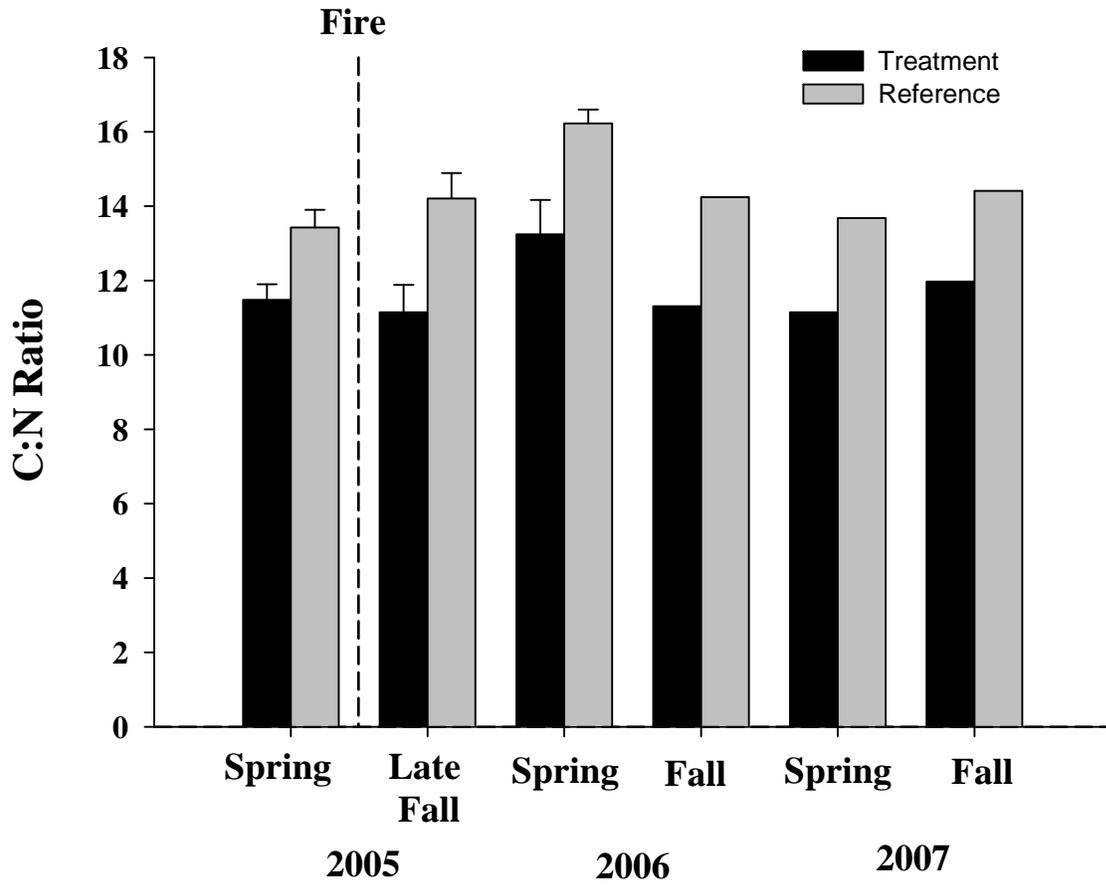
- Aquatic Macroinvertebrate and C:N Survey Site
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- ▒ Prescribed Burn
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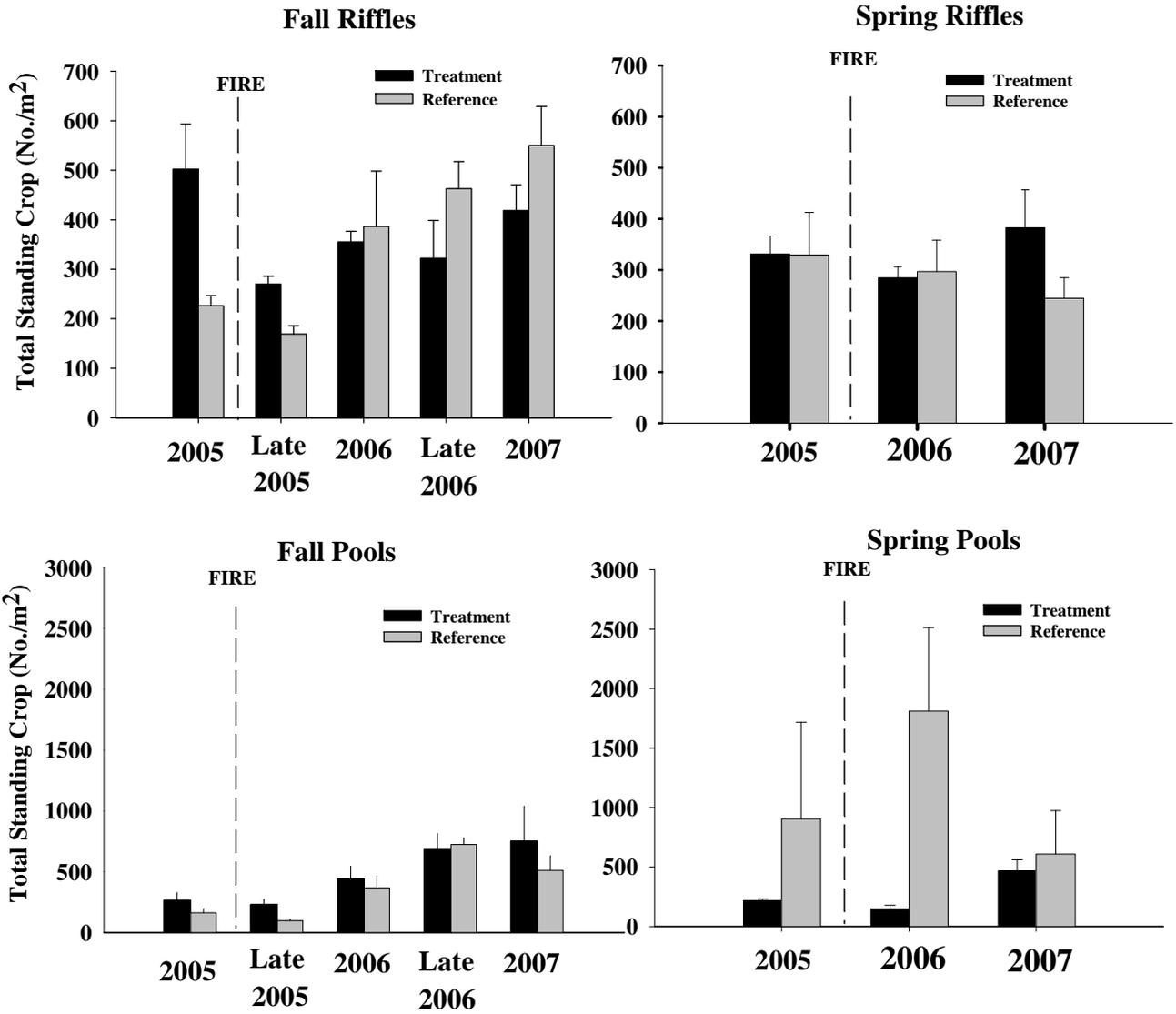
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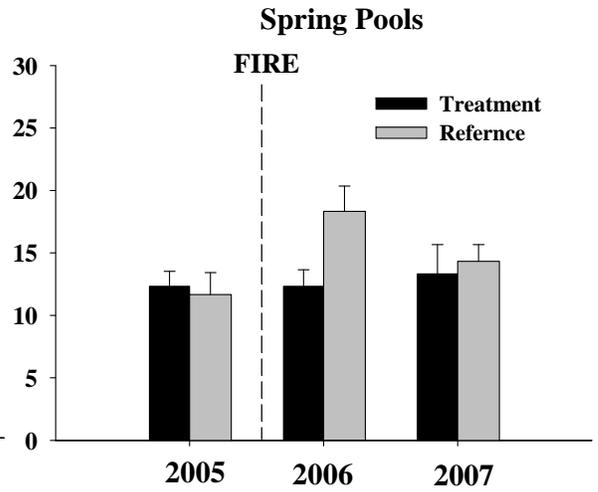
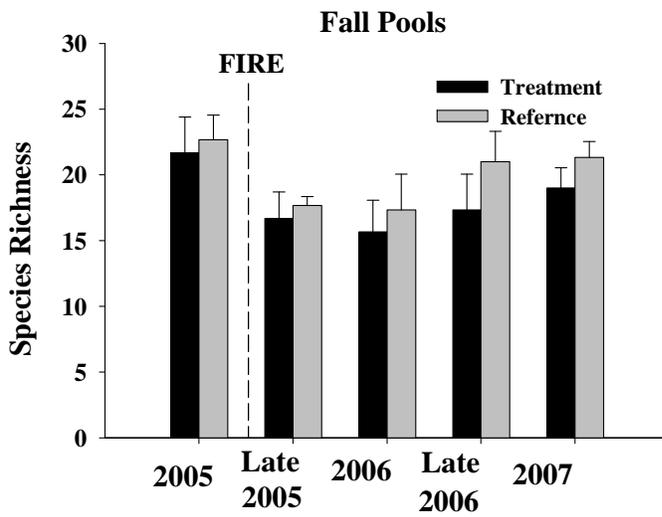
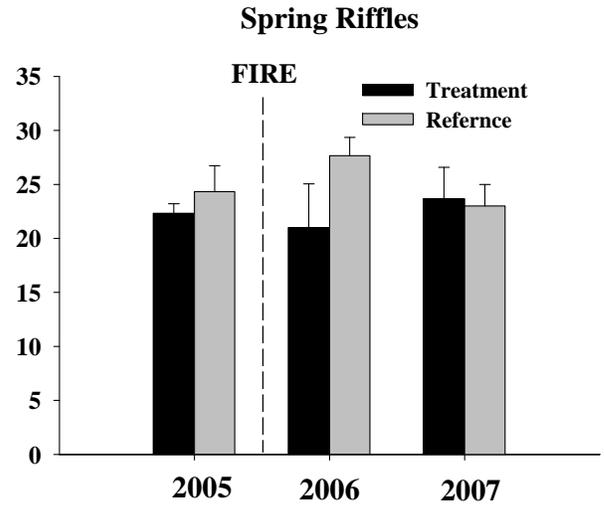
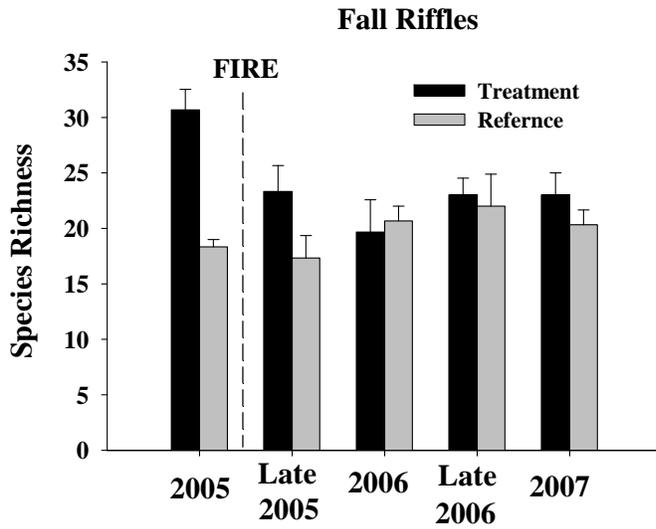


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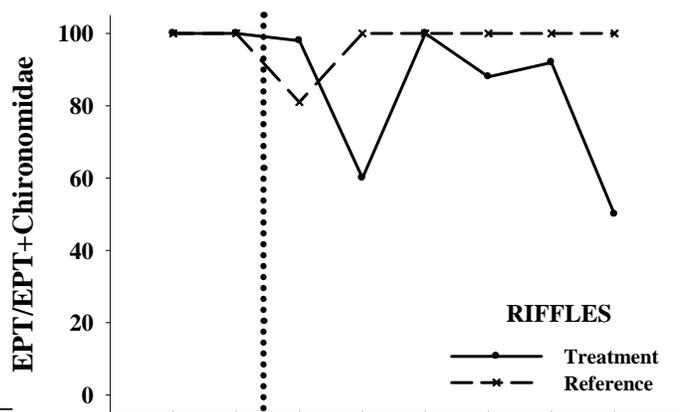
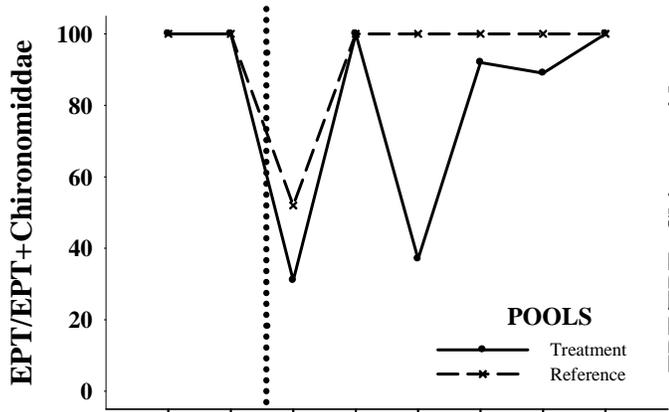
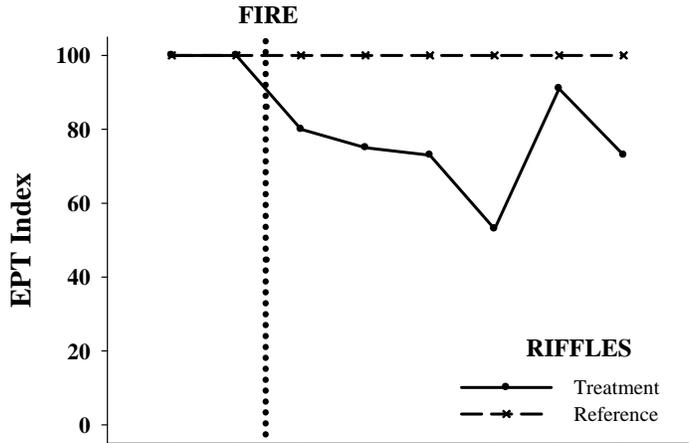
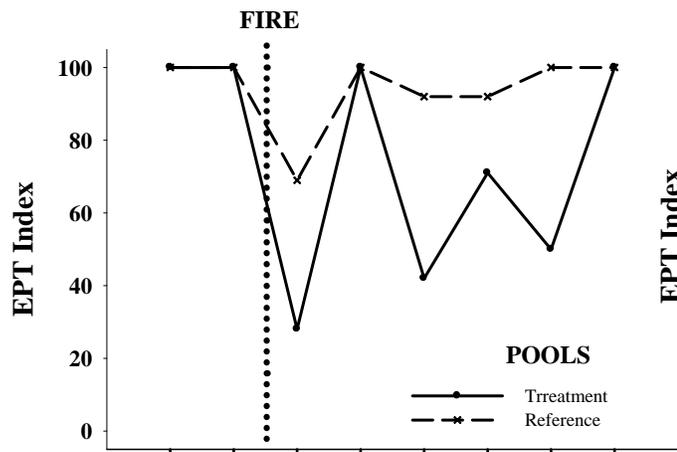


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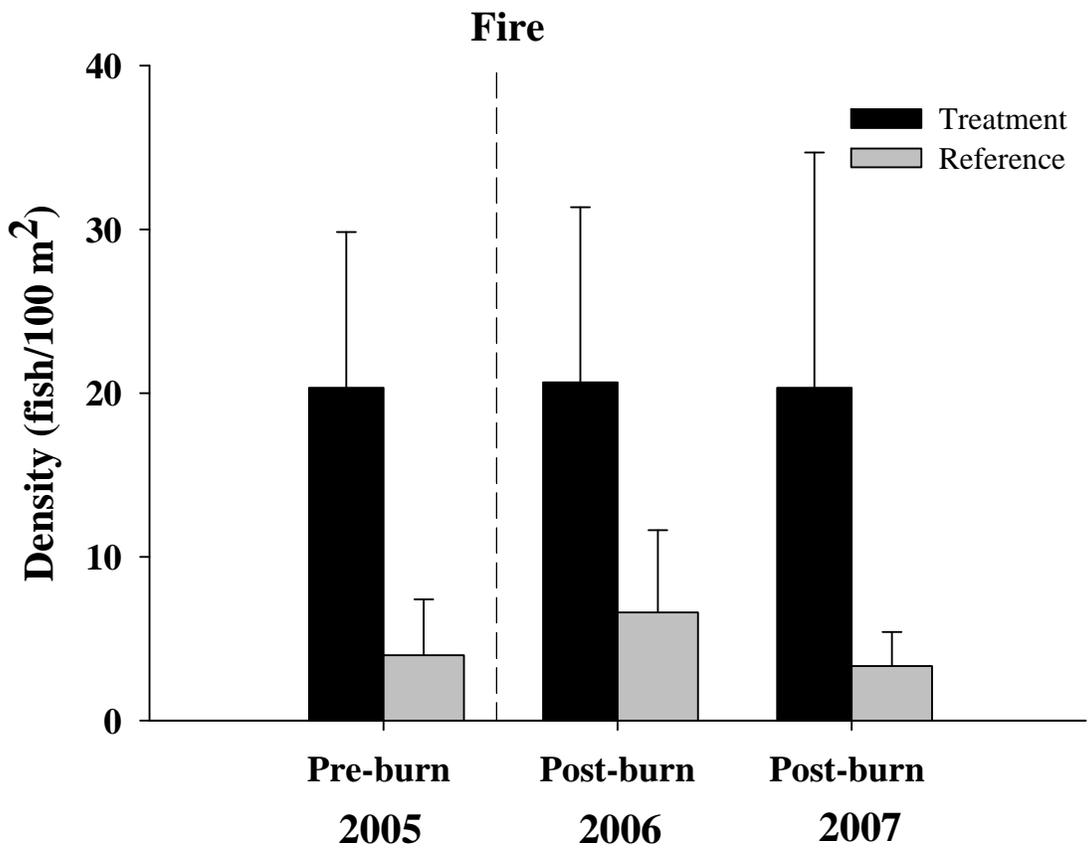
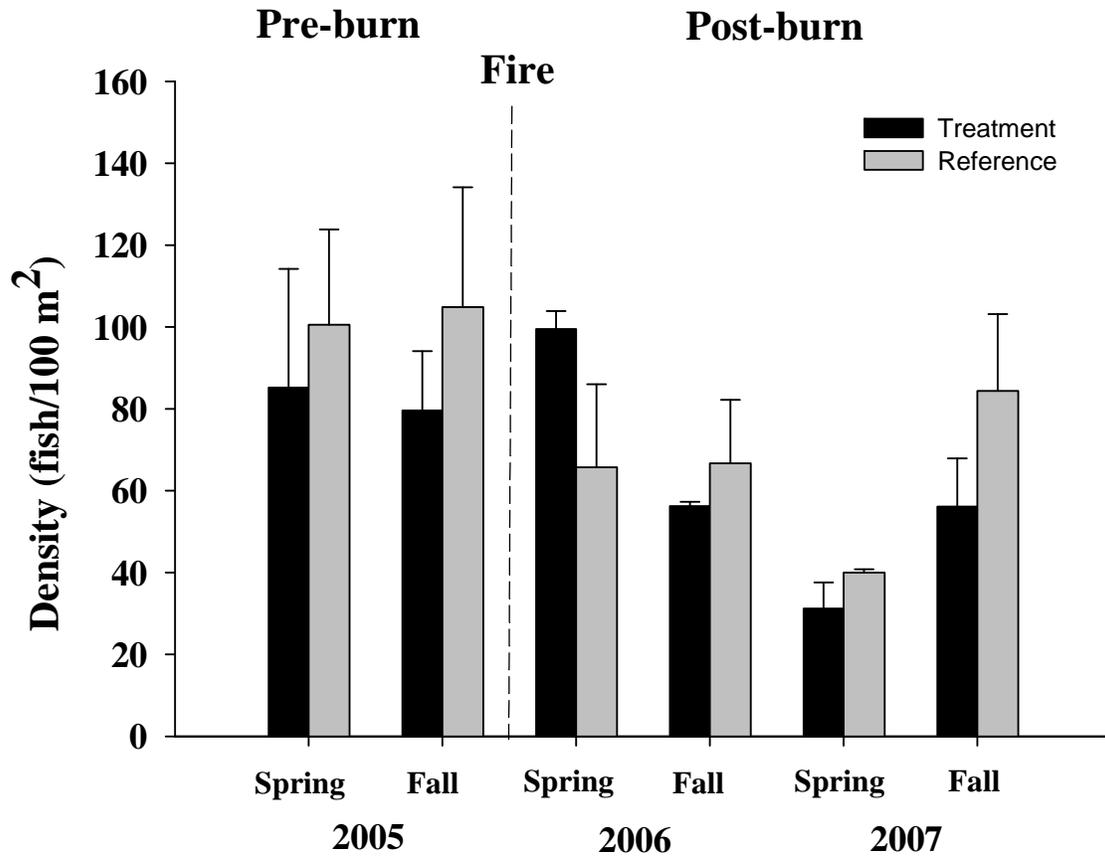
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Appendix. Taxa list of aquatic macroinvertebrates collected prior to the prescribed in 2005 through 2007 post-burn among riffles and pools of the treatment (Rio San Antonio) and reference (Rito de los Indios) streams on the Valles Caldera National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

Taxon	Family, Genus, Species	Taxon	Family, Genus, Species
Plecoptera - stoneflies	<i>Amphinemura</i> sp.	Diptera – true flies	
	<i>Pteronarcella badia</i>		<i>Chronomidae</i> – midges
	<i>Isoperla</i> sp.	<i>Radotanypus</i> sp.	
	<i>Sweltsa</i> sp.	<i>Procladius</i> sp.	
	<i>Hesperoperla pacifica</i>	<i>Thienemannimyia</i> sp.	
Ephemeroptera - mayflies	<i>Baetis tricaudatus</i>	<i>Diamesa</i> sp.	
	<i>Acentrella insignificans</i>	<i>Pagastia</i> sp.	
	<i>Ecdyonurus</i> sp.	<i>Pothastia</i> sp.	
	<i>Cinygmula</i> sp.	<i>Odontomesa</i> sp.	
	<i>Paraleptophlebia</i> sp.	<i>Prodiamesa</i> sp.	
	<i>Tricorythodes</i> sp.	<i>Thienemanniella</i> sp.	
	<i>Drunella grandis</i>	<i>Brillia</i> sp.	
		<i>Cricotopus</i> spp.	
Tricoptera - caddisflies	<i>Rhyacophila brunea</i> cpx.	<i>C. nostococladius</i>	
	<i>Glossosoma</i> sp.	<i>Eukiefferiella</i> sp.	
	<i>Protoptila</i> sp.	<i>Heterotrissocladius</i> sp.	
	<i>Hydropsyche</i> sp.	<i>Hydrobaenus</i> sp.	
	<i>Cheumatopsyche</i> sp.	<i>Orthocladius</i> spp	
	<i>Arctopsyche grandis</i>	<i>Parametricnemus</i> sp	
	<i>Hydroptila</i> sp.	<i>Paraphaenocladius</i> sp.	
	<i>Ithytrichia</i> sp.	<i>Psectrocladius</i> sp.	
	<i>Oxyethira</i> sp.	<i>Pseudosmittia</i> sp.	
	<i>Limnephilus</i> sp.	<i>Synorthocladius</i> sp.	
	<i>Hesperophylax</i> sp.	<i>Tvetenia</i> sp.	
	<i>Oecetis</i> sp.	<i>Cryptochironomus</i> sp.	
	<i>Brachycentrus</i> sp.	<i>Microtendipes</i> sp.	
	<i>Lepidostoma</i> sp.	<i>Paracladopelma</i> sp.	
	<i>Micrasema</i> sp.	<i>Paratendipes</i> sp.	
	<i>Helicopsyche</i> sp.	<i>Phaenopsectra</i> sp.	
	<i>Cernotina</i> sp.	<i>Polypedilium</i> sp.	
	<i>Oligophlebodes</i> sp.	<i>Pseudochironomus</i> sp.	
		<i>Cladotanytarsus</i> sp.	
		<i>Micropsectra</i> sp.	
	<i>Rheotanytarsus</i> sp.		

Appendix. (Continued)

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