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Is wild taro a suitable target for classical biological control in the United States?

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ABSTRACT

Weed biological control is a sustainable, cost-effective, and environmentally safe alternative to conventional practices such as chemical and mechanical control. However, before a biological control program is initiated, a feasibility study is conducted to determine whether a target weed is a good candidate for biological control. Therefore, the purpose of this review was to examine different aspects of the invasive wild taro, *Colocasia esculenta* (L.) Schott, as a potential target for biological control in the United States. Though cultivated in different regions of the world for its ornamental foliage and edible corms, wild taro in the United States is an aggressive weed that can form dense stands along waterways. Taro can displace native species, decrease scenic value of habitat, and colonize areas under a range of environmental conditions. We discuss pertinent aspects of the biology, ecology, and economics of wild taro, and used the Peschken-McClay scoring system to evaluate wild taro as a target for biological control. Pest records of wild taro in regions where it is cultivated provide a preliminary list of pathogens and herbivores that should be considered as potential biological control agents. Wild taro scored 139 on the Peschken-McClay scoring system. We argue that wild taro would be a good candidate for biological control in the United States based on biomass accumulation in wetlands, negative impacts to biodiversity, clogging of irrigation canals, and potential for future spread.

Key words: Araceae, *Colocasia esculenta*, invasive plant, Peschken-McClay.

INTRODUCTION

Exotic plant invasions can alter the abundance and composition of native plant communities, reduce biodiversity, cause negative long-term impacts on agricultural practices, and threaten human and animal health (Westbrooks 1998, Bryson and DeFelice 2009). In the United States, nonnative weeds spread at a rate of approximately 700,000 ha per year, with an estimated 5,000 exotic species invading natural areas (Pimentel 2011). Characteristics of invasive plants include early and consistent reproduction by seed and/or vegetative structures, rapid growth rates and survival, and prolific seed production (Rejmanek and Richardson 1996, Westbrooks 1998). Additionally, anthropogenic issues such as climate change, habitat disturbances, and changing landscape patterns may contribute to invasiveness (Batish et al. 2011).

Methods to control invasive plants include mechanical removal or shredding, herbicide application, and the introduction of host-specific weed biological control agents (DiTomaso 1998). Of these, biological control can be a sustainable, cost-effective, and environmentally safe alternative to chemical and mechanical pest control (McFadyen 1998). Classical biological control of weeds is the introduction of host-specific natural enemies to reduce exotic weed populations in the adventive range (Culliney 2005, Manrique et al. 2011). The aim of classical biological control is to restore the ecological balance of a habitat by reuniting an invasive pest and natural enemy that share an evolutionary history in the native range (Culliney 2005).

Before a biological control agent is released, feasibility studies evaluating the target pest must be conducted (Manrique et al. 2011, Wheeler and Ding 2014). These studies consider the taxonomy and distribution, ecological and economic damage, availability of natural enemies, potential spread, and currently available means to control the target pest (Manrique et al. 2011, Wheeler and Ding 2014). Documenting potential safety issues of biological control, such as risk to nontarget species and the potential spread of an agent beyond the intended range of introduction are also fundamental components of feasibility studies (Manrique et al. 2011), and these early studies increase the safety, efficacy, and transparency of proposed biological control programs (Wheeler and Ding 2014).

In the southeastern United States, the invasive perennial plant *Colocasia esculenta* var. *esculenta* (L.) Schott (Alismatales: Araceae, wild taro) is common to wet areas where it reduces water sources by altering the dynamics of riparian plant communities, inhibiting recreational activities, and reducing habitat for native insects, birds, or other wildlife (Radosevich et al. 2007, Moran and Yang 2012). Conventional management tactics can be inconsistent because of limitations associated with scale and nontarget effects; however, the integration of biological control with these methodologies may improve the success of control. The objective of this review is to examine wild taro as a potential target for biological control. The biology, taxonomy, distribution, economic, and ecological importance, current

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Figure 1. Minor wild taro infestation in Red River, LA.

means of control, and known natural enemies of wild taro are further discussed to build the case for biological control.

WILD TARO BIOLOGY

Wild taro has been cultivated around the globe for its ornamental foliage and edible corms (Nelson et al. 2011, Garcia-de-Lomas et al. 2012). However, in some cases, plants have invaded natural areas and have become aggressive weeds (Langeland et al. 2008) by forming dense stands along lakes and rivers, and displacing native flora. Wild taro successfully invades natural areas because of its ability to grow in a variety of conditions, reproduce vegetatively by corms, tubers, or root suckers, and reproduce sexually by seed propagation (CABI 2015). Wild taro can grow in a range of soil types; however, the variety *esculenta* is classified as obligate wetland/facultative wetland (OBL/FACW), usually occurring low-lying areas with an abundant water supply, especially soils in high-rainfall areas that are saturated for long periods of time (Plucknett 1983, DEP 1994, NWPL 2014, USDA-NRCS 2015). The growing season of wild taro lasts throughout the year in the tropics, with average daily temperatures in Hawaii and the Philippines ranging from 18 to 27°C (Wang 1983). Wild taro grown for ornamental purposes is recommended in zones 8b or higher in the U.S. Department of Agriculture (USDA) Plant Hardiness Zones (Moran and Yang 2012, USDA Plant Hardiness Zone Map 2012). A balanced nitrogen, phosphorous, and potassium (N–P–K) ratio, such as 10–10–10 every 3 wk, is suggested for optimal growth of wild taro (Dyer 2016). Wild taro is known as an adaptable crop because of its ability to survive in swampy conditions (Wang 1983), as well as in seawater diluted to 25% strength in the laboratory (Chang et al. 1984). Unlike most terrestrial plants, wild taro can tolerate long periods of standing water, similar to wetland habitats associated with rice crop production (USDA-NRCS 2015; Figure 1).

All parts of the wild taro plant contain an insoluble needle-like crystal, calcium oxalate, which causes irritation to mouth, throat, and stomach if not properly and fully cooked prior to ingestion (Iwuoha and Kalu 1995, Kumuro et al. 2014). Uncooked parts taste acrid and cause gastralgia, and the sap can cause skin irritation. Toxic effects range from acute irritations to chronic poisonings from calcium oxalate buildup in kidneys, which causes renal disorder (Kumuro et al. 2014). Raw wild taro has been the cause of cattle poisonings and is not considered an acceptable livestock feed because it requires costly high-energy preparation (Carpenter and Steinke 1983, Slaughter et al. 2012). As such, the importance of wild taro to wildlife in the United States is probably low. For human consumption, the mature root is prepared like a starchy vegetable and can be roasted, fried, boiled, sliced, grated, or mashed. Leaves, which are high in vitamins A, B, and C, are cooked and eaten like spinach, and young shoots are prepared to taste like mushrooms; however, all parts must be adequately cooked to remove undesirable components and acrid flavor (Greenwell 1947, Carpenter and Steinke 1983, Hussain et al. 1984).

Taxonomy and identification

The Araceae is a large monocotyledonous family of tropical and subtropical terrestrial plants found in moist or shady habitats (Plucknett 1976, Jianchu et al. 2001, Kreike et al. 2004). The edible species of Araceae are classified into five genera within two tribes, Lasioideae (*Cyrtosperma* and *Amorphophallus*) and Colocasioideae (*Alocasia*, *Colocasia*, and *Xanthosoma*) with genera *Colocasia* and *Xanthosoma* the most important (Plucknett 1983). Although the USDA plants database lists four botanical varieties (USDA-NRCS 2015), taxonomists agree that there are two botanical varieties of *Colocasia*, *Colocasia esculenta* var. *esculenta*, and *C. esculenta* var. *antiquorium*. Although native to Southeast Asia (Barrau 1965, Purseglove 1972, Lebot and Aradhya 1991), the dispersal and spread pathway from Asia to the Pacific and beyond is still unclear (Lebot and Aradhya 1991). Wild taro (*C. esculenta* var. *esculenta*) produces a large central corm with suckers and stolons, whereas *C. esculenta* var. *antiquorium* (referred to as *eddow* in the Pacific) produces a smaller corm surrounded by a large number of cormels (Plucknett 1983, Lebot and Aradhya 1991). Morphological and genetic characteristics mark differences between the two varieties. Previous studies found significant genetic diversity between *C. antiquorium* (triploid) and *C. esculenta* (diploid) (Strauss et al. 1980); whereas morphological differences can be distinguished only at the flowering stage (Strauss et al. 1980, Plucknett 1983). A comparison of inflorescences reveals that *C. esculenta* var. *antiquorium* has a sterile appendage of the spadix that is three times longer than that of *C. esculenta* var. *esculenta* (Lebot and Aradhya 1991).

Current and potential U.S. distribution

The origin of wild taro is likely Southeast Asia (Greenwell 1947, Wang 1983). Today, taro is found in over 100 countries throughout Asia, Africa, North America, Central America, the Caribbean, South America, Europe, and Oceania (CABI 2015, Global Biodiversity Information Facility [GBIF] 2015). The spread of wild taro has been facilitated by its domestication as a food crop (Greenwell 1947, Wang 1983). It was first brought to the United States from Africa as an inexpensive ration for slaves (Akridge and Fonteyn 1981). In 1910, the USDA introduced wild taro to the southern United States as a substitute crop for potatoes

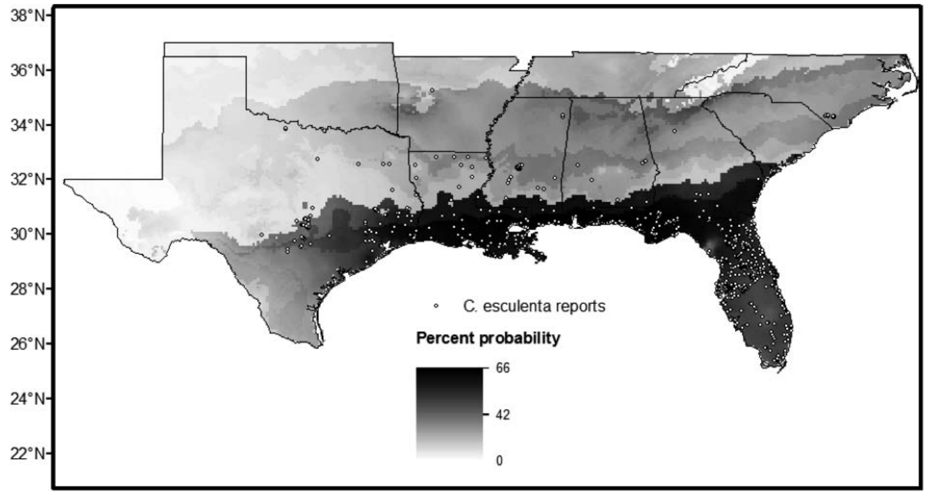

Figure 2. Probability of wild taro occurrence in the United States based on climate matching with worldwide distribution. Warmer colors (42 to 66%) indicate high probability of occurrence. White circles indicate known occurrences of taro.

(Greenwell 1947, Moran and Yang 2012). Wild taro has been reported in eight states in the continental United States, as well as Hawaii and Puerto Rico (EDDMapS 2015, USDA-NRCS 2015), and listed as invasive in Texas, Alabama, Georgia, and South Carolina (Swearingen and Bargeron 2016). Additionally, wild taro is present in almost every county in Florida, and is listed as a Category 1 invasive exotic in the Florida Exotic Pest Plant Council's (FLEPPC) 2015 List of Invasive Plant Species (EDDMapS 2015, FLEPPC 2015).

To assess whether wild taro has reached its full geographic potential in the United States, we used the maximum entropy species distribution model, MaxEnt, to create a potential distribution map. MaxEnt utilizes point occurrences (latitude and longitude) and bioclimatic environmental layers to predict a species' range (Phillips et al. 2006). We used 285 spatially unique point occurrences of wild taro occurring throughout the global distribution, excluding the United States, for the model. Ninety-five spatially unique point occurrences from the same extent were used to test the model quality. Point occurrences were acquired through distribution data from the Global Biodiversity Information Facility (GBIF 2015). For the model, we chose four bioclimatic environmental layers: 1) Annual mean temperature (BIO1), 2) mean temperature of warmest quarter (BIO10), 3) mean temperature of coldest quarter (BIO11), and 4) annual precipitation (BIO12) (Hijmans et al. 2005). These layers were generated by interpolating very high resolution (30 arc s) average climate data collected from weather stations around the world (Hijmans et al. 2005). In MaxEnt, every grid cell receives a nonnegative probability based on the chosen layers, where higher values represent higher probabilities (Phillips et al. 2006).

Higher probabilities of occurrence were predicted for the southeastern United States; southeastern Texas, southern Louisiana, Alabama, Mississippi, northern Florida, and southern Georgia display colors associated with approximately 42 to 66% probability of occurrence (AUC test value = 0.926; Figure 2). Existing point occurrences were projected on the map to compare wild taro locations with the predicted U.S. range. Although the majority of point occurrences are concurrent with the predicted distribution, there are several areas throughout the southeastern United States with high probability of occurrence with no record of existence yet. This means that wild taro has the potential to spread and expand its geographic range in the United States. More specifically, we speculate that wild taro has the potential to spread in the southeastern United States if it is not there already. Additionally, wild taro records from central and northeastern Texas, Arkansas, northern Alabama, Mississippi, Georgia, and North Carolina coincide with areas of low probabilities of occurrence according to EDDMapS. In these cases, wild taro has colonized areas predicted by the MaxEnt model to be unsuitable, which suggest that nonwild populations might be present instead of wild taro, or the climate variables we chose were not the most predictive of occurrence. Because wild taro cannot tolerate winter conditions in Plant Hardiness Zones lower than 8b, we speculate that the points found in areas with low probability of occurrence are ornamental taro, which are probably overwintered indoors (Romer 2005, Moran and Yang 2012).

Economic importance

Wild taro is used in many parts of the world as a starch crop, and ranks fourteenth of most consumed vegetables worldwide (Scott et al. 2000). Although not commonly farmed in the conterminous United States, wild taro is cultivated in Hawaii and American Samoa and contributed $2.32 million to the Hawaiian economy in 2005 (Cho et al. 2007). Additionally, wild taro is a popular ornamental species throughout the United States, sold through online and local garden centers (Moran and Yang 2012). In Florida, wild taro is available through commercial nurseries, accounts for approximately 0.05% of statewide nursery plant sales, and supports an estimated 20 jobs in the state (Wirth et al. 2004).

Despite the negative impact of wild taro on many ecosystems, economic impacts related to its invasion have not been measured. However, costs associated with invasive species are well known and can be substantial (Pimentel et al. 2005, National Invasive Species Council 2014). For example, federal spending on invasive species, including prevention, early detection and rapid response, research, restoration, education and public awareness, and leadership/ international cooperation was $2.2 billion in the United States in 2012 (National Invasive Species Council 2014). Since 1975, over $100 million has been spent on herbicide management of water hyacinth alone in Louisiana (Harms, unpublished data). The Aquatic Plant Control budget for the state of Louisiana is $7.9 million dollars annually, which includes salaries for plant control biologists and spray teams, equipment, supplies, herbicides, and contractor spraying (A. J. Perret, pers. comm.).

Ecological importance

Negative environmental impacts associated with wild taro populations in the United States include displacement of native species, decreased scenic value of habitat because of dense infestation, concerns over water uptake in arid climates, and facilitation of invasion by other nonnative species (Burks et al. 2010, Garcia-de-Lomas et al. 2012, Moran and Yang 2012, Wekiva River System Advisory Management Committee 2012). Wild taro outcompetes native plant species, becoming the dominant taxon in riparian habitats where it establishes (U.S. Fish and Wildlife Service [USFWS] 1996, Brown and Brooks 2003). For example, an increase in its abundance has been associated with a decline in native plant species richness in the San Marcos River, TX (USFWS 1995). Moreover, negative impacts of wild taro to endemic and rare species have been reported in important systems, such as the Atchafalaya Basin, LA and the San Marcos River system, in the United States (USFWS 1995, Piazza 2014), and decrease of native species diversity has been reported in the Hornillo Stream, Spain and Gingin Brook, Australia (Nelson and Getsinger 2000, Brown and Brooks 2003, Atkins and Williamson 2008, Garcia-de-Lomas et al. 2012).

To estimate the size of wild taro infestations in the United States, we used the EDDMapS database. EDDMapS is an online invasive species database, maintained by the Center for Invasive Species and Ecosystem Health at the University of Georgia (EDDMapS 2016). Our search returned 1,661 location records, 15% (256) included an observation of infestation size, and 57% (952) included a qualitative statement about infestation extent/abundance. Of the reports that included information on infestation severity, the majority (47%) was of “scattered patches,” followed by “scattered dense patches” (22%), “scattered plants” (14%), “linearly scattered” plants (11%), “dense monoculture” (4%), and “dominant cover” (2%) (Figure 3). Infestation size (256 observations) ranged from 0.000019 ha (~ a single plant) to 3,035 ha, though the upper report seems to be a summation of infestations associated with a waterway system (Tombigbee and Black Rivers, AL) and not a single infestation, as we could find no supporting information in subsequent literature and web searches of wild taro at the reported location. We also excluded a report of a 35,000-ha infestation in the Wekiva River as an apparent mistake in entry; the Wekiva River is only approximately 3,200 ha in total (Wekiva River System Advisory Management Committee 2012). Mean reported infestation size (disregarding the 3,035-ha infestation) was 6.85 ± 2.30 ha (mean ± SE).

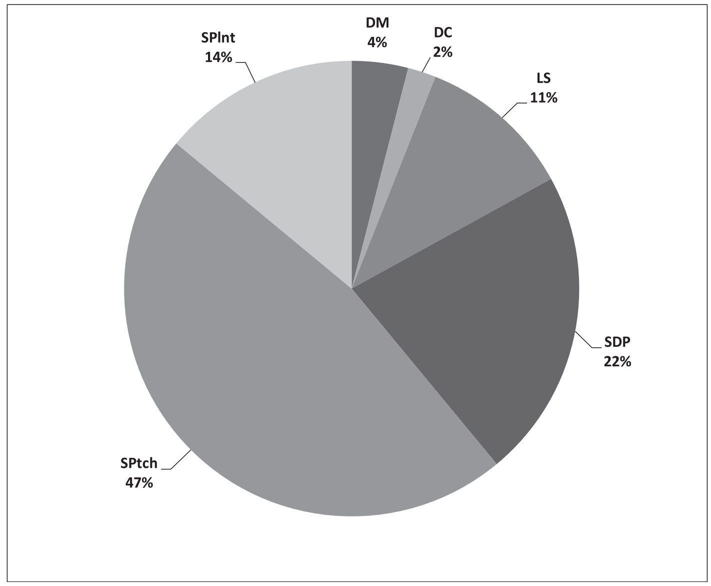

Figure 3. Breakdown of infestation severity in the United States based on reports in EDDMapS. DM = dense monoculture; DC = dominant cover; LS = linearly scattered; SDP = scattered dense patches; SPtch = scattered patches; SPlnt = scattered plants.

CURRENT MANAGEMENT OPTIONS

Chemical control

Little research has been published on the use of herbicides to control wild taro (Nelson and Getsinger 2000, Brown and Brooks 2003). Control of wild taro above- and belowground biomass was achieved with tricoplyr, 2,4-D, and glyphosate with limiting nontarget impacts on the endemic *Zizania texana* (Hitchc.) (Poaceae) (Texas wild rice) in the San Marcos River (Nelson and Getsinger, 2000). However, this study assessed the impacts on plants that were grown for 1 mo prior to treatment and not established populations. Additionally, imazamox was highly efficacious against wild taro in an unpublished Texas study (L. Glomski, pers. comm. [LeeAnn Glomski, Senior Project Manager, U.S. Army Corps of Engineers, St. Paul District, St. Paul, MN 55101]). Atkins and Williamson (2008) tested a variety of treatment strategies on wild taro *in situ* and found sustained control could be achieved with repeated applications of glyphosate, a systemic herbicide that is translocated in the plant, affecting both above- and belowground parts of the plant. Manual removal required the least amount of applications and application time, followed by herbicide treatment. However, neither of the two studies mentioned before directly addressed long-term dynamics of treated systems; i.e., how treatment affects regrowth in subsequent years.

Although a problem of using herbicides can be nonselectivity (e.g., Lym and Messersmith 1982), timing treatment to correspond to phenological weaknesses in target plants and to avoid nontarget impacts may be possible. Nelson and Getsinger (2000) suggested timing of wild taro herbicide treatments in the San Marcos River prior to flowering of the sensitive *Z. texana*. Additionally, with increased interest in wild taro management, selectivity of potential herbicide formulations can be assessed and treatment recommendations made prior to field implementation.

Mechanical control

Manual removal consists of hand-pulling of the entire wild taro plant, including the corm. This technique is labor-intensive, but may provide the most efficient method over the long term because it requires less follow-up management (Atkins and Williamson 2008). The amount of effort needed to remove plants manually probably limits the use of this practice, but might represent an opportunity to involve local community organizations, particularly in early-detection programs. Grassroots community participation in environmental stewardship is a positive phenomenon and has been harnessed effectively in a variety of invasive species-management programs (Briese and McLaren 1997, Carr 2001, Kwong 2003). Community involvement in a management program not only provides a large labor resource, but offers a sense of ownership to those involved, and a stake in the eventual outcome of the project.

In contrast to manual removal of whole plants from the soil, cutting removes aboveground tissues only. This technique has been shown ineffective and likely represents a waste of resources, even when reapplied throughout the growing season (Atkins and Williamson 2008). This technique may be effective in newly establishing populations. However, if populations are small enough, manual removal is likely to provide the best chance for sustained control.

Biological control

There is currently no active biological control program in the world for wild taro. However, because of its economic importance to agriculture, there are several potential agents known from pest records in cultivation areas (Appendix 1). The following is a summary of known natural enemies and potential candidates for biological control.

There are a number of fungi, bacteria, nematodes, and viruses associated with wild taro, of which fungi have been the most studied. The severity of damage caused by disease varies according to the environmental requirements of the pathogen agent, plant physiological status, and agent virulence (Wang 1983, Carmichael et al. 2008). The introduction of a pathogen for biological control requires extensive prerelease studies conducted in pathogen-approved quarantine facilities (Dylan and Manish 2015). Several fungal diseases have been reported in the literature to cause infection in wild taro; however, most are known to be generalists, which preclude them from being considered as biological control candidates. The only pathogen known to attack wild taro exclusively is *Cladosporium colocasiae* Sawada (Capnodiales: Davidiellaceae), which causes Cladosporium leaf spot and affects plants both in wetland and upland environments. There is a report of this fungus infecting celery (*Apium graveolens* L.) in New Zealand (Bensch et al. 2010). Despite its high virulence, it does not usually kill wild taro plants (Reddy 2015) but could be considered for further examination.

The most important viral disease of wild taro is caused by the co-infection of taro with Taro badnavirus (TaBV) and Colocasia bobone disease virus (CBDV), resulting in the lethal alomae disease, which is considered the most destructive viral disease of wild taro. The alomae disease has been reported on wild taro in the Solomon Islands and Papua New Guinea (Revill et al. 2005) and it is transmitted by piercing-sucking insects (Reddy 2015). Infection results in stunted, thickened, twisted, dark green leaves, arresting growth and causing systemic necrosis (Carmichael et al. 2008). Taro feathery mottled virus (TFMV) is transmitted by the plant hopper *Tarophagus colocasiae* (Palomar 1987). Infection of other plant species by these viruses is not known, however it should be considered that proper identification of virus species is difficult and mutations are common, so more work is needed to assess these viruses for biological control.

Many root-knot nematodes attack wild taro, but they have a wide range of hosts and only cause minor damage to wild taro (Reddy 2015). On the other hand, the nematode *Hirschmanniella miticausa* Bridge, Mortimer and Jackson (Tylenchida: Pratylenchidae) can cause severe damage by feeding endoparasitically in the corm. Infection produces irregular red or brown necrotic zones, leading to complete secondary brown soft rot of the basal portions of the corms (De Waele and Elsen 2007). This nematode is reported to exclusively attack wild taro in the Solomon Islands (Bridge 1978, Bridge et al. 1983, Mortimer et al. 1981, Patel et al. 1984) and Papua New Guinea (Bridge et al. 1983, Bridge and Page 1984) suggesting its potential as biological control agent. However, the damage caused by these nematodes has not been fully evaluated (Singh et al. 2013).

There are a number of invertebrate herbivores associated with wild taro in its native range, and many of them are quite damaging to the plant. Among the sap-sucking insects, *Tarophagus colocasiae* Matsumura (Hemiptera: Delphacidae) shows potential to be used in biological control because it has only been reported in association with *C. esculenta* (Matthews 2003). This plant hopper feeds on leaves and stems and can significantly damage host plants, especially under dry conditions; older leaves are most affected by high infestations, which can cause plant wilt and death (Carmichael et al. 2008). Taro feathery mottle virus (TFMV) has been reported to be transmitted by *T. colocasiae* (Palomar 1987), which also causes damage from feeding and piercing tissues to oviposit, which exposes the plant to secondary infections (Carmichael et al. 2008). However, *Tarophagus* sp.

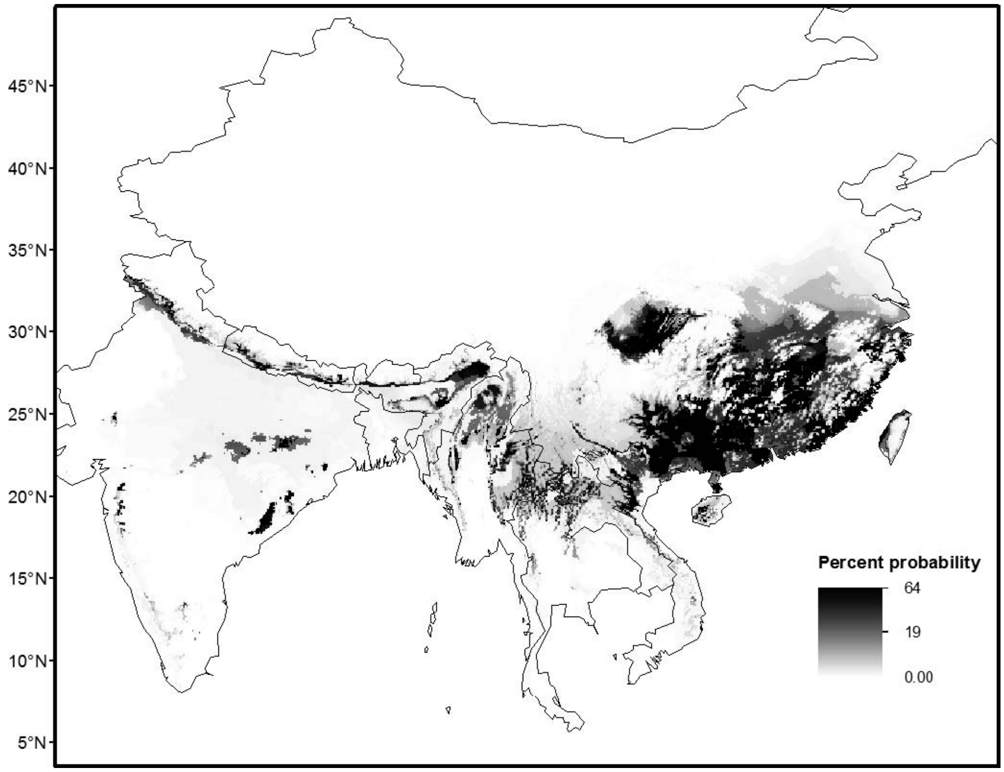

Figure 4. Probability of native wild taro occurrences in SE Asia based on climatic matching with the U.S. invaded range. Warmer colors (19 to 64%) indicate higher probabilities of occurrence.

have been reported feeding in plants in the aroid family, including caladiums (*Caladium* sp.) and malanga [*Xanthosoma sagittifolium* (L.) Schott] (Duatin and de Pedro 1986); therefore more research is needed to determine its potential as biological control agent.

Natural enemy exploration

Although a large number of natural enemies are known from the literature (Table A1), the great majority are generalist natural enemies and have been reported as pests in cultivated wild taro populations. Thus, foreign exploration should be undertaken to document species damaging wild taro in natural areas within its native range. To narrow the search, the geographic area of origin of U.S. wild taro, and locations with similar environmental conditions to the United States should provide the best chance at identifying suitable agents. Genetic studies will be needed to characterize wild taro populations in the United States and the native range in order to narrow the survey of natural enemies in areas likely to be the geographic origin of U.S. populations of wild taro (Goolsby et al. 2006, Manrique et al. 2008). Moreover, areas thought to be the center of evolution of wild taro should be prioritized, as these may provide the best chance for finding host specific natural enemies (Schroeder and Goeden 1986, Wapshere et al. 1989, Muller-Scharer et al. 1991).

Towards prioritizing overseas locations with matching climate, we used MaxEnt to generate a predictive distribution map using known point occurrences of wild taro from the United States. We used 368 U.S. point occurrences reported from EDDMapS (EDDMapS 2015) and four bioclimatic variables from the WORLDCLIM database (Hijmans et al. 2005). The four bioclimatic layers chosen were mentioned in the above section, "Potential U.S. distribution." Results from the MaxEnt model indicate that areas in southeastern Asia, particularly southeastern China, are most climatically similar to the southeastern United States (AUC test value = 0.973; Figure 4). Therefore, we suggest that searches for natural enemies of taro are prioritized towards southeastern China because of the climatic similarity with the United States.

Conflicts of interest

Wild taro is considered a popular ornamental species in some areas in the United States. This is especially evident by the number of popular "how-to" articles available online to assist home gardeners with cultivation (e.g., Southern Living 2015) and the number of online retailers willing to ship propagules throughout the United States. Any attempt to initiate a biological control program for wild taro will certainly have to balance the interests of retailers and home-gardeners. However, enhanced public education and outreach may provide the public support needed to move toward a biological management goal.

Peschken-McClay scoring

To assess the suitability of wild taro as a target for classical biological control, we used the Peschken-McClay

Table 1. The Peschken-McClay scoring system was used to assess the suitability of wild taro as a target for classical biological control. Descriptions of each category can be found in Peschken and McClay (1995). Scores are the average of those provided by five independent assessments; see Appendix 1 for individual author scores.

Category	Rank	Average Score	Standard Deviation
A. Economic aspects			
Economic losses	Light/severe	14	4.2
Infested area	Large/very large	7	2.7
Expected spread	Moderate	10	0.0
Toxicity	Severe/very severe	8.6	2.2
Available means of control			
Environmental damage	Medium/high	15	5.0
Economic justification	Medium/low	14	5.5
Beneficial aspects	Minor	− 8	2.7
B. Biological aspects			
Intraspecific variation	Small	9.4	1.3
Native range	Outside the United States	30	0.0
Relative abundance	Possibly more or not so	5	3.5
Success elsewhere	Not attempted	0	0.0
Number of known agents		3	0.0
Habitat stability	Moderate	25	5.0
Economic species in genus	> 1	0	0.0
Economic species in tribe	> 8	0	0.0
Ornamental species in genus	> 5	0	0.0
Ornamental species in tribe	> 15	0	0.0
Native species in genus	None	2	0.0
Native species in tribe	None	4	0.0
Total		139	

(1995) scoring system (Table 1). Each author independently scored each criterion and the results were averaged for all authors. This system provides a semiquantitative method for evaluating a potential target based on a suite of criteria, including economic (losses, infested area, expected spread, toxicity, available means of control, beneficial aspects, conflicts of interests) and biological (intraspecific variation, geographic area where weed is native, relative abundance, success of biological control elsewhere, among others). Wild taro invades a wide range of habitats with high capacity for spread and colonization (CABI 2015). Reports of toxicity to humans and livestock (Carpenter and Steinke 1983, Slaughter et al. 2012), the difficulty of control (Atkins and Williamson 2008) and its broad geographic distribution (Culliney 2005) confirmed the author's high scores in those categories. There were minimal beneficial aspects but ornamental importance is addressed in the biological aspects section of the Peschken-McClay model. The native geographical range is well defined in the literature, and our scores reflected this. The literature review revealed an abundance of documented natural enemies, but there are no known cases of biological control programs against this weed. From this review, at least one arthropod (*Tarophagus colocasiae*), a nematode (*Hirschmanniella miticausa)*, and a pathogen (*Cladosporium colocasiae*) may be suitable for biological control of wild taro, but extensive host range testing and efficacy should be conducted on these candidates. In addition, foreign explorations are needed to identify new candidates that are host-specific to wild taro and have potential to control the weed in the southeastern United States. The final score, averaged for all authors, was 139 out of possible 179. Although lower than assessments for other target weeds (Peschken and McClay 1995, Cuda and Sutton 1999, Cuda et al. 2007), this score is comparable to that given to *Matricaria perforata* Mérat (Asteraceae: Anthemideae), a weed that was considered a good target for biological control (Peschken et al. 1989). Our score mainly reflects a lack of empirical evidence for economic impacts, the need to assess specificity of known candidates, and the value of wild taro and relatives as an ornamental species.

CONCLUSIONS

The invasive plant wild taro is currently widespread in the southeastern United States. Management options are limited to herbicide or physical removal, and therefore alternatives are needed. Biological control is an environmentally safe and self-sustaining alternative against wild taro, but more research is needed to identify potential biological control agents. The literature surrounding pests of wild taro is informative and provides a starting point for prioritizing potential agents, including at least three reviewed here. However, because the literature is concentrated on pests of cultivated crops and not natural areas, it is important to conduct thorough overseas searches to identify additional damaging herbivores or pathogens that are host specific to wild taro. We suggest, based on our MaxEnt model predictions, that searches for natural enemies should be prioritized in southeastern China. The next steps in the initiation of a biological control program of wild taro in the United States should include molecular analysis and comparison of U.S. and overseas populations, outreach programs to educate the public on negative impacts of wild taro and build support, and quantification of environmental and economic impacts in the United States.

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Appendix 1. Natural enemies of wild taro reported in the literature, organized alphabetically by type (pathogen/insect).

Order: Family	Species	Common name/ disease caused	Plant part attacked	Distribution	Host range	References
Pathogens: Bacteria						
Enterobacteriales: Enterobacteriaceae						
	Erwinia chrysanthemi	Bacterial soft rot	Roots	Worldwide	Wide host range	Asthana (1946), Wang (1983), Reddy (2015)
Pathogens: Fungi						
Atheliales: Atheliaceae						
	Sclerotium rolfsii	Southern blight	Roots	Widespread in the Pacific	Wide host range	Reddy (2015)
Botryosphaeriales: Botryosphaeriaceae						
	Lasiodiplodia theobromae	Spongy black rot	Roots	Widespread in the Pacific	Wide host range	Reddy (2015)
	Phyllosticta colocasiophila	Phyllosticta leaf spot	Leaf	Hawaii and America Samoa	Araceae	Wang (1983), Plucknett et al. (1970)
Capnodiales: Davidiellaceae						
	Cladosporium colocasiae	Cladosporium leaf spot	Leaf	Hawaii, New Caledonia, New Hebrides, Western and American Samoa, the Carolines, and the Marianas	*C. esculenta*, *Apium graveolons*	Bugnicourt (1954), Parris (1941), Trujillo (1967)
Hypocreales: Nectriaceae						
	Fusarium solani	Fusarium dry rot	Roots	Widespread in the Pacific	Wide host range	Reddy (2015)
Mucorales: Mucoraceae						
	Rhizopus stolonifer	Rhizopus rot	Roots	Widespread in the Pacific	Wide host range	Reddy (2015)
Ophiostomales: Ceratocystidaceae						
	Ceratocystis fimbriata	Black rot	Roots	Widespread in the Pacific	Wide host range	Reddy (2015)
Peronosporales: Pythiaceae						
	Phytophthora colocasiae	Phytophthora leaf blight	Leaf	Africa, Caribbean, North America, Pacific Islands and South America	Associated with Araceae but can infect *Phaseolus vulgaris* (Fabaceae)	Cline et al. (2008), Reddy (2015), Wang (1983), Zentmyer (1988)
Pleosporales: Pleosporaceae						
	Curvularia sp.	Curvularia leaf blight	Leaf	Worldwide	Wide host range	Hunter et al. (2001)
Pythiales: Pythiaceae						
	Pythium aphanidermatum	Pythium rot	Roots	Worldwide	Wide host range	Reddy (2015)
	Pythium carolinianum	Pythium rot	Roots	Worldwide	Wide host range	Reddy (2015)
	Pythium graminicolum	Pythium rot	Roots	Worldwide	Wide host range	Reddy (2015)
	Pythium irregular	Pythium rot	Roots	Worldwide	Wide host range	Reddy (2015)
	Pythium myriotylum	Pythium rot	Roots	Worldwide	Wide host range	Reddy (2015)
	Pythium splendens	Pythium rot	Roots	Worldwide	Wide host range	Reddy (2015)
	Pythium ultimum	Pythium rot	Roots	Worldwide	Wide host range	Reddy (2015)
Pathogens: Viruses						
	Colocasia bobone disease virus (CBDV)	Colocasia bobone disease	Leaf	Solomon Islands and Papua New Guinea	*C. esculenta*	Reddy (2015), Revill et al. (2005)
Pathogens: Nematodes						
Tylenchida: Pratylenchidae						
	Hirschmanniella miticausa	Taro nematode	Corms and roots	Solomon Islands and Papua New Guinea	*C. esculenta*	Bridge (1978), Bridge and Page (1984), Mortimer et al. (1981), Patel et al. (1984)
Tylenchida: Heteroderidae						
	Meloidogyne incognita	Root-knot nematode	Corms and roots	Worldwide	Wide host range	Reddy (2015)
	Meloidogyne javanica	Root-knot nematode	Corms and roots	Worldwide	Wide host range	Reddy (2015)

Appendix 1. Continued.

Order: Family	Species	Common name/ disease caused	Plant part attacked	Distribution	Host range	References
Tylenchida: Hoplolaimidae						
	Pratylenchus coffeae	Lesion nematode	Corms and roots	Worldwide	Wide host range	Reddy (2015)
Arthropods						
Coleoptera: Chrysomelidae						
	Monolepta signata	White-spotted flea beetle	Leaf	Asia, India	Polyphagous	Hill (2008)
Coleoptera: Scarabaeidae						
	Papuana huebneri	Taro beetle	Corms and roots	Bismarck Archipelago, Gilbert Island, Indonesia, New Guinea, New Britain, Papuasia, Solomon Islands	Polyphagous	Waterhouse (1997), Waterhouse and Norris (1987)
	Papuana laevipennis	Taro beetle	Corms and roots	Bismarck Archipelago, Gilbert Island, Indonesia, New Guinea, New Britain, Papuasia, Solomon Islands	Polyphagous	Waterhouse (1997), Waterhouse and Norris (1987)
Hemiptera: Aleyrodidae						
	Aleurodicus dispersus	Whitefly	Leaf	Widespread	Polyphagous	Carmichael et al. (2008), Reddy (2015)
Hemiptera: Aphididae						
	Patchiella reaumuri	Taro root aphid	Roots	Asia, India	Olyphagous	Stroyan (1979), Sato et al. (1989), Carmichael et al. (2008)
Hemiptera: Fulgoridae						
	Tarophagus colocasiae	Planthopper	Leaf	Southeast Asia, Indonesian archipelago and Pacific islands	*C. esculenta*, caladiums and malanga	Asche and Wilson (1989), Carmichael et al. (2008), Duatin and de Pedro (1986), Halbert and Bartlett (2015), Palomar (1987)
Hemiptera: Pseudococcidae						
	Planococcus citri	Mealybugs	Leaf	Widespread	Polyphagous	Carmichael et al. (2008), Reddy (2015)
	Pseudococcus longispinus	Mealybugs	Leaf	Widespread	Polyphagous	Carmichael et al. (2008), Reddy (2015)
Lepidoptera: Arctiidae						
	Pericallia ricini	Moth	Leaf	Widespread	Polyphagous	Habeck (1974), Swezey (1942)
Lepidoptera: Noctuidae						
	Spodoptera litura	Moth	Leaf	Widespread	Polyphagous	Habeck (1974), Swezey (1942)
Lepidoptera: Sphingidae						
	Theretra gnoma	Moth	Leaf	Widespread	Polyphagous	Habeck (1974), Swezey (1942)
Orthoptera: Acrididae						
		Grasshoppers	Leaf	Widespread	Polyphagous	Wang (1983)
Orthoptera: Gryllidae						
		Crickets	Leaf	Widespread	Polyphagous	Wang (1983)
Thysanoptera: Thripidae						
	Organothrips bianchi	Thrips	Petioles	Hawaii, Palau, Samoa	Polyphagous	Ananthakrishnan (1993), Bhatti (1974), Mound (2000)
Trombidiformes: Tetranychidae						
	Tetranychus sp.	Spider mites	Leaf	Widespread in Pacific	Polyphagous	Carmichael et al. (2008), Krauss (1944)

Appendix 2. The Peschken-McClay scoring system was used to assess the suitability of wild taro as a target for classical biological control. Descriptions of each category can be found in Peschken and McClay (1995). Scores are the average of those provided by five independent assessments.

Category	Rank	AC	NH	AR	MdS	RD
A. Economic aspects						
Economic losses	Light/severe	15	10	10	20	15
Infested area	Large/very large	10	10	5	5	5
Expected spread	Moderate	10	10	10	10	10
Toxicity	Severe/very severe	10	8	5	10	10
Available means of control						
Environmental damage	Medium/high	20	10	20	10	15
Economic justification	Medium/low	10	10	20	20	10
Beneficial aspects	Minor	− 5	− 5	− 10	− 10	− 10
B. Biological aspects						
Intraspecific variation	Small	10	10	7	10	10
Native range	Outside the United States	30	30	30	30	30
Relative abundance	Possibly more or not so	5	10	5	0	5
Success elsewhere	Not attempted	0	0	0	0	0
Number of known agents		3	3	3	3	3
Habitat stability	Moderate	30	30	20	20	25
Economic species in genus	> 1	0	0	0	0	0
Economic species in tribe	> 8	0	0	0	0	0
Ornamental species in genus	> 5	0	0	0	0	0
Ornamental						
Species in tribe	> 15	0	0	0	0	0
Native species in genus	None	2	2	2	2	2
Native species in tribe	None	4	4	4	4	4
Total		154	142	131	134	134