Prioritizing and managing invasive plant species in Austin, TX
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Abstract

In an effort to minimize environmental and economic harm associated with exotic plant species invasion, land managers have focused their efforts on returning species composition to pre-development baselines in restoration areas. However, without a better understanding of the ecological linkages facilitating exotic plant invasion, efforts focused on eradication and control may be too expensive or unlikely to succeed. The goal of this paper is to review common components of species invasiveness in order to make management recommendations and prioritizations for all currently listed Central Texas invasive plants. Mechanisms of invasion, management strategies and prioritization are discussed in detail for individual species. Six of the 26 exotic species analyzed were determined to be a high management priority. The remaining 20 species should either be removed from management lists, or should be subject to additional monitoring and data collection to determine appropriate level of management effort. Current literature indicates that native and exotic species colonization processes do not differ and are controlled by propagule pressure (P), and the abiotic (A) and biotic (B) conditions of the site. Given that disturbance and spatial heterogeneity of resources are reliable factors explaining species invasion, in urban environments where disturbance and anthropogenic alterations to P, A, and B are common, species invasion will remain high. Focusing management strategies on treating the cause and not the symptoms of exotic plant species invasion may improve restoration success and minimize costs.

Introduction

An invasive species is defined as a species that is non-native (or alien) to the ecosystem under consideration and whose introduction causes or is likely to cause economic or environmental harm or harm to human health (Executive Order 13112). An introduction can be an intentional or unintentional escape, release, dissemination, or placement of a species into an ecosystem as a result of human activity (Executive Order 13112). In turn, managers in the United States have focused on removal of exotics, attempting to restore invaded ecosystems to a historical baseline consisting of a pre-European native species assemblage (Marris 2011). However, native standing is not a guarantee of species evolutionary fitness and positive ecological function, and continued eradication of exotic species may in fact impair the ecosystem services humans rely upon (Davis *et al.* 2011). For example, African honey bees are an important global pollinator of agricultural crops and isolated forest fragments (Schlaepfer *et al.* 2011). Their eradication could minimize crop production and gene flow between fragmented landscapes. Given that non-native species add to local biodiversity and may contribute to ecosystem resilience and stability, increased understanding of their ecological linkages is necessary. Invasion biologists have focused much effort on predicting invasive species spread and developing control methods rather than documenting economic and social damage to society from impacts on ecosystem services (Pejchar and Monney 2009).

Recently the City of Austin has developed a framework for evaluating ecological function of riparian zones and has identified situations where passive restoration of these areas is beneficial over active vegetation management (City of Austin 2012). Understanding the main disturbance drivers shaping the vegetation community can provide insight on whether invasive exotic removal is warranted. Age class distribution of the dominant native and exotic tree species, the surrounding vegetation community, defining plant species, soil integrity, and hydrology can all be used as indicators for deciding the extent of intervention required at a potential restoration site (City of Austin 2012). The goal of this paper is to discuss common methods of species invasion and to make recommendations on the likely threats from and potential management strategies for Central Texas invasive plants.

Discussion

The following summarizes the general scientific knowledge on the methods of exotic species invasion. Specific exotic species listed in the City of Austin Invasive Species Management Plan and the Central Texas Invasive Plants list (City of Austin 2007) are then analyzed within this framework. And finally likely threats to ecosystem function of each species as well as potential management strategies within the City of Austin will be discussed.

Methods of Invasion

Numerous studies have linked anthropogenic modified landscapes with increased exotic invasion potential (Moffatt and McLachlan 2004; Pennington *et al*. 2010; Richardson *et al*. 2011; Sung *et al.* 2011). In Austin, exotic plant invasion is most strongly correlated to percent impervious cover, due to altered hydrology and extended dry periods (Sung *et al.* 2011). In urban environments invasion is primarily determined by disturbance and not interspecific competition between native and alien species (Sung *et al.* 2011). But what are the likely mechanisms surrounding exotic species success and how can managers diagnose a potential loss of ecological function? Successful species invasion requires sufficient propagule pressure and favorable abiotic and biotic conditions (Catford *et al*. 2009). These mechanisms are discussed in the context of various invasion hypotheses that can provide insight into the causes of exotic species invasion.

Propagule Pressure

Propagule pressure (the number of individuals introduced in an event multiplied by the temporal frequency of these events) is required for successful exotic species invasion (Catford *et al*. 2009). High propagule pressure can intensify invasion by increasing genetic diversity and the likelihood of being introduced into a favorable environment (Catford *et al*. 2009). If invaders dominate the seed pool they are more likely to dominate colonization and eventually become established into a new environment (Catford *et al*. 2009). A recent review by Richardson and Rejmanek (2011) has demonstrated that the primary mode of exotic species introduction was for horticultural (62%) purposes and that dispersal was dominated by bird (50.8%) and wind (18%) vectors. These findings are significant because protected horticultural exotics, which are nurtured and allowed to produce large seed banks, live in close proximity to invasible (urban) environments where propagules can be dispersed ubiquitously throughout the city. Management strategies focused on invasive removal from public lands may be unable to combat propagule dispersal pressure from nearby private lands. In addition, the **global competition** hypothesis states that with an increasing number of species introductions, the higher the likelihood that a competitive species will be in the invading species pool (Catford *et al*. 2009). The global species pool is more diverse than the local, increasing the chance that it will contain species that have traits that enable them to out-compete local species (Catford *et al*. 2009).

Abiotic characteristics

Abiotic characteristics such as water availability, nutrients, soils, and light must be advantageous for a species invasion to occur. Many invasion hypotheses attribute exotic success to changing resources; specifically, when disturbance is combined with high ecosystem productivity (Catford *et al*. 2009). In Texas and the arid southwest, hydrology is one of the major limiting abiotic factors dictating the vegetation community and ecosystem services provided by riparian systems (Richardson et al. 2007). Specifically, the hydrologic associations with available nitrogen may be of biggest concern. Groffman *et al.* (2002) found that hydrologic changes associated with urbanization can cause riparian zones to be sources rather than sinks for nitrogen in urban watersheds. Lowering of the water table in riparian zones reduces denitrification (an anaerobic process that transforms nitrogen from a biologically available from to atmospheric nitrogen that is unavailable to most organisms) and increases nitrification (a microbial process transforms nitrogen from a form that is unavailable to most organisms to a biologically available one) (Groffman *et al.* 2002). Additional sources of nitrogen include urban landscapes receiving fertilizers that then are exported as runoff. Increasing the supply of available nitrogen in urban riparian zones can result in increased invasion potential. For example, *Arundo donax* may take advantage of anthropogenically enriched nitrogen levels in riparian ecosystems, thereby out-competing its native competitors (Coffman 2007). These findings are consistent with the **fluctuating resource** hypothesis which states that increasing resource levels provides opportunity for invasion and invaders only become dominant in high productivity systems (Davis *et al*. 2000). Resource levels can increase by either boosting supply (e.g. abiotic disturbance, eutrophication) or decreasing resource use (e.g. die back of resident plants) (Davis *et al.* 2000). Contrary to popular belief invasive exotic species do not necessarily exhibit hydraulic advantages when compared to co-occurring native species (Pratt and Black 2006). Pratt and Black (2006) found no significant difference in cavitation resistance (a functional trait representing how tolerant a woody plant is to water stress), hydraulic efficiency, and xylem density found between five North American exotic species and their co-occurring natives. In fact, invasive exotic species rarely outperform native species in any growth category. Based on 72 independent native-invasive plant comparisons, exotic species were not significantly more likely to have higher growth rates, competitive ability, or fecundity (Daehler 2003). Daehler (2003) concludes that there are no super-invaders and that resource availability and altered disturbance patterns associated with human activities differentially increase the performance of invaders over natives. This again stresses the importance of fluctuating resources but also supports the **empty niche** and **disturbance invasion** hypotheses (Catford et al. 2009). Increasing anthropogenic disturbance events will alter current successional paths creating conditions unfavorable to locally adapted natives. This loss of local biodiversity creates unused niches which are then filled by exotic species better adapted to current site conditions (Catford et al. 2009). Managers should view exotic invasions as a symptom and not a cause of ecosystem change.

Biotic characteristics

Classic niche theory predicts that a species will invade, excluding the local resident, when fitness advantages outweigh any niche differences (MacDougall *et al*. 2009). In the previous section we described how altered nutrient dynamics and disturbance can modify the local environment creating unused niches. But how can the biotic interactions (herbivory, pathogens, competition, etc.) of exotic species help increase their fitness advantages over natives? The **enemy release** hypothesis dictates that exotic species can, upon entry into a new range, lose its natural enemies (herbivores, pathogens) that formerly limited its population size in its home (native) range (Catford *et al.* 2009). Resources previously allocated for defense can be used for growth and reproduction thereby facilitating fitness advantages and invasion (Catford *et al.* 2009). Many exotic plants and animals have shown fewer fungal or viral pathogens in their new rather than source ranges (Alpert 2006). In addition, exotic species can possess novel weapons that limit competitive exclusion. Production of allelopathic chemicals is a common biological attribute potentially contributing to exotic species invasion (Ren and Zhang 2008). Although biotic differences between exotic and native species can facilitate invasion there are instances where these interactions can have negative consequences. For example, the **reckless invader** hypothesis predicts that characteristics that facilitate invasion under certain environmental conditions (rapid growth and high fecundity) may be disadvantageous to invaders when conditions change (Alpert 2006). Such tradeoffs may explain transient invasions and demonstrate the importance of understanding ecosystem drivers behind invasion. For example, introduced populations of *Silene latifolia* bloom earlier and longer than their native counterparts, which might expose them to early or late frosts (Alpert 2006). Missed mutualistic opportunities, biotic resistance and local adaptation of natives can also negatively impact exotic invasion potential (Alpert 2006).

Synthesis

Most evidence suggests that native and exotic colonization do not differ and both are affected by propagule pressure and the abiotic and biotic conditions of the local environment (Catford *et al*. 2009). Exotic invasion generally follows passive models whereby invaded communities are primarily structured by non-interactive factors, such as altered disturbance and dispersal limitation, which are less constraining on the exotics which then dominate (MacDougall and Turkington 2005). Overall, disturbance is the main driving force behind exotic invasion of natural ecosystems. Identifying the main disturbance driver is critical in determining the success of exotics and their impact on native plant communities (Lake and Leishman 2004). Research has shown that sites without any disturbance do not support exotic species invasion (Lake and Leishman 2004). In urban environments anthropogenic modification has facilitated exotic species invasion on public lands by increasing the propagule pressure of exotic plants through large horticultural seed banks on private and public lands, increased resource supply and open niches as a result of disturbance, and implementation of management strategies lacking understanding of biotic links and fitness advantages of exotics. Unless managers and policy makers consider all the ecological links facilitating exotic species invasions, efforts to control and eradicate may be futile.

Central Texas Invasives
Currently known ecological mechanisms facilitating the invasion of specific species of interest for Central Texas are discussed in Appendix A. Summary information and recommendations on their management are given in Table 1. Table 2 ranks the management priority of these species in Central Texas based on factors contributing to invasiveness and the documented ecological impacts of each species. It is important to note that the management strategies recommended here focus primarily on the potential impacts by exotic species to ecosystem function. Although we recognize the importance of preserving native species for cultural and aesthetic purposes, their consideration is outside the scope of this paper. We recommend that specific threats by exotic species to native species of concern, for reasons other than ecological function, be considered on a case by case basis. This plant species list was selected from the City of Austin's Invasive Species Management Plan and Central Texas Invasive Plants Volunteer Field Guide and is not meant to be an absolute resource.

Conclusions

Exotic species invasion is strongly linked to anthropogenic modification (Moffatt and McLachlan 2004; Pennington *et al*. 2010; Richardson *et al*. 2011; Sung *et al.* 2011) and its success is primarily driven by propagule pressure and alterations to the abiotic and biotic conditions of the environment (Catford *et al*. 2009). Exotic species that are most dominant in a new range have high propagule pressure and have the ability to alter or can take advantage of anthropogenically modified environments, where the abiotic and biotic conditions have been severely changed (Catford *et al*. 2009). As species invasion is expected to alter ecological structure and function (Davis 2009; Catford 2009), both positive and negative changes to ecological function must be considered throughout the management process. Therefore, management prioritizations of exotic species in modified environments should be based on answering the following questions:

1. Does the exotic species under consideration have high propagule pressure?
2. Does the exotic species under consideration alter or possess traits that allow it to take advantage of modified abiotic characteristics of the local environment?
3. Does the exotic species under consideration alter or possess traits that allow it to take advantage of modified biotic characteristics of the local environment?
4. Has the exotic species under consideration been shown to negatively alter ecological function of the local environment?

Exotic species that have high propagule pressure, either alter or take advantage of the modifications in the abiotic and biotic factors of an environment, and have been shown to negatively alter ecological function should be considered of highest management priority. Conversely, exotic species that do not show the above-mentioned characteristics and/or have insufficient data to make a conclusion should be considered lower management priority. Table 2 ranks the management priority of 26 central Texas exotic plants listed by the City of Austin Invasive Species Management Plan and/or the Central Texas Invasive Plants Volunteer Field Guide as invasive and in need of management. Results of this study have found that only 6 of the

26 species evaluated were identified as a High management priority, with none of the species having documented research providing evidence to answer yes to all four of the above mentioned questions (Table 2). Until further scientific data is gathered it is recommended that species receiving a Low or Medium ranking (Table 2) be carefully monitored and not actively controlled at this time. It is also recommended that any species that ranked none (Table 2), or were negative to all four considerations listed above, be considered for removal from local invasive species lists. Overall, spatial heterogeneity of resources has proven to be the most robust and reliable hypothesis for explaining species invasion (Davis 2009). In urban environments where nutrient enrichment and disturbance is the norm, species invasion is likely to remain high. We understand that there is value in removing exotic species for cultural, economic, and aesthetic purposes, and that control of low and medium priority species may be deemed necessary under certain situations; but, we encourage land managers to consider the ecological mechanisms involved in an exotic plant species invasion as well as the potential consequences of invasive species management practices on ecological function and ecosystem services before embarking on removal and/or control practices for these species.

Recommendations

- Management efforts should focus on exotic plant species that received a very high/ high ranking (Table 2). Giant cane, Chinese tallow, privets, Chinaberry, hydrilla, and catclaw vine are likely candidates for active control, management, and restoration.
- Until further scientific data is gathered it is recommended that species receiving a Low or Medium ranking (Table 2) be carefully monitored and not actively controlled at this time.
- Extensive data collection and scientific evaluation is necessary in order to ascertain where and when to actively manage exotic species. Identifying priority City of Austin parcels, where multiple ecological functions and invasibility is high, is necessary before management decisions can be finalized.
- Education campaigns are necessary to inform citizens and businesses on how management of exotic species on public lands is strongly influenced by private plant propagule sources. Species with high propagule pressure and a high ornamental/ horticultural value will continue to spread into public lands until private seed sources can be controlled. Species that are widely distributed by urban birds (Chinese tallow, privets, and Chinaberry), humans (hydrilla, Johnsongrass, Bermudagrass, KR blustem and Malta star-thstle), and wind (tree of heaven and Chinese parasol tree) will continue to establish on our public land until sources are controlled. Educational efforts, like the Grow Green Program, targeting the businesses involved in promoting or planting this species are paramount.
- Exotic species management practices should encompass more ecological approaches where treating the cause and not the symptom is the primary focus. For example, disturbance and anthropogenic alterations of abiotic factors (water, nutrients, etc.) are a leading cause of exotic plant invasion (Davis 2009). Restoring the physical processes, such as water holding capacity of the soil or nutrient decomposition, should precede attempts at species control when practicable. Successful restoration projects should focus on restoring physical as well as biological processes (Tongway and Ludwig 2011). An exotic species is likely to reinvade a site if the environmental conditions (abiotic and biotic) contributing to its establishment success and spread remain.

Table 1: Summary of Central Texas invasive plants invasion mechanisms and management strategies.

Species	Common Name	Methods of Invasion	Distribution	Management Recommendations
Ailanthus altissima	Tree of Heaven	Disturbance Mediated growth (ruderal).	Disturbed urban areas, along roadsides, forest edges, and open canopy environments	Focus efforts on reducing site disturbances and maintaining vegetation cover. Continued disturbance, through vegetation removal, that opens up available niches only facilitates the spread and establishment of ruderal plant species
Albizia julibrissin	Silk Tree	Disturbance, fluctuating resources (hydrologic alterations create open niche).	Stream banks, abandoned farmland, roadsides, and open sites	Management strategies should focus on research as information on Silk Tree ecological linkages and impacts to ecosystem function are limited. Reducing planting and spread of seeds through education, reducing ecosystem disturbances, and promoting dense forest canopies can reduce the likelihood of Silk Tree spread into central Texas environments.
Arundo donax	Giant Cane	Fluctuating resources (↑ N), enemy release	Lakes, streams, drainage channels and other riparian areas	Restoring the riparian zone's capacity to retain water or reducing nitrogen enrichment (increasing soil carbon) should precede any attempts at biological control or native species establishment. Herbicides coupled with mechanical removal can be effective.
Bothriochloa ischaemum var. *songarica*	King Ranch Bluestem	Altered disturbance regimes (reductions in natural, historic growing season wildfires)	Rangelands, along roadsides, and in other disturbed mesic habitats	Increasing canopy cover should be seen as a BMP for minimizing spread. In urban environments KR Bluestem should be considered a low management priority due to its inability to invade existing forest patches and riparian zones coupled with the routine mowing of most other upland and non-forested habitats.
Broussonetia papyrifera	Paper Mulberry	Disturbance	Disturbed sites and forest edges, major urban centers of Texas	Requires further research on ecological effects and distribution patterns. Drought stress and water availability in Texas may limit future expansion.
Centaurea melitensis	Malta Star-thistle	Disturbance (likely soil compaction/ decreased water holding capacity)	Disturbed areas such as roadsides, fields, pastures, and rangelands	Mechanical, chemical, and biological control can be effective but will require multiple applications. Minimizing soil disturbance and increasing the water holding capacity should favor native plants.
Colocasia esculenta	Elephant Ear	Disturbance (flooding)	Lakes, rivers and in wetland habitat	Focus efforts on reducing hydrologic disturbances and maintaining vegetation cover. If control is desired a combination of manual and chemical control has proven effective. Due to positive association with water quality, control efforts should be carefully considered.
Cynodon dactylon	Bermudagrass	Propagule pressure, disturbance, and Fluctuating resources (high nitrogen)	Disturbed sites, gardens, agricultural fields, turf areas, landscape and forestry areas	Minimize soil/ vegetation disturbance to reduce spread, promoting vegetation cover (shade). Cleaning mowers and agricultural machinery after use in Bermudagrass infested areas can prevent spread of rhizomes and stolons.
Cyrtomium falcatum	Holly Fern	n/a	Coastal cliffs, rocky slopes, stream banks, and moist, shady places	Slow growth coupled with lack of habitat in Austin make for minimal management concern as an invasive species in Austin.

Eichhornia crassipes	Common Water Hyacinth	Empty niche and/or limiting similarity (Water Hyacinth establishes in areas that lack significant aquatic vegetation)	Lakes, ponds, rivers, wetlands and marshes	Managers should consider water hyacinth's positive association with water quality, aquatic organisms, and habitat complexity before selecting a management strategy. Chemical and biological control can be effective, but it is recommended that monitoring and education in conjunction with eradication efforts.
Firmiana simplex	Chinese Parasol Tree	Not enough available info, likely a Disturbance Mediated growth (ruderal) response	Roadsides, riparian areas, and forest margins	Further information is needed before conclusions on invasion strategy and recommendations for management can be suggested.
Hydrilla verticillata	Hydrilla	Fluctuating Resources (↑ N, P, K in substrate)	Lakes, rivers, reservoirs, ponds, and ditches. generally rooted in depths of greater to 20 feet	Biological control with triploid grass carp is most effective for large populations. Small populations can be controlled manually, but be sure to remove all cut biomass to minimize spread. Managing for reduced substrate fertility should also be considered a BMP for hydrilla control and native aquatic plant establishment.
Ligustrum spp.	Glossy Privet, Chinese Privet	Propagule Pressure, Enemy Release, Fluctuating Resources (↑ N), Empty Niche (few extant shrubs), Limiting similarity (functionally distinct species), disturbance	Abandoned home sites and can readily invade riparian areas, forest edges, and abandoned lots and farmlands	Reduce propagule pressure; introduce biological controls, reduce soil nitrogen. When possible, mimic natural succession by creating intermediate site disturbances. Focus on increasing native understory/shrub community.
Lonicera japonica	Japanese Honeysuckle	Disturbance, Enemy Release	Edge habitat, positively correlated with urbanization	With minimal propagule pressure and range limitations (moisture), the ability of Japanese Honeysuckle to exploit the disturbed urban environments in Austin will be limited. Management strategies should focus on decreasing site disturbance; improve soil texture, and promoting native diversity.
Macfadyena unguis-cati	Catclawvine	Disturbance (likely loss of soil structure and compaction) and phenotypic plasticity (enhanced leaf adaptations)	Riparian areas, woodlands, plantations, orchards, and near human habitations and disturbed sites	Chemical and mechanical control is considered largely impractical. Biological control has been shown to be effective, especially in the early stages of an infestation. Reducing soil compaction/increasing organic matter content in the soil should be viewed as a BMP for Catclawvine control.
Melia azedarach	Chinaberry	Hydrologic alterations (urban drought), Fluctuating resources (↑ N), propagule pressure, disturbance	Roadsides, forest margins, abandoned homesteads, and other disturbed sites	Restoring the riparian zones capacity to retain water and reducing nitrogen enrichment should precede any attempts at Chinaberry removal or native species establishment. Control seed sources.
Nandina domestica	Heavenly Bamboo, Nandina	Increased competitive ability (light use efficiency), fluctuating resources (increased N and acidification in urban riparian zones)	Riparian areas, high nutrient and moisture requirements	Requires further research on ecology and invasion potential. Reduce soil nitrogen levels. Increase native understory/shrub diversity.
Photinia spp.	Taiwanese/ Fraser's Photinia	n/a	Open woods, thickets, disturbed sites, and riparian habitats	Requires further research on ecology and invasion potential. Based on the limited distribution and low seed production it seems unlikely that either photinia species will significantly expand their range in the future.

Phyllostachys aurea	Golden Bamboo	Disturbance	Old home sites, expands rapidly following disturbance	Ecological research, reduction of ecosystem disturbance, promotion of vegetation cover (shade). Erratic seed production, lack of sexual reproduction, and drought and shade intolerance may result in invasions being localized and potentially vulnerable to changing environmental conditions. More information needed on control mechanisms.
Pistacia chinensis	Chinese Pistache	n/a	Riversides and in cultivated areas	Limited information on its distribution and response to changing abiotic conditions do not allow for conclusions to be made. Based on growth characteristics it is likely that hydrologic disturbances in urban environments could facilitate the spread of this drought tolerant species.
Pueraria montana var. lobata	Kudzu	Disturbance, propagule pressure	Full sun and well-drained soils of right-of-ways, forests edges, stream banks, and disturbed eroded environments	Management strategies should focus on observation and research. The current lack of quantitative research on the ecological effects of kudzu is limiting the understanding of its current and future effects on ecosystem function. Drought related seedling mortality will limit spread to Central Texas.
Pyracantha coccinea	Scarlet Firethorn	Disturbance	n/a	Ecological research, reduction of ecosystem disturbance, promotion of vegetation cover (shade). Passive restoration following site disturbance has been shown to effectively reduce cover.
Rapistrum rugosum	Bastard Cabbage	Disturbance	Disturbed soils of agricultural fields, roadsides, forests, and some riparian locations	Focus efforts on reducing site disturbances and maintaining vegetation cover. Continued disturbance, through vegetation removal, that opens up available niches only facilitates the spread and establishment of annual plant species. Use of herbicides can facilitate spread of this species.
Sorghum halepense	Johnsongrass	Disturbance	Open Forest Patches, old agricultural fields, roadsides	Minimize soil/vegetation disturbance to reduce spread, promoting vegetation cover (shade). It follows similar successional paths as native herbaceous species and will not pose a lasting threat to restoration success in reforested floodplains.
Tamarix ramosissima	Salt Cedar	Fluctuating resources (hydrologic alterations, lowered water tables in urban and dammed watersheds' create open niche).	moist riparian areas	Limited propagule pressure should minimize spread into Central Texas environments. Drawdowns or mimicking/maintaining natural flood regimes could be utilized to limit the invasibility of riparian habitats. Mechanical and biological control are relatively ineffective, costly and time consuming.
Triadica sebifera	Chinese Tallow	Propagule Pressure (bird dispersed), Fluctuating resources (↑N, K, Mg), Enemy release, phenotypic plasticity, disturbance.	low, swampy places and riparian areas but can also invade dry upland environments	Control seed sources, minimize disturbance, and reduce soil nitrogen, potassium and magnesium. Biological control not feasible due phenotypic plasticity. Mechanical removal, herbicides, and fire have been employed with minimal success.

Table 2: Management Prioritization of 26 Central Texas exotic plants. Prioritization ranking based on number of yes answers to each category; Very High = yes in all 4 classes, High = yes in 3 classes, Medium = yes in 2 classes, Low = yes in 1 category, and none = yes in 0 categories.

Species	Priority Ranking	Propagule Pressure	Abiotic	Biotic	Impact to Ecological Function
Arundo donax (Giant Cane)	High	**No** - Does not produce viable seeds, lacks sexual reproduction, low genetic diversity (Bell 1997). However, rhizome fragments are dispersed through substantial flood events and disturbance associated with construction (Boland 2008)	**Yes** - Takes advantage of elevated nitrogen levels (Ambrose 2007; Coffman 2007; Quinn 2007).	**Yes** - Enemy release, lacks herbivores (Wijte et al. 2005).	**Yes** – Reduced arthropod diversity that may degrade habitat for birds (Herrera and Dudley 2003).
Triadica sebifera (Chinese Tallow)	High	**Yes** – Prolific seed production, grackle is primary disperser (USDA 2012). Popular landscape tree.	**Yes** – Nitrogen, potassium and manganese linked to accelerated invasion (Burns and Miller 2004; Siemann and Rogers 2007).	**Yes** - Enemy release, lacks herbivores (Rogers and Siemann 2004). Allelopathic (Conway et al. 2011)	**No** – Considered desirable plant for birds (USDA 2012). Does not stop native plant succession in Texas (Bruce et al. 1995).
Ligustrum spp. (Privet)	High	**Yes** – Prolific seed production (Morris et al. 2002), Birds primary distributer (USDA 2012). Popular landscape tree.	**Yes** – Distribution strongly linked to disturbance (Loewenstein and Loewenstein 2005). Most likely hydrologic (Sung et al. 2011)/nitrogen (Brantley 2008) alterations.	**Yes** - Enemy release, lacks herbivores (Morris et al. 2002). Limiting similarity hypothesis (Green and Blossey 2011).	**No** – Increases to system NPP is beneficial to ecosystem function (Ehrenfeld and Toth 1997; Costanza et al. 2007). Important food for deer and songbirds (Stromayer et al. 1998; Wilcox and Beck 2007)
Melia azedarach (Chinaberry)	High	**Yes** – Prolific seed production, birds primary distributer ((Miller 1998). Popular landscape tree.	**Yes** – Takes advantage of hydrologic disturbance, drier conditions in urban environments (Sung et al. 2011). Possibly elevated nitrogen (Groffman et al. 2002)	**Yes** – Allelopathic properties suppress radish growth in lab studies (Hung et al. 2003).	**No** – No scientific documentation linking Chinaberry to decreased ecological function at this time. Likely important food for some bird species
Hydrilla verticillata (Hydrilla)	High	**Yes** - High rate of growth and dispersal give it the ability to invade multiple ecological niches (Sousa 2011). Reproduces through fragmentation, tubers, turions, and seeds (Langeland 1996).	**Yes** - Soil fertility has been identified as a key mechanism facilitating the dominance of Hydrilla over co-occurring natives (Van et al. 1999).	**No** – No scientific documentation of biotic advantages at this time.	**Yes/ No** – Shown to reduce macroinvertebrate diversity (Colon-Gaud 2003), positive and negative effects on wading birds (Esler 1992).
Macfadyena unguis-cati (Catclawvine)	High	**No** – Does not have a persistent seed bank (Vivian-Smith and Panetta 2004). Limited info on seed production, viability and spread.	**Yes** – Enhanced leaf adaptations, such as light use efficiently, can increase competitive advantages (Osunkoya et al. 2010), can grow under poor soil structure conditions (Downey and Turnbull 2007)	**Yes** – Root tuber densities likely to pose competitive advantage by impacting native species (Downey and Turnbull 2007).	**Yes** - Suffocating mature trees, destabilizing banks, creating intense root competition, and reducing habitat for native species (Downey and Turnbull 2007).
Ailanthus altissima (Tree of Heaven)	Medium	**Yes** – Prolific seed production can grow both by seed and vegetatively (Kowarik and Saumel 2008; USDA 2012).	**No** – Growth associated with disturbance, but no specific variables have been identified (Landenberger et al. (2009).	**Yes** - Allelopathic properties shown to significantly influence surrounding plant populations (Lawrence et al. 1991).	**No** - Researchers in Greece have found higher overall species diversity in plots containing Tree of Heaven versus plots dominated by the native Downy Oak (Fotiadis et al. 2011).

Lonicera japonica (Japanese Honeysuckle)	Medium	**No** –drought related seedling mortality (Sasek and Strain 1990) and lack of pollination (Larson et al. 2002).	**Yes** – Tolerate elevated cadmium (Liu et al. 2009), and altered soil texture (Surrette and Brewer 2008).	**Yes** - Enemy release, lacks herbivores (Schierenbeck et al. 1994).	**No** – Not linked with native species loss (Surrette and Brewer 2008), negatively correlated to invasion potential (Loewenstein and Loewenstein 2005)
Albizia julibrissin (Silk Tree)	Medium	**Yes** – Abundant seed production and rapid growth (USDA 2012).	**Yes** – Nitrogen fixer (USDA 2012), can alter nutrient dynamics. Takes advantage of altered hydrology (Burton et al. 2005).	**No** – No scientific documentation of biotic advantages at this time.	**No** – No scientific documentation of negative impacts to ecological function at this time.
Nandina domestica (Heavenly Bamboo)	Medium	**Yes** – Abundant seed production and rapid growth (USDA 2012).	**Yes** – Enhanced light use efficiency can increase competitive advantages (Knox and Wilson 2006)	**No** – No scientific documentation of biotic advantages at this time.	**No** – No scientific documentation of negative impacts to ecological function at this time.
Phyllostachys aurea (Golden Bamboo)	Medium	**No** – Although rapid growth (USDA 2012), species rarely produces seeds (Filgueiras and Mango 2007).	**No** – No scientific documentation of abiotic advantages at this time.	**Yes –** Root crushing of tree seedlings reduces competition (Griscom and Ashton 2003).	**Yes –** Root crushing of tree seedlings and may arrest forest succession (Griscom and Ashton 2003).
Sorghum halepense (Johnsongrass)	Medium	**Yes** – Rapid growth, long lived, abundant seed production, wind and animal dispersed (Holm et al. 1991; USDA 2012)	**No** – No scientific documentation of abiotic advantages at this time.	**Yes** – Allelopathic properties, can alter soil biochemistry (Chrzanowski and Rout 2009)	**No** – Will not alter forest succession, does not pose a lasting threat to restoration success in reforested floodplains (McLane et al. 2011).
Tamarix ramosissima (Salt Cedar)	Medium	**No** – No known populations in Austin currently (Sung et al. 2011; City of Austin 2012). Limited info on seed production, viability and spread.	**Yes** – Can take advantage of altered hydrology, regulated flow (Beauchamp and Stromberg 2007; Huddle et al. 2011).	**No** – No scientific documentation of biotic advantages at this time.	**Yes** - alters decomposition rates of leaf litter (Kennedy and Hobbie 2004), decreases macroinvertebrate diversity (Bailey et al. 2001). Conversely, provides bank stabilization and habitat for birds (Ellis 1995).
Cynodon dactylon (Bermudagrass)	Medium	**Yes** - Stands of Bermudagrass can establish by seed and sprigs with rapid vegetative spread (USDA 2012).	**Yes** – Increased nitrogen in soil, shown to increase the spread and success of Bermudagrass (Cohn et al. 1989, Harp et al. 2008).	**No** – No scientific documentation of biotic advantages at this time.	**No** – No scientific documentation of negative impacts to ecological function at this time.
Bothriochloa ischaemum var. *songarica* (KR Bluestem)	Medium	**Yes** – Still widely planted for erosion control, re-vegetation projects, fodder, and rangeland improvement (Gabbard and Fowler 2006).	**No** – No scientific documentation of abiotic advantages at this time.	**No** – No scientific documentation of biotic advantages at this time.	**Yes/ No** – Depending on soil conditions, KR can impact plant production, soil organic carbon and nitrogen, nitrogen mineralization, soil microclimate, and litter decomposition rates (Ruffner et al. 2012).
Firmiana simplex (Chinese Parasoltree)	Low	**Yes** – Wind and water dispersed seeds are self-pollinated and widely distributed (Miller et al, 2010).	**No** – Growth associated with disturbance, but no specific variables have been identified.	**No** – No scientific documentation of biotic advantages at this time.	**No** – No scientific documentation of negative impacts to ecological function at this time.
Broussonetia papyrifera (Paper Mulberry)	Low	**No** – Although rapid growth occurs species rarely produces seeds (Miller et al. 2010).	**Yes** - Soil texture, soil moisture and organic matter content could aid establishment (Malik and Husain 2007).	**No** – No scientific documentation of biotic advantages at this time.	**No** – No scientific documentation of negative impacts to ecological function at this time.

Pyracantha coccinea (Scarlet Firethorn)	Low	**Yes** – Abundant seed production and rapid growth (USDA 2012).	**No** – No scientific documentation of abiotic advantages at this time.	**No** – No scientific documentation of biotic advantages at this time.	**No** – does not pose a lasting threat to ecological succession (Villalobos et al. 2010).
Pueraria montana var. lobata (Kudzu)	Low	**No** – drought related seedling mortality (Sasek and Strain 1990) Future spread not expected outside current range (Guertin et al. 2008).	**No** – No scientific documentation of abiotic advantages at this time.	**Yes** – Enemy release from herbivory (Li et al. 2011), Benefit from mycorrhizal fungi (Greipsson and DiTommaso 2006).	**No** – No scientific documentation of negative impacts to ecological function at this time.
Rapistrum rugosum (Bastard Cabbage)	Low	**Yes** - Rapid growth rate and high seed production (Simmons 2005)	**No** – No scientific documentation of abiotic advantages at this time.	**No** – No scientific documentation of biotic advantages at this time.	**No** – No scientific documentation of negative impacts to ecological function at this time.
Eichhornia crassipes (Common Water Hyacinth)	Low	**No** – Spread largely facilitated through human action (Williamson et al. 2009) and can be partially controlled.	**No** – No scientific documentation of abiotic advantages at this time	**Yes** –Often establishes in areas that lack significant aquatic vegetation and provides a novel increase to habitat complexity, which suggests an underutilized/ empty niche (Villamagna and Murphy 2010).	**No** - Increase water clarity and decreases dissolved oxygen, nutrient, and contaminant concentrations, benefit macroinvertebrates and fish abundance and diversity (Toft et al., 2003; Pimentel et al., 2005Villamagna and Murphy 2010).
Centaurea melitensis (Malta Star-thistle)	Low	**Yes**- Similar species (*Centaurea solstitialis)* has rapid growth and high seed production (USDA 2012). Seeds can be dispersed by humans and birds (USDA 2010).	**No** – No scientific documentation of abiotic advantages at this time	**No** – No scientific documentation of biotic advantages at this time.	**No** – No scientific documentation of negative impacts to ecological function at this time.
Pistacia chinensis (Chinese Pistache)	None	**No** – Propagation does occur by seed (Dunn et al. 1996), but productivity and dispersal is unknown.	**No** – Growth associated with disturbance, but no specific variables have been identified.	**No** – No scientific documentation of biotic advantages at this time.	**No** – No scientific documentation of negative impacts to ecological function at this time.
Cyrtomium falcatum (Holly Fern)	None	**No** – Slow growing (Peck 2011) with little info on dispersal and relative propagule pressure.	**No** – No scientific documentation of abiotic advantages at this time.	**No** – No scientific documentation of biotic advantages at this time.	**No** – No scientific documentation of negative impacts to ecological function at this time.
Photinia spp (Taiwanese/ Fraser's Photinia)	None	**No** – Although rapid growth, seed production is low (USDA 2012).	**No** – No scientific documentation of abiotic advantages at this time.	**No** – No scientific documentation of biotic advantages at this time.	**No** – No scientific documentation of negative impacts to ecological function at this time.
Colocasia esculenta (Elephant Ear)	None	**No** –Rarely produces seed, relies on vegetative growth and corm dispersal (Atkins and Williamson 2008).	**No** – No scientific documentation of abiotic advantages at this time	**No** – No scientific documentation of biotic advantages at this time.	**No** – Shown to reduce nitrate and phosphate concentrations and improve water quality (Bindu et al., 2008)

APPENDIX A

INVASION MECHANISMS AND MANAGEMENT STRATEGIES FOR CENTRAL TEXAS INVASIVES

***Ailanthus altissima* (Tree of Heaven)**

Tree of heaven is a rapidly growing deciduous tree introduced from China in 1784 as an ornamental (Texas Invasive Plant and Pest Council 2012). It possesses a relatively short lifespan, can grow both by seed and vegetatively, is drought tolerant (Albright et al. 2010), produces an abundant supply of seeds, has known allelopathic properties, and is shade and flood intolerant (USDA 2012). Distribution commonly occurs in disturbed urban areas, along roadsides, forest edges, and open canopy environments (Texas Invasive Plant and Pest Council 2012). Tree of heaven seeds can be dispersed by both wind and water, making riparian corridors especially vulnerable to propagule pressure (Kowarik and Saumel 2008). However, flood intolerance (Albright et al. 2010) would limit spread in riparian systems that maintain lateral connectivity, experiencing periodic flooding. Overall, the invasiveness of tree of heaven seems to be determined by disturbance and the availability of open niches. Although the allelopathic properties of the species have been shown to significantly influence surrounding plant populations, relatively rapid adaptation by native species has been suggested as a mechanism to nullify any competitive advantages gained (Lawrence *et al*. 1991). In fact, researchers from Greece have found higher overall species diversity in plots containing tree of heaven versus plots dominated by the native downy oak (Fotidis *et al.* 2011). They conclude that disturbance is the primary driving force dictating species composition in urban environments where tree of heaven is benefiting from the novel environments created by humans (Fotidis *et al.* 2011). Similarly, Landenberger *et al.* (2009) found that tree of heaven distribution was associated with urbanization and concluded that continued disturbance (e.g. development and vegetation removal) is likely to further its spread. Current management strategies designed to control tree of heaven spread focus on mechanical removal and herbicidal application (Texas Invasive Plant and Pest Council 2012). However, studies of succession (Li et al. 2008; Moore and Lacey 2009) demonstrate that the rapid growth, short life, high light requirements, and disturbance-mediated growth response of tree of heaven are characteristic of a ruderal life stage that represents minimal long-term threat to native species diversity and ecosystem function (Barbour *et al.* 1987). Continued disturbance, through vegetation removal, that opens up available niches only facilitates the spread and establishment of ruderal plant species (Barbour *et al.* 1987).

***Albizia julibrissin* (Silk Tree)**

Silk tree is a deciduous, leguminous tree introduced from Asia in 1745 for forage, biofuel, and ornamental purposes (Miller *et al*. 2010). Characterized as invasive throughout much of the southern and eastern United States, silk tree is characterized by rapid growth, a short lifespan, shade intolerance, nitrogen fixing ability and abundant seed production (USDA 2012). This species is commonly found growing along stream banks, abandoned farmland, roadsides, and open sites (Miller *et al*. 2010). A study of riparian woody plant diversity in western Georgia found silk tree to be a dominant tree species in the forest regeneration layer of the most urbanized study location (Burton *et al*. 2005). Burton *et al*. (2005) suggest that hydrologic shifts in urban riparian communities may result in upland species invasion. In addition, silk tree has been shown to occupy watersheds with less forest cover in western North Carolina (Kuhman *et* *al*. 2010). Kuhman *et al.* (2010) concluded that invasive plants are more common in formerly disturbed reforested locations rather than historically undisturbed locations. These findings suggest that silk tree invasion is most closely linked to the disturbance invasion and fluctuating resource hypotheses. The altered hydrologic conditions coupled with increased availability of open/edge habitat in urban environments results in increased silk tree invasion in urban environments. Currently, distribution in Texas is limited. A recent study by the Texas Forest Service found Silk Tree inhabiting less than 5 percent of U.S. forest lands in east Texas (Oswalt *et al.* 2011). The City of Austin failed to locate any specimens in a 2011 survey of 12 urban riparian forests (City of Austin 2011). However, preferred urban habitat in Austin will likely see additional spread of silk tree beyond its current range. Any invasion is likely to be temporally brief as both its short lifespan and shade intolerance would likely reduce long-term fitness advantages over natives. Management strategies should focus on research as information on ecological linkages and impacts to ecosystem function are limited. Reducing planting and spread of seeds through education, reducing ecosystem disturbances, and promoting dense forest canopies can reduce the likelihood of Silk Tree spread into central Texas environments.

***Arundo donax* (Giant Cane)**

Giant cane is a tall, perennial grass, typically found along lakes, streams, drainage channels and other riparian areas (Bell 1997). Introduced to North America in the early 1800s for bank stabilization, this plant is currently listed as an invasive pest throughout the warmer coastal freshwaters of the United States (Bell 1997, Texas Invasive Plant and Pest Council 2012). Patches dominated by giant cane have been shown to negatively impact arthropod diversity (Herrera and Dudley 2003). Giant cane grows rapidly, spreads vegetatively mostly through rhizome fragments during significant flood events and soil disturbance related to construction (bulldozers) (Boland 2008). It is tolerant of fire, and has a high C:N ratio (USDA 2012). Fire has been linked as a potential colonization mechanism giving the plant a foothold in ecosystems by clearing out native vegetation (Ambrose 2007). Overall, the invasiveness of giant cane can be linked to altered nutrient dynamics, growth form and reproductive strategies. Floodplains anthropogenically enriched with nitrogen have been shown to support *A. donax* invasion whereas sites with low levels did not (Ambrose 2007; Coffman 2007; Quinn 2007). Elevated levels of available nitrogen, typical of urban areas, increases the invasion potential of riparian sites by creating an unused niche for colonization (Coffman 2007). However, the growth form of giant cane coupled with a lack of herbivory increases its competitiveness over native species, allowing it to become invasive. A combination of enemy release, vegetative growth, and production of unviable seeds (Wijte *et al.* 2005) affords it the ability to allocate more resources to growth than its co-occurring natives. However, a lack of sexual reproduction could be considered a reckless invasion strategy where giant cane is unable to adapt to changing environmental conditions. If vegetation management is desired, mechanical removal coupled with herbicide use has been shown to be an effective control method (Bell 1997; Texas Invasive Plant and Pest Council 2012). Laboratory tests have shown that biological control with the Arundo scale may also be an effective management strategy (Goolsby *et al.* 2009). Additionally, lowering available nitrogen in the soil could help reduce future spread and colonization (Ambrose 2007). The addition of soil carbon (e.g. sawdust) can be an effective strategy for reducing soil nitrogen levels and reducing exotic species invasion (Averett et al. 2004). Increasing denitrification by improving the water holding capacity of riparian soils could also reduce available nitrogen (Groffman *et al.* 2002).

Bothriochloa ischaemum var. *songarica* (KR Bluestem)

KR (King Ranch) bluestem is a perennial grass species that was introduced into the United States from Europe and Asia in the 1930s (USDA 2012; Gabbard and Fowler 2006). It is still commonly used for erosion control, re-vegetation of plowed or graded areas, fodder, and rangeland improvement (Gabbard and Fowler 2006). KR bluestem is characterized as fast growing, shade intolerant, and moderately fire and drought tolerant (USDA 2012). Commonly found growing in rangelands, along roadsides, and in other disturbed mesic habitats, it is distributed throughout the Southern United States (Texas Invasive Plant and Pest Council 2012). Its spread and invasiveness in Central Texas environments has been attributed to a combination of deliberate introduction and broad environmental tolerances (Gabbard and Fowler 2006). Although study plots dominated by KR bluestem have been shown to have a 30% reduction in species diversity (Gabbard and Fowler 2006), its impacts to ecological function are highly variable. Ruffner *et al*. (2012) found that patches dominated by KR bluestem may not significantly impact ecological function, depending on soil type. Factors such as plant production, soil organic carbon and nitrogen, nitrogen mineralization rates, soil microclimate, and litter decomposition rates are not significantly different between KR bluestem dominated patches and native dominated patches on clay soils (Ruffner *et al*. 2012). However, significant differences in these functional parameters occur on sandy loam soils, suggesting that control or restoration efforts may be necessary under coarse-textured soil conditions (Ruffner *et al*. 2012). Overall, it appears that alteration of disturbance regimes, predominantly reductions in natural, historic growing-season wildfires, is the primary mechanism facilitating the spread and invasion of KR bluestem into Central Texas environments (Fowler and Simmons 2008). In fact, only shade and growing season wildfires have been shown to effectively control its spread (Fowler and Simmons 2008, Simmons *et al.* 2007). In urbanized areas, increasing canopy cover should be seen as a best management practice for minimizing spread of KR bluestem.

Broussonetia papyrifera (Paper Mulberry)

Paper mulberry is a deciduous tree introduced from Japan and China in the mid-1700s as a rapidly growing, ornamental shade tree (Miller *et al*. 2010). Found growing on disturbed sites and forest edges, this species colonizes by root sprouts and occasionally by animal dispersed seeds (Miller *et al*. 2010). An investigation of the vegetation composition of cedar glades of the southeast United States found paper mulberry distribution occurring in the most anthropogenically-impacted sampling site (Baskin and Baskin 2003). The authors suggested that human disturbance was the main driver facilitating the high number of invasive species observed at this site (Baskin and Baskin 2003). Current distribution in Texas is clustered around the major urban centers of Austin, Dallas, and Houston (USDA 2012; Texas Invasive Plant and Pest Council 2012). Factors such as soil texture, soil moisture and organic matter content are potential factors controlling paper mulberry distribution (Malik and Husain 2007). Invasion into the Himalayan foothills of Pakistan was inhibited by the availability of moisture (Malik and Husain 2007), suggesting that drought stress and water availability in Texas should limit future expansion. However, there still remains a lack of quantitative research on the ecological effects and distribution patterns of this plant.

Centaurea melitensis (Malta Star-thistle)

Malta star-thistle is a noxious annual weed that commonly invades disturbed areas such as roadsides, fields, pastures, and rangelands and is thought to have originated in the U.S. from

Europe sometime in the early 1700s (Ashigh 2010; Donaldson and Rafferty 2011). It reproduces by seeds, which can remain dormant for up to 10 years (USDA 2010), and can tolerate drought conditions, often appearing to spread more quickly in drier areas (Donaldson and Rafferty 2011). In grazing areas, Malta star-thistle outcompetes native grass species and can be poisonous to grazing animals when ingested (USDA 2010). Similarly to most annual plant species, Malta star-thistle invasion is dependent on disturbance creating open patches. As an early successional annual species, invasion is likely to be transient, lasting only as long as the disturbance regime is maintained. Therefore, treating the land and not the plant should be viewed as best management practice. Increasing the water holding capacity of the soil, through inputs of organic matter, and minimizing excessive disturbance and compaction will likely favor native plans. Additional research is needed to understand the abiotic and biotic factors leading to Malta star-thistle dominance.

Although few studies have been done that describe the different control methods of Malta star-thistle, it can likely be controlled in similar ways to the yellow star-thistle, a close relative (Donaldson and Rafferty 2011). Mowing has been successful in the past as a means of mechanical control (Donaldson and Rafferty 2011). Biological control may be an option as well; Balciunas and Villegas (2007) found that an unintentionally introduced fly species (*Chaetorellia succinea*) destroyed the seeds of yellow star-thistle, a close relative to *C. melitensis*. Another management technique uses prescribed burning; it kills the seeds before they become viable plants or else hinders the seeds from dispersing (Di Tomaso et al., 2006; USDA 2010). However, a single burn in early summer will not be enough to completely control this species within a certain region (Di Tomaso *et al.* 2006). Previous study has shown that yellow star-thistle was 99% controlled after three years of burning, and therefore Malta star-thistle may be able to be controlled in the same way (Di Tomaso *et al*. 2006). However, an increase in Malta star-thistle has been observed in some cases as well after a fire (Parsons and Stohlgren 1989). Chemical control could be used at different stages to stop growth and spread (USDA 2010; Donaldson and Rafferty 2011); the USDA recommends reducing invasive biomass first such as by mechanical removal or herbicide use followed by prescribed burning followed by the implementation of monitoring programs (USDA 2010).

***Colocasia esculenta* (Elephant Ear)**

Elephant ear is a wetland herbaceous perennial that was introduced from Asia in 1910 as an edible crop and later an ornamental (Bindu *et al*. 2008, Texas Invasive Plant and Pest Council 2012). Commonly found growing along lakes, rivers and in wetland habitat in the Southeastern United States, it prefers warm freshwater environments where it can form dense stands once established (Garcia-de-Lomas *et al*. 2012; USDA 2012; Texas Invasive Plant and Pest Council 2012). Along the San Marcos in Central Texas River elephant ear can be found growing in high and low light environments and all types of substrate; but seems to grow best in silty anaerobic soils lining the riverbank (Atkins and Williamson 2008). It primarily reproduces vegetatively, via culm fragmentation and budding, with disturbance greatly enhancing its ability to spread (Texas Invasive Plant and Pest Council 2012). Elephant ear has been shown to withstand long dry summers of Seville, Spain, which suggest some drought resistance (Garcia-de-Lomas et al., 2012). Although the presence of elephant ear in a wetland region can reduce the nitrate and phosphate concentrations and improve water quality (Bindu *et al*., 2008), it is still considered an invasive species due to its ability to displace native aquatic plants along streams (Atkins and

Williamson 2008; Garcia-de-Lomas *et al*. 2012). It has also been shown to cause river narrowing along the San Marcos and other stream systems (Atkins and Williamson 2008; Garcia-de-Lomas *et al*. 2012). Although almost no research has been performed investigating the invasion biology of elephant ear, its dominance is likely facilitated by alterations to the natural disturbance regime of the hydrologic system, which has been suggested as a mechanism for increasing spread (Atkins and Williamson 2008). If vegetation management is desired, chemical control can be a viable method; the use of 1% solutions of triclopyr, glyphosate, or 2, 4-D has proven to be effective within 6 weeks (Galveston Bay Field Guide 2010). Mechanical methods of removal must be thorough because even remnants of leaves and roots can germinate (Galveston Bay Field Guide 2010). Atkins and Williamson (2008) found that mechanical cutting did not control the spread of this invasive even when followed by an application of herbicide. However, they found that manual removal combined with herbicide use was an effective method (Atkins and Williamson 2008). However, there still remains a lack of quantitative research on the ecological effects and distribution patterns of elephant ear in Central Texas environments.

***Cynodon dactylon* (Bermudagrass)**

Bermudagrass is a perennial grass species with probable Asian origin that was documented as an important grass in the United Sates in the early 1800s (USDA 2012). Commonly utilized for turf grass, Bermudagrass is characterized as fast growing and long lived, having high moisture and fertility requirements, shade intolerance, and drought, fire, and salt tolerance (USDA 2012). Stands can establish by seed and sprigs with rapid vegetative spread (USDA 2012). Commonly found growing in disturbed sites, gardens, agricultural fields, turf areas, landscape and forestry areas, Bermudagrass can quickly spread under irrigated/moist soil environments (Texas Invasive Plant and Pest Council 2012). However, shade intolerance should limit spread into established forest systems (Guglielmini and Satorre 2001). Overall, the invasiveness of Bermudagrass can be explained by high propagule pressure, disturbance, and nutrient enrichment. The likelihood of establishment on public lands is increased in urbanized environments where its use as a turf grass greatly increases its propagule pressure. Mowing practices can increase the spread of seed and sprigs into unintended environments. Vegetation control in urban environments is a common practice, causing disturbed environmental conditions, which can decrease canopy cover and open available habitat for Bermudagrass spread. Finally, increased available nitrogen in the soil, common in urban environments, has been shown to increase the spread and success of Bermudagrass (Cohn *et al.* 1989, Harp *et al.* 2008). Due to altered hydrology and reduced nitrification rates in urban environments (Groffman *et al.* 2002) spread into riparian zones is likely to increase. However, maintaining a dense canopy as well as reducing disturbances to the soil and vegetation should minimize its spread into these areas. Hand pulling for small infestations and herbicide has been effectively used for control efforts (Texas Invasive Plant and Pest Council 2012). Also cleaning mowers and agricultural machinery after use in Bermudagrass-infested areas can prevent spread of rhizomes and stolons (Texas Invasive Plant and Pest Council 2012).

***Cyrtomium falcatum* (Holly Fern)**

The holly fern is a widely-planted ornamental species across much of the Southeastern United States that originated from Japan (Peck 2011). This plant is found growing on coastal cliffs, rocky slopes, stream banks, and moist, shady places (Rotherham 2009). In Europe, the distribution and spread appears to coincide with the low winter temperature where its growth is hardiest where temperatures do not drop below -7ºC (Rotherham 2009). Due to the plant's affinity for moist environments, water is most likely the limiting factor for distribution and spread in Central Texas. Overall, holly fern's slow growth coupled with lack of habitat should limit its establishment. However, further information is needed before conclusions on invasion strategy and recommendations for management can be suggested.

***Eichhornia crassipes* (Common water hhyacinth)**
Water hyacinth is a free-floating aquatic plant originating from Brazil that has spread to over 62 countries, generally appearing between 40°N and 45°S (Lei and Bo 2004). Its spread is facilitated mainly through human action, often entering a new territory through ballast water and occupying reservoirs following dam construction (Williamson *et al*. 2009). Once established, Water hyacinth's rapid growth and ability to reproduce both sexually and asexually allow it to quickly spread, forming dense mats that float on the water's surface (Howard and Harley 1998; Padilla and Williams 2004). It is considered an invasive aquatic plant species largely due to its ability to alter transportation pathways, tourism activities, fisheries, wetland biodiversity, and the efficiency of irrigation and water power facilities (Lei and Bo 2004). Once introduced, it has been shown to alter ecological structure of aquatic systems by increasing water clarity and causing decreases in phytoplankton, dissolved oxygen, and nutrient and contaminant concentrations (Toft *et al.* 2003; Pimentel *et al.* 2005; Villamagna and Murphy 2010). These alterations in aquatic systems can have both positive and negative effects on the aquatic community. Diversity and total number of aquatic species can decrease due to limited availability of phytoplankton but can also increase as a result of a more complex habitat (Howard and Harley 1998; Villamagna and Murphy 2010). Water hyacinth has been shown to positively benefit macroinvertebrates and fish abundance and diversity which suggest a likely positive association with water bird abundance and diversity (Villamagna and Murphy 2010). The soci-economic and environmental effects on a specific waterbody are dependent upon the extent of invasion, use of the system, and control methods; thus careful consideration must take place prior to designing a management strategy. Although few studies have reported on the ecological condition prior to establishment, making it difficult to understand invasion mechanism and how it impacts ecological function, the fact that its presence increases ecological structure and can take advantage of a limited nutrient supply (Villamagna and Murphy 2010) suggests an empty niche and/or limiting similarity invasion mechanisms. Water hyacinth often establishes in areas that lack significant aquatic vegetation and provides a novel increase in habitat complexity, which suggests an underutilized niche (Villamagna and Murphy 2010).

A combination of mechanical, biological, and chemical controls is utilized for water hyacinth control. Eradication is difficult, so management efforts have focused on limiting both economic and ecological impacts. Mechanical control involves physically removing plants; however, studies have observed alterations in water quality surrounding these removal sites (Greenfield *et al.* 2007; Villamagna and Murphy 2010). It is labor intensive, costly, and only effective on small scales (Lu *et al.* 2007). Chemical control is costly with multiple treatments and can pollute the surrounding habitat (Lu et al., 2007; Villamagna and Murphy 2010). Because biological control has been found to be more cost efficient (Zimmermann *et al.,* 2004), this management method has been widely used (Evans and Reeder 2001; Sosa *et al.* 2004; Wilson *et al.* 2007). It is not toxic or labor intensive, but introduced species brought in as a control agent can attack non-target plants and alter community composition in the area (De Groote *et al.* 2003; Villamagna and

Murphy 2010). Thus, the effectiveness of biological control will vary by region (Lu *et al.* 2007). The joint use of chemical and biological control has been successful (Center *et al.* 1999). Lu *et al*. (2007) recommends a sustainable science approach involving monitoring, education, as well as integrative and adaptive management plans to control and attempt to eradicate this species. Lacking an effective restoration strategy following the removal of water hyacinth could lead to a decrease in water quality and a shift toward a cyanobacteria-dominated system (Villamagna and Murphy 2010).

***Firmiana simplex* (Chinese Parasol Tree)**

Chinese parasol tree is a deciduous, upright tree reaching 50 feet in height that was introduced from China in 1757 as an ornamental (Miller *et al.* 2010). Frequently found growing in coastal states, it inhabits roadsides, riparian areas, and forest margins (Miller *et al.* 2010). Its growth is characterized as fast growing with self-pollinating seeds being dispersed by both wind and water vectors (Miller *et al.* 2010). Little is known about the ecological linkages and invasion mechanism of this species. Based on its affinity for disturbed habitat (roadsides, riparian areas, and forest margins) it seems likely that a positive correlation with urbanization exists. However, further information is needed before conclusions on invasion strategy and recommendations for management can be suggested.

***Hydrilla verticillata* (Hydrilla)**

Hydrilla is a submersed macrophyte native to Asia; it first appeared in the U.S. in 1960 in Florida and is now found in all Gulf Coast states and as far west as California (Langeland 1996). Hydrilla was introduced mainly through ports in ballast water. (Ruiz *et al*. 1997; Monterroso *et al.* 2011). Its high rates of growth and dispersal give it the ability to invade multiple ecological niches (Sousa 2011), and these high growth rates allow it to compete effectively for needed sunlight (Langeland 1996). However, hydrilla can grow in low-light conditions as well and can therefore establish in deep water in addition to shallow, warm habitats (Langeland 1996). It can reproduce through fragmentation, tubers, turions, and seeds, thus increasing the likelihood of establishment (Langeland 1996). This species grows quickly and can deplete oxygen levels in the water, causing harm to aquatic invertebrates and plants (Madsen 1997). Certain species of wading birds have been negatively affected by the presence of hydrilla while other species have benefitted from the appearance of this invasive (Esler 1992). Hydrilla also affects macroinvertebrate biodiversity; less diversity is observed in areas with high hydrilla density (Colon-Gaud 2003). The plant shows high levels of polymorphism, varying its appearance depending on the surrounding environment, and can adapt to a wide range of environmental conditions (Sousa 2011). Generally, hydrilla starts sprouting around mid-April with water temperatures around 16°C (Spencer and Ksander 2001). Increased water fluxes usually limit macrophyte growth, but hydrilla plants are flexible and roots develop quickly giving it the ability to grow even in fast-flowing water (Sousa 2011). Additionally, substrate fertility has been identified as a key mechanism facilitating the dominance of hydrilla over co-occurring natives (Van *et al*. 1999). Van *et al*. (1999) found that hydrilla was dominant, outcompeting native *Vallisneria americana*, under high fertility levels (6.3, 2.1, and 4.2 g of N, P, K), whereas Vallisneria became the dominant species under low (< 1.6, 0.5, and 1.1 g of N, P, K) fertility levels, suggesting that anthropogenic enrichment of nitrogen, phosphorous and potassium could be facilitating hydrilla's competitive advantage. Once established, hydrilla has the potential to cause large economic impacts due to the high cost of management (Langeland 1996).

Biological control methods include the introduction of sterile triploid grass carp. However, they are non-specific herbivores that may begin to target other vegetation (Langeland 1996; Hershner and Havens 2008). No extensive damage of hydrilla has been reported with insects or other biological control agents (Langeland 1996), although the use of fungal pathogens has shown some positive results (Shearer 1998). Herbicides can be used to manage hydrilla invasions, although there are potential environmental impacts associated with their use (Hershner and Havens 2008). In addition, studies have shown that herbicide-resistant hydrilla can develop as a result of chemical control efforts (Michel *et al*. 2004; Arias *et al*. 2005). Mechanical removal is plausible on small scales, but because the species reproduces so quickly, these efforts will rarely be successful long-term or on large scales (Hershner and Havens 2008). Therefore, biological control may be the most efficient means of management, although there are still unknown effects that come with this method. Managing for reduced substrate fertility could also be considered a BMP for hydrilla control and native aquatic plant establishment.

***Ligustrum sinense* (Chinese Privet)**

Chinese privet grows as a shrub or small tree, typically reaching heights ranging from 5 to 12 feet (USDA 2012). It was introduced as an ornamental from China in 1852 and has become invasive throughout much of the Southern United States (USDA 2012). Privet is commonly found growing on abandoned home sites and can readily invade riparian areas, forest edges, and abandoned lots and farmlands where it forms dense thickets (Chess 2011; USDA 2012). Riparian areas are specifically vulnerable to invasion due to the plant's ability to tolerate both flooding and low light levels (Green and Blossey 2011). Chinese privet reproduces by sexual and vegetative means, producing abundant seed stocks which are readily distributed by birds (USDA 2012). Landscape plantings in urban environments provide seed sources for establishment in nearby disturbed environments, such as cleared forests, abandoned agricultural lands, and construction sites (USDA 2012). The intense propagule pressure resulting from large private seed banks coupled with vegetative growth ability results in populations being nearly impossible to control or eliminate (USDA 2012). Studies of distribution patterns have found that Chinese privet is positively correlated with increasing urbanization and negatively correlated with species richness (Loewenstein and Loewenstein 2005). Loewenstein and Loewenstein (2005) suggest that the proportion of edge habitat, small size of forest fragments, and disturbance are the potential ecosystem drivers of invasiveness. Locally, a recent study by Sung *et al.* (2011) found that invasion of exotic species, including Chinese privet, in Austin riparian forests was related to urbanization and most likely resulting from hydrologic disturbances. Sung *et al*. (2011) conclude that biological invasion is primarily determined by the disturbance (hydrology) and not interspecific competition between species. Hydraulic disturbances in urban riparian environments have been associated with elevated nitrogen levels due to a lowered water table reducing bacterial denitrification rates (Groffman *et al.* 2002). Evidence for a positive association between Chinese privet invasion and elevated nitrogen can be made from an analysis of net primary production (NPP) by Brantley (2008). Brantley (2008) found that Chinese privet invasion in Georgia Piedmont riparian forests was accompanied by a significant increase in total site NPP. An increase in NPP is an indication of nitrogen enrichment in terrestrial environments (Alberti 2009). This link between NPP increases and nitrogen enrichment gives support for the fluctuating resource hypothesis as the likely method of Chinese privet invasion.

Additional advantages for Chinese privet are likely attained through enemy release, empty niche, and limiting similarity hypotheses. Research on comparisons between *Ligustrum sinense* and *Forestiera ligustrina* growing in Tennessee found significantly lower levels of insect damaged leaves and higher stem elongation, leaf area and leaf mass ratios in *L. sinense* (Morris *et al.* 2002). These results suggest that enemy release from herbivory could result in greater resource use efficiency leading to a fitness advantage by Chinese privet over co-occurring natives (Morris *et al.* 2002). High resource use efficiency is the likely mechanism behind its elevated fruit production and could lead to increased propagule pressure and invasion potential to surrounding environments (Morris *et al.* 2002). Additionally, the empty niche and limiting similarity hypotheses have been suggested as mechanisms of privet invasion. Green and Blossey (2011) have suggested that with few extant shrub species in floodplain forests, Chinese privet is invading an open/underutilized niche; and, as a novel genus to North America, Ligustrum is taxonomically distinct from other species which may lead to functional differences that increase its invasiveness (Green and Blossey 2011). If an invading species is functionally distinct in its new environment it encounters minimal competition from native species and can fill an empty niche (Catford *et al.* 2009). Typically, management practices targeting Chinese privet have focused on mechanical removal and herbicide treatments (USDA 2012). Although Chess (2011) demonstrated that cutting followed by herbicidal application was effective at reducing Chinese privet re-growth in a North Carolina park system, overall invasive species cover was not significantly reduced. The niche created by Chinese privet removal was quickly filled by the invasive Japanese siltgrass and after eight years of monitoring Chess (2011) found no improvements in native species cover. Therefore, management strategies beyond mechanical and chemical methods are likely needed if improvements to species diversity and ecological function are desired. The use of biological control agents and soil amendments need to be investigated. Studies have shown that *Argopistes tsekooni*, a beetle native to China, is host specific and that grazing pressure limits growth of Chinese private demonstrating its potential as a promising candidate for biological control (Zhang *et al*. 2011). The addition of soil carbon (e.g. sawdust) can be an effective strategy for reducing soil nitrogen levels and reducing exotic species invasion (Averett et al. 2004). Any removal strategies should attempt to mimic natural succession and intermediate disturbance patterns. An intermediate level of anthropogenic disturbance in riparian and upland plant communities has been correlated to increased plant functional richness and diversity (Biswas and Mallik 2010).

***Ligustrum vulgare* (Common Privet) and *Ligustrum japonicum* (Japanese Privet)**

Common rivet and Japanese privet are naturalized species to many temperate areas of the eastern United States and both are similar to the Chinese privet (*Ligustrum sinense*) (Texas Invasive Plant and Pest Council 2012: USDA 2012). Both species possesses similar growth requirements and reproductive strategies as Chinese privet and management recommendations and invasion strategies suggested above will likely apply to all *Ligustrum spp*. in Central Texas.

***Lonicera japonica* (Japanese Honeysuckle)**

Japanese honeysuckle is an evergreen woody vine introduced into North America from Japan in the early 1800s as an ornamental species with valued deer browse and erosion control attributes (Miller *et al*. 2010). This climbing vine is characterized by rapid vegetative growth reaching heights of up to 80 feet tall in forest canopies (Miller *et al*. 2010). Japanese honeysuckle is both shade and fire tolerant, is relatively short lived, and produces bird-dispersed seeds (USDA 2012).

Although commonly found growing in the southeastern United States, drought related seedling mortality (Sasek and Strain 1990) and lack of pollination (Larson *et al.* 2002) has been linked to western range expansion limits. Studies of distribution patterns have found that Japanese honeysuckle is positively correlated with urbanization in central Georgia (Loewenstein and Loewenstein 2005). Loewenstein and Loewenstein (2005) suggest that the proportion of edge habitat, small size of forest fragments, and disturbance are the potential ecosystem drivers of invasiveness. In addition, enemy release and toxic tolerance could increase Japanese honeysuckle's advantage over natives. Research has shown that Japanese honeysuckle's compensatory response to herbivory allows for greater year-round production of leaf tissue when compared to a native South Carolina honeysuckle (Schierenbeck *et al*. 1994). The authors suggest that escape from herbivores is a partial explanation for invasion (Schierenbeck *et al*. 1994). Research by Liu *et al*. (2009) has shown that Japanese honeysuckle has a strong tolerance to cadmium and is a potential hyperaccumulator. Ability to tolerate changing environmental conditions could provide Japanese honeysuckle with a competitive advantage. However, research by Surrette and Brewer (2008) conclude that Japanese honeysuckle cover is the least important predictor of native species diversity in northern Mississippi forests. In other words, disturbance and soil texture, not competition from Japanese honeysuckle, is the main driver of native species decline (Surrette and Brewer 2008). In fact, the presence of Japanese honeysuckle has been shown to have a negative impact on invasion potential by limiting the establishment of other non-native plant species (Loewenstein and Loewenstein 2005). Although Japanese honeysuckle is the most frequently detected invasive species on east Texas forest lands (Oswalt and Oswalt 2011), drought and limited precipitation in Austin should limit seed viability and spread (Sasek and Strain 1990).

***Macfadyena unguis-cati* (Catclawvine)**
Catclawvine is a slow growing, woody, perennial vine that is native to tropical America (Vivian-Smith and Panetta 2004; Texas Invasive Plant and Pest Council 2012). Commonly found growing on fertile, well-drained alluvial soils (Vivian-Smith and Panetta 2004), catclawvine can form dense mats along river banks, riparian areas, woodlands, plantations, orchards, and near human habitations and disturbed sites (Williams 2002; Texas Invasive Plant and Pest Council 2012). It can grow in both low and high light environments reaching canopy heights of 15 meters or more (Vivian-Smith and Panetta 2004). Its ability to cause tree mortality is the main reason it is considered invasive in Australia (Downey and Turnbull 2007). In addition, catclawvine has been characterized as drought and saline tolerant, having an extensive root system with nodes capable of re-sprouting, rapidly growing, and able to grow in areas with poor soil structure and fertility (Downey and Turnbull 2007). In riparian areas, where it is considered at high risk of invading, catclawvine has the potential to alter ecological function by suffocating mature trees, destabilizing banks, creating intense root competition, and reducing habitat for native species (Downey and Turnbull 2007). Currently, there is relatively little scientific information on catclawvine's invasion mechanisms in riparian or other invaded systems (Vivian-Smith and Panetta 2004, Raghu and Dhileepan 2005). Osunkoya *et al*. (2010) found that several leaf physio-chemical traits were significantly different between catclawvine and other co-occurring natives, suggesting that phenotypic plasticity of traits may be partially responsible for invasion success. Catclawvine's ability to grow under poor soil structure conditions (Downey and Turnbull 2007) may also afford it a competitive advantage in highly disturbed or compacted environments. Therefore, reducing soil compaction and increasing organic matter content in the

soil should be viewed as a best management practice for controlling spread of catclawvine in urban environments. Although some control success has been achieved through the use of herbicides (Downey and Turnbull 2007), both chemical and mechanical control methods have been deemed largely impractical (Williams 2002; Downey and Turnbull 2007). Therefore biological control can play a crucial role in the control of catclawvine, especially in the early stages of an infestation (Williams 2002). Several species of insects have been successfully utilized to control catclawvine in both laboratory and field tests (Williams 2002; Downey and Turnbull 2007).

Melia azedarach **(Chinaberry)**
Chinaberry is a medium sized tree that was introduced from Asia as an ornamental in the mid-1880s (Miller 1998; Texas Invasive Plant and Pest Council 2012). Its growth is characterized as rapid with reproduction both occurring vegetatively and through seeds using birds as its primary dispersal method (Miller 1998; Texas Invasive Plant and Pest Council 2012). Chinaberry produces an abundant amount of seeds, is drought tolerant, and possesses known allelopathic properties (USDA 2012). It commonly inhabits roadsides, forest margins, abandoned homesteads, and other disturbed sites in the Southeast United States (Miller 1998; Texas Invasive Plant and Pest Council 2012). A 2008 survey of Texas Forest Service lands found Chinaberry to be sparsely distributed, occurring at 53 of 2,258 sampled sub-plots throughout east Texas (Oswalt and Oswalt 2011). Locally, Sung *et al*. (2011) found Chinaberry to be the most widely distributed exotic tree surveyed in Austin riparian forests. Sung *et al.* (2011) concluded that hydrologic disturbance, specifically drier conditions as a result of urbanization lowering the riparian water table, was the main ecosystem driver facilitating exotic species spread. From these results, the authors suggest a causal pathway from watershed urbanization to the invasion of alien species in riparian forests: (1) urbanization alters stream hydrology of a watershed by lengthening dry periods; (2) the altered hydrologic regime disturbs riparian forests and reduces ecosystem resistance; (3) alien species replace a native competitor within a same regenerating cohort; and (4) after several generations, the riparian forests become dominated by the alien invaders (Sung *et al*. 2011). Although Chinaberry is known to possess allelopathic properties (Hung *et al*. 2003; USDA 2012) invasion success appears to be primarily facilitated by ecosystem disturbance and not interspecific competition between native and alien species (Sung *et al*. 2011). Hydrologic disturbance in urban environments may be favoring Chinaberry invasion by fluctuating resource supply and creating open niches. Groffman *et al.* (2002) found that hydrologic changes associated with urbanization can cause riparian zones to be sources rather than sinks for nitrogen in urban watershed. Increasing nitrogen supply causes an ecosystem to be more susceptible to invasion, offering additional resources for Chinaberry establishment and growth. Additionally, increasing drought conditions in formerly mesic riparian environments decreases the success of hydrophilic vegetation and favors the growth of drought tolerant species, creating an open niche for colonization. As dry conditions worsen in riparian habitats the success of Chinaberry and other drought tolerant plant species is expected to increase. Management strategies designed to control Chinaberry should focus on restoring physical processes (e.g. water retention) prior to restoring biological processes (Tongway and Ludwig 2011). Increasing perennial plant cover, surface roughness, and soil porosity can increase water infiltration and groundwater recharge (Tongway and Ludwig 2011). Allowing for increased duration and frequency of localized flooding will increase lateral connectivity and groundwater recharge (Woosely *et al*. 2007). The addition of soil carbon (e.g. sawdust) can be an effective

strategy for reducing soil nitrogen levels and exotic species invasion (Averett *et al*. 2004). Due to prolific seed production (USDA 2012) leading to intense propagule pressure, source populations surrounding managed areas should be investigated prior to any vegetation removal. As long as the intense propagule pressure and hydrologic disturbances remains, mechanical removal may be ineffective at eliminating Chinaberry from Austin riparian forests (Sung *et al.* 2011).

***Nandina domestica* (Heavenly Bamboo, Nandina)**

Nandina is a bamboo-like evergreen shrub introduced from Asia and India in the Early 1800s as an ornamental (Texas Invasive Plant and Pest Council 2012). This rapidly growing species can reach heights up to 10 feet and is considered invasive throughout much of the southeastern United States (Texas Invasive Plant and Pest Council 2012). Nandina is characterized as having abundant seed production, fast vegetative growth, high fertilization and moisture requirements, low drought and fire resistance, shade tolerance, and an affinity for acidic soils (USDA 2012). A recent study by the Texas Forest Service found nandina inhabiting less than 0.01 percent of U.S. forest lands in east Texas (Oswalt *et al.* 2011). While nandina's ability to capture and efficiently use light has been suggested as a mechanism for competitive advantages (Knox and Wilson 2006), relatively little information exists related to its ecology and invasion potential. However, due to nandina's high moisture requirements and low resistance to drought, availability of water in central Texas will likely limit its spread. Growth and establishment would likely occur in moist riparian ecosystems flowing through urban environments. The increased nutrient enrichment (Groffman *et al.* 2002) and acidification (Driscoll *et al*. 2003) characteristic of urban soils in riparian zones could provide beneficial habitat for nandina in Austin. Overall, in Texas and the arid southwest, hydrology is the major abiotic factor dictating the vegetation composition and ecosystem services provided by riparian systems (Richardson et al. 2007). As urbanization is expected to continue in Central Texas, reductions in riparian groundwater levels may pose growth limitations and reduce habitat for nandina. Recent surveys by the City of Austin failed to document naturalized nandina specimens in all sampled urban riparian forests (City of Austin 2011). Management strategies should focus on researching the ecology and invasion potential of nandina in Austin. Further information is needed before conclusions on invasion strategy and recommendations for management can be suggested.

***Photinia serratifolia* (Taiwanese Photinia); *Photinia* x *fraseri* (Fraser's Photinia, Red-tip)**

There is limited scientific information on the ecology, distribution and invasion potential of both the Taiwanese and Fraser's (red-tip) photinia. Both are rapidly growing evergreen shrubs listed as invasive in Travis County, Texas (Texas Invasive Plant and Pest Council 2012). Taiwanese photinia was introduced from Asia as an ornamental, is shade tolerant, and has been found growing in juniper-oak woodlands on limestone slopes and near older residential developments (Texas Invasive Plant and Pest Council 2012). Current U.S. distribution is limited to a few counties in Georgia, Alabama, Mississippi, Louisiana, and Texas (USDA 2012; Texas Invasive Plant and Pest Council 2012). Fraser's photinia is a hybrid cross of Taiwanese photinia and Japanese photinia (*Photinia glabra*) and was discovered in an Alabama nursery in the 1940s (Texas Invasive Plant and Pest Council 2012). Fraser's photinia reaches heights of 12 feet, is shade intolerant, produces low amounts of seed, and has an affinity for acidic soils (USDA 2012). Preferred habitat for both species in Austin has been suggested to be open woods, thickets, disturbed sites, and riparian habitats (Nesom 2008). Based on the limited distribution and low seed production it seems unlikely that either species will significantly expand their range in the future. However, further information is needed before conclusions on invasion strategy and recommendations for management can be suggested.

***Phyllostachys aurea* (Golden Bamboo)**

Golden bamboo is a perennial member of the grass family introduced from Asia in 1882 as an ornamental (USDA 2012, Texas Invasive Plant and Pest Council 2012). Commonly found growing on old home sites, colonization can occur by both rhizomes and stolons with infestations rapidly expanding after disturbance (Miller *et al*. 2010). Golden bamboo is characterized as rapidly growing, long-lived, fire tolerant, drought and shade intolerant, and low moisture and fertility requirements (USDA 2012). In Southeastern Peru, Griscom and Ashton (2003) found that dense seands of golden bamboo have been shown to crush roots of native seedlings, thereby limiting growth and survival. They suggest that this mechanism may arrest forest succession by not allowing seedlings to recruit into larger size classes (Griscom and Ashton 2003). Although a native forest system will likely return following monocarpic flowering/die-off events (Griscom and Ashton 2003), the longevity of golden bamboo will likely reduce native species diversity and some aspects of ecological function in the short term. Current distribution of golden bamboo occurs in the southeast United States, California and Oregon (USDA 2012). In east Texas, recent surveys of U.S. forest lands found Golden Bamboo on 2 of the 2,258 sampled forest plots (Oswalt *et al*. 2011). Despite continued widespread urban ornamental planting, limited flowering and seed production (Filgueiras and Mango 2007) should result in low propagule pressure and spread of Golden Bamboo into Central Texas environments. This lack of propagule pressure coupled with potential drought and shade intolerance of golden bamboo should result in a limited number of invadable habitats. However, further information is needed before conclusions on invasion strategy and recommendations for management can be suggested.

***Pistacia chinensis* (Chinese Pistache)**

Chinese pistache is a commonly planted urban street tree that reaches heights of 25 feet (Texas Invasive Plant and Pest Council 2012). Introduced from China, this dioecious species is found in Texas, California, Georgia and Alabama occurring along riversides and in cultivated areas (Texas Invasive Plant and Pest Council 2012, USDA 2012). Chinese pistache flourishes in full sun and is tolerant of drought, saline conditions, and extreme heat and wind (Dunn *et al*. 1996). These characteristics suggest that hydrologic disturbances in urban environments would likely facilitate the spread of this drought tolerant species. However, limited information on its distribution and response to changing abiotic conditions do not allow for conclusions to be made at this time.

***Pueraria montana var. lobata* (Kudzu)**

Kudzu is a deciduous, climbing leguminous vine introduced into North America from Japan and China in the early 1900s for erosion control and livestock feed, and as an ornamental (Miller *et al.* 2010; Texas Invasive Plant and Pest Council 2012). Reports of kudzu cultivation for clothing and food production appear as early as 494 B. C. in parts of China (Li *et al*. 2011). This long history of cultural and economic associations with humans has led to the development of cultivars that are fast growing, adaptable to harsh environments, and that have developed resistance to some natural enemies (Li *et al*. 2011). Li *et al.* (2011) suggest that the repeated introduction, cultivation, and escapes back into the wild of these anthropogenically-selected cultivars have increased the invasion success of kudzu. Overall it appears that disturbance resulting from human activities is driving invasiveness in the Southeastern United States. However, Liu *et al.* (2012) also suggest that lack of human harvesting of kudzu in the United States is the main driver facilitating invasion, when compared to wild populations in China. In the United States kudzu grows best in full sun on well-drained soils commonly located along right-of-ways, forests edges, stream banks, and disturbed eroded environments (Miller *et al.* 2010; Texas Invasive Plant and Pest Council 2012). Although commonly found growing in the southeastern united states, drought related seedling mortality has been linked to western range expansion limits (Sasek and Strain 1990). Future spread of kudzu is not expected outside its current range (Guertin *et al.* 2008). With very few current reports of kudzu growing in Travis County and limited distribution throughout East Texas (Oswalt *et al*. 2011), the risk of invasive spread in Central Texas appears minimal. However, the current lack of quantitative research on the ecological effects of kudzu is limiting the understanding of its current and future effects on ecosystem function (Forseth and Innis 2004; Devine and Fei 2011).

***Pyracantha coccinea* (Scarlet Firethorn)**

Scarlet firethorn is a large evergreen shrub native to Southern Europe and Western Asia (Texas Invasive Plant and Pest Council 2012). This rapidly growing species can reach heights up to 8 feet and is considered invasive throughout much of the southern and eastern United States (Texas Invasive Plant and Pest Council 2012). Found in Travis and surrounding Texas Counties (Texas Invasive Plant and Pest Council 2012), scarlet firethorn is characterized as having abundant seed production, fast growth, high fertilization requirements, and low drought, fire, shade and salt tolerance (USDA 2012). A recent evaluation of scarlet firethorn invasion in southern France demonstrated that soil disturbance and removal of vegetation cover promote species establishment (Villalobos *et al*. 2010). The authors suggest that disturbance conditions immediately following field abandonment enhance survival and growth of scarlet firethorn. However, as passive recolonization of vegetation proceeds, future establishment of the invasive may be prevented (Villalobos *et al*. 2010). These results give support to the disturbance invasion hypothesis and demonstrate the potential ecosystem benefits of passive restoration. While increasing ecosystem disturbance is likely to cause additional species invasion in central Texas, threat from scarlet firethorn spread seems minimal. A study by Leidolf and McDaniel (1998) made comparisons between the vegetation composition of various Blackland Prairie regions and concluded that the plant composition found in Mississippi and Arkansas differs from Texas. They suggest that differences in soils, parent material, and moisture are responsible for the variation in vegetation, including scarlet firethorn (Leidolf and McDaniel 1998). With differing ecosystem processes driving the vegetation composition of the Blackland Prairie ecosystem of Central Texas, restoration and invasive species management needs will also likely differ (Leidolf and McDaniel 1998). Although additional research is need on the ecological linkages of scarlet firethorn in the United States, invasion will likely be transient and dependent upon frequent disturbance and moisture availability to facilitate spread.

***Rapistrum rugosum* (Bastard Cabbage)**

Bastard cabbage, also known as turnip weed, is an annual herb originating from Eurasia that has sporadically naturalized in North America (Simmons 2005). In Central Texas, it is commonly found growing in disturbed soils of agricultural fields, roadsides, forests, and some riparian locations (Texas Invasive Plant and Pest Council 2012, Simmons 2005). Bastard cabbage germinates in the fall or early winter, flowers relatively early in the spring, has a high growth rate, high seed production, and large rosette diameter (Simmons 2005) and can grow in a wide range of soil pH conditions (Chauhan *et al.* 2006). These characteristics, although not unusual among annual plant species, have been suggested as contributing to its invasion success (Simmons 2005). However, as an early-successional, annual species, bastard cabbage invasion is likely to be transient, lasting only as long as the disturbance regime is maintained. Chemical control is not recommended due to documented resistance to herbicides (Adkins et al., 1997; Simmons 2005). It has been suggested that continued herbicidal treatment may only exacerbate the spread of bastard cabbage and other undesirable species by maintaining an ecologically disturbed site where further infestation from the exiting seed bank is likely (Simmons 2005). Mechanical removal should also not be utilized because it is only effective if the entire taproot is removed (Texas Invasive Plant and Pest Council 2012) and it will also perpetuate further spread by maintaining a disturbed ecological condition. The best method for minimizing persistence and spread is to minimize disturbance and/or overseed with native species of a similar phenology. Simmons (2005) found that overseeding a site with similar competitive annual species (such as *Gaillardia pulchella*) is an effective mechanism for reducing the productivity of bastard cabbage and does not threaten non-target, native vegetation.

Sorghum halepense **(Johnsongrass)**

Johnsongrass is a perennial, warm-season grass introduced to the U.S. from Africa and temperate Asia in the early1800s for use as a pasture and forage grass (Anderson 1999, Miller *et al.* 2011). It is commonly found growing in old fields, rights of way, and open forest patches and is now considered invasive throughout the continental United States (Miller *et al.* 2011, USDA 2012). Johnsongrass generally appears in late April to early May (Anderson 1999) and grows best in warm, humid regions with mild winters (Holm *et al.* 1991). Its spread is facilitated mainly by seed dispersal, carried by water, wind, and animals; but, it can also grow and spread via rhizomes (Holm *et al.* 1991). Johnsongrass is characterized as rapidly growing, long-lived, fire tolerant, drought and shade intolerant, having a high moisture usage, allelopathic, and an abundant seed producer (USDA 2012). Although the allelopathic properties and ability to alter soil biochemistry of Johnsongrass have been suggested as a mechanism for increased invasion pressure (Chrzanowski and Rout 2009), disturbance is likely the main invasion driver. Thriving in old agricultural fields and open forest patches, Johnsongrass is an early successional species reliant on ecological disturbance and formation of open patches for its persistence. McLane *et al*. (2011) found that an exotic herbaceous species, including Johnsongrass, follow similar successional paths as native herbaceous species and may not pose a lasting threat to restoration success in reforested floodplains. In other words, Johnsongrass follows a similar successional pathway to other weedy disturbance tolerant herbaceous species. Newman (1993) suggests that maintaining undisturbed land around areas dominated by Johnsongrass can be used as a mechanism to limit its spread. Conversely, heavily managed, frequently disturbed landscapes will have a high probability of facilitating Johnsongrass spread. In these environments control can often be achieved through the use of herbicides (Knowles and Benson 1983). However, Johnsongrass is beginning to develop resistance to certain herbicides (Binimelis *et al*. 2009; Bradley and Hagood 2001; Reid *et al*. 1997), and its use may no longer be a best management option. The use of mycoherbicides (fungi) to control the spread of Johnsongrass has been documented (Chiang *et al.* 1989; Winder and van Dyke 1990) and may prove to be an effective biological control technique. Johnsongrass is more easily removed when young (Martin 2002);

thus small infestations can be controlled by removing the plants as early as possible. Larger infestations can be dealt with by mowing, heavy grazing, or repeated tillage/plowing (Chambers and Hawkins 2002). Xeriscaping is an option when trying to prevent further spread because Johnsongrass tends to thrive in moist habitats (Martin 2002); with plants that require little to no water, the need to irrigate is eliminated. Habitat types that are managed to maintain heavy shade and woody plant cover, such as riparian forests, do not need to manage Johnsongrass as invasions are likely to be transient and dependent on continued site disturbance (Newman 1993).

Tamarix ramosissima **(Salt Cedar)**
Salt Cedar is a phreatophytic shrub or small tree native to Eurasia that was introduced into North America in the mid-1800s for ornamental use and erosion prevention along streams and lakes (Di Tomaso 1998; Kennedy and Hobbie 2004). It is commonly found growing in moist riparian areas in the southwestern United States and is dependent upon groundwater for its growth and survival (James and Evans 1991). Salt Cedar spread is facilitated by water-driven seed dispersal into favorable bare, moist soil environments (James and Evans 1991). Once established, Salt Cedar has been shown to tolerate to a wide range of environmental stresses; such as drought, fire, grazing, cold, elevated salt concentrations and water inundation (Frasier and Johnsen 1991; Busch and Smith 1993; Hughes 1993; Vandersande *et al.* 2001). This species has been successful at colonizing high latitude regions as well, through phenotypic plasticity and adaptive evolution (Sexton *et al*. 2002). It can also re-sprout from roots quickly if the above-ground growth is damaged (Frasier and Johnsen 1991). Salt Cedar has been shown to alter decomposition rates of leaf litter thereby modifying the organic composition of surrounding soils (Kennedy and Hobbie 2004). This decrease in litter quality has been shown to negatively affect the abundance and overall diversity of stream macroinvertebrates that depend on leaf litter as a source of food (Bailey *et al*. 2001). Conversely, Salt Cedar trees provide important bank stabilization functions and habitat for many bird species (Ellis 1995); including, the endangered southwestern willow flycatcher (Sogge *et al*. 2008, Davis *et al*. 2011). Sogge *et al*. (2008) concluded that removal or control of Salt Cedar, in the absence of effective restoration activities, may actually decrease the net riparian habitat value for birds. Overall, hydrologic disturbance appears to be the most likely mechanism behind Salt Cedar's current invasion success (Huddle *et al.* 2011). Beauchamp and Stromberg (2007) found that Salt Cedar stand density is greatest in reaches where water flow is regulated and concluded that variation in flow and depth to groundwater are the main factors influencing its spread. Altering natural flood flows in the Southwestern United States is leading to cottonwood and willow-dominated forests being replaced by salt cedar stands (Beauchamp and Stromberg 2007). This in turn has led some researchers to suggest that drawdowns or mimicking natural flood regimes may be used as an effective control measure (Stromberg 2001; Bhattacharjee *et al*. 2006; Beauchamp and Stromberg 2007). Additionally, Harms and Hiebert (2006) found that cutting and burning after the application of herbicide reduced salt cedar cover; however, they observed regrowth because new seedlings were not treated. Depending on the characteristics of the site, mechanical removal or biological control may be implemented, but may be impractical. The use of biological control may only show gradual and temporary effects (Dudley and DeLoach 2004), and mechanical removal of the salt cedar stands can be costly and time-consuming (Shafroth *et al*. 2008). Lack of propagule pressure will likely limit spread into Central Texas environments. However, given the large amount of managed surface water along the Colorado River and its tributaries there is ample habitat for salt cedar to become established.

***Triadica sebifera* (Chinese Tallow)**

Chinese tallow is a fast-growing medium sized tree introduced to the United States from China in 1776 by Benjmain Franklin (USDA 2012). Tallow is generally found growing in low, swampy places and riparian areas but can also invade dry upland environments (USDA 2012). It can become the dominant plant when growing in disturbed vacant lots, abandoned agricultural lands, natural wet prairies, and bottomland forests (USDA 2012). Once established, Chinese tallow is virtually impossible to eliminate (USDA 2012). Birds, such as grackles and cardinals, and water are the major dispersal methods (USDA 2012). The invasiveness of tallow can be attributed to increasing propagule pressure, altered resource dynamics, and evolutionary adaptation to herbivory. Due to its colorful autumn foliage the plant has become a dominant ornamental tree in urban and suburban environments (USDA 2012). This large protected population coupled with ubiquitous dispersal by great-tailed grackles affords tallow intense seed bank pressure on public lands. However, simply depositing a seed at a site is no guarantee that it will germinate, establish, or thrive. Changes in nutrient dynamics and strategies to cope with herbivory have also contributed to establishment and eventually invasion. Increased nitrogen, potassium, and manganese have been shown to lead to tallow spread and invasion (Burns and Miller 2004; Siemann and Rogers 2007). Siemann and Rogers (2007) found that potassium and/or nitrogen addition has the potential to accelerate the invasion of tallow into Texas coastal prairies. Burns and Miller (2004) found a positive correlation between tallow and elevated manganese concentration in the soil. These findings support the fluctuating resource hypothesis whereby elevated levels of available nutrients, typical of urban areas, increases the invasion potential of riparian sites by creating an unused niche (Davis *et al.* 2000). Tallow appears to have developed an evolutionary adaptation to herbivory that also contributes to its success. Rogers and Siemann (2004) have found that Texas ecotypes of Chinese tallow have undergone an evolutionary shift away from possessing costly herbivore defenses to producing relatively inexpensive tissues that are capable of rapidly compensating for damage. This advantage nearly eliminates the option of using traditional biological control methods and demonstrates the and adaptability of Chinese tallow trees in Texas (Rogers and Siemann 2004). Current management strategies focusing on mechanical removal, herbicides, and fire have been employed with minimal success (USDA 2012). Based on the intense propagule pressure and large seed bank any niche created by the removal of tallow trees will potentially be quickly re-colonized. Management effort should instead focus on controlling seed sources on private lands (Gan *et al.* 2009) and altering the abiotic conditions facilitating tallow establishment and growth. The addition of soil carbon (e.g. sawdust) can be an effective strategy for reducing soil nitrogen levels and reducing exotic species invasion (Averett *et al.* 2004). Minimizing site disturbances that create available niches for tallow can limit invasion (Gan *et al*. 2009). Special attention should be given to recently disturbed sites on low flatlands, areas adjacent to water and roadways, sites recently harvested or disturbed, younger stands, and private forestlands (Gan *et al*. 2009).

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