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EFFECT OF HIGH-INTENSITY DIRECTED FIRE IN DIFFERENT SEASONS ON SURVIVAL AND SPROUTING OF ROYAL PAULOWNIA (*PAULOWNIA TOMENTOSA* (THUNB.) STEUD.)

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Abstract

Invasive plant species are widely recognized as a serious environmental threat. They can cause local extinction or a vast decline in some native species populations by increasing competition and through modifications of their habitat. The currently used control measures commonly involve herbicides that can be toxic to other plants, people, and wildlife. Some of the undesired side effects include reduction of nutrients to non-target species, mortality in tadpoles, deformation of fish, and reduced fertility and sexual development in frogs. There is a need for new methods for invasive species removal that do not use herbicides, but whose efficiency is similar to that of herbicide-based techniques. In this study we tested the effectiveness of high-intensity fire directed at the base of the stem. We used external fuel source and applied the fire to royal paulownia trees to examine the effects of tree diameter, length of time of fire application, and season of application. Overall, greatest mortality occurred when the studied treatments were applied in the summer and when the exposure to fire continued for 30 s in diameters from 3.8 cm to 20.1 cm and for 60 s in diameters from 20.2 cm to 68.6 cm. The season of burning had no effect on sprouting. Post-burn sprouting was greater following longer fire exposure. Additional treatment of the sprouts may be necessary using this or other methods. Nevertheless, the percent of trees that sprouted was relatively low. The tested treatment can be scaled up and its cost decreased by employing already available flaming equipment currently used in agriculture.

Key words: invasive**,** *Paulownia tomentosa*, mortality, sprouting, burn.

Introduction

Most non-native species are often brought into new habitats for forage or for ornamental purposes, while others are brought accidently (Miller 2003). Some non-native species can become invasive in the absence of their main natural predators. This results in greater survival of the invasive species than the indigenous species. Non-native invasive woody species are of growing concern because of the negative impact they have on ecosystems including hindering forest use and management, and reduction of biodiversity and wildlife habitat (Burton et al. 2005). Control of non-native species is often needed in order to restore indigenous species.

Invasive species rank second, after habitat destruction, as a threat to biodiversity of imperiled groups of plants, mammals, invertebrates, fish, reptiles, and amphibians (Wilcove et al. 1998). Other threats include overharvest, pollution, and diseases. Not only do invasive species pose a threat to biodiversity and increase competition, but they are also expensive to control once established. The detrimental effects from invasive species are estimated to cost \$138 billion·year^{-1} in the United States (Pimentel et al. 1999). There is no direct way to assign monetary value to all the negative effects such as extinctions, losses in biodiversity, ecosystem services, and aesthetics, so the above cost estimate is rather conservative.

There are numerous management techniques for control of unwanted woody species which include herbicide use, mechanical control, prescribed fire, grazing, biological control, and soil solarization (Green and Newell 1982, Hartman and McCarthy 2004, Miller 2003, Tu et al. 2001). Each control method has advantages, but also disadvantages, e.g., there may be risks for damage to non-target species, or the method may be time consuming, labor intensive, or expensive to apply. Herbicide use, prescribed fire, and mechanical control are among the most common control methods.

The average yearly amounts of herbicide used in forestry over a period of a rotation are much smaller than in agriculture. Nevertheless, some herbicides have a number of negative effects on other plants, humans, and wildlife (Neumann et al. 2006, Acquavella et al. 2004). Herbicide application is also less desirable in valuable natural areas (Groves 1989).

Some of the commonly used herbicides in forestry have been glyphosate formulations commercialized as Roundup® by the Monsanto Company. Next we examine some of the issues with this widely used and relatively well studied herbicide.

The active ingredients of Roundup® have been detected in urine of applicators whether or not prevention was taken to minimize herbicidal contact (Acquavella et al. 2004). Glyphosate is also frequently found in non-target plant species after application (Neumann et al. 2006). The glyphosate overspray can reach the rhizosphere where it can reduce a non-target species ability to adequately absorb nutrients. In addition, glyphosate can be harmful to microorganisms and animals as well (Neumann et al. 2006).

Evidence for toxicity of the Roundup® formulations has been accumulating in the literature for over a decade. The toxicity of the active ingredient and especially of the inactive ingredients, adjuvants, and surfactants are now well known and documented (Peluso et al. 1998). Research demonstrated that Roundup® induced a dose-dependent formation of DNA adducts (a complex that forms when a chemical binds to the DNA) in the kidneys and liver of mice (Peluso et al. 1998). The formation of such adducts in the organs is considered damage to the DNA. The DNA adducts were related to some unknown compound in the mixture rather than to the active ingredient, isopropylammonium salt of glyphosate.

A study on a coastal microbial community showed that Roundup® modifies natural coastal microbial communities of prokaryotes and some pico-eukaryotes after a 7-day exposure (Stachowski-Haberkorn et al. 2008). Relyea (2005) showed that Roundup® and its surfactant Polyoxiethyleneamine (POEA) affect amphibians causing high mortality rates. Within three weeks, 98 % of tadpoles were killed, and within 24 hours 79 % of juvenile frogs and toads were killed. Roundup® starts to affect young and adult tilapia after approximately 96 hours of exposure (Jiraungkoorskul et al. 2002). The fish gills were found to have filament cell proliferation (a chain-like series of cells increasing at a fast rate), lamellar cell hyperplasia (abnormal increase of cells in the scale layer), and aneurysm. Kidney lesions consisted of dilation of Bowman's space and buildup of hyaline droplets in the tubular epithelial cells. This study showed clearly that the presence of Roundup® causes degenerative formation of organs in fish.

Roundup® also disrupts the expression of the steroidogenic acute regulatory (StAR) protein, inhibits steroidogenesis, and causes decrease in progesterone production (Walsh et al. 2000). Progesterone is a hormone involved in pregnancy (by supporting gestation) and in embryogenesis of animal species and humans. Therefore, Roundup® can impact fertility in wildlife and humans. Work by Paganelli et al. (2010) demonstrated that Glyphosate and Glyphosate based herbicides (GBH) impair the mechanisms needed for regulating early development in frog and chicken embryos when exposed at sub lethal levels, which leads to concerns about possible similar effects on human embryos.

Glyphosate is also toxic to human placental JEG3 cells and this toxicity was observed less than 18 hours after treatment (Richard et al. 2005). More importantly, this toxicity occurred at concentrations lower than the concentrations found in agricultural use. The observed toxic effect went up with the increases in concentration as well as in the presence of Roundup adjuvants.

The known negative effects for glyphosate-based herbicides listed above and the potential detrimental effects of less extensively studied herbicides create a need to consider other methods for control of invasive plants. Two widely used methods that do not rely of chemical applications are prescribed fire and mechanical control.

Some of the major disadvantages of implementing a prescribed burn are the impact on air quality and visibility, as well as possible lack of, or excess of, fuel on the forest floor. When fire has been suppressed for long and the fuel buildup is very large, a prescribed fire will burn too intensely and cause mortality in desired species (Stephens and Ruth 2005). It can indiscriminately kill non-target native species even at low intensity and often stimulates vigorous sprouting from the invasive species (Heffernan 1998). Prescribed burning requires a crew of qualified personnel, expensive equipment, construction of fire breaks, and can pose risk to property and human life, which raises liability and public concern (McCaffrey 2006).

Mechanical control methods are expensive, generally ineffective (Tu et al. 2001) and often disturb the soil (Evans et al. 2006). These methods of control have many advantages, but are often time consuming and not as effective when used without additional measures of control, such as herbicide application or burning (Miller 2003).

We explored a new and simple technique that poses minimal risk to the environment and may be effective at controlling the spread of invasive woody species. We examined the effectiveness of high-intensity fire directed at the base of the stem to kill and prevent stump resprouting. Unlike prescribed burning, this method requires one worker instead of a full crew, does not require the use of expensive fire control equipment, and can be applied anytime when there is no risk of starting a wildfire (i.e., during or soon after rain, snow, or when air humidity and fuel moisture are high). We used the method with the non-native (to North America) invasive plant species royal paulownia (P*aulownia tomentosa* (Thunb.) Sieb. & Zucc. ex Steud.). We investigated the effectiveness of the treatments when they are applied in different seasons (winter, spring, and summer), and we compared the effectiveness of the treatments for different lengths of exposure.

Materials and Methods

Species of study

Royal paulownia is a common invasive tree species in the eastern United States (Miller 2003). It is native of East Asia and belongs to the Scrophulariaceae family. It is fast growing and reaches heights of over 25 meters and diameters of over 70 cm. It is capable of growing up to 5.0 meters in one year, and reproduces from seeds, root sprouts (Bartlow et al. 1997), or stump sprouts (Miller 2003). Paulownia can invade a variety of different habitats including roadsides, cliffs, riparian areas, open woodlands, highway embankments, stream banks, forest edges, landslides, burned-over areas, rocky-outcroppings, mine spoils, old home sites, and other disturbed sites (Evans et al. 2006).

Royal paulownia was ranked as the twelfth most problematic invasive species by the National Environmental Research Park (Drake et al. 2003). Paulownia infestations occur in scattered and localized areas in managed forests, as well as in natural areas and parks (Alabama ... 2007). It is an undesirable species in forest management, it causes difficulty for establishing fire adapted species like Table Mountain pine (*Pinus pungens* Lamb.), because of paulownia's ability to quickly establish itself after a disturbance such as fire (Evans et al. 2006).

The species is widespread in part because it produces an abundance of seeds (Drake et al. 2003), estimated to be up to 20 million per plant (Remaley 2005), which are small, lightweight, and easily spread by wind or water. The seeds germinate quickly when in contact with mineral soil (Remaley 2005) and seeds remain viable in the soil for many years (McCarthy 2005). Paulownia's continued use as an ornamental, for wood production (Miller 2003), and mined land reclamation (Zhao-hua et al. 1986) has increased its opportunity to spread and outcompete native species.

Control methods used for paulownia include mostly herbicidal control using Arsenal AC® or a formulation of glyphosate applied by stem injection into the main stem of mature trees (Miller 2003). Prescribed fire has not been successful in the removal of this species (Evans et al. 2006).

Study site

The study site was at the Winfred Thomas Agricultural Research Station in Hazel Green, Alabama (34°53’51’’N, 86°34’25’’W). The elevation is 236 m. The paulownia trees were planted approximately 20 years ago as part of an agroforestry research project. The trees were planted in rows along with Tulip poplar (*Liriodendron tulipifera* L.), Red oak (*Quercus rubra* L.), and Pecan (*Carya illinoinensis* (Wangenh.) K. Koch).

Table 1. Paulownia treatment design.

Season and year of burn, replications	Diameter class, cm	Burn time, s
Control n=10	DBH ≤ 20.1	None
Control n=13	DBH ≥ 20.2	None
Winter 2010		
n=21	DBH ≤ 20.1	15
n=22	DBH ≤ 20.1	30
n=23	DBH ≥ 20.2	40
n=25	DBH ≥ 20.2	60
Spring 2010		
n=21	DBH ≤ 20.1	15
n=17	DBH ≤ 20.1	30
n=23	DBH ≥ 20.2	40
n=23	DBH ≥ 20.2	60
Summer 2010		
n=19	DBH ≤ 20.1	15
n=17	DBH ≤ 20.1	30
n=23	DBH ≥ 20.2	40
n=24	DBH ≥ 20.2	60

Experimental design and sampling

Two hundred fifty-eight royal paulownia trees were treated, while 23 trees served as the control. The 258 trees and the control trees were divided prior to treatment into two diameter classes (diameter at breast height (dbh) of 1.30 m above ground): 1) dbh between 3.8 and 20.1 cm and 2) dbh between 20.2 and 68.6 cm. Ten trees from the dbh ≤ 20.1 cm size class and 13 trees from the dbh ≥ 20.2 cm size class were randomly selected as controls and were left unburned (Table 1). In both diameter classes the trees were randomly selected to be burned during winter, spring, or summer. In the dbh ≤ 20.1 cm size class, 61 trees were randomly selected to be burned for 15 seconds and 56 trees for 30 seconds. In the dbh ≥ 20.2 cm size class, 69 trees were randomly selected to be burned for 40 seconds and 72 trees for 60 seconds. All trees were flagged and tagged with numbers for future identification. The fire was powered using a torch (Red Dragon, model VT 3-30 C 527.5 MJ·h^{-1}) and a 7 kg propane tank. The burn time was recorded using a stopwatch. Treatments were applied in the winter (February and March) of 2010, spring (April and May) 2010, and summer (August) 2010. Mortality was examined and documented in the summer of 2011. All stump sprouts from the winter, spring, and summer 2010 burn were recorded in August of 2010, and examined again one year later.

Hypothesis and statistical analysis

Chi-square test was used to determine if there was a difference in mortality and sprouting among the seasons of treatments for each length of burning. It was also used to test the differences in mortality and sprouting among each treatment, including the control, regardless of season. We employed Bonferroni correction for multiple comparisons. Fisher's Exact Test was used instead of Chi-square in instances when cells had a count of less than five. We considered results to be significant if $p<0.1$. The experimental design was a split plot design.

Results and Discussion

The mortality data that we collected in August of 2011, which was at least one full year after completion of burning in winter, spring, and summer of 2010, indicated that not all treatments were able to cause mortality in the experimental trees over the period of time that we examined (Table 2). There was no tree mortality among the trees from the control treatment, or among the trees from the small diameter class that were burned for 15 s, regardless of the season of burn (Table 2). Mortality did occur, however, as a result of the longer burn of 30 s in the small diameter class, as well as 40 s and 60 s in the larger diameter class. Greater mortality occurred in the small diameter trees after 30 s burn and in the large diameter trees after 60 s burn than in the control (p=0.035, p=0.005, respectively, Table 3).

However, the percent mortality after burning trees in the large diameter class for 40 s was not different from the percent mortality in the control (p=0.333, Table 3). When we combined the mortality data from all seasons, we found as expected that for the small diameter trees there was higher mortality after burning for 30 s than for 15 s, and for the large diameter trees there was higher mortality after burning for 60 s than 40 s (p<0.001 in both size cases, Table 3, Figure 1 and 2). The mortality after 30 s treatment was an average of 25 % and after 60 s treatment was an average of 31.9% (Table 3).

There was no difference in mortality

Table 2. Percent tree mortality ± standard error in August of 2011 for treatments applied in winter, spring, and summer of 2010.

Burn length, s for two size classes	Season of burn			Average
	Winter	Spring	Summer	
	Mortality, % ± standard error			
Small Diameter Class				
0	0.0±0.0	0.0±0.0	0.0±0.0	0.0
15	0.0±0.0	0.0±0.0	0.0±0.0	0.0
30	13.6±7.5	17.7±9.5	47.1±12.5	25.0
Large Diameter Class				
0	0.0±0.0	0.0±0.0	0.0±0.0	0.0
40	0.0±0.0	0.0±0.0	17.4±8.1	5.8
60	4.0±4.0	26.1±9.4	66.7±9.8	31.9

Table 3. Differences in the percent mortality of paulownia trees following burning the main stem for different lengths of time.

Pairwise treatment comparisons	Difference in mortality, %	*P*-value
Small Diameter Class		
Control vs. 15 s	0.0	N/A
*Control vs. 30 s	–25.0	0.035
15 s vs. 30 s	–25.0	<0.001
Large Diameter Class		
*Control vs. 40 s	–5.8	0.333
*Control vs. 60 s	–31.9	0.005
40 s vs. 60 s	–26.1	<0.001

* Indicates Fisher's Exact Test analysis. In all other comparisons Chi-square tests were used. We used Bonferroni correction for multiple comparisons.

between trees burned in the winter for 15 s and those burned for 30 s (p=0.233, Table 4). However, when the burn was carried out in the spring or summer, the 30 s burn resulted in greater mortality than the 15 s burn (p=0.081, p<0.001, respectively, Table 4). We observed similar results for the large diameter class, where burning for 40 s or 60 s in the winter resulted in the same percent mortality (p=1.000), but burning in the spring and summer resulted in greater mortality of the trees burned for 60 s than those burned for 40 s (p=0.022 and p=0.001, respectively, Table 4).

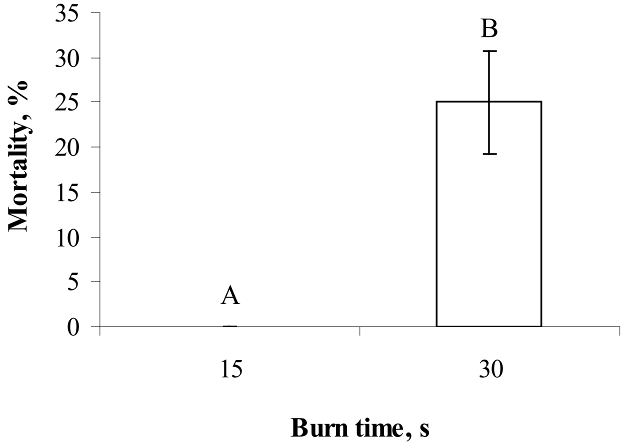

Fig. 1. Paulownia mean percent mortality and standard error in the small diameter class after a 15 s and 30 s burn. Different letters indicate significant difference at the p<0.1 level.

None of the trees died if they were burned for 15 s, regardless of the season of burn, so we did not look for a season effect (Table 5). Similarly, we did not compare the season effect on mortality of the trees burned for 40 s in the winter and spring, because no mortality was observed. There was no difference in percent mortality after burning for 30 s in the winter versus the spring (p=0.333, Table 5). However, there was a difference in percent tree mortality in each season for each of the other burn lengths. For the 60 s burn lengths, mortality after spring burn was greater than after winter burn. After the 40 s and 60 s burn lengths, mortality after summer burn was greater than after spring burn, and mortality after summer burn was greater than after winter burn.

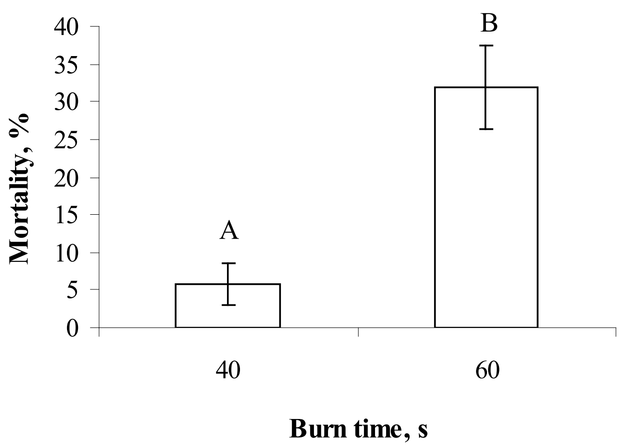

Fig. 2. Paulownia mean percent mortality and standard error in the large diameter class after a 40 s and 60 s burn. Different letters indicate significant difference at the p<0.1 level.

Table 4. Differences in the percent mortality of paulownia trees following burning the main stem for different lengths of time.

Size and season of treatment	Pairwise treatment comparisons	Difference in mortality, %	*P*-value
Small Diameter Class			
*Winter	15 s vs. 30 s	–13.6	0.233
*Spring	15 s vs. 30 s	–17.7	0.081
*Summer	15 s vs. 30 s	–47.1	<0.001
Large Diameter Class			
*Winter	40 s vs. 60 s	–4.0	1.000
*Spring	40 s vs. 60 s	–26.1	0.022
Summer	40 s vs. 60 s	–49.3	0.001

*Indicates Fisher's Exact Test analysis. In all other comparisons Chi-square tests was used.

Table 5. Differences in tree mortality following burning (comparison of the effect of season of burn).

Size and burn time, s	Pairwise season comparison	Difference in mortality, %	*P*-value
Small Diameter Class			
15	Winter vs. Spring	0.0	N/A
15	Winter vs .Summer	0.0	N/A
15	Spring vs. Summer	0.0	N/A
*30	Winter vs. Spring	–4.1	0.333
*30	Winter vs. Summer	–33.5	0.011
30	Spring vs. Summer	–29.4	0.022
Large Diameter Class			
40	Winter vs. Spring	0.0	N/A
*40	Winter vs. Summer	–17.4	0.036
*40	Spring vs. Summer	–17.4	0.036
*60	Winter vs. Spring	–22.1	0.015
60	Winter vs. Summer	–62.7	<0.001
60	Spring vs. Summer	–40.6	0.005

* Indicates Fisher's Exact Test analysis. In all other comparisons Chi-square tests was used. We used Bonferroni correction for multiple comparisons.

The results were similar with the 30 s burn, but with the exception that mortality after the winter burn was not different from mortality following the spring burn.

Stump sprouting observed in August, 2011, which was a minimum of one full year after burning that took place in winter, spring, and summer of 2010, showed that sprouting occurred after each of the burns, but no sprouting was observed in the control (Table 6). Sprouting after burning for 40 s and 60 s was more common than in the control trees (p=0.067, p=0.011, respectively), but no difference was found between the percent of trees with new sprouts in the control and the 15 s burn or the control and 30 s burn (p=0.333, p=0.114, respectively, Table 7). The greatest percent of trees with stump sprouts, 29.6 %, occurred after the 60 s burn.

There was no difference in percent of trees with sprouts after burning for 15 s and for 30 s during the winter or summer (p=0.664, p=0.324, respectively, Table 8). However, greater sprouting occurred after the spring 30 s burn than the spring 15 s burn (p=0.081, Table 8). There was no difference in percent of trees with stump sprouts after burning for 40 s or 60 s during the spring (p=0.153, Table 8). The same was observed after burning during summer as well (p=0.956, Table 8). Differences in percent trees with sprouts were found in the 40 s treatment and 60 s treatment during winter (p=0.047, Table 8).

No differences were found in sprouting occurrence when the main stem was burned for 30 s between the following: winter and spring, winter and summer, and spring and summer (p=0.212, p=0.212, p=0.333, respectively; Table 9, Figure 3). Similar lack of differences between seasons was found in the results for 60 s burns (Table 9, Figure 4). The same sprouting average of 30.4 % was observed after the winter and spring 60 s burn.

The 60 s summer burn caused, on average, 28 % of the trees to sprout (Table 6). Similarly, seasonal differences were not found in comparisons of sprouting after winter and summer burn for 15 s, spring and summer burn for 15 s, and winter and spring burn for 40 s (p=0.202, p=0.158, p=0.203, respectively; Table 9). Seasonal differences were observed in comparisons of sprouting after spring and winter burn for 15 s, winter and summer burn for 40 s, and spring and summer burn for 40 s (p=0.077, 0.032, and p=0.095 Table 9). No sprouting occurred after burning for 15 s in the spring, but 14.3 % average sprouting did occur after the 15 s winter burn (Table 6).

Table 6. Mean percent ± standard error of paulownia trees with stump sprouts after each treatment.

Burn length for two size classes, s	Season of burn			Average
	Winter	Spring	Summer	
	Mortality, % ± standard error			
Small Diameter Class				
0	0.0±0.0	0.0±0.0	0.0±0.0	0.0
15	14.3±7.8	0.0±0.0	5.3±5.3	6.6
30	9.1±6.3	17.7±9.5	17.7±9.5	14.3
Large Diameter Class				
0	0.0±0.0	0.0±0.0	0.0±0.0	0.0
40	4.6±4.5	13.0±7.2	27.3±9.7	14.9
60	30.4±9.8	30.4±9.8	28.0±9.2	29.6

Table 7. Differences in the percent of paulownia trees with new stump sprouts following burning the main stem for different lengths of time.

Pairwise treatment comparisons	Difference, %	***P*-value**
Small Diameter Class		
*Control vs. 15 s	–6.6	0.333
*Control vs. 30 s	–14.3	0.114
15 s vs. 30 s	–7.7	0.056
Large Diameter Class		
*Control vs. 40 s	–14.9	0.067
*Control vs. 60 s	–29.6	0.011
40 s vs. 60 s	–14.7	0.013

* Indicates Fisher's Exact Test analysis. In all other comparisons Chi-square tests was used. We used Bonferroni correction for multiple comparisons.

Table 8. Differences in the percent trees with new sprouts as a result of burning for the particular lengths of time during each season.

Size and season of treatment	Pairwise treatment comparisons	Difference, %	*P*-value
Small Diameter Class			
*Winter	15 s vs. 30 s	5.2	0.664
*Spring	15 s vs. 30 s	-17.7	0.081
*Summer	15 s vs. 30 s	-12.4	0.324
Large Diameter Class			
*Winter	40 s vs. 60 s	-25.8	0.047
Spring	40 s vs. 60 s	-17.4	0.153
Summer	40 s vs. 60 s	-0.7	0.956

* Indicates Fisher's Exact Test analysis. In all other comparisons Chi-square tests was used.

Table 9. Differences in the percent trees with new sprouts as a result of burning (comparison of the effect of season of burn).

Size and burn time, s	Pairwise season comparison	Difference, %	*P*-value
Small Diameter Class			
*15	Winter vs. Spring	14.3	0.077
*15	Winter vs. Summer	9.0	0.202
*15	Spring vs. Summer	–5.3	0.158
*30	Winter vs. Spring	–8.6	0.212
*30	Winter vs. Summer	–8.6	0.212
*30	Spring vs. Summer	0.0	0.333
Large Diameter Class			
*40	Winter vs. Spring	–8.4	0.203
*40	Winter vs. Summer	–22.7	0.032
*40	Spring vs. Summer	–14.3	0.095
60	Winter vs. Spring	0.0	0.333
60	Winter vs. Summer	2.4	0.284
60	Spring vs. Summer	2.4	0.284

* Indicates Fisher's Exact Test analysis. In all other cases we used Chi-square tests and Bonferroni correction for multiple comparisons.

Sprouting occurred almost 6 times more often after the summer 40 s burn, an average of 27.3 %, than after the winter 40 s burn, and occurred at almost double the rate of sprouting after the spring 40 s burn (Table 6).

Results from our work demonstrate that applying high-intensity directed fire to the base of the stem is indeed able to cause mortality in royal paulownia, even in trees up to 68 cm in diameter. Burning paulownia trees with diameters from 3.8 cm to 20.1 cm for 30 seconds and burning trees with diameters from 20.2 cm to 68.6 cm for 60 seconds may be preferable as it results in greater mortality than the shorter fire exposure. Burning paulownia stems with diameters smaller than 20.1 cm for 15 seconds should not be used because this length of burning was not effective. In a similar study, stems of *Acacia* species burned with higher intensity fire had greater stem mortality (Wright and Clarke 2007). The most adequate season to burn in order to cause the greatest mortality in paulownia is summer, which could be because of active growth and less carbohydrate reserves in the roots during summer than spring (Bowen and Pate

1993). Successful mortality can be obtained by burning in spring as well, but burning in winter for the examined lengths of time results in low levels of mortality. Similarly to our results, greater morality occurred in ponderosa pine (*Pinus ponderosa* C. Lawson) when burning took place in the growing seasons rather than in the dormant seasons (Harrington 1993).

Paulownia trees burned in the summer for 40 s sprouted more frequently than trees burned in the winter or spring for the same length of time. However, this difference among seasons was not observed for the 30 second treatment of the smaller trees or 60 second treatment of the larger trees during any season. This could be a result of the extremely hot fire that we applied. Other studies have used prescribed burning which does not normally get as hot and burn as long as the high-intensity directed fire in our study. Due to the intense heat, the basal buds may have been damaged, as was desired, which resulted in similar sprouting for among the treatments and among the seasons. Since sprouting occurs at similar rates regardless of season, burning should be applied in frequent intervals during each of the seasons to determine how many repetitions should be applied to successfully eradicate these species. Others report that it may take five annual burnings to finally reduce the stem

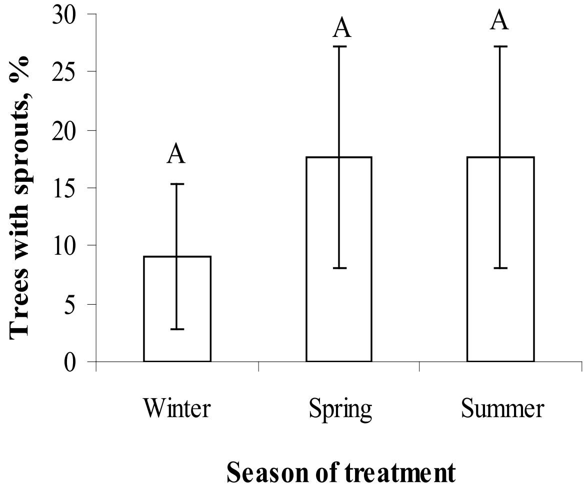

Fig. 3. Mean percent and standard error of small diameter paulownia trees with sprouts after burning for 30 s in winter, spring, and summer.

The same letters indicate no difference at the *p*<0.1 level.

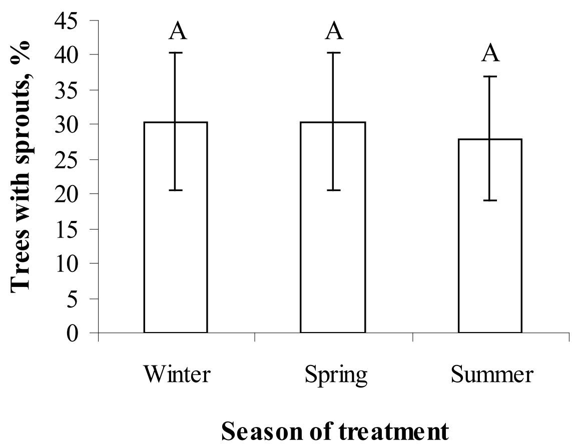

Fig. 4. Mean percent and standard error of large diameter paulownia trees with sprouts after burning for 60 s in winter, spring, and summer.

The same letters indicate no difference at the *p*<0.1 level.

numbers of shrub live oak to what they were before the first treatment, while desert ceanothus sprouts were almost all killed by a second burn (Pond and Cable 1961).

Conclusion

The method used in this study was successful at causing mortality even in large size trees of the studied species. To successfully kill paulownia, burning should take place in the summer. Additionally, trees between 3.8 cm and 20.1 cm in dbh should be burned for 30 seconds and trees between 20.2 cm and 68.6 cm for 60 seconds. In our study these treatments caused the most tree mortality, although stump sprouting was stimulated. Although we did not burn the sprouts, burning them after the following growing season may reduce the resprouting rate of this species. Repeated burning of the sprouts should be further studied for this species. Pond and Cable (1961) reported that sprouts from Wright's silktassel (*Garrya wrightii* Torr.) and Hollyleaf buckthorn (*Rhamnus crocea* Nutt. ssp. *pilosa*) were completely eliminated by four annual burns by a torch with flame temperature of 1,500 °C.

The cost for the equipment used was a total of $92.95. The torch cost was $63.00 and the propane tank $29.95. Assuming labor wages at $7.25·hr^{-1}, the approximate average cost for supplies and labor was $1.46 per paulownia tree. However, the cost can be lowered with scaling up of the treatment. This can be achieved with some of the already widely available flaming equipment currently used in agriculture (e.g., Red Dragon® vegetable bed flamer). Some adaptations in the design may be needed for forestry settings. Future studies of this method may try to develop a more accurate cost estimate. We also recommend testing how many repeated burns of the sprouts are needed for the plant to no longer be able to carry out vegetative reproduction.

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