RESEARCH ARTICLE

FUEL CONDITIONS ASSOCIATED WITH NATIVE AND EXOTIC GRASSES IN A SUBTROPICAL DRY FOREST IN PUERTO RICO

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ABSTRACT

Exotic grasses capable of increasing frequency and intensity of anthropogenic fire have invaded subtropical and tropical dry forests worldwide. Since many dry forest trees are susceptible to fire, this can result in decline of native species and loss of forest cover. While the contribution of exotic grasses to altered fire regimes has been well documented, the role of native grasses in contributing to fuel loads in dry forest has received little attention. We assessed differences in fuel conditions among native and exotic grasses within a subtropical dry forest preserve in Puerto Rico. We quantified fine fuel loads, fuel continuity, and seasonal changes in percent dead grass among the following grass patch types: (1) native grass with no known history of recent fire, (2) exotic grass that had burned once (single burn), and (3) exotic grass that burns frequently. Sampling was conducted during one wet season (August to October 2008) and again in the following dry season (February to March 2009). Overall, fine fuel loading was highest in native grass, but this was due to woody fuels rather than grass fuels. Percent of dead grass fuels increased with the transition from wet to dry season, and this increase was more pronounced for exotic grasses. Fuel continuity was highest in frequently burned exotic grass. Differences in grass phenology and fuel continuity may contribute to differences in fire frequency among native and exotic grass patches. Fuel management focused on prescribed fire should be used in conjunction with restoration of tree canopy to reduce fuels and limit development of a grass-fire cycle.

Keywords: alien species, fuel loads, fuel management, grass, Guánica Commonwealth Forest, invasive species, Puerto Rico, subtropical dry forest

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INTRODUCTION

Invasive non-native plant species have been linked to altered fire regimes in ecosystems worldwide (Brooks *et al.* 2004). One commonly cited example is that of introduced pasture grasses that alter anthropogenic fire regimes in dryland systems, particularly those with little history of natural fire (D'Antonio and Vitousek 1992, Williams and Baruch 2000). By affecting fuel loads and structure, invasive grasses can alter the frequency, timing, intensity, and spatial extent of fires. This can result in altered fire effects that can ultimately change ecosystem structure and function. This phenomenon has been described as the grass-fire cycle, in which invasive grasses and fire generate a positive-feedback cycle that increases dominance of fire-promoting grasses (D'Antonio and Vitousek 1992, Brooks *et al.* 2004).

As such, invasive exotic grasses, through their effects on fire, represent a threat to tropical dry conservation and management. Although tropical dry forests make up 42 % of all tropical forests worldwide (Van Bloem *et al.* 2004), 97 % of these forests are at risk from multiple threats (Miles *et al.* 2006). Invasive exotic grasses are one of the most immediate and damaging of these threats, as they both degrade tropical dry forests and limit their restoration (Cabin *et al.* 2000). Exotic grass invasion has been associated with significant losses in forest cover, particularly on islands. For example, <5 % of the original dry forests remain in Puerto Rico (Murphy *et al.* 1995) and <10 % in Hawaii (Bruegmann 1996). This represents a significant loss of biodiversity as tropical dry forests on islands often contain significant numbers of endemic or rare species (Gentry 1995).

In contrast to the relationship between invasive non-native grasses and altered fire regimes, the relationship between native grasses and fire in tropical dry forests has received little attention. In some tropical dry forest landscapes, native grasses can also be a significant component of the ecosystem, particularly where past disturbances have decreased native tree canopy (Kellner *et al.* 2011) or where forest cover is naturally limited (Daehler and Goergen 2005). Little is known about how fuel characteristics of native grasses may compare to those of invasive exotic grasses within tropical dry forest landscapes. We compared fuels among areas dominated by either native or non-native grasses within a tropical dry forest reserve in Puerto Rico. Human-caused fires are a major management problem in dry forest in Puerto Rico (Brandeis and Woodall 2008). Previous studies have quantified fuel loadings within some Puerto Rico forests (Brandeis and Woodall 2008, Gould *et al.* 2008*a*), but no studies have focused directly on grass-dominated patches. We measured fuel loads, fuel continuity, and grass phenology (seasonal change in percent dead) by sampling in the field. We hypothesized that native and exotic grasses differed in fuel loads and rates of drying during the transition from wet to dry season. Our results are relevant to assessing the effectiveness of ongoing fuel management activities (e.g., prescribed fire in exotic grass patches) that are being conducted on the site.

METHODS

Study Site

This study was conducted in the 4500 ha Guánica Commonwealth Forest Biosphere Reserve in southwestern Puerto Rico (17°58' N, 66°55' W). Annual rainfall averages 860 mm, with most rainfall occurring during the peak of the Atlantic hurricane season from July through October (Murphy and Lugo 1986). Vegetation is primarily semi-deciduous subtropical dry forest (*sensu* Holdridge 1967), with smaller areas of evergreen dry forest and coastal scrub all on limestone substrate. Previous mapping studies (Lugo *et al.* 1978) have shown that 18 % of Guánica Forest includes vegetation associations with substantial grass components (scrub, thicket, and savanna). Island-wide, 52% of the dry forest life zone that is not active agriculture or urban development is now either grassland or open forest with grass understory (Gould *et al.* 2008*b*). Most trees are multi-stemmed with canopy height tending to be less than 10 m (Murphy and Lugo 1986). Grass-dominated patches with little to no tree cover occur throughout the forest, particularly in association with roads, trails, and areas prone to human-caused fires (Murphy *et al.* 1995, Wolfe and Van Bloem 2012).

Guánica Forest has a history of both natural and anthropogenic disturbance. Hurricanes are the primary natural disturbance, with an estimated average return interval of every 20 years (Van Bloem *et al.* 2005). Tree mortality during hurricanes is low, with most trees recovering through resprouting (Van Bloem *et al.* 2006). The last major hurricane to affect Guánica Forest was Hurricane Georges in 1998. The forest received sustained winds of 176 km h^{-1} to 184 km h^{-1}, but tree mortality was <2% (Van Bloem *et al.* 2006). Anthropogenic disturbances prior to the establishment of the forest reserve in the early 1900s primarily included cutting wood for charcoal production, digging of charcoal pits, farming, and grazing (Molina-Colon and Lugo 2006). Within the past few decades, human-caused fires that encroach into the forest have become an increasing problem (Wolfe and Van Bloem 2012). Few native woody species are capable of surviving even low-intensity fire (Wolfe 2009). Some species resprout following fire, but repeated fires kill sprouts and tend to decrease forest cover and promote exotic grass invasion (Santiago *et al.* 2008).

Open patches dominated by grasses (either native or exotic), ranging in size from <100 m^2 to up to 2.5 ha, occur within the boundaries of the Guánica Forest Reserve. These grass patches are primarily in the southern coastal part of the forest, which receives the most human use and is adjacent to 8 km of paved highway. Human-caused fires occur annually in this part of the forest and are a major management concern (M. Canals, Guánica Forest Manager, personal communication). For this study, we identified three principal grass patch types that occur in this section of the forest. Two of the grass patch types are dominated by exotic grasses while one is dominated by a native grass. For each type, patches as large as 2 ha occurred within the study area. Grass nomenclature follows Axelrod (2011).

Two exotic grass patch types were distinguishable on the basis of fire history and dominant grass species. Frequently burned exotic grass patches are dominated by two C_4 African grasses invasive in dry tropical regions worldwide (Williams and Baruch 2000): *Cenchrus ciliaris* L. (buffel grass) and *Megathyrsus maximus* (Jacq.) B.K. Simon and S.W.L. Jacobs (= *Panicum maximum* Jacq., guinea grass). These grass species were originally introduced into the area as pasture grasses and then later spread into the forest (Monsegur-Rivera 2009). Frequently burned exotic grass patches experience human-caused fires every 1 to 3 years, are almost completely devoid of trees other than *Leucaena leucocephala* (Lam.) de Wit, and tend to be located near roadsides and areas where human access is frequent (Wolfe and Van Bloem 2012). The second type of exotic grass patches are dominated by *Megathyrsus* and have burned once. Patches of this type tend to have a more developed woody component than do frequently burned patches, but tree canopy remains limited. Scattered non-native trees such as *Leucaena* and *Prosopis juliflora* (Sw.) DC. regularly occur, as well as occasional natives including: *Bucida buceras* L., *Bursera simaruba* (L.) Sarg., and *Exostema caribaeum* (Jacq.) Roem. & Schult. (Wolfe 2009). Patches sampled in this study were known to have burned once 20 to 30 years ago, which coincided with initial grass invasion.

A third grass patch type is dominated by a native grass, *Uniola virgata* (Poir.) Griseb., a large C_4 bunchgrass that is found throughout the greater Antilles in dry coastal areas (Monsegur 2009). Tree canopy cover is limited where *Uniola* is dominant, but native woody species that commonly co-occur with *Uniola* include: *Croton discolor* Willd., *Comocladia dodonaea* (L.) Urb., *Exostema caribaeum* (Jacq.) Roem. & Schult., *Lantana involucrata* L., and *Pictetia aculeata* (Vahl) Urb. (Thaxton and Van Bloem, University of Puerto Rico, unpublished data). Little is known of the ecology of *Uniola* or the ecological factors that promote formation of *Uniola* patches. However, the *Uniola* patches used in this study were not known to have burned within the past 30 years (M. Canals, personal communication).

Field Methods and Statistical Analyses

We sampled fuel biomass (live and dead from 0 m to 2 m in height) within 3 to 4 patches (each at least 0.25 ha in size) of each grass cover type (native grass, frequently burned exotic grass, and once burned exotic grass). All patches were located within 6 km of each other within the southern part of Guánica Forest Reserve. We used a stratified random sampling approach by randomly locating 7 to 10 sample plots of 0.25 m^2 (50 cm × 50 cm) each within each patch. This resulted in 30 sample plots per grass cover type and a total of 90 sample plots for the entire study. Fuels were collected during one wet season (September to October 2008) and one dry season (February to March 2009). Samples were dried to constant weight in a forced-air drying oven at 70 ºC; sorted into categories of live and dead grass, litter, live shrubs, downed dead wood (1 hr, 10 hr, and 100 hr timelag [TL]); and weighed. For each grass cover type, the proportion of samples containing grass fuel was used as a measure of fuel continuity.

We analyzed differences in fuel loads, seasonal changes in percent dead grass, and grass fuel continuity among grass patch types. We used PROC MIXED in SAS (Vers. 9.1; SAS Institute, Cary, North Carolina, USA) to assess the effect of grass cover type (fixed effect) on fuel loads within each fuel category. The largest wood category (100 hr TL) was absent from all but one sample plot, so it was not included in analyses. We generated linear contrasts to test two *a priori* hypotheses regarding the influence of grass type (exotic or native). The first contrast compared native grass patches with all exotic grass patches combined, while the second compared fuel loads between exotic grass patches that had burned once with those that experienced frequent fire (every 1 to 3 years). In a second analysis, we used PROC MIXED to compare the percent dead grass among patch types and between dry and wet seasons. For both analyses, we included patches as a random effect and a Satterthwaite approximation was used to generate appropriate error degrees of freedom. Finally, we used a Cochran-Mantel-Haenszel test in PROC FREQ to determine if grass fuel continuity, as measured by the frequency of samples containing grass fuel, differed among patches of each grass cover type (native grass, frequently burned exotic grass, and once burned exotic grass).

RESULTS

Total surface fuel biomass differed significantly among patch types ($P = 0.05$), with average biomass of native grass patches (1.28 ± 0.2 kg m^{-2}) 24 % higher than that of exotic single-burn patches (1.03 ± 0.17 kg m^{-2}) and 73 % higher than that of frequently burned exotic grass (0.74 ± 0.07 kg m^{-2}) (Figure 1a, Table 1). Linear contrasts indicated that the overall difference between native grass and exotic grass pooled was significant ($P = 0.044$), but the difference between frequently burned and single burn exotic grass was not ($P = 0.185$).

Differences in fuel biomass among grass patch types related to differences in the amount of small dead wood (1 hr TL fuels) and live shrubs, rather than differences in grass fuels. Both 1 hr TL fuels ($P = 0.006$) and shrub bio-

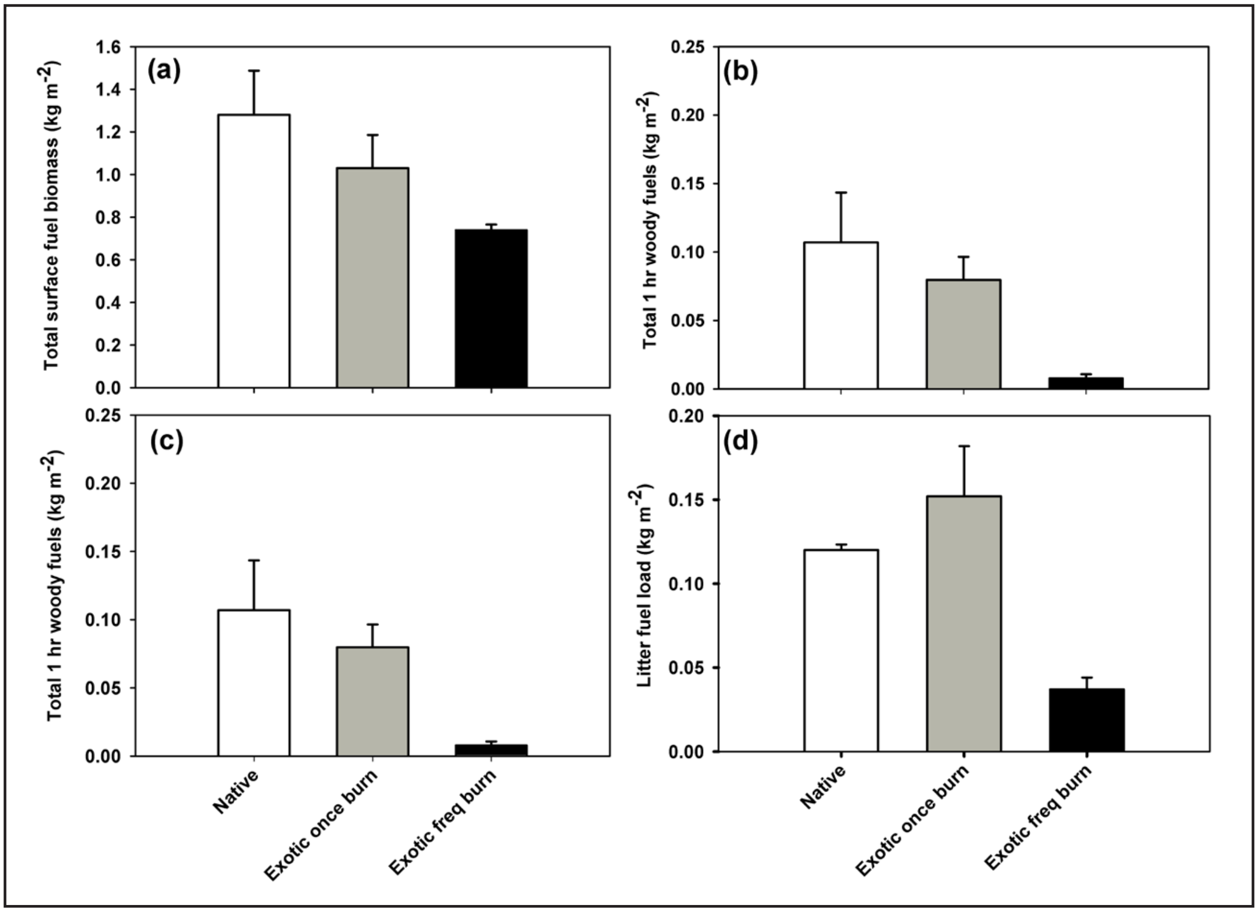

Figure 1. Surface fuels in patches dominated by native grass, frequently burned exotic grass, and once burned exotic grass: (a) total fine surface fuels, (b) live shrub biomass, (c) small dead (1 hr TL) woody fuels, and (d) litter.

Table 1. ANOVA results (main effects and linear contrasts) for fuel load by type among native and exotic grass patches. Sample size equals 30 for each grass cover type (native, exotic frequently burned, exotic single burn) and 60 for pooled exotic. Significant *P*-values are indicated by bold text.

	Grass cover type		**Native vs. exotic (pooled)**		**Exotic (frequent vs. single burn)**	
	$F_{(2,76)}$	P	$F_{(1,76)}$	P	$F_{(1,76)}$	P
Total biomass	3.04	**0.05**	4.18	**0.044**	1.79	0.185
Grass	0.69	0.503	1.38	0.243	0.01	0.924
Litter	8.56	<**0.001**	1.01	0.318	15.93	<**0.001**
Shrub	4.21	**0.018**	5.94	**0.018**	2.33	0.13
1 hr TL wood	5.41	**0.006**	5.24	**0.025**	5.35	**0.024**
10 hr TL wood	0.32	0.73	0.04	0.848	0.59	0.445

mass (live branches and leaves) (*P* = 0.018) differed significantly among site types and in a pattern similar to that of total biomass (Figure 1b and 1c, Table 1). Small dead wood fuels were 34 % higher in native grass patches than in exotic single burn patches, which in turn were nearly 10 times greater than that of frequently burned exotic grass patches. Linear contrasts indicated that both differences between native and exotic, and differences between single burn and frequently burned exotic grass patches were significant (Table 1). In contrast, total standing grass biomass averaged 0.71 ± 0.09 kg m^{-2} ($P = 0.50$), but did not differ significantly among site types (Table 1).

Litter fuels differed significantly among site types ($P < 0.001$), but the pattern was not the same as that for fine live and dead woody fuels. There was no overall significant difference between native and exotic patches (linear contrast $P = 0.32$), but among exotic grass patches, single burned contained >4 times more litter fuel than frequently burned (Figure 1d).

The percent of dead grass fuels tended to increase with the transition from wet to dry season, but this increase was more pronounced for exotic grass patches (Figure 2). This resulted in a significant site-by-season interaction ($P = 0.004$). The main effects of site type and season were also significant ($P = 0.004$ and $P < 0.001$, respectively). Percent dead in exotic grass in the wet season was 40 % and 53 % (exotic single burn and exotic frequently burned, respectively) and increased to 77 % and 90 % in the dry season. These increases were significant ($P < 0.001$ for the test of simple main effects for both variables). In contrast, percent dead for native grass was higher than that of exotic grass during the wet season, but did not increase significantly with the transition to the dry season ($P = 0.13$).

Overall, 70 % of sample plots contained grass fuels, but this measure of fuel continuity differed significantly among grass cover types when adjusted for patches (Cochran-Mantel-Haenszel test: $\chi_2^2 = 15.16$, $P < 0.001$). Fuel continuity was highest in frequently burned exotic grass (96 %). This was nearly twice the fuel continuity in either exotic single burn (58 %) or native grass (52 %).

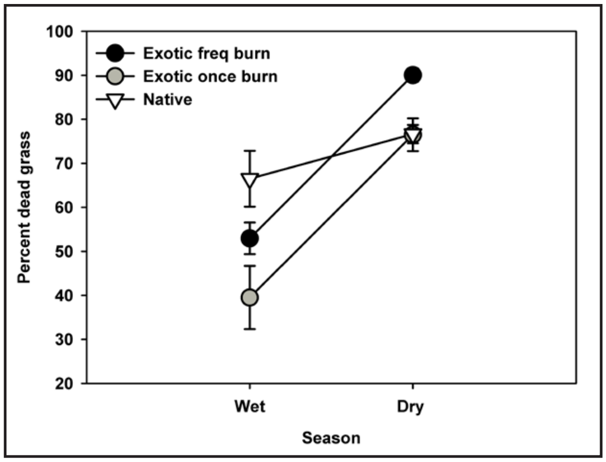

Figure 2. Change in percent dead grass from wet to dry season within patches dominated by native grass, exotic grass burned frequently, and once burned exotic grass.

DISCUSSION

The presence of large C_4 grasses (either native or exotic) within this subtropical dry forest reserve produces patches of flammable fine fuels, but exotic grass invasion does not increase total fuel loads above that of native grasses. Both native grass- and exotic grass-dominated patches contained substantial amounts of continuous fine fuels capable of producing damaging fires. However, in contrast to subtropical and tropical savannas in which exotic grass invasion increased fuel loads above that of native grasses (Lippincott 2000, Platt and Gottschalk 2001, Rossiter *et al.* 2003), native grass patches in our study site supported total fine fuel loads higher than (and grass fuel loads equal to) those of exotic grass patches. This suggests that observed differences in recent fire frequency among grass patch types is probably not directly due to differences in fuel loading.

Differences in fuel loading between native and exotic grass patches were caused by differences in woody fuels rather than grass fuels. Native grass patches supported higher surface fuel biomass than did exotic grass patches.

This appeared to result from native grass patches having higher shrub and dead wood biomass. This may result from differences in fire history among grass types. Few native woody species in Puerto Rican dry forest are fire adapted (Wolfe 2009), thus the prevalence of woody species, particularly natives, tends to decrease with fire frequency. The presence of a well-developed native shrub community in association with native grass patches suggests that they have burned infrequently, if at all. Furthermore, fuel loads in grass-dominated patches tended to be lower than total fine fuel loads previously reported for more intact tropical dry forest in Puerto Rico (Brandeis and Woodall 2008, Gould *et al.* 2008*a*). This suggests that the primary effect of exotic grass invasion in this site has been not to increase total fuel amounts, but rather to alter fuel structure and phenology.

Differences in phenology and fuel continuity may contribute to differences in fire frequency among native and exotic grass patches. Native and exotic grasses exhibited differences in phenology during the transition from rainy to dry season. Exotic grasses, independent of species or fire history, displayed a more pronounced shift towards a higher proportion of dead grass per area during the dry season than did native grass. Furthermore, fuel continuity was higher in exotic grass than native grass, particularly in frequently burned patches. Finally, qualitative observations of *Uniola virgata* patches following fire suggest that its ability to survive topkill and resprout immediately after fire is limited (J. Thaxton, personal observation). In contrast, the two exotic grasses at our sites are both extremely fire tolerant and resprout rapidly following fire (Williams and Baruch 2000). All of these factors have likely contributed to exotic grass invasion triggering a grass-fire cycle that causes substantial loss of forest cover, while the presence of native grass, even over extensive areas of the forest, does not appear to have contributed to the development of this cycle.

Anthropogenic ignition sources are widespread in Guánica Forest, and fire driven encroachment of exotic grass into forest is a persistent problem. For this reason, forest managers at the site have used prescribed fire along roadsides during the beginning of the dry season in an attempt to reduce grass fuels and limit fire spread from roadside ignitions. This program has been carried out since the mid-1990s (M. Canals, personal communication) on a near-annual basis. Our data suggest that, while prescribed burning does temporarily reduce fuel loads, regrowth to the level that fuels will carry fire is rapid, and increased fire frequency may enhance fuel continuity. These results point to the importance of using a variety of fuel management strategies, including prescribed burning and mechanical control, as well as forest restoration. Santiago-Garcia *et al.* (2008) showed that the presence of *Leucaena leucocephala* canopy cover in a nearby site that was grass invaded and repeatedly burned significantly reduced fuel loads, decreased fire intensity, and lessened fire effects on native woody species. Since the non-native tree *Leucaena leucocephala* readily invades grass-dominated sites following fire and appears to have beneficial effects on native woody species (Wolfe and Van Bloem 2012), it may be advisable to use this species in a fuel management program for Guánica Forest.

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LITERATURE CITED

Axelrod, F.S. 2011. A systematic vadecum to the vascular plants of Puerto Rico. Brit Press, Fort Worth, Texas, USA.

Brandeis, T.J., and C.W. Woodall. 2008. Assessment of forest fuel loadings in Puerto Rico and the US Virgin Islands. Ambio 37: 557-562. doi: 10.1579/0044-7447-37.7.557

Brooks, M.L., C.M. D'Antonio, D.M. Richardson, J.B. Grace, J.E. Keeley, J.M. Ditomaso, R.J. Hobbs, M. Pellant, and D. Pyke. 2004. Effects of invasive alien plants on fire regimes. Bioscience 54(7): 677-688. doi: 10.1641/0006-3568(2004)054[0677:EOIAPO]2.0.CO;2

Bruegmann, M.M. 1996. Hawaii's dry forests. Endangered Species Bulletin 11: 26-27.

Cabin, R.J., S.G. Weller, D.H. Lorence, T.W. Flynn, A.K. Sakai, D. Sandquist, and L.J. Hadway. 2000. Effects of long-term ungulate exclusion and recent alien species control on the preservation and restoration of a Hawaiian tropical dry forest. Conservation Biology 14: 439-453. doi: 10.1046/j.1523-1739.2000.99006.x

D'Antonio, C.M., and P.M. Vitousek. 1992. Biological invasions, the grass-fire cycle and global change. Annual Review of Ecology and Systematics 23: 63-87.

Daehler, C.C., and E.M. Goergen. 2005. Experimental restoration of an indigenous Hawaiian grassland after invasion by buffel grass (*Cenchrus ciliaris*). Restoration Ecology 13: 380-389. doi: 10.1111/j.1526-100X.2005.00047.x

Gentry, A.H. 1995. Diversity and floristic composition of neotropical dry forests. Pages 145-194 in: S.H. Bullock, H.A. Mooney, and E. Medina, editors. Seasonally dry tropical forests. Cambridge University Press, England. doi: 10.1017/CBO9780511753398.007

Gould, W.A., G. Gonzalez, A.T. Hudak, T. Nettleton-Hollingsworth, and J. Hollingsworth. 2008*a*. Forest structure and downed woody debris in boreal, temperate, and tropical forest fragments. Ambio 37: 577-587. doi: 10.1579/0044-7447-37.7.577

Gould, W.A., C. Alarcón, B. Fevold, M.E. Jiménez, S. Martinuzzi, G. Potts, M. Quiñones, M. Solórzano, and E. Ventosa. 2008*b*. The Puerto Rico Gap Analysis Project. Volume 1: land cover, vertebrate species distributions, and land stewardship. USDA Forest Service General Technical Report IITF-GTR-39, Río Piedras, Puerto Rico, USA.

Holdridge, L.R. 1967. Life zone ecology. Tropical Science Center, San Jose, Costa Rica.

Kellner, J.R., G.P. Asner, K.M. Kinney, S.R. Loarie, D.E. Knapp, T. Kennedy-Bowdoin, E.J. Questad, S. Cordell, and J.M. Thaxton. 2011. Remote analysis of biological invasion and the impact of enemy release. Ecological Applications 21: 2094-2104. doi: 10.1890/10-0859.1

Lippincott, C.L. 2000. Effects of *Imperata cylindrica* (L.) Beauv. (Cogongrass) invasion on fire regime in Florida sandhill (USA). Natural Areas Journal 20: 140-149.

Lugo, A.E., J.A. Gonzalez-Liboy, B. Cintron, and K. Dugger. 1978. Structure, productivity, and transpiration of a subtropical dry forest in Puerto Rico. Biotropica 10: 278-291. doi: 10.2307/2387680

Miles, L., A.C. Newton, R.S. DeFries, C. Ravilious, I. May, S. Blyth, V. Kapos, and J.E. Gordon. 2006. A global overview of the conservation status of tropical dry forests. Journal of Biogeography 33: 491-505. doi: 10.1111/j.1365-2699.2005.01424.x

Molina-Colon, S., and A.E. Lugo. 2006. Recovery of a subtropical dry forest after abandonment of different land uses. Biotropica 38: 354-364. doi: 10.1111/j.1744-7429.2006.00159.x

Monsegur-Rivera, O. 2009. Vascular flora of the Guánica dry forest. Thesis, University of Puerto Rico, Mayagüez.

Murphy, P.G., and A.E. Lugo. 1986. Structure and biomass of a subtropical dry forest in Puerto Rico. Biotropica 18: 89-96. doi: 10.2307/2388750

Murphy, P.G., A.E. Lugo, A.J. Murphy, and D.C. Nepstad. 1995. The dry forests of Puerto Rico's south coast. Pages 178-209 in: A.E. Lugo and C. Lowe, editors. Tropical forests: management and ecology. Springer-Verlag, New York, New York, USA. doi: 10.1007/978-1-4612-2498-3_7

Platt, W.J., and R.M. Gottschalk. 2001. Effects of exotic grasses on potential fine fuel loads in the groundcover of south Florida slash pine savannas. International Journal of Wildland Fire 10: 155-159. doi: 10.1071/WF01016

Rossiter, N.A., S.A. Setterfield, M.M. Douglas, and L.B. Hutley. 2003. Testing the grass-fire cycle: alien grass invasion in the tropical savannas of northern Australia. Diversity and Distributions 9:169-176. doi: 10.1046/j.1472-4642.2003.00020.x

Santiago Garcia, R.J., S. Molina Colón, P. Sollins, and S.J. Van Bloem. 2008. The role of nurse trees in mitigating fire effects on tropical dry forest restoration: a case study. Ambio 37: 604-608. doi: 10.1579/0044-7447-37.7.604

Van Bloem, S. J., A. E. Lugo, and P. G. Murphy. 2004. Tropical dry forests. Pages 1767-1775 in: J. Burley, J. Evans, and J. Youngquist, editors. Encyclopedia of Forest Sciences. Elsevier, Oxford.doi: 10.1016/B0-12-145160-7/00176-9

Van Bloem, S.J., S. Molina-Colon, A.E. Lugo, P.G. Murphy, R.A. Ostertag, M. Rivera-Costa, I. Ruiz-Bernard, and M. Canals-Mora. 2005. The influence of hurricane winds on subtropical dry forest structure and nutrient pools. Biotropica 37: 571-583. doi: 10.1111/j.1744-7429.2005.00074.x

Van Bloem, S.J., A.E. Lugo and P.G. Murphy. 2006. Structural response of Caribbean dry forests to hurricane winds: a case study from Guánica Forest, Puerto Rico. Journal of Biogeography 33: 517-532. doi: 10.1111/j.1365-2699.2005.01450.x

Williams, D.G., and Z. Baruch. 2000. African grass invasion in the Americas: ecosystem consequences and the role of ecophysiology. Biological Invasions 2: 123-140. doi: 10.1023/A:1010040524588

Wolfe, B.T. 2009. Post-fire regeneration in subtropical dry forest of Puerto Rico. Thesis, University of Puerto Rico, Mayaguez.

Wolfe, B.T., and S.J. Van Bloem. 2012. Subtropical dry forest regeneration in grass-invaded areas of Puerto Rico: understanding why *Leucaena leucocephala* dominates and native species fail. Forest Ecology and Management 267: 253-261. doi: 10.1016/j.foreco.2011.12.015