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RESEARCH ARTICLE

Restoration impacts on fuels and fire potential in a dryland tropical ecosystem dominated by the invasive grass *Megathyrsus maximus*

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Ecological restoration often attempts to promote native species while managing for disturbances such as fire and non-native invasions. The goal of this research was to investigate whether restoration of a non-native, invasive *Megathyrsus maximus* (guinea grass) tropical grassland could simultaneously promote native species and reduce fire potential. *Megathyrsus maximus* was suppressed with herbicide, and three suites of native species—each including the same groundcover and shrub, and one of three tree species—were outplanted in a randomized, complete block design that also included herbicide control (herbicide with no outplantings) and untreated control treatments. Fuels were quantified 27 months after outplanting, and potential fire behavior (rate of spread and flame length) was modeled with BehavePlus. Compared with untreated controls, native outplant treatments reduced *M. maximus* cover by 76–91% and *M. maximus* live and dead fuel loads by greater than 92 and 68%, respectively. Despite reductions in *M. maximus* fuels, neither treatment-level (grass + native) total fuel loads and fuel moistures, nor modeled fire behavior differed between outplant treatments and controls. The best performing native woody species (*Dodonaea viscosa*) had significantly lower average individual plant live fuel moisture (84%) than *M. maximus* (156%) or other native woody outplant species (201–328%), highlighting the need for careful species selection. These results demonstrate that restoring native species to degraded tropical dry forests is possible, but that ecological restoration will not necessarily alter the potential for fire, at least in the short term, making selection of species with beneficial fuel properties and active fire management critical components of ongoing restoration.

Key words: BehavePlus, *Dodonaea viscosa*, fire modeling, guinea grass, native species outplanting

Implications for Practice

- Suppression of *Megathyrsus maximus* with repeated herbicide application may be necessary until a woody canopy is restored.
- Restoring native vegetation can suppress fire-promoting non-native grasses, but this may not change fuel properties enough to reduce potential for fire in early stages of succession.
- *Dodonaea viscosa*, a shrub species, has much lower live fuel moisture than other native species, and its use in ecological restoration should be balanced with complementary fire mitigation.
- *Myoporum sandwicense*, a canopy species, has high live fuel moisture, and should be considered in ecological restoration where fire is a threat.
- Selection of species with beneficial fuel properties and ability to survive in harsh environments combined with ongoing fire mitigation are complementary toward effective restoration in non-native grass-invaded landscapes.

Introduction

A primary goal of ecological restoration is to reestablish self-sustaining aspects of a natural ecosystem in degraded areas (Hobbs & Norton 1996; SERI 2006), which is often accomplished by promoting native and controlling undesirable species. Disturbance is a critical ecosystem component that plays a large role in controlling species interactions and ecosystem structure and function (Turner 2010). As such, restoration should involve managing for disturbances in addition to returning native species to the site (MacDougall & Turkington 2005), either by returning natural disturbance regimes or by removing disturbances that lie outside of the natural (historical) regime.

Where fire regimes have been altered, fuels and fire management should be primary considerations of ecological restoration projects (Baker 1994; Brooks et al. 2004). The quantity, arrangement, chemical content, and moisture content of fuel

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loads (burnable plant material) are important determinants of fire regimes. As a result, the alteration of fuels can result in either ecosystem degradation or restoration, depending on the characteristics of the system and the restoration activities. In ponderosa pine ecosystems, for example, fire suppression has resulted in large fuel accumulations. Prescribed fire and mechanical fuel treatments are often employed to return a low-intensity surface fire regime (Baker 1994; Allen et al. 2002), thereby restoring ecological structure and function (Fule et al. 2001). In fire-adapted temperate tall grass ecosystems, restoration objectives often include managing for both heterogeneous disturbance regimes and fire-adapted species (Moranz et al. 2012). Globally, contemporary restoration efforts often need to modify fuels to restore ecosystem structure, but are challenged by climate change, invasive species, and altered disturbance trajectories (D'Antonio & Vitousek 1992; Vallejo et al. 2012).

In seasonally dry tropical ecosystems, widespread grass invasion complicates restoration efforts by both outcompeting natives (Ammondt et al. 2013), and increasing fire occurrence and intensity (D'Antonio & Vitousek 1992). *Megathyrsus maximus* [Jacq.] (guinea grass; formerly *Urochloa maxima* and *Panicum maximum*), an African pasture grass, was introduced to Hawaii as livestock forage, quickly invaded, and became naturalized by 1871 (Motooka et al. 2003) across a majority of coastal, low elevations landscapes across the Hawaiian islands (D'Antonio & Vitousek 1992; Portela et al. 2009). Invasive grasses like *Megathyrsus maximus* now dominate approximately 25% of total land area in Hawaii and have led to a large increase in the number and size of wildfires in the state (Trauernicht et al. 2015, in press). This species recovers rapidly following fire (Vitousek 1992; Williams & Baruch 2000), contributing to a positive feedback between invasion and wildfire (D'Antonio & Vitousek 1992). *Megathyrsus maximus* and similar invasive, tropical grass species are common in areas that have been degraded by grazing, fire, agriculture, and development, and interactions between these invasions and altered fire regimes are prevalent throughout the tropics (D'Antonio & Vitousek 1992; Mack & D'Antonio 1998; Brooks et al. 2004). Ecological restoration in invaded areas therefore necessitates careful consideration of fire along with promotion of native species in these degraded areas.

Globally, tropical dry forests are among the most endangered ecosystems (Vieira & Scariot 2006). Hawaii is no exception, where less than 10% of these forests remain intact (Bruegmann 1996), with the majority of these historical dry forest ecosystems converted to invasive grasslands. Fires in these ecosystems in Hawaii occur year-round, are almost exclusively anthropogenic in origin (with the exception of very rare lightning ignitions), and few native plants survive in the postfire environment, particularly after repeated fires (LaRosa et al. 2008; Trauernicht et al. 2015, in press). Remnant dry forest stands are typically invaded by grass understory, making them highly susceptible to wildfire, further invasion, and ultimately, leading to type conversion (Ellsworth et al. 2014). Priority restoration objectives in this ecosystem, as in many systems globally, include removal of the processes causing degradation (i.e. altered fire regime and grass invasion), followed by reintroduction and promotion of native species to establish new trajectories toward stable, functional ecosystems (Cordell et al. 2008).

The overarching objective of this research was to use herbicide control and native species outplantings to simultaneously promote native species and reduce *M. maximus* cover and fuel loads, with the overall goal to reduce potential fire behavior (i.e. Rate of spread [ROS] and fire intensity). To address this objective, we used a randomized complete block field experiment with three different native species outplant and herbicide treatments (referred to hereafter as outplant treatments), an herbicide control (grass herbicide with no native outplantings), and an untreated control (no herbicide or native outplantings) to monitor survival and fuel characteristics. Four specific hypotheses were tested: (1) native outplantings with grass herbicide would suppress *M. maximus* cover and fuel loads (Ammondt & Litton 2012); (2) total fuel loads (non-native grass + native outplants) would be lowest in outplant treatments due to control of the invasive grass (Motooka et al. 2002); (3) shading by woody species in outplant treatments would result in higher fuel moisture content (Bigelow & North 2012); and (4) outplanting native species would result in decreased potential fire behavior due to the reduction of non-native grass fuels (Griscom & Ashton 2011; Bigelow & North 2012). These hypotheses were tested by quantifying fuel loads and moistures, and modeling fire behavior 27 months after treatment implementation.

Methods

Study Site and Restoration Treatments

This study was conducted in the Waianae Kai Forest Reserve on leeward Oahu, Hawaii (300 m.a.s.l., 158°9′181″W, 21°28′53″N; Fig. 1) in a highly degraded area with a dense monoculture of *Megathyrsus maximus* and limited non-native woody species (*Leucaena leucocephala*, *Grevillea robusta*, and *Prosopis pallida*) in the general area. Site selection was based on access, ownership (State Forest Reserve with permission for fence establishment and outplanting), and observed spatial variability in *M. maximus* cover and fuel properties where this site is approximately in the middle of a range of grass cover values across the island (Ellsworth et al. 2013). The study area last burned in 2002, approximately 8 years before this study began (R. Peralta 2008, State of Hawaii Department of Land and Natural Resources, personal communication). Mean annual precipitation is 1,258 mm, with drier conditions from April to September but at least 50 mm precipitation on average in every month (Giambelluca et al. 2011). Mean annual temperature is 22°C, with an average monthly low in January of 20°C and an average monthly high of 24°C in August (Giambelluca et al. 2014). Soils are mollisols in the Ewa series (fine, kaolinitic, isohyperthermic Aridic Haplustolls) (Natural Resources Conservation Service 2006). In July 2009, the study area was mowed to an estimated height of 20 cm aboveground level exposing individual *M. maximus* crowns, which were treated with glyphosate application following regrowth 8 weeks later. A 0.13-ha fence was erected to exclude feral ungulates (pigs, goats,

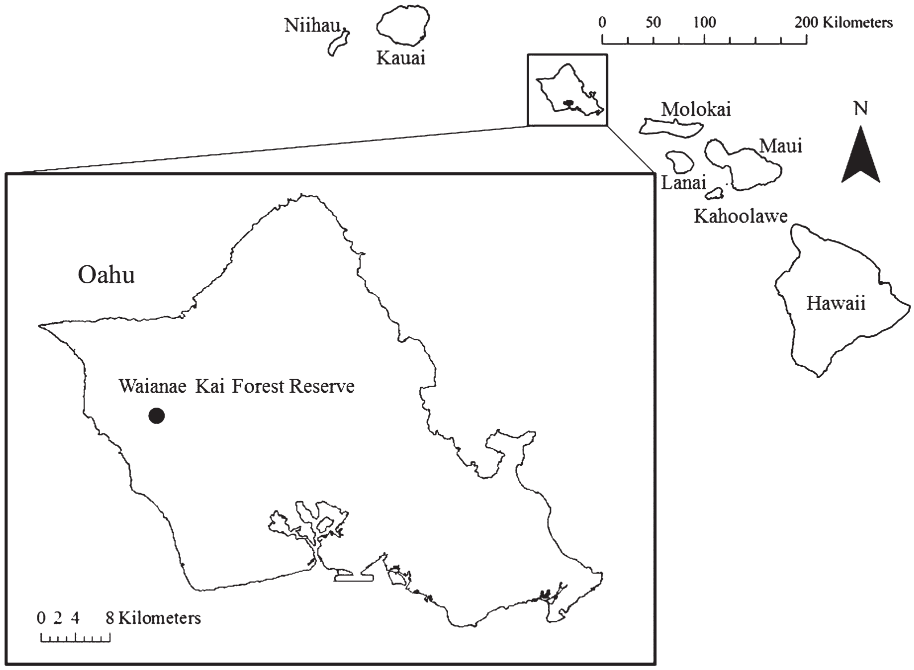

Figure 1. Restoration site in the Waianae Kai Forest Reserve on the Island of Oahu, Hawaii.

cattle; see more details on site preparation in Ammondt et al. 2013).

Four blocks were established within the exclosure, with five 9-m^2 plots in each block. Treatments were randomly assigned to one of five plots (four replicates per treatment). Herbicide control and untreated control plots were also randomly assigned within each block. On 7 January, 2010, native species were outplanted at a density of four trees per plot for overstory trees (4,444 trees/ha), and 9 and 12 plants per plot (10,000 and 12,000 plants per hectare) for shrub and ground cover species, respectively. Planting densities were based on the desire to obtain canopy closure and shading of *M. maximus* as early as possible in stand development, balanced with what would be realistic for land managers.

Species selection was based on local knowledge of prior outplanting success, rapid growth rates, potential for competition with non-native grass, and commercial availability. All three outplant treatments included the shrub *Dodonaea viscosa* (L.) Jacq. and the ground cover *Plumbago zeylanica* L., and each treatment contained one of three canopy trees: *Thespesia populnea* (L.) Sol., *Cordia subcordata* Lam., or *Myoporum sandwicense* (A. DC.) A. Gray. *D. viscosa* (Sapindaceae) is a rapidly growing shrub, typically reaching a height of 1–3 m, but rarely up to 10 m. It is pantropical in distribution and commonly used in restoration and revegetation projects in tropical dry forests in Hawaii and throughout the tropics due to rapid growth and reproductive rates as well as some capacity to survive in a postfire environment (Cabin et al. 2002; Ainsworth and Kauffman 2009). *Plumbago zeylanica* (Plumbaginaceae) is a ground cover that reaches a spread of 5 m. It successfully establishes into disturbed areas and is frequently used for soil stabilization projects. *Thespesia populnea* (Malvaceae) is pantropical in distribution and reaches a height of 6–10 m. It can survive a wide range of soil, precipitation, and salinity conditions and was historically commonly used as a shade tree by native Hawaiians (Stone et al. 1995). *Cordia subcordata* (Boraginaceae) is a fast-growing evergreen tree that reaches 8–10 m in height. *Myoporum sandwicense* (Scrophulariaceae) is endemic to Hawaii and has rapid growth rates and high tolerance to drought conditions. This species has been shown to effectively suppress *M. maximus* in greenhouse trials (Ammondt & Litton 2012).

Outplants were commercially produced and purchased from a local source (Hui Ku Maoli Ola Kaneohe, Hawaii) in 10-cm containers, and were approximately 1-year-old at the time of outplanting. A total of 25 plants were outplanted uniformly in each outplant treatment plot (Fig. 2; 12 *P. zeylanica*, 9 *D. viscosa*, and 4 trees of the same species) in early January 2010, coinciding with the beginning of the wetter winter season, and each plant was given 1 L water at planting and once each week for 3 weeks. Plants that died within the first month (21% mortality) were replaced. On 5 January, 2010, 12 April, 2010, 30 November, 2010, and 21 May, 2011 (i.e. 0, 3, 10, and 16 months after outplanting), the grass-specific herbicide fluazifop *p*-butyl (Fusilade® DX) was reapplied to *M. maximus* regrowth in all treatments except the untreated control.

Survival and Cover

Survival of native woody species was measured on 24 April, 2012, 27 months after outplanting at the beginning of the drier summer season. Because *P. zeylanica* produced a continuous cover at the soil surface where individuals could no longer

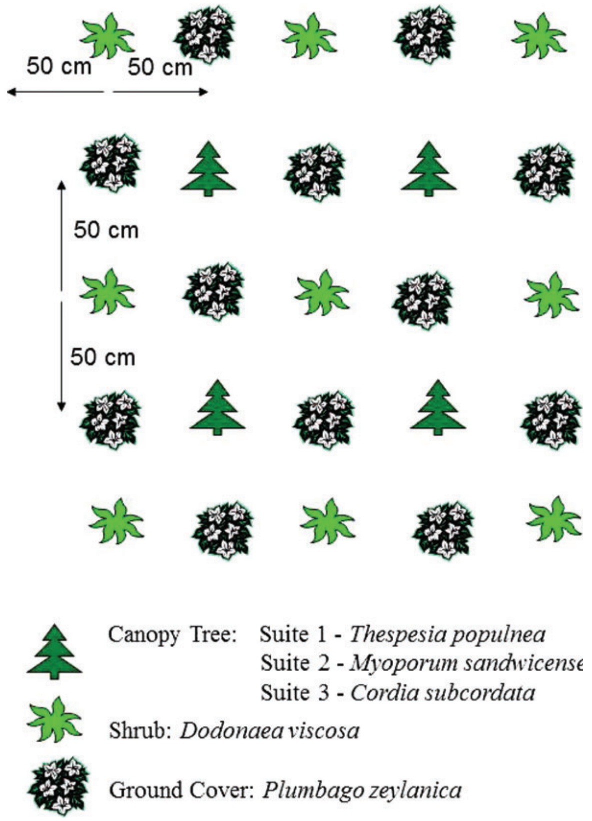

Figure 2. Experimental layout of each outplanting plot. Each plot contained the shrub *Dodonaea viscosa*, the ground cover *Plumbago zeylanica*, and one of the three canopy trees: *Thespesia populnea*, *Myoporum sandwicense*, or *Cordia subcordata*.

be distinguished, survival was not measured for this species. Percent cover of native woody species, *P. zeylanica*, *M. maximus*, and surface litter was measured on 25–26 April, 2012 using a point-intercept method with an 81-point plot frame in each 9 m^2 plot.

Fuels

Surface litter and standing live and dead fuel loads of *M. maximus* and *P. zeylanica* were measured on 25–26 April, 2012 in each plot by collecting litter and clipping vegetation in four randomly located 25 × 25 cm subplots, and compositing by species and fuel type (live, standing dead, litter). Samples were transported in plastic bags and kept in a cooler to minimize moisture loss. Within 5 hours of collection, samples were weighed, dried at 70°C for 48 hours, and reweighed to determine dry fuel mass and fuel moisture relative to dry fuel mass.

Live fuel loads of *C. subcordata* and *D. viscosa* were estimated with species-specific allometric models developed in Hawaii (Litton & Kauffman 2008; Ammondt et al. 2013). New allometric equations were developed in this study to quantify *T. populnea* and *M. sandwicense* fuel loads. Individuals of each woody species were planted near the study area during the initial outplanting, and destructive harvest occurred throughout the experiment to obtain individuals from a range of size classes spanning the size of outplanted individuals. Plants were harvested at the soil surface, separated into foliage and wood components, dried at 70°C to a constant mass, and weighed.

Live moisture content for individual native woody species was measured by clipping ≥3 leaves and ≥1 woody stem from three individuals of each species in each plot. Samples were immediately placed into plastic bags and kept in a cooler. Within 5 hours of collection, samples were sorted into leaf, wood, and reproductive (flowers and seeds) components, weighed, dried at 70°C for 48 hours, and reweighed. Moisture content for each species was weighted by the proportional mass of each plant component to estimate overall individual plant moisture content. Fuel moisture was then scaled from individual plant to the treatment level by weighting the relative proportions of *M. maximus*, litter, and native outplant (*P. zeylanica*, *D. viscosa*, *T. populnea*, *M. sandwicense*, and *C. subcordata*) fuel mass by their respective fuel moistures.

Fire Modeling

BehavePlus (version 5.0.5) fire modeling software was used for simulating fire behavior in the experimental treatments. Fuel inputs were collected as above. Treatment-level weighted fuel moisture measurements were used in modeling simulations for each plot ($N = 4$ per treatment). Height of the tallest plant (grass or native) was measured in each plot, and mean fuel height was estimated as 70% of the maximum fuel height (Burgan & Rothermel 1984). Microclimate variables (temperature, wind speed) were obtained from an adjacent (approximately 50 m) remote-automated weather station (RAWS). Live and dead fuel heat contents were measured by bomb calorimetry (Hazen Research, Inc., Golden, CO, U.S.A.). Previously published values were used for dead fuel moisture of extinction for *M. maximus* (Beavers 2001) and woody surface area to volume ratios for humid tropical grasslands (Scott & Burgan 2005). One-hour surface area to volume ratios were quantified using a LI-3100C portable leaf area meter (Li-Cor, Inc., Lincoln, NE, U.S.A.) and water displacement. Wind speeds of 15 and 30 km/hour at 6.1 m height were used to simulate moderate and severe fire danger scenarios, and a wind adjustment factor of 0.3 was used to adjust the windspeed collected by the RAWS (6.1 m wind speed) to vegetation height (surface wind speed) (Andrews et al. 2005). Fire behavior outputs analyzed included probability of ignition (POI: the likelihood that a firebrand will cause an ignition when it lands on combustible fuels) (%), maximum ROS (m/minute), and flame length (m; an estimate of fire intensity) (Rothermel 1972).

Statistical Analyses

Analysis of variance (ANOVA) was used to quantify differences in treatment-level fuel loads (total, *M. maximus*, native plants), fuel moisture (*M. maximus*, individual native plants, and weighted treatment-level moisture), and predicted fire behavior (POI, ROS, flame length) between treatments. Block was a random factor, and treatment was a fixed factor. For all tests, there are four replicates of each treatment. Tukey's multiple comparison tests were used to separate treatment means following significant ANOVAs. ANOVA was also used to test for differences among native species in individual plant survival, cover, fuel load, and fuel moisture. For allometric modeling, predictive relationships between stem basal diameter and leaf, wood, and total biomass were developed using nonlinear regression. Power, quadratic, and cubic models were explored, and final model selection was based on R^2 and p values and visual inspection of residual plots. As we were interested both in the treatment effect (different groupings of species, as might be used in a restoration outplanting scenario) as well as the response of individual species (i.e. survival, species-level fuel moisture, and fuel load), we discuss analyses as occurring at the individual plant level versus the treatment level. IBM SPSS v.20 (IBM SPSS, Inc., Chicago, IL, U.S.A.) was used for all analyses. Results were considered significant at $\alpha < 0.05$.

Results

Survival and Cover

Survival of native plants averaged 51% across species, with 57% of *Dodonaea viscosa* ($n = 62$), 56% of *Thespesia populnea* ($n = 9$), 38% of *Myoporum sandwicense* ($n = 6$), and 19% of *Cordia subcordata* ($n = 3$) individuals surviving 27 months after outplanting. *Plumbago zeylanica* ranged from 68 to 92% cover, with no differences among treatments (Table S1, Supporting Information). *Megathyrsus maximus* cover was significantly lower ($p < 0.01$) in outplant treatments (9–24%) than in herbicide control or untreated control plots (91–100%) (Fig. 3). There was no recruitment of native or non-native species in herbicide control plots, so cover in this treatment was limited to regrowth of *M. maximus*. Litter cover ranged from 97 to 100% and did not differ by treatment ($p = 0.67$; Table S1).

Fuels

Basal diameter accurately predicted aboveground individual plant live fuel loads (i.e. live biomass) for native woody species ($R^2 \geq 0.61$; $p < 0.01$), with the exception of *T. populnea* foliage ($R^2 = 0.38$) (Table S2). Log transformation of dependent and/or independent variables, and inclusion of tree height did not improve model fits. Using these equations, individual woody plant live fuel loads averaged across all treatments were calculated at 738, 335, 87, and 30 g for *D. viscosa*, *M. sandwicense*, *T. populnea*, and *C. subcordata*, respectively.

At the treatment level, *M. sandwicense* outplant treatments had more total native woody fuels (6.47 Mg/ha) than *C. subcordata* treatments (3.3 Mg/ha; $p < 0.05$), but neither differed from *T. populnea* treatments (3.8 Mg/ha; $p > 0.08$; Table 1). *Cordia subcordata* treatments had more *P. zeylanica* ground cover (4.9 Mg/ha) than *T. populnea* (2.6 Mg/ha; $p < 0.01$) and *M. sandwicense* treatments (3.7 Mg/ha; $p > 0.06$; Table 1).

Mean individual plant moisture content differed considerably by outplant species (Fig. 4). *Dodonaea viscosa* moisture content (84%) was lowest ($p < 0.01$) but did not differ across treatments

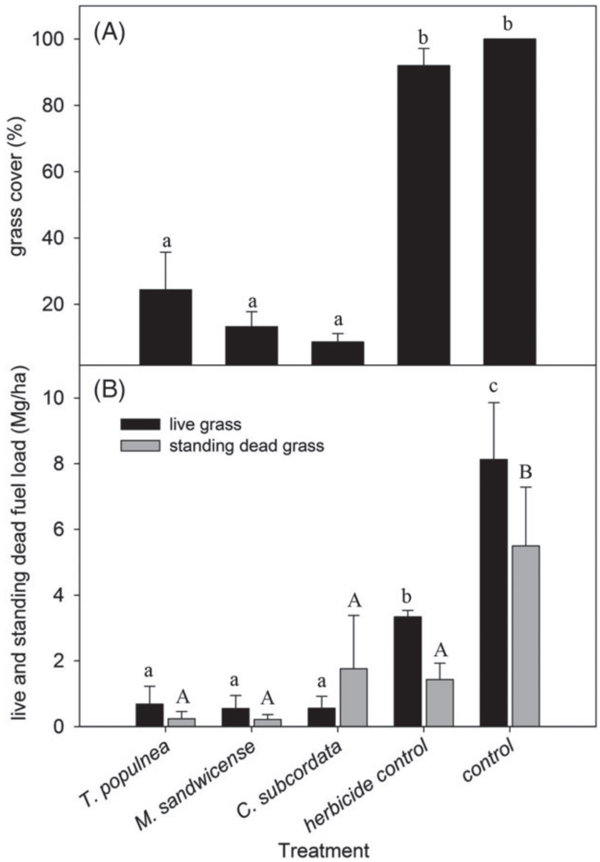

Figure 3. *Megathyrsus maximus* (A) cover and (B) live and standing dead fuel load. Bars are treatment means and error bars are +1 SE. Significant differences between treatments at the $\alpha = 0.05$ level are indicated by different lowercase (A and B) and uppercase (B) letters.

($p = 0.45$), and *M. sandwicense* moisture content (328%) was highest ($p < 0.01$). *Thespesia populnea* (201%) and *C. subcordata* (203%) moisture content did not differ ($p = 0.99$). Moisture content of *P. zeylanica* (120–165%) did not differ across treatments ($p = 0.27$).

Megathyrsus maximus was greatly reduced by outplant treatments (Fig. 3; Table 1). Treatment-level live grass fuel loads ranged from 0.6 to 0.7 Mg/ha in outplant plots, whereas herbicide control plots averaged 3.4 Mg/ha and untreated control plots averaged 8.1 Mg/ha. Standing dead *M. maximus* fuel loads were also higher in untreated control plots (5.5 Mg/ha) than outplant treatments and herbicide control plots (0.2–1.8 Mg/ha). *Megathyrsus maximus* moisture content was 103–189% for live and 25–48% for dead grass, and did not differ across treatments ($p = 0.36$).

Total plot fuel loads (native plant + *M. maximus* fuels) were not different between outplant treatments and either untreated or herbicide control plots, though herbicide controls had lower total fuels than untreated controls (Table 1). When moisture content for the entire plot was weighted by the biomass of each species, treatment-level live fuel moisture was 118–182%, and dead fuel moisture was 25–48%, with no differences across outplant or control treatments (live, $p = 0.10$; dead, $p = 0.38$;

Table 1. Mean plot-level fuel loads (SE) for all outplant, herbicide control, and untreated control treatments at Waianae Kai Forest Reserve, Island of Oahu, Hawaii. *Thespesia populnea*, *Myoporum sandwicense*, and *Cordia subcordata* were planted as canopy species in three separate outplant treatments. A consistent midstory of *Dodonaea viscosa* and a groundcover of *Plumbago zeylanica* were planted in each outplant treatment plot. Herbicide control and untreated control plots were excluded from analyses of differences between native outplants as there was no natural recruitment into control plots. Bold indicates statistically significant model results. Different letters indicate statistically significant differences between treatments at the $\alpha < 0.05$ level. *df*, degree of freedom.

Parameter	*T. populnea*	*M. sandwicense*	*C. subcordata*	*Herbicide Control*	*Untreated Control*	*Model* R^2	*Block* p-*value (df, F-statistic)*	*Treatment* p-*value (df, F-statistic)*
Native tree leafy fuel load (Mg/ha)	0.83 (0.22)	1.54 (0.27)	0.73 (0.21)	0.00 (0.00)	0.00 (0.00)	57.57	0.62 (3, 0.63)	0.11 (2, 3.13)
Native tree woody fuel load (Mg/ha)	3.16 (0.87)	5.13 (1.50)	2.63 (0.86)	0.00 (0.00)	0.00 (0.00)	43.89	0.57 (3, 0.73)	0.35 (2, 1.26)
P. zeylanica fuel load (Mg/ha)	2.63 (0.53)	3.68 (0.52)	4.87 (0.46)	0.00 (0.00)	0.00 (0.00)	64.15	0.61 (3, 0.66)	0.07 (2, 4.38)
Megathyrsus maximus live fuels (Mg/ha)	0.68 (0.54)[a]	0.55 (0.40)[a]	0.55 (0.36)[a]	3.39 (0.20)[b]	8.13 (1.73)[c]	83.75	0.45 (3, 0.95)	**<0.01** (4, 14.75)
M. maximus standing dead fuels (Mg/ha)	0.24 (0.22)[a]	0.22 (0.15)[a]	1.76 (1.62)[a]	1.43 (0.50)[a]	5.50 (1.79)[b]	54.03	0.81 (3,0.32)	**0.05** (4, 3.29)
Litter fuels (Mg/ha)	6.66 (1.15)	5.06 (1.75)	5.05 (1.09)	5.69 (1.06)	8.70 (2.12)	68.37	**0.01** (3, 5.89)	0.15 (4, 2.07)
Total fuels (Mg/ha)	14.2 (1.69)[a,b]	16.18 (3.35)[a,b]	15.59 (0.35)[a,b]	10.51 (1.43)[a]	22.33 (4.89)[b]	71.26	**0.02** (3, 4.60)	**0.03** (4, 3.99)

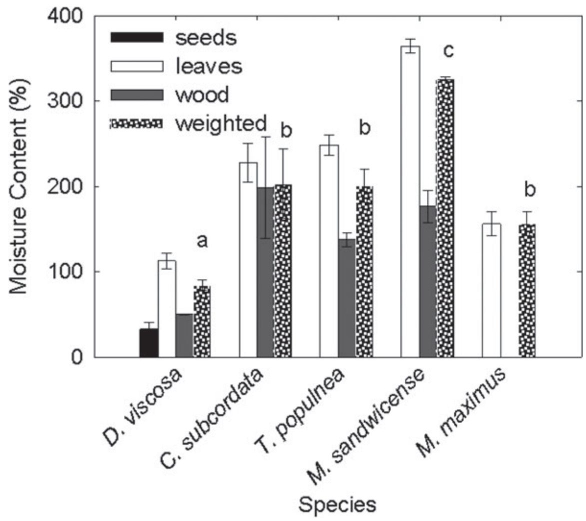

Figure 4. Species-level plant live fuel moisture content for *Megathyrsus maximus* and native outplant species at Waianae Kai Forest Reserve. Moisture content for seed, leaves, and wood for each species are given, as well as overall moisture content for each species weighted by the proportional mass of each plant component. Bars are means for each species and error bars are ±1 SE. Different letters denote statistically significant differences in overall weighted moisture content between species at the $\alpha = 0.05$ level.

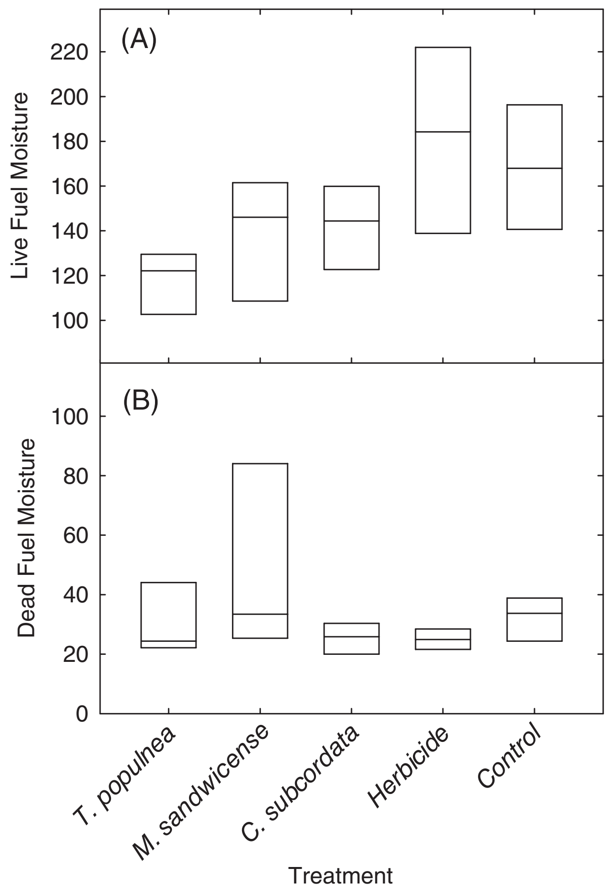

Figure 5. Treatment-level (A) live and (B) dead fuel moisture weighted by relative proportions of *Megathyrsus maximus*, litter, and native outplant (*Plumbago zeylanica*, *Dodonaea viscosa*, *Thespesia populnea*, *Myoporum sandwicense*, and *Cordia subcordata*) fuels. The lower boundary of the box indicates the 25th percentile, the line within the box marks the median, and the upper boundary of the box is the 75th percentile. No significant differences between treatments were found for either live or dead fuel moisture.

Fig. 5). Average fuel height ranged from 41.8 to 44.4 cm in herbicide control, *T. populnea* and *M. sandwicense* treatment plots, which was lower than that of untreated control plots (64.1 cm; $p < 0.05$). Average fuel height in *C. subcordata* treatments (49.9 cm) did not differ from any other treatment ($p > 0.18$).

Fire Modeling

Due to few differences in treatment-level fuel loads and fuel moistures, there was no difference in predicted fire behavior between any treatment ($p > 0.40$; Table 2). When

Table 2. Predicted fire behavior under both moderate (15 km/hour) and severe (30 km/hour) wind conditions in outplanting, herbicide control, and untreated control treatment plots in *Megathyrsus maximus*-dominated, non-native grass ecosystems on leeward Oahu, Hawaii. Means (SE) are given for fire behavior parameters for each treatment ($n = 4$). *Thespesia populnea*, *Myoporum sandwicense*, and *Cordia subcordata* were planted as canopy species in three separate treatments. A consistant midstory of *Dodonaea viscosa* and a groundcover of *Plumbago zeylanica* were planted in each outplant treatment plot. There were no differences between treatments for any fire behavior parameters.

Parameter	*Wind Conditions*	*T. populnea*	*M. sandwicense*	*C. subcordata*	*Herbicide Control*	*Untreated Control*	*Model* R^2	*Block p (df, F-Statistic)*	*Treatment* p *(df, F-Statistic)*
ROS (m/minute)	Moderate	0.90 (0.3)	0.30 (0.1)	1.13 (0.7)	1.75 (0.3)	1.50 (0.7)	28.79	0.93 (3, 0.14)	0.40 (4, 1.11)
	Severe	1.83 (0.5)	0.53 (0.2)	2.13 (1.58)	3.48 (0.6)	2.95 (1.4)	28.80	0.92 (3, 0.16)	0.40 (4, 1.09)
Flame length (m)	Moderate	0.78 (0.2)	0.33 (0.1)	0.75 (0.4)	1.15 (0.2)	1.25 (0.6)	24.64	0.98 (3, 0.07)	0.48 (4, 0.93)
	Severe	1.10 (0.3)	0.40 (0.2)	1.03 (0.6)	1.60 (0.3)	1.73 (0.8)	25.58	0.99 (3, 0.04)	0.43 (4, 1.03)
POI (%)	Moderate	2.75 (1.8)	0.75 (0.8)	2.25 (1.0)	2.50 (0.9)	0.75 (0.8)	31.14	0.50 (3, 0.84)	0.59 (4, 0.73)
	Severe	2.75 (1.8)	0.75 (0.8)	2.25 (1.0)	2.50 (0.9)	0.75 (0.8)	31.14	0.50 (3, 0.84)	0.59 (4, 0.73)

surface windspeeds were simulated at moderate conditions (15 km/hour), ROS was 0.3–1.8 m/minute, and flame length was 0.3–1.3 m. When surface windspeed simulations were increased to severe conditions (30 km/hour), ROS was 0.5–3.5 m/minute, and flame length was 0.4–1.7 m. Probability of ignition was 0.8–2.8% for both windspeed scenarios (Table 2).

Discussion

The results from this study and others in heavily invaded tropical dry ecosystems suggest that reintroduction of native species can be successful with substantial initial investments including ungulate exclusion, invasive grass control, fire exclusion, and native outplanting (Cabin et al. 2002; Daehler & Goergen 2005; Thaxton et al. 2012). Even with modest survival rates of native outplants (approximately 50% for all species except *Cordia subcordata*), we documented significant reductions of *Megathyrsus maximus* in the first two years, as hypothesized. Furthermore, the significant reductions in *M. maximus* in native outplant plots compared with herbicide control plots indicates that native species outplanting following herbicide application more effectively suppresses this invasive grass than herbicide alone. Given that survival was relatively low for the canopy species *C. subcordata* (19%) and *Myoporum sandwicense* (38%), reduction of *M. maximus* cover and fuels in these treatments during the first 27 months after outplanting was largely accomplished with the shrub and groundcover species. In contrast to our expectation, however, outplant treatments did not reduce total plot-level fuel loads.

From a native species restoration perspective, modest survival and rapid growth of native species and decreased invasive grass biomass would be considered a success. From a fuels and fire management perspective, however, the moisture content, continuity, and arrangement of fuels must also be considered (Pyne et al. 1996). At 27 months following outplanting, we saw no vertical separation of surface and canopy fuels, so at this early stage of development, native plants contributed to, rather than hindered, fire spread via their fuel properties (i.e. fine fuel accumulations and, in the case of some native species, low live fuel moisture). As the woody species grow into the canopy, lower surface wind speeds and shading of the understory can be expected (Freifelder et al. 1998), resulting in separate surface and canopy fuel considerations (Scott & Reinhardt 2001; Scott & Burgan 2005), and potentially reduced surface fire spread (Stephens 1998; Pollet & Omi 2002; Reinhardt et al. 2008). Also contrary to our hypotheses, there was no difference in fuel moistures or modeled fire behavior between control and treatment plots. Tree canopy species had smaller canopies at the end of the experiment than the intended midstory shrub species (*Dodonaea viscosa*), so they have not yet provided the expected shading effect previously documented (Freifelder et al. 1998).

Selection of appropriate species for outplanting is critical, and when objectives include both native species restoration and fire management, important trade-offs must be considered. *Dodonaea viscosa* is a pantropical species and one of a few fire adapted native Hawaiian plants (Hughes et al. 1991; Ainsworth & Kauffman 2009). The rapid growth and copious seed production demonstrated by this species has made it a frequent choice in restoration projects in Hawaii (D'Antonio et al. 1998; Medeiros & Von Allmen 2006; Ammondt et al. 2013). However, we document here that this species has very low live fuel moisture. As a result, a species that is preferred in restoration for increasing native species cover due to high survivorship and rapid growth rates does not appear to be ideal from a fire prevention perspective.

Myoporum sandwicense exhibited very high fuel moisture, making it a good species to consider in restoration projects where fire is a concern. There was a trend toward decreased flame lengths and ROS in treatments that included *M. sandwicense*. As a hypothetical modeling exercise, we replaced the moisture content for *D. viscosa* with that measured for the outplant tree species in each plot and simulated fire behavior at moderate wind speeds. Under this scenario, fire behavior decreased markedly in all outplant treatments, with lower ROS than untreated control plots ($p < 0.05$). Although this *post hoc* simulation was conducted for outplant treatments that were not included in the field experiment, it demonstrates that careful assessment of the moisture content of potential restoration species is an important consideration when fire management and native species restoration are dual objectives.

Probability of ignition was low (<3%) for all treatments. This measure varies with dead fuel moisture, air temperature, and canopy shading (Andrews et al. 2005). Dead fuel loads, which were primarily *M. maximus*, had fuel moistures ranging from 25 to 48% in this study, and previously reported dead fuel moistures for this species have been as low as 10% (Ellsworth et al. 2013). Fuel moisture of extinction for *M. maximus* is reported at 40% (Beavers 2001), meaning that fire will not carry through dead fuels with a moisture content over 40%. Samples for this study were collected in April, at the beginning of the driest part of the year on average for this site and ecosystem type that typically occurs in summer to early autumn. In this study, some of the dead grass fuels measured were above this moisture of extinction threshold. We would expect that probability of ignition would increase markedly during drier periods when dead fuel moisture is at the lower end of the expected range for this species (Ellsworth et al. 2013). As a result, the fire behavior predictions described here likely represent flammability conditions expected during most of the year, but also represent a best-case scenario for fire occurrence and severity. In addition, we recognize that fire modeling programs such as BehavePlus are simplified representations of ecosystem processes on complex landscapes. As such, they do not account for variability in fuels characteristics, such as the clumpy pattern of individual bunchgrasses as we have on our site, microclimate differences due to a discontinuous canopy, rapidly fluctuating weather characteristics, or other heterogeneity at small spatial and temporal scales (Alexander & Cruz 2013). Despite these limitations, models such as BehavePlus are excellent tools for comparing across plot-level treatments, while holding other potentially confounding variables constant.

The synergistic impacts of altered fire regimes and grass invasion are detrimental to tropical ecosystems globally (D'Antonio & Vitousek 1992; Williams & Baruch 2000). Returning native components to the landscape is feasible, albeit management intensive, in tropical dry ecosystems (Cabin et al. 2002; Ammondt et al. 2013). Perhaps an even greater challenge is altering the positive feedback between non-native grass invasion and repeated wildfires (D'Antonio & Vitousek 1992) to ensure that restoration activities meet long-term objectives. An initial step is an explicit consideration of survival and growth rates and fuel characteristics of the native species chosen for restoration, particularly where wildfire tends to degrade native vegetation and promote non-native grass invasion.

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Supporting Information

The following information may be found in the online version of this article:

Table S1. Mean percent cover (SE) 27 months after outplanting for all outplant, herbicide control, and untreated control treatments at Waianae Kai Forest Reserve, Oahu, Hawaii. *Thespesia populnea*, *Myoporum sandwicense*, and *Cordia subcordata* were planted as canopy species in three separate outplant treatments. A consistent midstory of *Dodonaea viscosa* and a groundcover of *Plumbago zeylanica* were planted in each outplant treatment plot. Herbicide control and untreated control plots were excluded from analyses of differences between native outplants as there was no natural recruitment. Different letters indicate statistically significant differences between treatments at the $\alpha < 0.05$ level from Tukey's multiple comparison tests.

Table S2. Allometric models for predicting native species standing live fuels (i.e. leaf, wood, and total biomass) from basal diameter in Hawaiian dry lowland ecosystems (individuals harvested from outside treatment plots). All models for *Dodonaea viscosa*, *Thespesia populnea*, *Myoporum sandwicense*, and *Cordia subcordata* are power functions ($Y = aX^b$), where Y is the dependent variable (g dry weight), X is the predictor variable (basal diameter, mm), and a and b are constants. *Dodonaea viscosa* equations are from Litton and Kauffman (2008) and *C. subcordata* equations from Ammondt et al. (2013).

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