THE EFFECTS OF WILDFIRE ON NATIVE TREE SPECIES IN THE MIDDLE RIO GRANDE BOSQUES OF NEW MEXICO

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Introduction

The cottonwood bosques along the Middle Fork of the Rio Grande (MRG) form a ribbon of surviving habitat in this once vast ecosystem. Historically, the channel had a multi-threaded and braided configuration that created a rich mosaic of habitats, including mixed-aged cottonwood forests, meadows, and willow-dominated riparian wetlands and backwaters (Crawford et al. 1993). While the characteristic cottonwood gallery forest still exists throughout the remaining corridor, these systems have been highly modified and many stands of cottonwood survive only as relics of a past ecological regime.

The stressors on the Bosque ecosystems have varied through history, but the underlying thread has been the direct and indirect impacts of water management; in fact, the natural history of the MRG is largely one of water management. Although the forests were used and managed since prehistoric times, settlement by Europeans in the 16th century marked the beginning of a period of more intensive use of the river and its resources, particularly in support of the irrigation-based agriculture that developed in the valley (Wozniak 1987). At the beginning of the 20th century relatively passive irrigation practices gave way to more intensive storage projects seeking to control flooding, secure water supplies and reduce sediment loads. The trend towards intensive water management culminated in 1973 with the construction of the massive Cochiti dam, one of the largest dams of its kind on Earth (Fig. 1).

In conjunction with water storage projects, levees and low flow channels were constructed in the 1920s and '30s to confine the river to a relatively narrow channel and convey a large percentage of flow in controlled channels outside of the active floodplain (Fig. 2). Finally, during the 1950s and '60s over 100,000 Jetty Jacks (Fig. 3), were installed to great effect to stabilize the meandering channel and protect levees (Grassel 2002). The end result of these modifications is the creation of a tightly managed and constrained riverine system that scarcely resembles its natural ecohydrologic condition (Fig. 5).

While these modifications have wrought a myriad of consequences for the MRG's natural system, one fundamental alteration that has ensued is the disruption of the reproductive capacity of the cottonwoods that so characterize and help create the unique environment of the bosque. It is well known that cottonwoods are reliant upon seasonal flooding to create the moist, exposed conditions necessary for their reproduction. Under current conditions, areas supporting successful cottonwood regeneration are scattered and the necessary conditions commonly occur only rarely. Moreover, it is well known that the physical conditions present along much of the river promote the establishment and growth of exotic species - most notably tamarisk and Russian olive. These species often come to dominate the understory, displacing natives such as sandbar willow and New Mexico olive, and may form almost monotypic stands when the cottonwood canopy fails. Moreover, presence of aggressive exotics further depresses the chance of cottonwood seedling establishment when the required conditions do arise.

Acknowledging the reality that widespread restoration of the historical hydrogeomorphic conditions that would support natural cottonwood reproduction is highly unlikely, management

Fig. 1. View SE along Cochiti dam.

Fig. 2. View W from a levee across a low-flow channel towards irrigated agricultural fields.

Fig. 3. Paired rows of jetty jacks installed in the 1950s and 60s served to narrow the channel and elevate its floodplain.

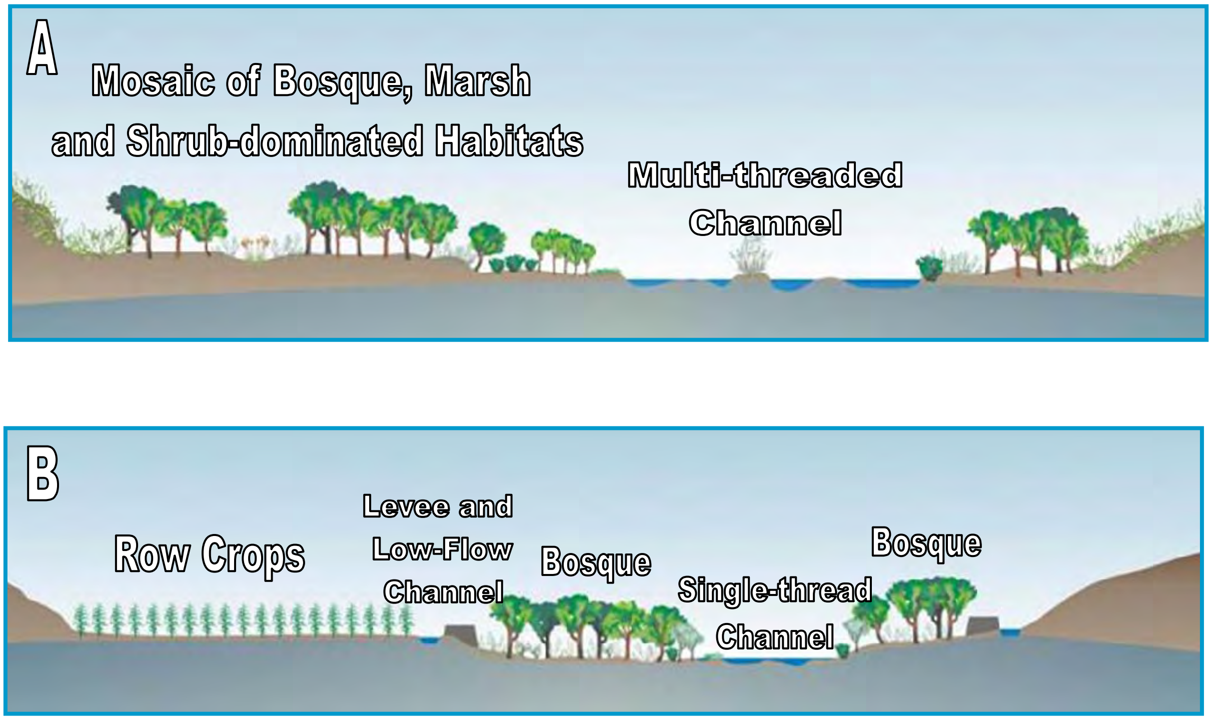


Fig. 5. Illustration of a hypothetical cross-section through the Middle Rio Grande floodplain in its historical natural condition (panel A). Panel B shows the same section as it would currently exist under contemporary management conditions.

strategies for bosque forests have seen an increasing emphasis on preservation of the existing habitats and the established trees within. Under this paradigm, perhaps the greatest natural threat to these forests is wildfire. While exhibiting a natural resistance to fire damage owing to a thick bark layer and the ability to resprout from roots, wildfires have been observed to produce devastating effects on cottonwood populations (Figs. 6 and 7). Two primary factors conspire to cause the observed wildfire impacts to the riparian forests. First, along most reaches of the river, a dense understory of tamarisk and Russian olive has developed below the overarching cottonwood forest. These species produce copious quantities of volatile fuels, which, once ignited, are able to ladder flames up through the understory and into the cottonwood canopy above, causing highly lethal crown fires.

Secondly, once the forests are subjected to such destructive fires, current environmental conditions retard regeneration. Without regeneration-facilitating floods, new cottonwood seedlings have very little chance of establishment. Resprouting, the second mode of cottonwood reproduction, is also commonly not productive owing to the rapid re-growth of exotic shrubs and trees which have the capacity to overtop and out compete the young sproutlings (Fig. 7). When individuals are able to successfully re-establish through resprouting, the ramifications on the resulting forest structure are also not well known. For instance, do suckered individuals develop the same robust stature and canopy structure as seed-grown individuals? Natural functions aside, the cottonwood gallery forest architecture is highly prized by society and a large percentage of urban parks are sited within them. Thus, a single wildfire event, a process natural to these ecosystems, can have essentially irreversible consequences for these forests and their dependent faunal populations, including human ones.

An array of programs, agencies and researchers are actively investigating ways in which cottonwood bosques can be better managed and preserved. Not surprisingly such efforts have generally focused on removal of the exotic understory canopy species, in what are commonly known as fuels reduction treatments. A notable current example of this approach involves the USDA Forest Service, in cooperation with a large number of partners who developed an ecosystem-scale experiment evaluating the outcomes of fuels reductions treatments. Using a randomized block design with three blocks (North, Middle and South) stands were untreated (Figs. 8 - 10), treated through either mechanical removal of all exotic trees and shrubs (Fig. 9 and 11), or mechanical exotics removal accompanied by planting of native shrubs. All cuttings were ground on site and the chips were unevenly distributed (Fig. 11). Larger trunks were often cut into short logs and stacked (Fig. 12). Using a multidisciplined team of researchers from agencies and academia, this study is examining a diverse array of ecological factors to characterize the ecosystem-scale ramifications of fuels reduction treatments.

The overarching objective of our study was to examine the effects of wildfires on cottonwood forests along the MRG to help inform the best management practices for these highly valued natural systems. Because it has been convincingly shown across the country and in a variety of ecosystems that forest fuels have a strong effect on the severity of wildfires, our investigations focused on (1) the effect of ambient fuel load and composition on fire severity as exhibited by cottonwood bole mortality, (2) characterization of individual cottonwood vigor, as expressed as live canopy volume, in relation to fuel loading and other site-level factors and (3) the patterns of woody vegetation regeneration via resprouting in wildfire-effected forests.

Fig. 6 (above). Cottonwood bosque approximately one year post-burn. Mortality was nearly 100 percent in many areas.
Fig. 7 (below). Cottonwood bosque several years post-fire showing the complete loss of the cottonwood canopy and the aggressive reestablishment of the a tamarisk understory that will likely replace the forest.

Fig. 8. Typical bosque site not having undergone exotic removal treatment.
Fig. 9. A partially treated site, showing the results of exotic clearing in the foreground and to the right. In the background at center is an untreated patch for comparison.
Fig. 10. Running a transect tape through a characteristic dense thicket of tamarisk and Russian olive that often exists in untreated sites.
Fig. 11. View of a bosque stand post-treatment showing a characteristic lack of vertical structure and uneven distribution of chips.
Fig. 12. View of larger logs stacked post-treatment.

A serendipitous opportunity provided us with a fourth objective as well. Six wildfires occurred within the acceptable timeframe for this study. One of these had to be excluded because the site lacked suitably sized stands of cottonwoods. A second site had to be excluded owing to a flood that occurred following the fire which made it impossible to infer pre-fire fuel characteristics. Of the remaining four sites, by chance two of them occurred within plots that had recently been subjected to fuels reduction treatments as part of the USDA study described above. The other two wild fires burned untreated, control plots of that same study. This situation provided a rare and important opportunity to also evaluate the efficacy of fuels reduction treatments as a means of preserving the MRG's cottonwood bosques.

Methods

Setting

This study took place in the cottonwood-dominated bosques (forests) associated with the historical floodplain of the Middle Rio Grande River in New Mexico. The approximately 160 km study reach ran from the southern edge of the City of Albuquerque (N 35' 01", E 106' 40") to the Bosque del Apache Wildlife Refuge (N 33' 42", E 106' 56") (Fig. M).

Target habitats consisted of cottonwood gallery forests with canopy coverages greater than 50%. In order to be selected, burned areas needed to possess an adjacent unburned reference area that appeared to possess a canopy structure similar to that of the burned one.

As described above, by chance our study sites corresponded to those used in an ecosystem based study of the effects of exotic clearing. Table 1 lists the sites included in this study along with the treatments, if any, that they received. Figure 13 provides a map to site locations.

Table 1. List of wildfire study sites including the date of the fire, the sample date, whether the site had been subjected to exotics removal and the number of points sampled (Brn. = burn site, Ref. = unburned, adjacent reference site).

Site Name	Fire Date	Sample Date	Fuels Reduction Treatment	Number of Sample Points	
N2	June 2004	July 2005	Yes	**Brn.**	15
				Ref.	12
N3	June 2006	October 2006	Yes	**Brn.**	15
				Ref.	12
M4	June 2004	July 2005	No	**Brn.**	10
				Ref.	15
S1	June 2004	July 2005	No	**Brn.**	10
				Ref.	14

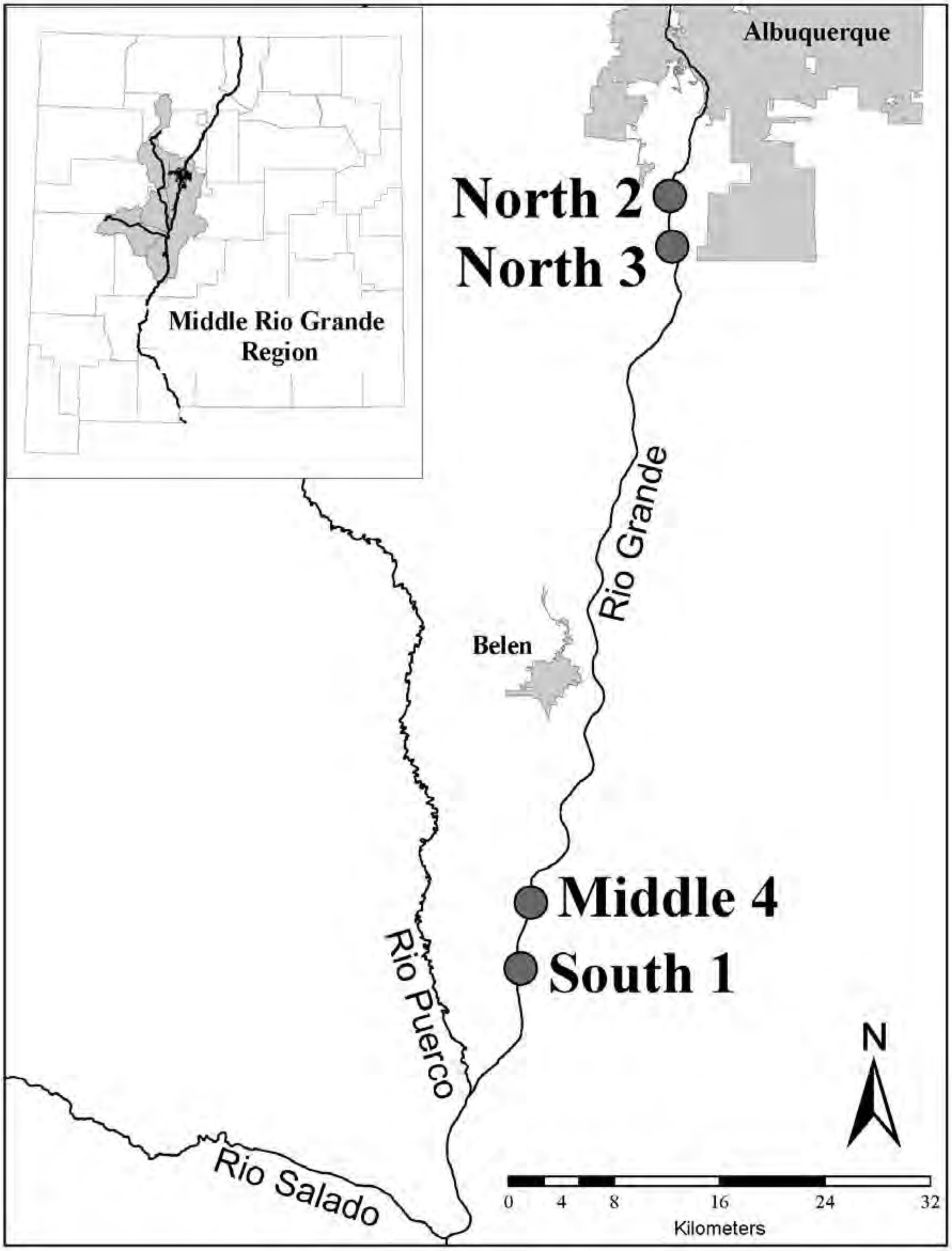


Fig. 13. Overview map of the study region showing the locations of wildfire sites. Unburned reference stands were located adjacent to burn sites (See Figs. 17– 20) for site specific maps. The inset map shows the boundary of the Middle Rio Grande Basin that encompasses the study reach.

Climate

The MRG basin encompasses a large range of climatic conditions influenced by topography and larger scale atmospheric circulations (Dorroh 1946 [from Finch et al. 1995]). In its headwater regions, the mountain environment receives in excess of 127 cm of annual precipitation and supports subalpine ecosystems. Much of the basin, including the area considered in this study, is considerably more arid, however. According to NOAA data (http://www7.ncdc.noaa.gov/CDO/CDODivisionalSelect.jsp) New Mexico's Central Valley region receives annual precipitation amounting to 24.1 cm (1895 – 2007 average).

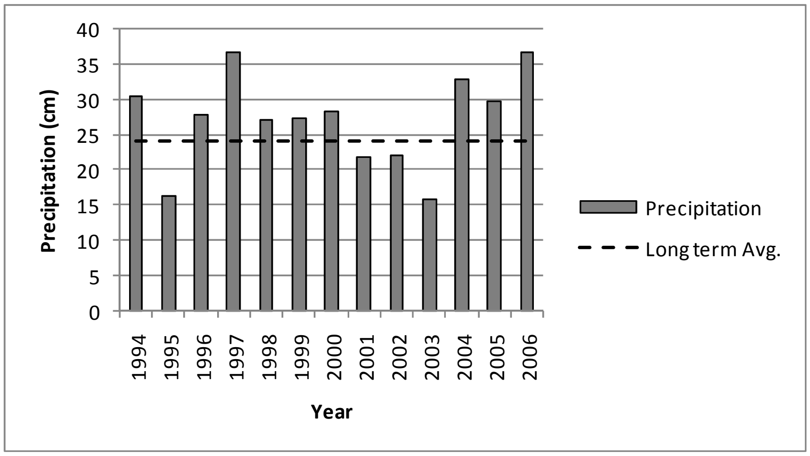
Figure 14 presents precipitation data comparing annual precipitation from 1994 through 2006, to the long-term average. The two fire years, 2004 and 2006, were relatively wet years that followed three years of below average precipitation. Figure 15 shows monthly rain fall during 2004 and 2006, in comparison to long-term monthly averages. In 2004 heavy storms in the spring led to the higher than average precipitation. In 2006, severe mid-summer storms brought precipitation to above normal levels.

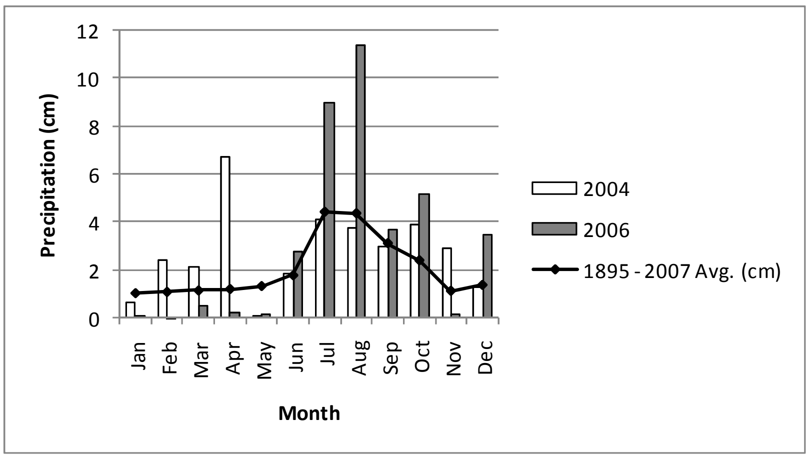
The Palmer Drought Severity Index (PDSI) models the balance between moisture supply and demand, to provide a measure of the relative wetness or dryness of a period. Figure 16 presents monthly PDSI for the Central Valley region. The index generally ranges symmetrically from -6 to +6, with negative values indicating dry periods and positive values signifying wet spells. PDSI values are interpreted as follows: 0 to -0.5 = normal moisture conditions; -0.5 to -1.0 = incipient drought; -1.0 to -2.0 = mild drought; -2.0 to -3.0 = moderate drought; -3.0 to -4.0 = severe drought; and greater than - 4.0 = extreme drought. The same scale and adjectives are applied to positive values indicating wetter than normal conditions. The monthly PDSI values closely follow those of precipitation showing that the three-year period preceding the 2004 fires was dominated by moderate to severe drought conditions. The period after 2004 was dominated by wetter the typical conditions, except a notable period of drought conditions during the 7 – 10 months preceding the 2006 fire.

Study Approach

In 2005, a list of recent fires was complied based on U.S.F.S. records and staff surveys. Twelve fires on accessible land were identified. Nine of these sites were rejected for study because of one or more of the following reasons: 1) the fire event was too old and pre-fire conditions could no longer be reliably interpreted, 2) lack of comparable adjacent unburned reference, 3) insufficient coverage of cottonwood, and/or 4) post-fire flooding. Three sites meeting selection criteria were sampled in July, 2005 (Table 1). In 2006, one additional fire occurred in the study reach (Site N3) and that site was sampled in October.

At each site the extent of the burn was delineated and an adjacent unburned reference was designated (Figs. 17 – 20). Unburned areas were judged as appropriate references based on visual





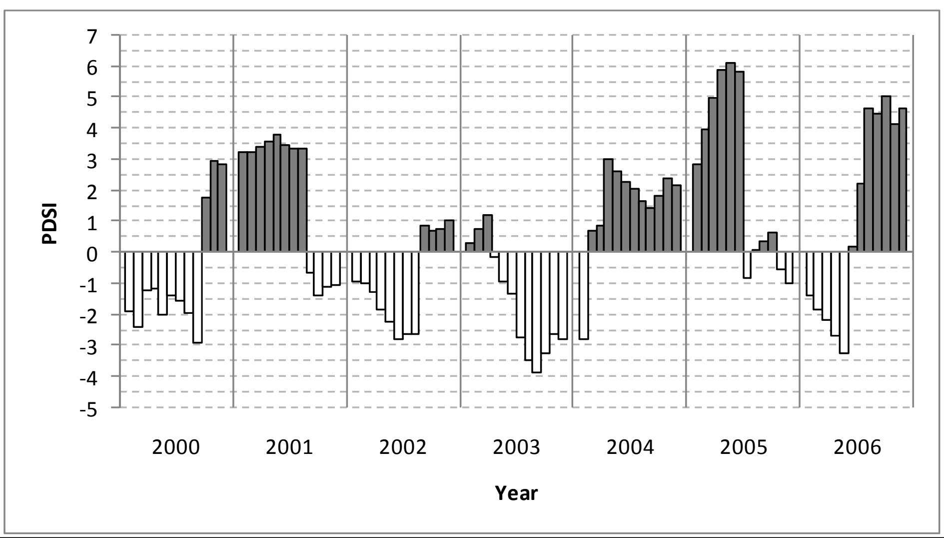


Fig. 14. Annual precipitation measured in the years preceding the wildfires examined in this study as compared to the long-term annual average.
Fig. 15. Monthly precipitation during 2004 and 2006 fire years as compared to average monthly precipitation.
Fig. 16. Palmer Drought Severity Index (PDSI) vales by month (bars) for the years leading up to and including the years during which the studied fires occurred (2004 and 2006).

Fig. 17. A 2006 aerial photograph of the North 2 site showing the extent of the wildfire study area ("Burned") and the unburned reference stand.

Fig. 18. A 2005 aerial photograph of the North 3 site showing the extent of the wildfire study area ("Burned") and the unburned reference stand. This photograph was taken prior to the fire.

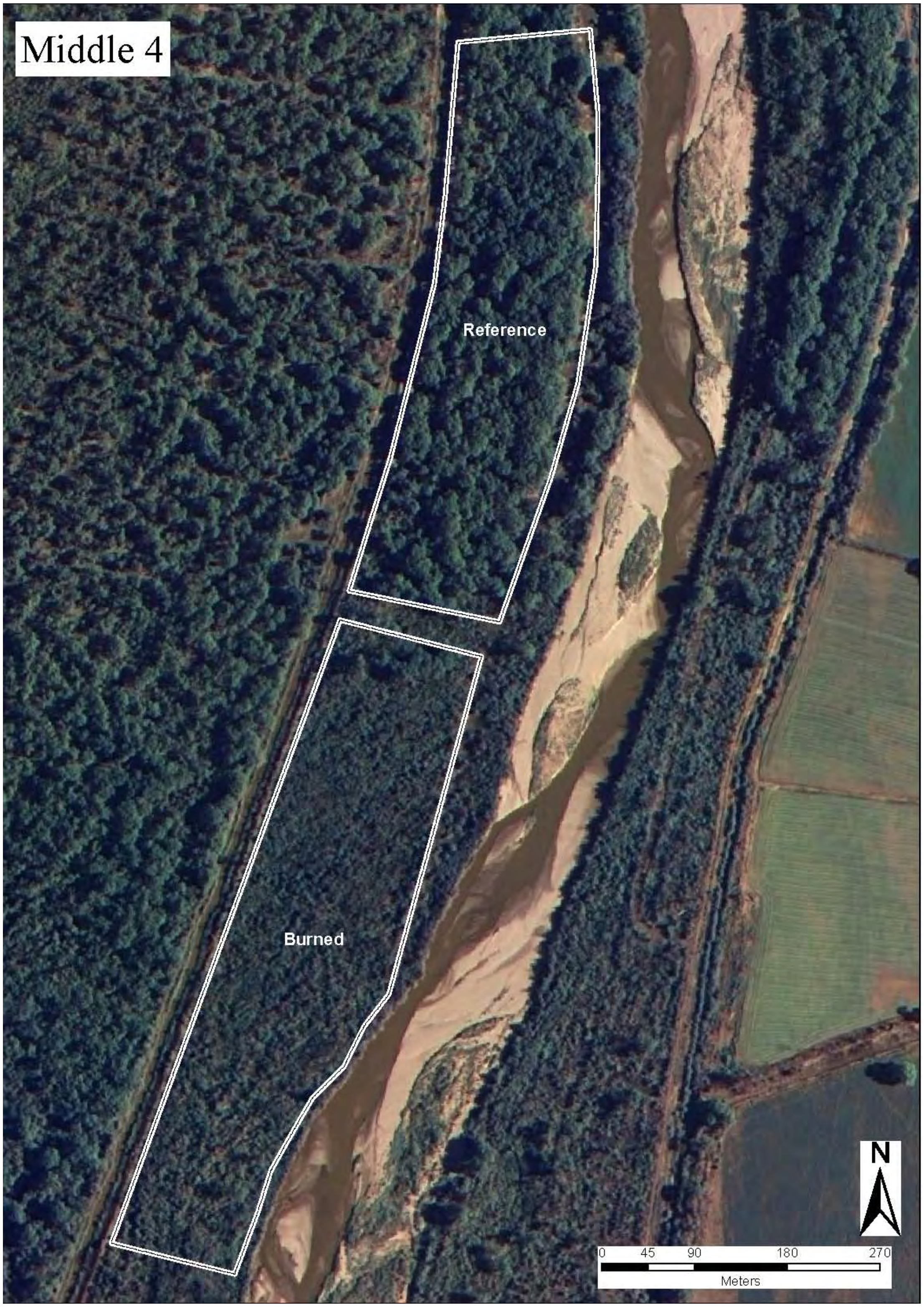


Fig. 19. A 2006 aerial photograph of the Middle 4 site showing the extent of the wildfire study area ("Burned") and the unburned reference stand.

Fig. 20. A 2006 aerial photograph of the South 1 site showing the extent of the wildfire study area ("Burned") and the unburned reference stand.

comparison of cottonwood stem density and size structure. Quantitative sampling and analyses were later used to evaluate the level of similarity between burned and reference stands.

Sampling focused on characterizing the stand characteristics of native tree species and resprouting woody plants and inference of pre-fire fuel loading. Table 2 lists the specific stand attributes that were measured and the methodology used to do so.

Table 2. Stand attributes measured and the methods used at each study site. All of the attributes except pre-wildfire fuel loading were measured at both the burned and unburned reference.

Stand Attribute	Sample Method
Native tree species density	Point-center quarter
Size structure of native tree species	Point-center quarter with DBH measurement
Vigor and health of individual native trees	Canopy volume method
Stand-level mortality	Point-center quarter
Resprout Density	Point-center quarter (only in burned stands)
Pre-wildfire fuel loading	Brown Method (only in unburned reference stands)

At each stand, five transects were laid out perpendicular to a baseline running roughly parallel to the MRG channel (Fig. 21). All stand attributes except resprout density were sampled in the reference stands. Conversely, in burned plots, fuel loading was not measured since fuels were consumed by the fire. Methods used in each pursuit are described below.

Evaluation of Cottonwood Stand Structure

The structure of native canopy species was evaluated using the point center-quarter method (Cottom and Curtis 1956); Figs. 22 and 23). In each stand, sample points spaced at 40m intervals were laid out along five transects. We found that a 40 m sample spacing was the minimum distance which consistently allowed four unique trees to be sampled. As it turned out, this spacing was a bit more than twice the mean maximum distance measured at each point.

The total number of points sampled at each stand varied according to the width of the wildfire or reference habitat patch (Table 1). At each point, the distance to the nearest native tree, live or dead, in each quarter was measured using a Sonin® Combo Pro distance meter and its diameter at breast height (DBH) was measured using a calibrated diameter tape. When individuals possessed multiple stems, the diameter of each was recorded.

The overall health and vigor of selected trees was assayed by visually estimating the percentage of the potential canopy volume that was still living as described by Scott et al. (1999). If a tree had no live canopy, it was recorded as “bole dead”. It is acknowledged that trees which experience bole death may still have live underground parts, and the individual might survive via root sprouting. Since it was not generally possible to determine which tree a given resprout belonged to, and also because

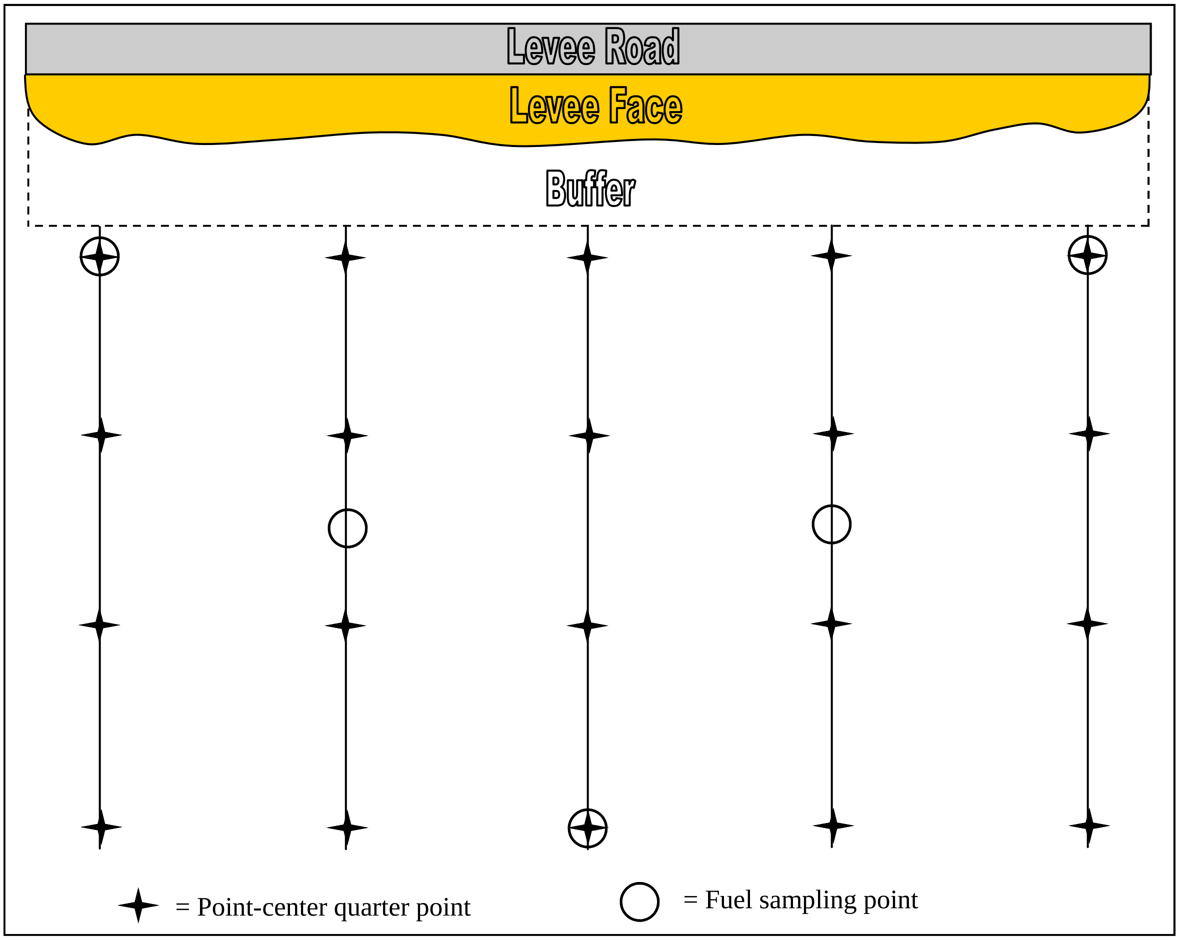


Fig. 21. Schematic diagram of the stand sampling design.

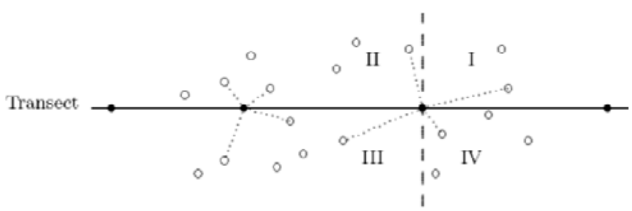


Fig. 22. Diagram showing the point-centered quarter sampling approach. Hollow circles represent trees. In each quarter, the tree nearest the point is sampled.

Fig. 23. The point-center quarter frame in position along a transect (yellow tape line to lower right).

conservation of existing canopy structure is a fundamental management goal, our sampling focused on the measuring the survival of the tree bole.

Within each point quarter, the distance to the nearest resprouting tree or shrub, regardless of nativity, was also measured. Species specific resprout density was used to gauge the probable trajectory of site succession.

Fuels Analysis

Inference of pre-burn fuel loading was based on a space-for-time substitutive approach (e.g., Whittaker 1975, Pickett 1989), where unburned areas adjacent to the burn sites were used as references to infer the character and composition of burned areas prior to fire. In reference stands, one fuel inventory point was placed along each of the five sampling transects in a stratified fashion to characterize fuels across the range of conditions found at the site (Fig. 21).

Brown et al.'s (1982) method ("Brown Method") for inventorying fuel biomass was used to quantify and characterize fuel loading. Because the Brown Method was developed in northern coniferous forests, the method required some modification and further development as described in the sections below.

The Brown Method utilizes a combination of variously sized plots and transects arrayed in a standardized format. Figure 24 shows the layout of the Brown Method's sampling regime. The narrative below is not intended to describe the Brown Method in its entirety, but rather outline the approach to readers that are not familiar with it; and more so to describe the way in which the method was refined to work under the specific conditions of this study and the MRG ecoregion.

The Brown Method considers six components of total fuel load: downed woody debris, duff, litter, herbaceous vegetation, shrubs and small trees. Because herbaceous cover was very patchy, and at sites N2 and N3, where quantitative data were available, it only averaged about 13 % (Johnson et al. 2010 – herbaceous vegetation at seven other sites included in that study possessed similar average cover values, as well), herbaceous vegetation was assumed to be a trivial source of fuel loading and so was not included in analyses. Measurement and analytical procedures for the six other components are described below. Because the Brown Method contains empirical constants, we used the prescribed imperial units during calculations and then converted to S.I. units as a final step.

Downed Woody Debris (DWD)

The DWD component of fuel load was measured using transect lengths of 2.0 m, 3.7 m and 15.24 m, for 0 – 2.5 cm, >2.5 – 7.5 cm, and > 7.5 cm diameter material, respectively. The mass of DWD was calculated using Brown's (1974) formula:

$$\boldsymbol{Component\ Loading\ (^{T}/_{ac}) = \frac{11.64 \times n \times D^2 \times s}{l \times N}}$$

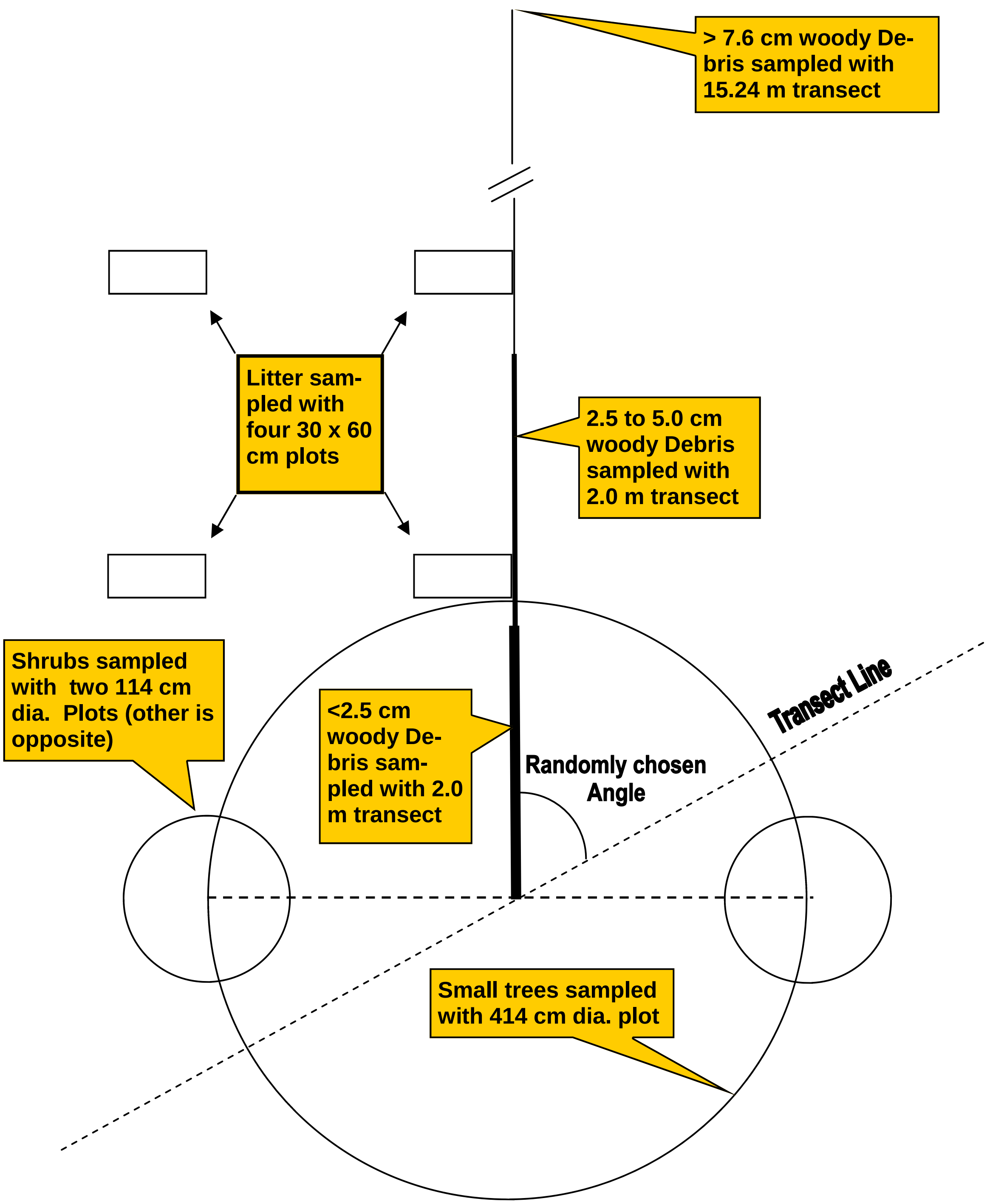


Fig. 24. Schematic diagram of the Brown Method's fuel inventory protocol. Objects indicated with solid lines are the entities actually used during sampling. Dashed lines indicate layout guide lines (After Brown et al. 1982. See that text for additional details).

Where:

11.64 = an empirical constant

n = the number of intersections for a given size class

D^2 = the average squared diameter for the size class in inches

s = the average specific gravity of material within a size class

l = the length of the transect (feet)

N = the number of sample points

No correction factor for topographical slope was used since all sites were essentially level. The default values representing empirical averages that are provided in Brown (1974) were used for D^2 and s. The mass of each DWD size class was initially calculated as tons/acre to utilize Brown's empirical constants and then converted to kg/ha.

Duff

Duff, comprised of the fermentation and humus layers, was measured using a ruler to determine its depth. To extrapolate to the stand-level volume the following formula was applied.

$$\boldsymbol{Component\ Loading\ \left(T/_{ac}\right) = 3,360Bd}$$

Where:

3,360 = conversion factor

B = Bulk Density = 6.36 lb/ft^3 (composite value, Brown et al. 1982)

D = average duff depth (in.) across all points within a stand

Litter

Litter was sampled in four plots 30 x 60 cm plots as shown in Fig. 24. Litter sampling followed a modified version of the Brown Method. Litter was first classified into a cover type. The percent coverage of the litter was then estimated by eye and its average depth was measured using a steel rule. Volumetric samples of each cover type were then collected and transported to USDA Forest Service facilities where they were oven-dried to a constant weight and weighed to derive cover type-specific densities. Within litter cover types, the average value of measured density values was used to determine this component fuel loading (Table 3).

Plot-level component loading was determined using the following formula:

$$\textit{Component Loading}\ \left({}^{g}/_{30\ x\ 60\ cm\ plot}\right) = 1800 \times \%\ \textit{litter cover} \times \textit{litter depth}\ (cm) \times\ \textit{Density}_{\ litter\ cover\ type}\left({}^{g}/_{cm^3}\right)$$

Plot-level component loadings were then converted to kg/ha units, averaged across plots within a sampling point and then between points within a stand to derive the litter component loading.

Table 3. Density values used to calculate the litter component of total fuel load.

Litter Cover Type	Density (g/cm^3)
Populus Leaf Litter	0.03
Populus/grass	0.05
Fine chips	0.26
Fine chips and sticks	0.16
Populus Leaf Litter and Chips	0.14
Populus Leaf Litter and twigs	0.09
Twigs	0.15

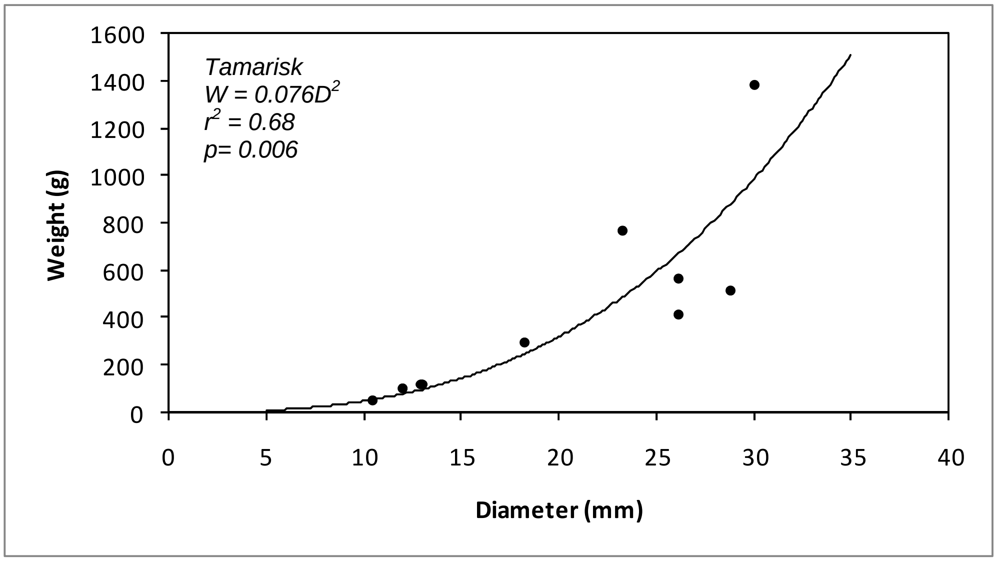
Shrubs

At sites M4 and S1 which had not undergone exotic cutting, shrubs were predicted to be the dominant source of fuels. Shrubs were sampled using two 114 cm diameter plots at each point as prescribed by Brown et al. (1982). In each plot, stems were counted and their basal diameters measured with calipers.

Because the Brown Method was developed in a different ecoregion, we were uncertain as to the applicability of the allometric relationships between shrub size and biomass presented in the method's description. To contend with this, whole stems of the dominant shrubs in a variety of age classes were collected and dried to a constant weight at a U.S. Geological Survey facility. The basal diameter and weight of each sample were then measured to model the allometric relationship between size and weight (Fig. 25). Our empirical models were compared to those included in Jenkins et al. (2004) and Brown et al. (1982). From these sources, species-specific formulae were selected to relate stem basal diameter to total above ground biomass (TAGB; Table 4). The measured relationship for *Ailanthus* fit Brown et al.'s (1982) model for medium shrubs well, therefore it was also applied to *Lycium pallidum* and *Amorpha fruiticosa,* which were present in three plots.

Table 4. Modeled relationships between basal diameter and Total Above Ground Biomass (TAGB).

Species	**Modeled diameter – TAGB relationship**	**Source of Model**
Tamarix ramossissimum	TAGB(g) = $0.076D^{2.8}$	Fig. Y
Prosopsis pubescence	TAGB(g)=$10^{-1.25+2.546LOG10(Dia(cm)*1000}$	Jenkins et al. (2004)
Lycium pallidum	TAGB (g) = $0.104(dia(cm)X10)^{2.5}$	Fig. Y
Amorpha fruiticosa	TAGB (g) = $0.104(dia(cm)X10)^{2.5}$	Fig. Y



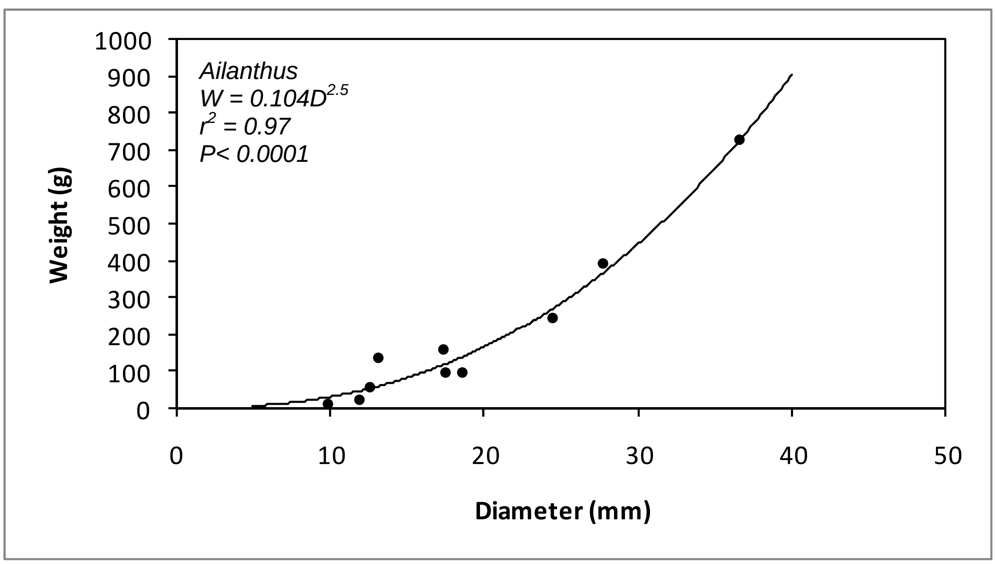


Fig. 25. Two examples of the species-specific allometric relationships between basal diameter and weight that were calculated based on samples collected at MRG study sites.

Because at three of the wildfire sites (N2, N3, and S1) we had intensively sampled the shrub canopy prior to burning as part of a different study (Johnson et al. 2010), those cover data were deemed to be the most accurate representation of shrub component loading. Evangelista et al. (2007) present an empirical formula modeling the relationship between biomass and Tamarix canopy cover, where:

$$\boldsymbol{Log_{10}(Total\ Above\ Ground\ Biomass) = -0.0518 + (1.5445)Log_{10}(Cover)}$$

In southwestern Colorado this model yielded an R^2 value of 0.946, describing the relationship of the variables. At the three stands where it was available, the measured cover of *Tamarisk*, the overwhelming dominant shrub, was used to estimate this species' contribution to fuel loading. To these values were added the contribution of other species as characterized using the Brown Method.

Small Trees

The final component of fuel loading considered by the Brown Method are small trees. Trees were sampled using a 414 cm diameter plot as shown in Fig. 24. Each tree included in a plot was identified to species and its diameter was measured. As with shrubs, Brown et al. (1982) do not include allometric relationships for the species of small trees found along the MRG or other species with comparable growth forms. To deal with this information gap, we used our own data collected for this purpose to create a model or we utilized diameter - weight equations found in the literature (as summarized in Jenkins et al. 2004). Table 5 shows the equations used. Since it was beyond our capacity to collect and process large individuals and no species-specific relationships were identified in the literature, for *Salix gooddingii* and *Elaeagnus angustifolia* we used a general relationship published in Jenkins et al. (2004) for woodland habitats.

Table 5. Allometric models used to estimate biomass of small trees based on the diameter at breast height (dbh) or basal diameter (dia).

Species	Modeled diameter – TAGB relationship	Source of Model
Ailanthus altissima	TAGB (g) = 0.104(dia(cm)X10)$^{2.5}$	Fig. Y
Salix gooddingii	TAGB(g)=Exp (-0.7152+1.7029Xln(dbh))	Jenkins et al. (2004)
Elaeagnus angustifolia (<2.5 cm dia)	TAGB (g) = 0.013(dia(cm)X10)$^{3.3}$	Merritt and Johnson (unpubl. Data)
Elaeangnus angustifolia (>2.5 cm dia)	TAGB(g)=Exp (-0.7152+1.7029Xln(dbh))	Jenkins et al. (2004)

Data Analysis

Fuel loading and phytosociological data from individual points were aggregated by site. Because of the stochastic occurrence of wildfires only four sites were available to study. With four study sites, the power of statistical inference and hypothesis testing were limited. As such our investigations focused on identifying trends in the data. In particular, analyses sought to explore five basic questions:

(1) Is there a relationship between fuel loading and fire-induced bole mortality of native tree species?
(2) Is there a relationship between fuel loading and canopy vigor following fire?
(3) Are there specific components of the fuel load that have a disproportionate effect on native tree mortality?
(4) How effective is removal exotics from the understory at preserving the existing cottonwood canopy?
(5) Given that resprouting is an important means of recolonization following wildfire, what are the density and species composition of resprouting woody plants?

Relationships between fuel loading, mortality and canopy vigor were investigated with linear regression using the SPSS statistical package (SPSS 2008). Two-sample t-tests were used to compare the means of site attributes.

Results

Comparison of Paired Sites

Unburned areas adjacent to wildfire sites were used generate estimates of fuel loading prior to fires. To gauge the appropriateness of these comparisons, the burned and unburned portions of each site were compared based on a number of descriptive metrics. Because native woody species other than cottonwood, including Goodding's willow and screwbean mesquite, were so uncommon at the study sites their phytosociological attributes could not be meaningfully captured; thus these species were excluded from analyses.

Mean DBH of *Populus* was equivalent in all paired sites except at M4 where the unburned mean was marginally lower than in the burned (35.7 vs. 27.7 cm; $p = 0.04$; Fig. 26). *Populus* size structures reveal additional detail about the composition of sites (Fig. 27). All sites lack the reverse-J size class distribution indicative of regenerating forests (Leak 1975), with sapling-sized (<10 cm DBH) always comprising less than 10% of the individuals sampled (Fig. 27). Site M4 represented the most extreme example of this age class depletion with only one individual sampled with a DBH of less than 5 cm. In general each site displays a maximum in the 30 – 60 cm size class and size class distributions are similar

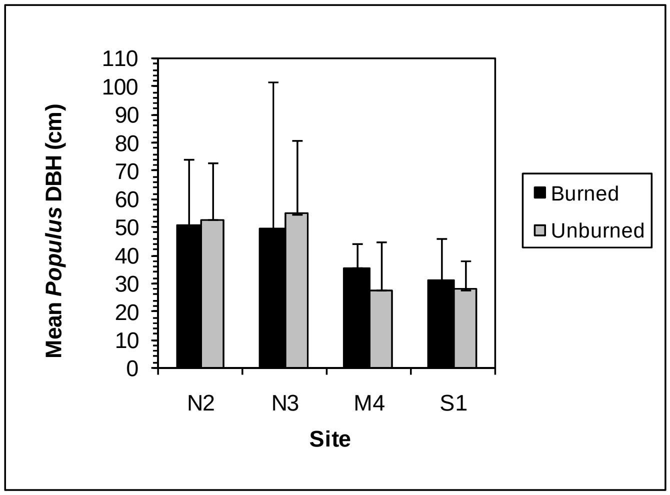


Fig. 26. Comparison of mean DBH between burned and unburned reference sites. For individuals with multiple trunks, the largest stem was used to calculate the site mean. All means are equivalent except at site M4 where the mean DBH in the unburned site was marginally significantly lower than in the burned one. Error bars represent one standard deviation.

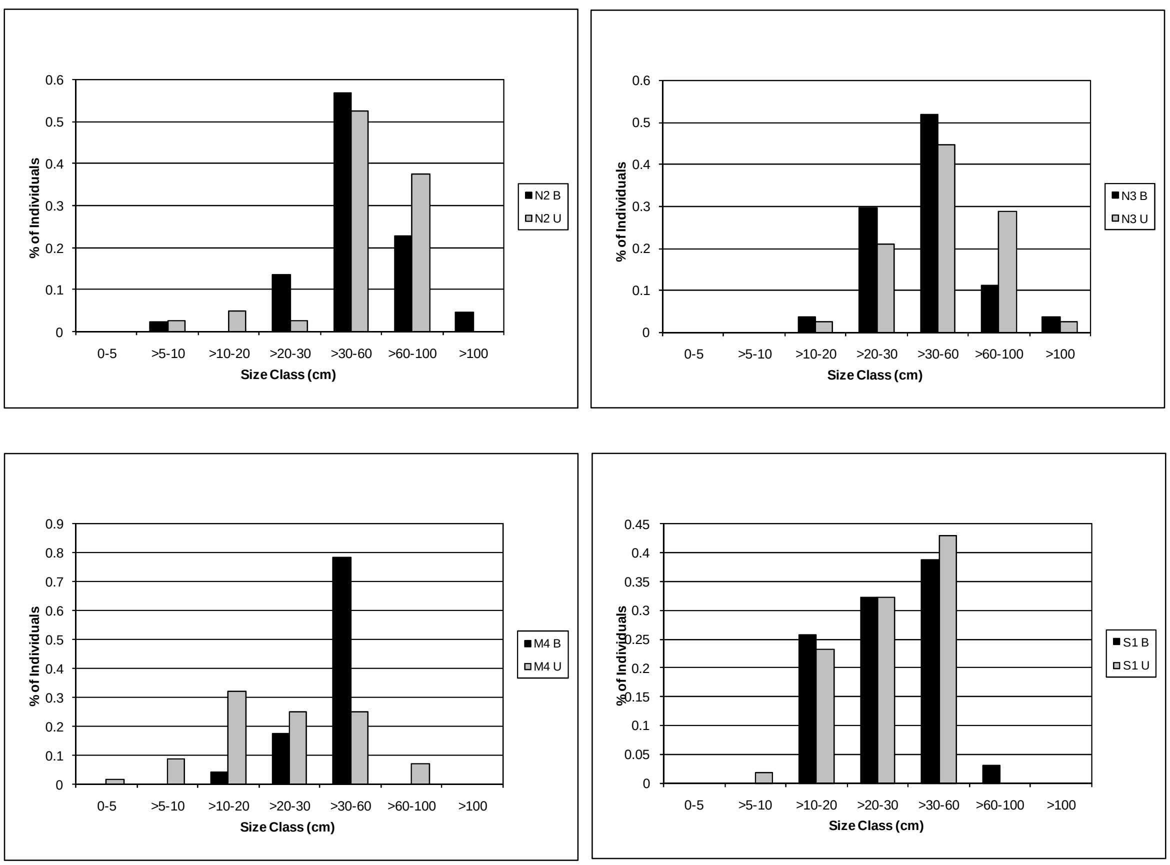


Fig. 27. Histograms displaying comparisons of the size structure of *Populus deltoids* in burned and unburned reference sites. In general, size structures were similar between sites, except in the M4 burn site which contained a higher proportion of 30—60 cm individuals.

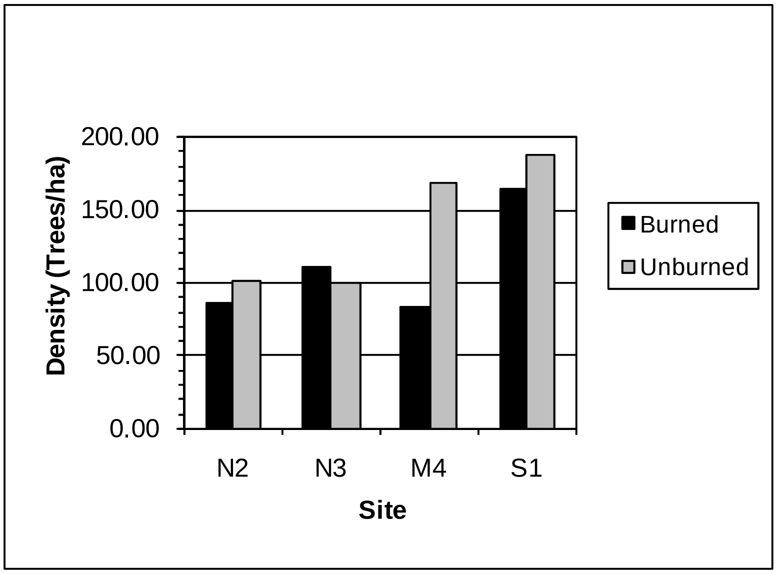


Fig. 28. Comparison of *Populus* density in burned and unburned reference sites. Each pair is very similar except M4 whose tree density in the unburned reference is almost double that of burned area.

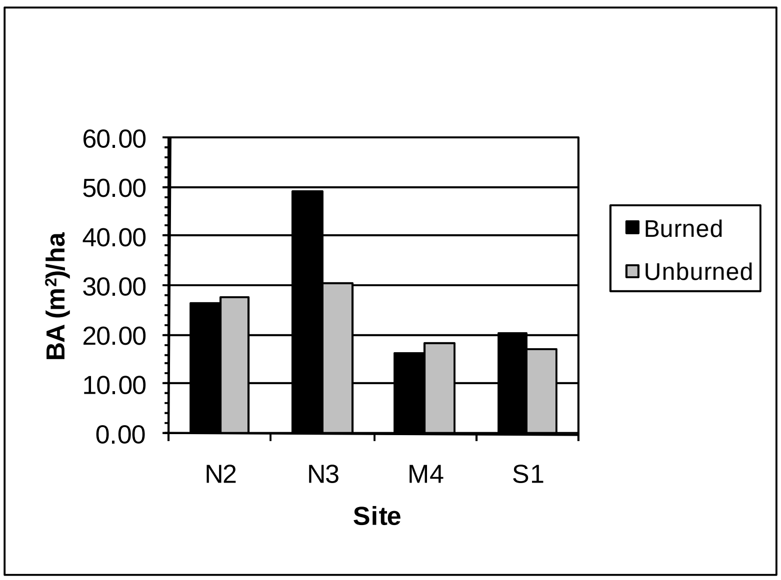


Fig. 29. Comparison of basal area per ha in burned and unburned reference sites. This metric is very similar between paired sites, except in N3 which has approximately 40% more basal area/ha in the burned area than does its reference.

between burned and unburned reference sites. Again the exception is found in M4, wherein the unburned site lacks the 30 – 60 cm peak and displays a relatively flat size class distribution. The low number of such individuals caused the disparity in mean DBH.

Populus density was also similar between burned and unburned references, again except in the case of M4 (Fig. 28). M4's unburned reference had nearly twice the tree density of the burned area. This pattern would be expected given the younger status of the unburned area. Comparing basal area/ha (i.e., dominance), it can be seen that M4's unburned and burned sites are quite similar, with the higher density compensating for the lack of larger individuals (Fig. 29). Dominance values were also similar between the unburned and burned areas of sites N1 and S1. At site N3, dominance in the burned area is about 40% higher than in the adjacent reference, despite the similarities in density, mean DBH and size structure. The higher dominance in N3's burned area is apparently caused by the preponderance of multi-stemmed individuals. There 33% of trees sampled had more than one stem, compared to only 11% in N3's unburned reference.

Fuel Loading

Total fuel loading ranged from 96,000 to 172,000 kg/ha (abbreviated as 96 – 172 kg x 10^3/ha). There were no clear trends in total fuel loading across sites (Fig. 30), with N2 and M4 having nearly identical fuel loads (172 and 169 kg x 10^3/ha, respectively). Fuel load at M4 was somewhat lower at 142 kg x 10^3/ha, while the lowest loading occurred at site S1 where loading was only 96 kg x 10^3/ha. This finding is at first surprising since sites N2 and N3 had been subjected to fuels "reduction" treatments one to two years before sampling.

Consideration of the distribution of fuel types, along with known treatment practices explains this apparent paradox (Fig. 31). In this distribution, total woody debris (TWD) includes naturally downed sticks and branches, as well as logs and slash created by clearing. Similarly, the litter category includes natural leaf litter and other recalcitrant poorly decomposed detritus as well as chips and fine material created by the treatment process. Duff is comprised of the well decomposed humus layer that typically exists below the stratum of litter (Brown et. al 1982). The shrub and small tree categories are self-explanatory.

In the treatments employed at sites N2 and N3, exotic shrubs and trees were cut and chipped. Smaller stems and branches were then chipped, while the larger components were cut into approximately 50 – 100 cm lengths. Chips were unevenly spread on sites and logs were stacked or piled (Figs. 11, 12, and 32). As is clear in Fig. 31, in the treated sites (N2 and N3) TWD and litter comprised the vast majority of fuel load (93 and 91%, respectively), whereas at the other sites these categories made up a third or less of the ambient fuel load. Instead, in the untreated M4 and S1 sites, the shrub category – which was almost entirely comprised of *Tamarisk* – strongly dominated the fuel load. As shown by Johnson et al. (2010), small trees, most commonly *Elaeagnus angustifolia*, can also be an

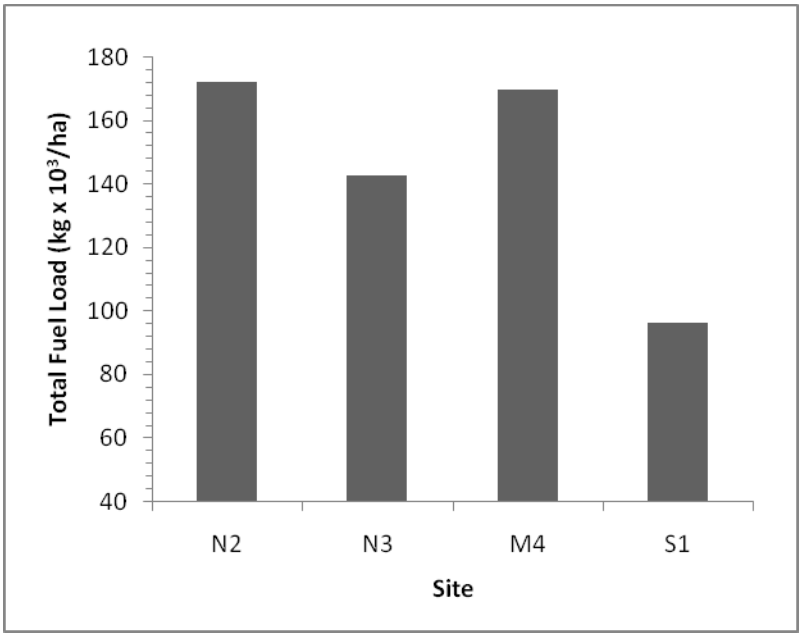


Fig. 30. Total fuel load measured at each of the four unburned reference stands.

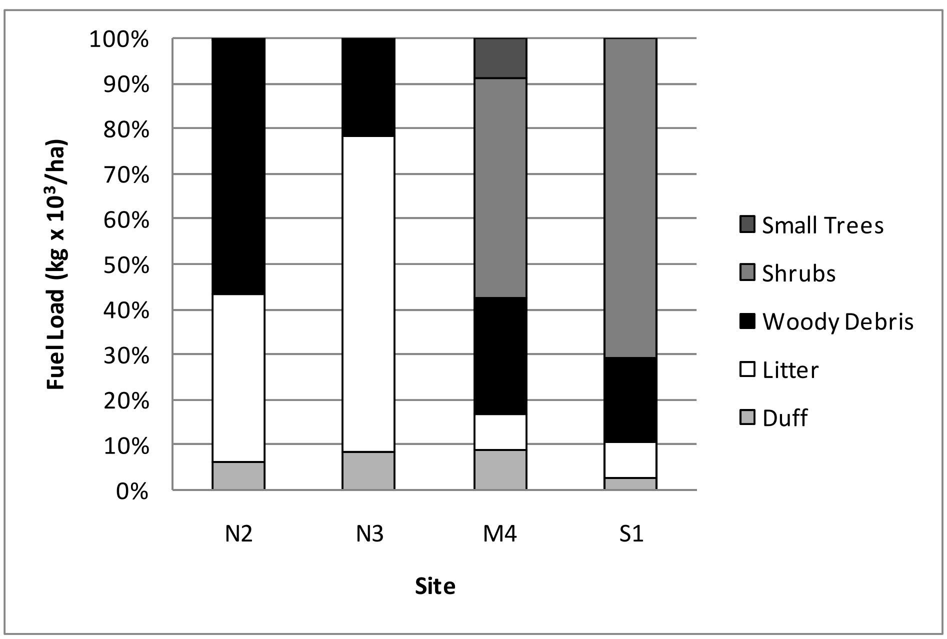


Fig. 31. Relative contribution of each fuel component to total fuel load.

important component of the fuel load in untreated forests, but this species was not prevalent at these wildfire sites.

The vast majority of litter and TWD at sites N2 and N3 originated from the exotic shrubs mitigated by site treatments. Thus, our results reveal an incongruity between the term "fuels reduction" as commonly applied to such treatments and the actual outcome of the treatment. At sites where exotics clearing follows the treatment prescriptions described here, understory fuels components are actually retained at the treated sites and are simply subjected to a transformation of structure and distributional characteristics.

Because fuel loading is strongly dictated by understory structure and composition, growing conditions in that stratum were hypothesized to play a role in influencing the amount of fuel present, secondary to shade-induced depression of undersotry biomass production. Figure 33 provides support for this hypothesis, showing a very tight negative relationship (R^2 = 0.99; p = 0.003) between *Populus* density and fuel loading in the four sites sampled regardless of treatment.

Effects of Fuel Loading on Populus Bole Mortality and Vigor

Populus has the ability to sprout from roots, thus the entire above ground portion of the tree can be killed and yet the individual may survive, persisting in its underground structures, and resprout. In this section, our investigations focused on mortality of the bole and crown. This is because when resprouts emerge it is often difficult to ascertain which tree they are linked to. Moreover, management goals in the region are focused on preserving the existing above ground biomass which takes many years to develop and which is obviously important to maintaining ecological integrity, as well as imbuing the forests with their societally-desirable characteristics. It is also not clear how widespread bole death may affect long-term forest viability and ecological dynamics. Later, we consider the frequency of resprouting, however, no attempt is made to determine absolute mortality of individual trees.

Wildfires had an obvious negative effect on the cottonwoods of the MRG bosques. Mortality was up to 83 percent greater in wildfire sites than in unburned reference stands, and the difference in mortality was highly statistically significant ($p \ll 0.01$) (Fig. 34). Similarly, considering trees that were not killed outright, the mean live canopy volume was significantly lower at three of the four burned sites (Fig. 35). Interestingly, at site S4, the remaining live trees in the burned stand maintained canopy volumes indistinguishable from those in the reference stand.

There was a strong positive correlation between total fuel load and bole mortality, with an R^2 of 0.98 (p = 0.01) (Fig. 36), however, there were no significant correlations between the individual components of the fuel load and bole mortality (Table 6). This was probably a result of the disparate patterns of fuel component loading observed in the treated and untreated sites. There was no evidence that removal of the exotic understory as executed at our study sites increased bole survivorship as indicated by the interspersion of treated and untreated sites along the regression line in Fig. 36.

Fig. 32. View of a stand following exotics cutting and chipping, showing the uneven and often deep cover of chips produced by the operation. The inset picture shows an excavation into a 20 cm deep chip layer.

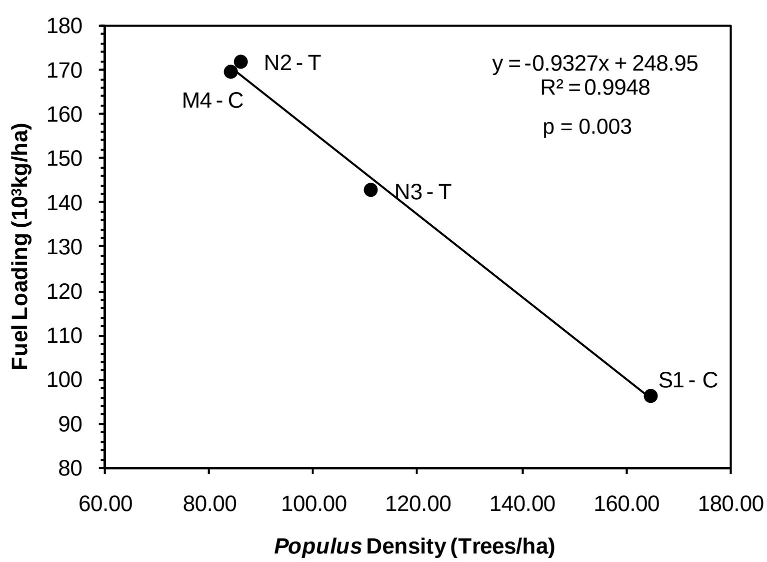


Fig. 33. Relationship between *Populus* density and total fuel loading. Points are labeled with site codes followed by a "T" if the site had been subjected to exotics clearing, and a "C" if it was untreated.

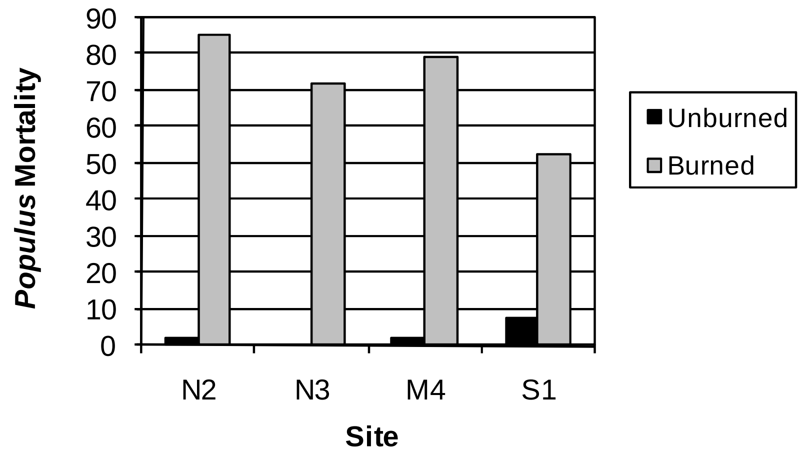


Fig. 34. Comparison of *Populus* mortality in burned and unburned reference sites.

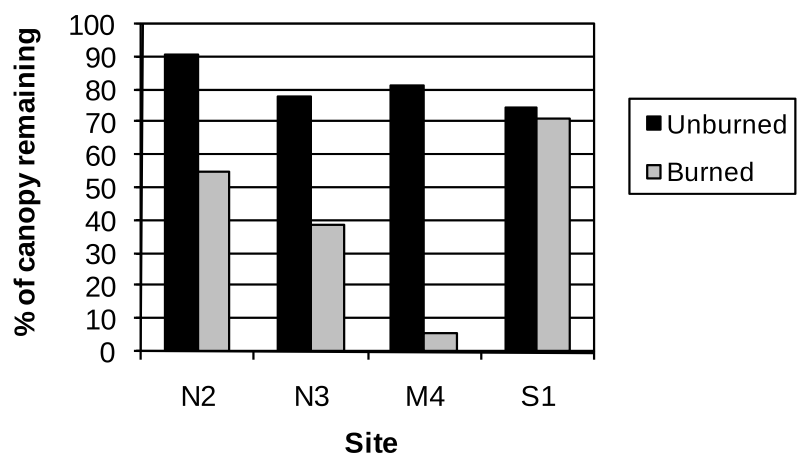


Fig. 35. Comparison of *Populus* canopy vigor in burned and unburned reference sites.

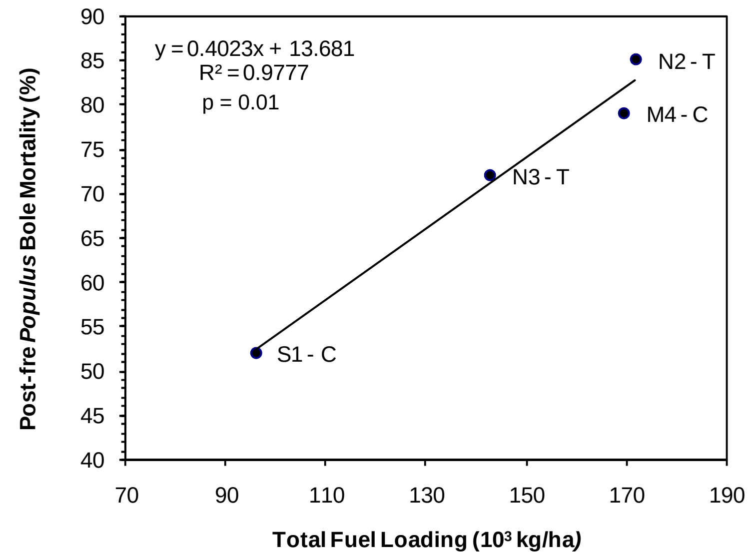


Fig. 36. Relationship between total fuel loading and *Populus* bole mortality. Points are labeled with site codes followed by a "T" if the site had been subjected to exotics clearing, and a "C" if it was untreated.

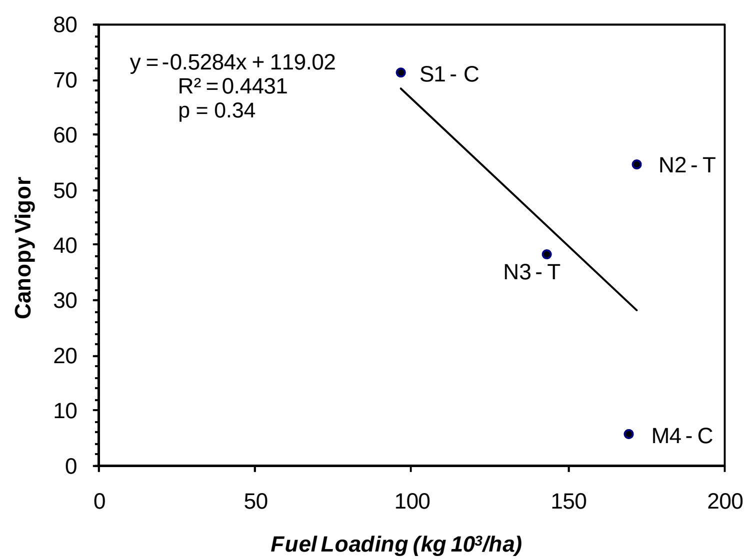


Fig. 37. Relationship between total fuel loading and vigor as expressed by the percentage of the total potential canopy that was alive at the time of sampling. Points are labeled with site codes followed by a "T" if the site had been subjected to exotics clearing, and a "C" if it was untreated.

Table 6. Correlations and significance values for regressions of fuel loading vs. Populus bole mortality.

Fuel Component	R^2	p
Woody Debris	0.65	0.19
Duff	0.70	0.16
Shrubs	0.17	0.59
Litter	0.17	0.59
Small Trees	0.11	0.66
Total Fuel Load	0.97	0.01**

Similar patterns were observed when canopy vigor was regressed against fuel loading and its components (Table 7). In this case, no significant correlations were found, although high R^2 values were calculated for duff, small trees and total fuel load. While not statistically significant, the relationship between total fuel load and canopy vigor is nonetheless compelling (Fig. 37). Three of the sites display a tight relationship between fuel load and vigor, with N2 being an outlier. When N2 is removed from the analysis, R^2 rises to 0.97, and the correlation is marginally significant ($p = 0.09$). It is unclear why vigor in N2 is higher than predicted by the relationship exhibited by the other three sites; however, it is not difficult to speculate that the individual characteristics of that fire spared the crowns of more trees than at the other sites. The relatively large size of the trees (Fig. 26) may also have made them more resistant to fire damage. Regardless of the above relationships (or lack thereof) between canopy vigor and fuel, again the data suggest that at these sites clearing of exotic understory species as presently practiced did not have a positive effect on cottonwood survivorship.

Table 7. Correlations and significance values for regressions of fuel loading vs. Populus canopy vigor.

Fuel Component	R^2	P
Woody Debris	0.0	0.99
Duff	0.79	0.11
Shrubs	0.09	0.70
Litter	0.001	0.97
Small Trees	0.77	0.12
Total Fuel Load	0.44	0.33

Resprouting Patterns

Resprouting density varied, from a low of 150 resprouts/ha at site N3, to 1762 resprouts/ha at site M4 (Fig. 38). The density at M4 was more than four times that of the next densest stand. Across all sites tamarisk was the most frequently sampled resprouting species, with essentially double the frequency of the next most common resprout species, *Populus* (Fig. 39 and 40). At individual sites tamarisk was the dominant resprouting species, except at N3 where *Populus* more than doubled its frequency (Fig. 41).

Based on these data it appears that exotics clearing may have a beneficial effect on forest regeneration. At the two treated sites, N2 and N3, tamarisk frequency was only 35 and 20 percent, respectively, compared to the 66 and 81 percent relative frequency measured at the untreated sites (Fig. 41). In this case, the apparent suppression of tamarisk regeneration may have occurred owing to the relatively rapid succession of severe stressors to which they were exposed (i.e., cutting, chemical treatment, then wildfire). Species diversity at treated sites was also as high, or higher, than at the

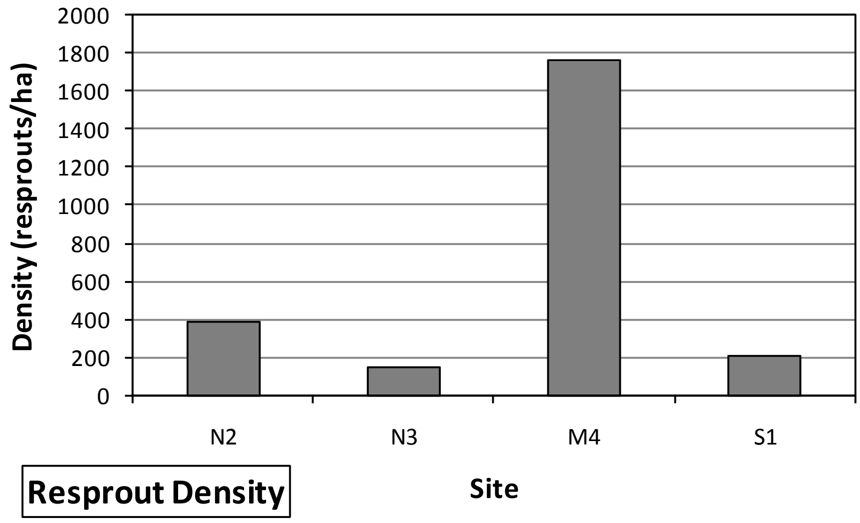
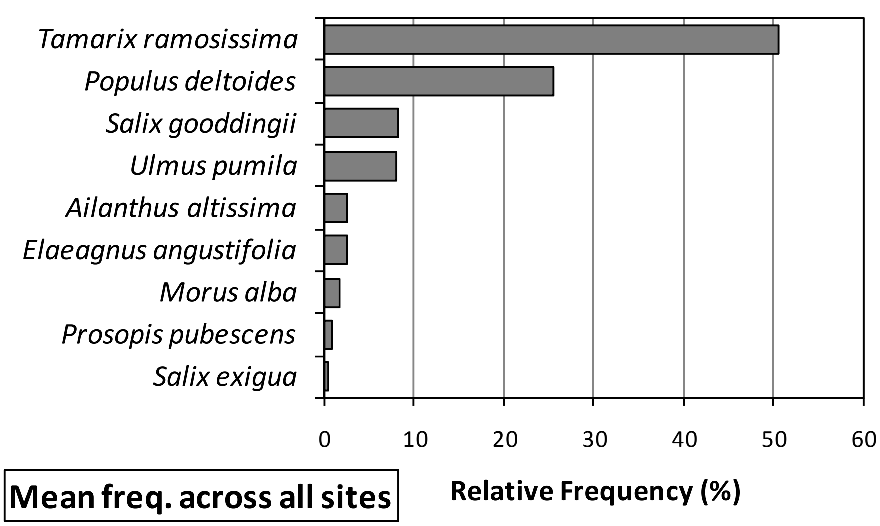


Fig. 38 (Top). Total density of resprouts at each site.

Fig. 39. (Middle). Tarmarisk resprouting after a burn.

Fig. 40 (Bottom). The relative frequency of resprout species averaged across all wildfire sites.



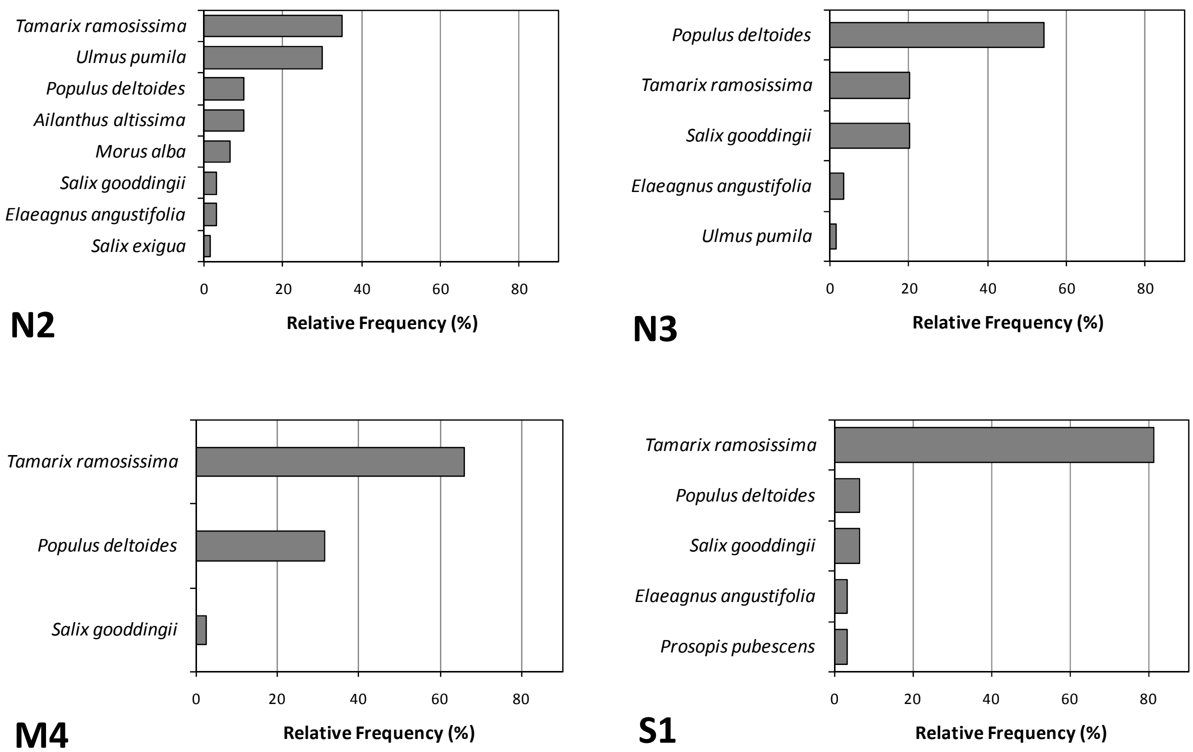


Fig. 41. Relative frequency of resprouting species measured at the four wildfire sites, using the point-centered quarter method.

untreated sites, with a higher density of resprouting native species. These patterns may have been the result of competitive release following removal of the overwhelming canopy of tamarisk, which allowed other woody species to become robust enough prior to the fire to survive it. The observed patterns of species diversity may also be an artifact of ambient site characteristics, since the two treated sites were located near one another at the northern extreme of the study region.

Discussion

It is well known, through both anecdotal observations (Figs. 6 and 7) and existing studies, that wildfires can have devastating effects on cottonwood bosques of the MRG. Once an intense wildfire burns through a bosque, under the existing physio-hydrological conditions, the site may be unable to regenerate a cottonwood canopy with any resemblance to what was lost.

Because ambient fuel load and composition is well known to influence fire intensity (Agee and Skinner 2005), and consequently mortality, programs aimed at protecting the remaining cottonwood bosques direct sizeable funds towards “fuel reduction programs”, wherein exotic canopy and sub-canopy woody species are cut and processed on site. Although applied in various forms throughout the Southwest as well as across the US, information on the actual effectiveness of such treatments is sparing (Wimberly et al. 2009). Wisely, the outcomes and effects of fuels reduction are now being studied by a variety of investigators and institutions in large in response to the Salt Cedar and Russian Olive Control Demonstration Act of 2005. One of the largest efforts spawned by the Act in the Southwest US involves the multi-agency collaborative study headed by the U.S.D.A. Forest Service Rocky Mountain Research Station that was described earlier.

In step with many of the current recommendations for carrying out demonstration projects for the Act, the RMRS ecosystem study is a broad-based one focused mainly on determining the multi-faceted ramifications of exotics clearing for plant and animal populations (Bateman et al. 2008a,b,c, Finch and Hawksworth 2006, Chung-MacCoubrey and Bateman 2006, Bateman et al. 2009, Chung-MacCoubrey et al. 2009, Bateman et al. 2010, Johnson et al. 2010). One key question which is scarcely addressed, however, is the relative effectiveness of exotics clearing in actually preserving the cottonwood canopy from fire. This lack of information stems, of course, first from the obvious difficulties posed in studying naturally stochastic wildfires, and secondly from the difficulty in determining pre-fire fuel load.

In the case of the current study, a three-year investigation of wildfire effects was initiated to evaluate burns in cottonwood bosques along the study reach of the MRG. Our approach was reference based, essentially employing a time-for-space substitution (Whittaker 1975) to characterize fuel loading and composition in structurally similar, adjacent stands and from that infer the pre-fire fuel conditions.

This approach to reconstructing pre-burn fuel loading based on nearby conditions appears to have been fruitful. The assumptions behind the strategy are two. The first is that fuel loading was comparable between the burned stand and its unburned reference stand. The second is that fuel loading did not change significantly between the time of the wildfire and the time of sampling.

The first assumption appears valid in light of the tight negative correlation between cottonwood density and fuel loading. Since the vast majority of fuel load is derived from the understory woody components, this relationship is consistent with the commonly documented inverse relationship between understory and canopy cover, and the one reported by Johnson et al. (2010) at these same sites. The method also showed the positive relationship between fuel load and mortality, which is essentially dogmatic in its level of acceptance. Moreover, analyses show a high degree of similarity between burned and reference sites based on cottonwood stand characteristics (Figs. 26 – 29), which suggests that canopy-controlled fuel loading was likely similar between burned and reference areas.

The assumption regarding the equivalency of fuel load between the time of the fire and that of sampling also appears reasonable. This is attributed to the fact that the majority of the fuel load at all sites was derived from relatively recalcitrant material, such as woody debris, chips and sub-canopy woody plants. Since sites were sampled within approximately one year of the wildfire we would not expect the volume of these materials to change dramatically within the study window.

The circumstances of this study, in which the only available study sites meeting selection criteria were part of a large scale study of the outcomes of exotics clearing, provided a rare and serendipitous opportunity to evaluate the efficacy of such treatments in preserving existing cottonwood bosques. This situation also allowed us to utilize data on shrub coverage that was collected at burn sites prior to the wildfire to bolster the accuracy of fuel loading estimations. Although we were restricted by chance as to the total number of sites that could be included and sample size is consequently modest, data reconstruct well established relationships in forest fire dynamics, such as between fuel loading and mortality. Thus, despite the limited number of sites, we suggest that the patterns exhibited likely portray a reliable representation of typical processes occurring in the MRG's cottonwood bosques.

Our data reconstruct the expected relationships between fuel load and bole mortality and canopy vigor. While it is important to demonstrate that such dynamics occur in the MRG's bosques as in other forested ecosystems and the fit with well-founded theory is an important gauge of data quality, the real value of these findings is in what they reveal about the efficacy of commonly applied measures to reduce fuel loading and mitigated the destructive force of wildfires.

Our data and analyses lead us to two primary conclusions. First, unexpectedly, our study found no evidence that treatments improved cottonwood survivorship or individual vigor within the first few years after exotics clearing (Figs. 36 and 37). Secondly, our data illustrate that approaches to exotics clearing that are commonly billed as "fuels ***reduction*** treatments", do not actually reduce the amount of available fuel (Fig. 30). Rather they are fuels ***transformation*** treatments in which vertically-structured biomass in the form of shrubs and small trees is converted into horizontally-concentrated layer of debris. The relationship between fuel load and cottonwood mortality coupled with the lack of a detectable treatment effect, suggest that the critical issue in cottonwood survival of wildfires is not the form of fuel, but rather the sheer amount available to burn.

We suggest that this conclusion can be explained by considering the way in which different fuel compositions burn and the mechanisms of tree death. In untreated sites, an abundance of fuel in the form of shrubs and small trees allows flames to ladder-up and enter into the cottonwood canopy which causes significant mortality and crown loss. Clearing of exotics should greatly reduce the chance of crown fire owing to removal of the flame-laddering shrub canopy. However, observations show that chipped and cut material is still consumed by wildfires and consequently serve as fuel (Fig. 6). As explained earlier, in the clearing process, chips are unevenly spread across sites (Fig. 32), but very commonly those materials are distributed around and in contact with the base of tree trunks. Similarly, cut materials are commonly stacked in close proximity to trees and the trunk may even be used to support stockpiles (Figs. 42 - 43). We posit that when the biomass of shrubs and small trees sites is left on-site in this cut and masticated form the primary mechanism of bole mortality shifts from the more readily discernable crown fire to girdling (Figs. 44 - 45).

Our hypothesis is that chips or logs, which are a concentrated source of fuel, burn for an extended duration and may create hot coals in direct contact with the trunk. When these materials are present in sufficient quantities, they contain enough energy to eventually burn through the cottonwood's protective bark layer and destroy the integrity of the cambium ring. When girdling is complete the tree bole has no opportunity for survival. In less severe circumstances, partial disruption of the cambium would result in the loss of the limbs and a reduction of canopy volume.

Regardless of the precise mechanism of death or decline, we conclude that exotic woody species very likely create unnaturally high fuel loads and thereby pose a significant threat to the ecological integrity of the MRG's bosques, maintaining the fuels on-site following treatment is not an effective means of increasing survivorship and vigor following fire. Haphazard disposal of the biomass generated by clearing to a large degree negates one of the primary objectives of exotics clearing efforts. If management programs truly seek to protect and maintain the prized gallery canopy of MRG bosques it is imperative that clearing efforts actually reduce the amount of fuel present on a site, or otherwise manage the bi-products of clearing, not simply transform it to concentrated form on the forest floor. This necessity poses a difficult challenge in terms of economics and feasibility, nonetheless it must be tackled head on else vast effort, funds and the forests themselves stand to be lost.

The magnitude of the biomass management issue was not lost on the National Committee on Energy and Natural Resources who drafted the Salt Cedar and Russian Olive Control Demonstration Act of 2005, who noted on p. 2:

> One important challenge to controlling the invasion of salt cedar and Russian olive has been dealing with the biomass of the vegetation that is removed. Finding economical means of using or disposing of this biomass is another key to formulating an effective regional strategy for dealing with the infestation.

If effective approaches to biomass management are not developed and implemented in conjunction with clearing, continued loss of these irreplaceable ecosystems will certainly be the price.

Fig. 42 (above). Woody exotic biomass stockpiled as logs piled amongst and against cottonwood boles.

Fig. 43 (below). Slash and logs resulting from clearing activities piled around cottonwood.

Fig. 44 (above). Cottonwood stems killed by girdling.

Fig. 45 (below). Site N2 several months post wildfire. Notice the distribution of fire scarring which is concentrated at the base of trunks. Many of these boles were killed despite the lack of crown fire secondary to girdling.

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