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# Bioenergy and Invasive Plants: Quantifying and Mitigating Future Risks

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The United States is charging toward the largest expansion of agriculture in 10,000 years with vast acreages of primarily exotic perennial grasses planted for bioenergy that possess many traits that may confer invasiveness. Cautious integration of these crops into the bioeconomy must be accompanied by development of best management practices and regulation to mitigate the risk of invasion posed by this emerging industry. Here I review the current status of United States policy drivers for bioenergy, the status of federal and state regulation related to invasion mitigation, and survey the scant quantitative literature attempting to quantify the invasive potential of bioenergy crops. A wealth of weed risk assessments are available on exotic bioenergy crops, and generally show a high risk of invasion, but should only be a first-step in quantifying the risk of invasion. The most information exists for sterile giant miscanthus, with preliminary empirical studies and demographic models suggesting a relatively low risk of invasion. However, most important bioenergy crops are poorly studied in the context of invasion risk, which is not simply confined to the production field; but also occurs in crop selection, harvest and transport, and feedstock storage. Thus, I propose a nested-feedback risk assessment (NFRA) that considers the entire bioenergy supply chain and includes the broad components of weed risk assessment, species distribution models, and quantitative empirical studies. New information from the NFRA is continuously fed back into other components to further refine the risk assessment; for example, empirical dispersal kernels are utilized in landscape-level species distribution models, which inform habitat invasibility studies. Importantly, the NFRA results in a relative invasion risk to known species (e.g., is giant reed a higher or lower invasion risk than johnsongrass). This information is used to design robust mitigation plans that include record keeping, regular scouting and reporting, prudent harvest and transport practices that consider species biology, and eradication protocols as an ultimate precaution. Finally, a socio-political balance must be struck (i.e., a cost-benefit analysis) among our energy choices that consider the broader implications, which includes the risk of future invasions.

**Nomenclature:** Giant miscanthus, *Miscanthus* × *giganteus* J. M. Greef and Deuter ex Hodk. and Renvoize; giant reed, *Arundo donax* L.; Johnsongrass, *Sorghum halepense* (L.) Pers.

**Key words:** Best management plan, biofuel, eradication, weed risk assessment.

Renewable energy in the transportation and power sectors has exploded in the last decade as a result of international conventions on mitigating climate change (e.g., Kyoto Protocol), as well as domestic policy intended to invigorate rural economies and move toward energy independence (EISA 2007). One component, the bioeconomy (i.e., an economy based on biological goods) has a variety of environmental, economic, and societal benefits that has received the support of policymakers, environmental organizations, the general public, and industry (Robertson et al. 2008). There exists tremendous potential in bioenergy (i.e., biomass used in electricity or fuel production) to mitigate greenhouse gases, re-establish agriculture in portions of the country that have seen recent declines (e.g., the southeastern United States), and develop jobs in the clean energy sector promoted by several U.S. Administrations. However, rapid development of the bioenergy sector may also bring about adverse environmental consequences through land-use change, extensive introduction of exotic and potentially invasive species across huge areas of the United States, and concomitant greenhouse gas emissions through corrective measures. The concern of solving one problem (renewable energy) by creating another (broad scale introduction of harmful species) has been voiced all over the globe (e.g., Low and Booth 2007). One of the grand challenges to this emerging industry is how to meet biomass demands efficiently and sustainably on the least

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amount of land, while mitigating negative externalities (i.e., unintended effects or byproducts of an activity), including, but not limited to invasiveness (Robertson et al. 2008).

The amount of land required to meet the biomass demands of power plants or liquid fuel conversion facilities is directly proportional to the yield of the chosen crop. Thus, bioenergy crop developers are selecting for species that are capable of being maximally productive with the fewest required resources. As many have identified previously (Barney and DiTomaso 2008; Raghu et al. 2006), achieving this goal requires that crops be highly competitive, rapid establishers, produce high annual yields of aboveground biomass, be resource-use efficient, and be tolerant of poor growing conditions and periodic disturbance. Herbert Baker (1965) would undoubtedly read this trait list and conclude that we were designing the ideal weed. In fact, some concerned groups view exotic and potentially invasive bioenergy crops as a "Clear and Present Danger" to conservation, and are actively working to prevent their adoption (Glaser and Glick 2012). Not to mention there exists a potential policy clash between preventing the support and dissemination of invasive species by Executive Order 13112, and the Energy Independence and Security Act mandating 136 billion liters of renewable transportation fuel, 44% of which will be derived from cellulosic crops by 2022 (DiTomaso et al. 2010). Important societal, environmental, economic, energy, and national security decisions are imminent as the bioenergy industry matures and competes with other energy sources that are coming online—hydrologic fracturing (fracking), oil shales, etc.—and reducing the cost of petroleum. This review focuses on one aspect of the bioeconomy; identifying, quantifying, and mitigating the invasive potential of bioenergy crops. This issue threads policy, conservation, ecology, and bioethics, of which I will cover in detail in an attempt to provide an unbiased assessment of the issue of the invasive potential of bioenergy crops.

## Humans and Exotic Species

Humans have a long history of introducing organisms to meet our dietary, fiber, aesthetic, and companion needs, which has fostered our ability to successfully colonize nearly every habitable region of the globe. Humans have a seemingly insatiable demand for the rare, the familiar, and the productive that has homogenized our flora and fauna in a snippet of geologic time (Mack et al. 2000). By moving species far beyond their historic limits and modifying the environment to our benefit, humans have given rise to a new geologic epoch—The Anthropocene (Rosenzweig 2001). Invasive species are a primary component of this anthropocentric epoch, and have attendant negative effects on the global economy through yield loss, management costs, and lost revenue (Keller et al. 2009).

Humans move species for primarily economic means in the hope of increased agricultural profits, or that people will be motivated to purchase a novel species, or that the new species will perform a function better than existing native species (McNeely 2001). In general, these introductions are without consideration to the potential negative effects to the environment or human well-being. Historically, the economic incentives for introduction have not been balanced by consequent responsibility for negative externalities since the "cost" of invasion (e.g., management) is borne by the public, not by the introducer. For much of history, we were unaware of the negative consequences of exotic species, and our market system favored economic growth, often at great future expense; thus we have typically weighed potential economic development over potential ecological and economic damage. This has changed to a degree with regulation intended to palliate harmful invasions (Harl 2010), but by and large we are free to introduce, breed, and cultivate most anything we choose, unless they are genetically modified. An ironic major exception to this pattern is bioenergy crops. Environmental groups and sustainability advocates have long called for renewable energy to become the foundation for our next great energy economy. However, it is often these same groups calling foul against the suite of crops that the bioenergy industry has selected as top performers (Glaser and Glick 2012), which are primarily large-statured exotic perennial grasses. Thus, the bioenergy industry is a rare exception to our socioeconomic history of introduce first, manage the damage later. If done pragmatically, this caution could usher development of a new sustainable agriculturally-based energy sector. On the other hand, belligerent employment of the precautionary principle could hamstring this nascent bioeconomy or force adoption of lower productivity crops necessitating additional land conversion to meet yield demands (Marchant et al. 2013).

## Policy as a Driver of the Bioeconomy (and Potentially New Invasions)

Bioenergy has been an integral component of human society since the development of fire, fostering civilization to the Industrial Revolution when wood was replaced with petroleum products (also biologically-derived). As we approach "peak oil" and attempt to stem climate change, tremendous interest has been expressed in integrating new bioenergy back into our energy portfolio away from its "boutique" status. There has always existed a grassroots interest in biodiesel, methane digesters, and ethanol production, but much of the recent push towards bioenergy has come as top-down policy from lawmakers. Policy initiatives have come from both federal and state



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institutions, with the primary driver being the Bush-era Energy Independence and Security Act (EISA 2007), which sets annual targets of various renewable fuels. In fact, large-scale projects have already been approved with the planting of thousands of hectares of exotic bioenergy crops (see Biomass Crop Assistance Program [BCAP] below).

The Bush Administration supported development of bioenergy through several programs, building on decades of "back burner" research on biofuel crops. Switchgrass (*Panicum virgatum* L.) was mentioned by President Bush in two State-of-the-Union addresses, and had been anointed as the "model bioenergy crop" in the United States (Parrish and Fike 2005). This eastern native warm-season grass was a favored early winner in the bioenergy crop race, but interest has since turned to larger exotic grasses. Interestingly, following EISA, and the likelihood of broad scale switchgrass planting, the California Department of Food and Agriculture added this western exotic to their state noxious weed list (DiTomaso et al. 2013); thus precluding commercial development. This listing has since been reversed, but it highlights the knee-jerk reaction to the issue of preventing future invasions, even from this bioenergy poster child.

To stimulate the bioenergy industry, the 2008 U.S. Farm Bill, or the Food, Conservation, and Energy Act, included a program aimed at supporting grower adoption of bioenergy crops. BCAP provides up to 75% of establishment costs for perennial biomass crops, as well as annual payments for enrolled land—up to two years for herbaceous crops and 15 years for woody crops. There are currently 11 project areas supported by BCAP from Kansas to New York with crops ranging from camelina [*Camelina sativa* (L.) Crantz], switchgrass, native prairies, and the "Illinois" and "Freedom" clones of giant miscanthus (*Miscanthus* × *giganteus* J.M. Greef and Deuter ex Hodk. and Renvoize) (www.fsa.usda.gov). This program is projected to have thousands of hectares of land in production by 2020. BCAP requires an Environmental Assessment of each project, which includes addressing the invasive potential of the included crops, although information is lacking on the procedure for making this determination. Additionally, plants that are noxious or invasive are not eligible for BCAP. Thus, giant reed (*Arundo donax* L.) is not eligible for BCAP in California where it is a state listed noxious weed, but would be eligible in North Carolina where it was ruled by Department of Agriculture and Consumer Services to not be added to their noxious weed list. Clarification of criteria and implementation of this rule would aid interpretation and transparency.

The Environmental Protection Agency (EPA) is the primary supervisor of the clean fuel programs and manages the Renewable Fuel Standard (RFS). To ensure compliance with the RFS, the EPA uses Renewable Identification Numbers (RIN), which are primarily an assessment of the life cycle greenhouse gas balance, and RINs can be traded on the open market. Several bioenergy crops are eligible for RINs—switchgrass, giant miscanthus (the sterile triploid), camelina, energycane (*Saccharum* spp.)—paving the way for commercial adoption. However, two crops in particular—giant reed and napiergrass or elephantgrass (*Pennisetum purpureum* Schumach.)—have caused tremendous controversy regarding their RIN status. Despite vocal public concerns, the EPA made a final ruling allowing giant reed and napiergrass to be eligible for RINs, but included several safeguards to mitigate escape outside cultivation. These include "clear and compelling evidence" that these crops do not "present a significant likelihood of spread beyond the planting area", which requires evidence from similar environments as the proposed production area. The producer must also supply a Risk Mitigation Plan detailing actions to reduce unintentional escape. Importantly, the plan must also include a closure plan for the production area should the field be abandoned. The EPA also has the right to suspend generating RINs if the feedstock has spread beyond the intended planting area. The criteria outlined in the EPAs final rule on giant reed and napiergrass RIN eligibility thoroughly covers the risk mitigation procedures called for by others (e.g., DiTomaso et al. 2010). The crux of this ruling will be implementation and enforcement of these important safeguards against invasion. I would suggest that such protocols be adopted for all bioenergy crops as invasiveness is context-dependent (Barney and Whitlow 2008), and we would be remiss to assume any species inherently safe (DiTomaso et al. 2007).

**What Are the States Doing?** Outside of the EPA ruling on giant reed and napiergrass, federal regulations on bioenergy invasion mitigation is unlikely, at least in a relevant time frame, especially considering the pace of BCAP projects. Thus, states are likely to be the first line of regulatory defense outside of voluntary measures (e.g., Barney 2012). State noxious weed lists are the primary regulatory mechanism to guard against weedy and invasive species. However, a recent analysis of these rules reveals a national "patchwork quilt" of legal protections against largely agricultural weeds (McCubbins et al. 2013; Quinn et al. 2013a). The historical focus on agricultural protection combined with the glacial pace of legal modification will likely make existing noxious weed lists impotent for effective invasion mitigation. Thus, some states have used other legal protections against bioenergy-based invasion (McCubbins et al. 2013). For example, both Mississippi and Florida have employed a permit and bond procedure for any biofuel planting $> 0.4$ or 0.8 ha ($> 1$ or 2 ac), respectively (Quinn et al. 2013a). This approach provides a stop gap measure against large-scale introduction of bioenergy crops, but may (1) unduly limit commercial



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production caused by the economic burden of the surety bond sometimes valued at 150% of the cost of eradication, and (2) are too piecemeal and opaque to have broad and meaningful invasion mitigation while supporting commercial-scale development.

Quinn et al. (2013a) proposed that states adopt a tiered approach to their noxious weed lists that would add transparency via a science-based process vetted by stakeholders to ensure a balanced and fair assessment to regulatory listings. The tiered approach, a variation of which was also suggested by Davis et al. (2010), is centered around weed risk assessments (see below) and *in situ* field data, which are then weighed against the economic and other benefits of introduction by a group of stakeholders. This is followed by a negligence liability scheme designed as an incentive for introducers to act responsibly, and relocate the economic burden of invasion cleanup away from taxpayers and back onto responsible parties (Quinn et al. 2013a).

**Why Not Use Existing Invasions to Make Bioenergy?** A common refrain from the public is "Why plant new exotics when there are endless acreages of existing invasive species?" Invasive plants infest millions of hectares of land, and are spreading faster than we can manage them (DiTomaso 2000). Anyone driving through any highway in the southeastern United States has witnessed the "kudzu [*Pueraria montana* var. *lobata* (Willd.) Maesen & S.M. Almeida] corridor" bordered by johnsongrass [*Sorghum halepense* (L.) Pers.]. Or float any river in the Upper Midwest and the banks are lined with tamarisk or saltcedar (*Tamarix* spp.), or ride through the Great Basin and witness the fields of downy brome (*Bromus tectorum* L.). Surely we can just harvest this and make something useful?

In fact, a joint venture between the Missouri River Watershed Coalition (MRWC) and the Center for Invasive Plant Management at Montana State University are attempting to do just that: manage exotic invasives and turn them into renewable energy (www.weedcenter.org/mrwc). This project is attempting to address a conservation and energy challenge by managing riparian infestations of Russian-olive (*Elaeagnus angustifolia* L.) and saltcedar, and turn them into renewable fuel using existing commercial technologies—primarily pelletizing. It is unclear as to the long-term success of this project, and whether the methods would translate to other systems. In an innovative analysis, Nackley et al. (2013) determined that Russian-olive and saltcedar (*Tamarix* spp.) could be profitably harvested and used as a bioenergy source in Washington State as a restoration byproduct. However, they also concluded that the suboptimal thermochemical properties may limit commercial viability.

Several factors limit the pragmatic application of managing invasives and turning them into bioenergy. While abundant, invasive plants rarely occur in sufficiently high density to supply, if harvested, a relevant amount of biomass to a commercial plant (Quinn et al 2013b). Invasive plants also commonly occur not in open agricultural fields, but in forest, rights-of-way, riparian areas, or far from civilization (e.g., downy brome in the western United States). This presents a logistical issue of harvesting and transport. Most commercial conversion facilities are likely to be centralized among concentrated bioenergy plantings, whereas invasions are likely spread out and would require smaller scale decentralized conversion facilities as in the MRWC project. Lastly, conversion facilities would require regular and reliable biomass deliveries, which would disincentivize eradication, and could potentially lead to planting of the invasive species. While practical in motivation, large-scale application of using existing invasions for bioenergy is unlikely.

## Ecological Risk Assessment of Bioenergy Crops

Predicting which species will become invasive, or in the case of biofuels, "separating the weeds from the crops", is a complex challenge of interacting factors. Thus, lacking a prescription for determining invasiveness we are faced with using existing information combined with expert opinion to make judgments on risk acceptability. To reduce subjectivity and promote equitability, Davis et al. (2010) and Quinn et al. (2013a) each promoted a tiered risk assessment approach that spans qualitative weed risk assessments to quantitative *in situ* field evaluations. Several studies follow weed risk assessments with post-entry evaluations, and focus on demographic variables as indicators of potential invasiveness (Davis et al. 2010; 2011; Matlaga and Davis 2013). Demographic variables such as population growth rates ($\lambda$) are heavily influenced by geography and community (Horvitz and Schemske 1995), so as Davis et al. (2010) suggest, should be evaluated in relevant habitats and locations. However, this could turn out to be dozens of combinations of habitats and geographies; making this parameter no less relevant, but less amenable to rapid quantification.

Below I will review existing methods to quantify invasive potential, in line with recommendations of others (Barney and DiTomaso 2010b; Flory et al. 2012), and follow with a suggested modification for invasiveness forecasting and mitigation.

**Weed Risk Assessments (WRA).** The challenges to predicting invasiveness are well documented (Moles et al. 2008), formidable, and pervasive; but have not intimidated attempts at this "futile" enterprise (Lonsdale and FitzGibbon 2011). Despite these challenges, a field of risk assessment has developed that is largely driven by meeting regulatory needs, but is founded on 50 years of invasion ecology. Weed risk assessments are biosecurity tools using



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existing information on history of introduction, distribution, species traits, and impacts that are scored to yield a relative risk of invasiveness (Koop et al. 2012; Pheloung et al. 1999). The Australian WRA (A-WRA) has been the global standard, and has a stated efficacy of identifying major weeds at > 90% (Gordon et al. 2008). The United States released an improved WRA through the Plant Protection and Quarantine Division of APHIS (PPQ-WRA) that boasts no false-negative or false-positives, and importantly, includes an estimate of analyst uncertainty (Koop et al. 2012). The A-WRA and PPQ-WRA have been applied to bioenergy crops both in specific geographies (Barney and DiTomaso 2008; Buddenhagen et al. 2009; Gordon et al. 2011[all A-WRA]), as well as more broadly (Smith and Barney 2012a [PPQ-WRA]).

The general conclusion of these assessments is that bioenergy crops have a high invasion potential. However, as Hulme (2012) has pointed out, there are serious limitations to the predictive ability of the WRAs, which are actually hindcast systems calibrated with taxonomically biased datasets with unknown base rates (i.e., percentage of introductions becoming invasive). Additionally, the most ecologically relevant component of invasions, their impact, are largely ignored by WRAs; or when incorporated lack sufficient species-level information to be relevant (Koop et al. 2012). Thus, to facilitate meaningful interpretation of WRA results, Smith and Barney (2012a) compared PPQ-WRA results of bioenergy crops against agronomically-introduced weedy/invasive plants (e.g., johnsongrass; orchardgrass, *Dactylis glomerata* L.) and common row crops [e.g., maize, *Zea mays* L.; soybeans, *Glycine max* (L.) Merr.] not considered weedy. They found that in this context, the bioenergy crops were nearly as "weedy", as many of the agronomic crops yielding high risk results. Buddenhagen et al. (2009) compared suitable bioenergy crops to a random selection of Hawaiian exotics, and found a two to four-fold greater probability of establishment for the bioenergy crops. Gordon et al. (2011) found equivocal invasion risk among a group of likely bioenergy crops in Florida. Thus, WRAs remain an excellent first step in evaluating invasiveness (Davis et al. 2010; Quinn et al. 2013a), but are limited in utility at sub-specific taxonomic levels (Flory et al. 2012), and do not include all information relevant to stakeholders (e.g., is risk primarily associated with previous ill-advised introductions, as with giant reed). The issue of cultivars or varieties is also likely to become increasingly important.

**Species Distribution Modeling.** The primary foundation for species distribution models (SDM) is that ecological niches exist as long-term stable constraints on the spatial extent of a taxon (Peterson 2003). Spatio-temporal niche conservatism has been recently challenged especially in the application of SDMs to biological invasions (Pearman et al. 2008). However, in an elegant analysis, Petitpierre et al. (2012) found broad support for niche stability across large geographic regions, thus supporting the application of SDMs, which can be used as management and regulatory tools to identify habitats susceptible to invasion at various spatial scales (Peterson and Vieglais 2001).

SDMs vary widely in their spatial and temporal resolution, which is dependent on data availability and study objectives. For example, Barney and DiTomaso (2011) conducted a global climate niche analysis, which accounts for moisture and temperature only, of 17 bioenergy crops. This coarse analysis is useful for assessing broad geographic potential (i.e., the southeastern United States vs. Mediterranean California), but is unsuitable for identifying specific habitats susceptible to invasion. However, Pattison and Mack (2008) paired a climate-niche analysis of Chinese tallowtree [*Triadica sebifera* (L.) Small] with *in situ* germination trials to test the predictive capacity of the SDM, which is an excellent approach to evaluate model predictions by combining empirical studies with SDMs (see below for further discussion of this approach).

For SDMs to be useful for management and mitigation, they must be conducted at finer spatio-temporal resolutions. There are several approaches for conducting SDMs at higher resolution, the most popular of which is MaxEnt, which requires presence-only data and can be parameterized with a broad selection of environmental, landscape, and climate variables (Phillips et al. 2006). This approach has not been employed for bioenergy crops, but has been used extensively to model invasive species (e.g., Nuñez and Medley 2011), which should be parlayed into bioenergy crops. Important steps towards utilizing SDMs for bioenergy crop invasion mitigation would be to begin with coarse-scale models (e.g., Barney and DiTomaso 2010a) to identify regions to focus landscape-level modeling. Spatial models developed for agronomic site selection should also be leveraged when employing SDMs for invasion mitigation (e.g., Jager et al. 2010), as these likely identify high-productivity sites that will be targeted for production. Additionally, regions approved under BCAP should be the primary focus of invasion susceptibility modeling as these locations will suffer the immediate potential flux of unintentionally dispersed propagules.

**Ecological Evaluations: An Invasion Stage Approach.** Most concerns with bioenergy crop invasion risk are focused on the production fields, which are likely to be large and spatially intermixed within a diverse landscape matrix, and will serve as an annual reservoir of seed or vegetative propagules. Others have parsed the invasion risk of bioenergy crops in relation to the biofuel supply chain (Barney 2012; Barney and DiTomaso 2010b); stating that in addition to the production field there exists important elements that should not be ignored; harvest method,



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transport system, and storage siting and management. Thus, empirical studies and model development should consider all invasion pathways.

The first wave of empirical studies is emerging following the 2006 clarion call regarding the invasive potential of bioenergy crops (Raghu et al. 2006). These ''higher-tier'' studies are quantitative attempts to identify the invasive potential of a range of bioenergy crops: switchgrass (Barney et al. 2012), giant miscanthus (Barney et al. 2012; Matlaga and Davis 2013; Matlaga et al. 2012; Quinn et al. 2011; Smith and Barney 2014), camelina (Davis et al. 2011), and giant reed (Mann et al. 2013a). These early studies can be categorized based on which invasion stage (*sensu* Theoharides and Dukes 2007) they are directly (or indirectly) addressing: (1) escape and dispersal; (2) establishment and demography; (3) landscape spread; and (4) ecological impact.

The most critical step facilitating an invasion is the escape and establishment of a propagule, which is characterized by species traits, propagule pressure, dispersal dynamics (i.e., kernels), and the receiving environment (Barney and Whitlow 2008). Each propagule type—seed, caryopsis, stem fragment, rhizome node—from each species will have a unique biology, dispersal kernel, and establishment potential that will need to be characterized, and will be foundational for development of effective management/mitigation plans (NRCS 2011), as well as to parameterize spatial-demographic models (see below). For example, Quinn et al. (2011) conducted a spikelet dispersal study of giant miscanthus (sterile 'Illinois' clone) and silvergrass or eulaliagrass (*Miscanthus sinensis* Anderss.) in a soybean field in Central Illinois. They found that 95% of the heavier silvergrass spikelets landed $< 50$ m ($< 164$ ft) from the source, while only 77% of the lighter (unfilled) giant miscanthus spikelets fell within the same distance. Some spikelets of both species were collected up to 400 m from the source. This information should be followed with germination studies to identify realized dispersal distances (Bullock et al. 2006), which could be set at management thresholds (e.g., 95% realized dispersal) to design production field set backs. Similar information should be generated for all propagule types and bioenergy crops, as current information is extremely scarce on dispersal kernels.

Following dispersal, propagule biology will interact with the receiving environment to determine the success (or failure) of escaped disseminules. Important metrics to quantify would be minimum viable propagules (especially for vegetative fragments), basic germination requirements (e.g., hydrothermal time models), and seedling stress tolerance (e.g., moisture stress, burial).

Even 'sterile' species should be assessed for potential establishment. The two sterile grasses giant miscanthus and giant reed are limited to stem and rhizome vegetative fragments to spread, which has not limited successful colonization of western riparian areas for giant reed (Bell 1997). Mann et al. (2013b) evaluated the seasonal viability of whole culms and stem fragments of both species in California, and found that unlike giant miscanthus which senesces in winter, giant reed culms are capable of producing new plants all year. We have observed similar phenology in Virginia where giant reed culms go dormant in the winter, and sprout new culms from each aboveground node (J. N. Barney, personal observation). Thus, the potential vegetative propagule number for giant reed per hectare far exceeds sterile giant miscanthus, and likely switchgrass (not counting seeds). In the same study, Mann et al. (2013b) also found that giant miscanthus was capable of emerging from a 10 cm (4 inches) burial depth even with rhizome fragments of only 1 g (0.035 oz). Vegetative fragment biology will be important to design harvest and transport methods that reduce escape of potentially viable propagules.

Germination requirements can be an important tool in modeling emergence for early detection rapid response, as well as characterizing niche requirements. For example, Clifton-Brown et al. (2011) calculated base temperatures for silvergrass, switchgrass, reed canarygrass (*Phalaris arundinacea* L.), perennial ryegrass (*Lolium perenne* L.), and maize, which they then applied to a climate map of Europe to identify suitable seed planting areas, but this same method could be applied for invasion mitigation. For giant reed, Graziani and Steinmaus (2009) calculated a base temperature of 12.7 C (54.9 F) and a hydrothermal time to sprouting of 124.1 MPa C days. We have found that young plants of switchgrass, giant reed, giant miscanthus, and silvergrass all have considerable drought tolerance (Barney et al. 2009; Dougherty et al. 2013; Mann et al. 2013a); information important to inform SDMs (see below). However, little information exists on seed germination or vegetative propagule sprouting for most bioenergy crops.

Following establishment, the escaped individuals must produce self-sustaining populations. Several studies have focused on this post-establishment phase, using population growth rate ($\lambda$) as a key determinant of invasiveness. Davis et al. (2011) estimated demographic rates of the oilseed crop camelina under four disturbance regimes in a rangeland for both spring and fall seedings. Interestingly, they found that population growth rates never exceeded 0.3, and all populations would go extinct within six years. This study is an excellent example of a first tier WRA, where they found high risk (reject via A-WRA), followed by empirical studies that better elucidate invasion risk; in this case a predicted low risk of establishment outside cultivation. In an agronomic setting, Matlaga et al. (2012) calculated age-dependent demographic rates for giant miscanthus populations one to seven years old. They also estimated vegetative expansion rates of 0.15 m $yr^{-1}$ (0.5 ft $yr^{-1}$) with a reproductive potential of 180,000



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spikelets plant$^{-1}$, although none germinated. In a subsequent analysis, Matlaga and Davis (2013) integrated population demographic parameters, clonal spread rates, and reproductive parameters to estimate invasiveness. They concluded that sterile giant miscanthus has a low spread potential without large disturbances that fragment rhizomes; while fertile giant miscanthus has a high spread potential even under low rates of seed viability and seedling survival. It should be noted that this study was receiving habitat agnostic, and we know that not all habitats are equally invasible (Davis et al. 2000). Much remains unanswered: Will giant reed perform in North Carolina as it does in San Diego? In what habitats is giant miscanthus likely to be successful outside of cultivation? Should we not worry about switchgrass invasion?

The final stage of invasion, ecological impact, is arguably the most important, but as Simberloff (2005) argues; the "impacts of introduced species are idiosyncratic…and our ecological knowledge of most species is insufficient to allow a well-informed opinion without substantial study." We recognize the important role of plant functional traits in invasiveness, but have an extremely poor understanding of their role in ecological impact (Drenovsky et al. 2012). Additionally, choosing the endpoints (i.e., which impacts to measure) is not at all straightforward, and has historically been driven by researcher bias (Hulme et al. 2013). To date, no studies have been conducted outside the agronomic context for the consequences of bioenergy crops. This is not to argue that we should ignore the impacts of escaped bioenergy crops, rather that we must strategically and thoughtfully design studies to estimate the potential consequences of escape (Barney et al. 2013). The ecological community should coalesce around specific metrics (e.g., native plant fitness, stream nutrient loading) to be studied in depth across a range of systems. Perhaps the most critical information when making decisions about bioenergy are (1) what are the ecological impacts, (2) what is the probability they will occur, and (3) how easily can they be mitigated?

**A Nested-Feedback Risk Assessment Approach.** The tiered risk assessments discussed above are improved models for assessing invasion risk, but are implemented in a linear fashion (e.g., nested-sieve approach of Davis et al. 2010), which precludes the ability to re-assess earlier tiers when new information becomes available. Considering the general lack of ecological knowledge for most bioenergy crops, which impedes the ability to conduct robust ecological risk assessment, a feedback approach is more appropriate; new information is continuously integrated back into other components to further refine risk analysis (Figure 1). This system should address both the bioenergy supply chain, as well as the stages of invasion to holistically assess invasion risk. I propose a nested-feedback risk assessment (NFRA) that begins with a qualitative WRA, in this case the PPQ-WRA is most appropriate (Koop et al. 2012), followed with SDMs, which can be conducted at various spatial scales, and finally quantitative site-appropriate empirical studies. For example, information derived from empirical studies on giant miscanthus dispersal (Quinn et al. 2011) and demography (Matlaga and Davis 2013; Matlaga et al. 2012) should be integrated into landscape-level SDMs, which may further inform which habitats empirical establishment studies should be conducted. Each iteration further refines our predictive ability, which leads to appropriate mitigation. The NFRA could be employed throughout the crop development and early deployment phase, and be continuously updated as new data are generated.

Importantly, the final outcome of the analysis is a relative risk of invasiveness that is parameterized proportionally to similar high and low risk species. For example, Smith and Barney (2014) evaluated the establishment potential of fertile giant miscanthus in seven habitats in three geographies, and found very low germination and survival; suggesting a low invasive potential. However, they also included two species known to be invasive in each habitat, as well as two species that are not invasive to serve as positive and negative controls, respectively. Overall, both negative and positive control species did poorly in most environments, suggesting that the poor performance of fertile giant miscanthus was more likely driven by environmental factors (dry conditions) than a true low invasive potential. Without this relative outcome, inappropriate conclusions regarding invasiveness may have drawn. In a related study, Smith and Barney (2012b) compared the establishment, population growth, performance, and spread potential of giant miscanthus (fertile and sterile), switchgrass, giant reed, and sugarcane (*Saccharum officinarum* L.) against invasive (e.g., johnsongrass) and native (e.g., big bluestem, *Andropogon gerardii* Vitman) species (Figure 2). As above, this relative design is intended to give context to the desired parameters.

Despite the ecological lexicon of invasive species, only specific populations are causing ecological or economic impact (Barney et al. 2013). Thus, the NFRA approach should be applied in the appropriate context—relevant species and geographic combinations. For example, giant reed is considered one of the worst invaders of the U.S. Southwest (Bell 1997), and many are using that information to argue against its application as a bioenergy crop in the Southeastern US. However, there are giant reed populations in upland settings in California that are not spreading (J.N. Barney, personal observation) because of the relatively slow lateral spread rate (Boland 2006), and lack of seeds. To presume that managed upland plantings of giant reed in the Southeast would similarly result in the invasions of the Southwest (which were the result



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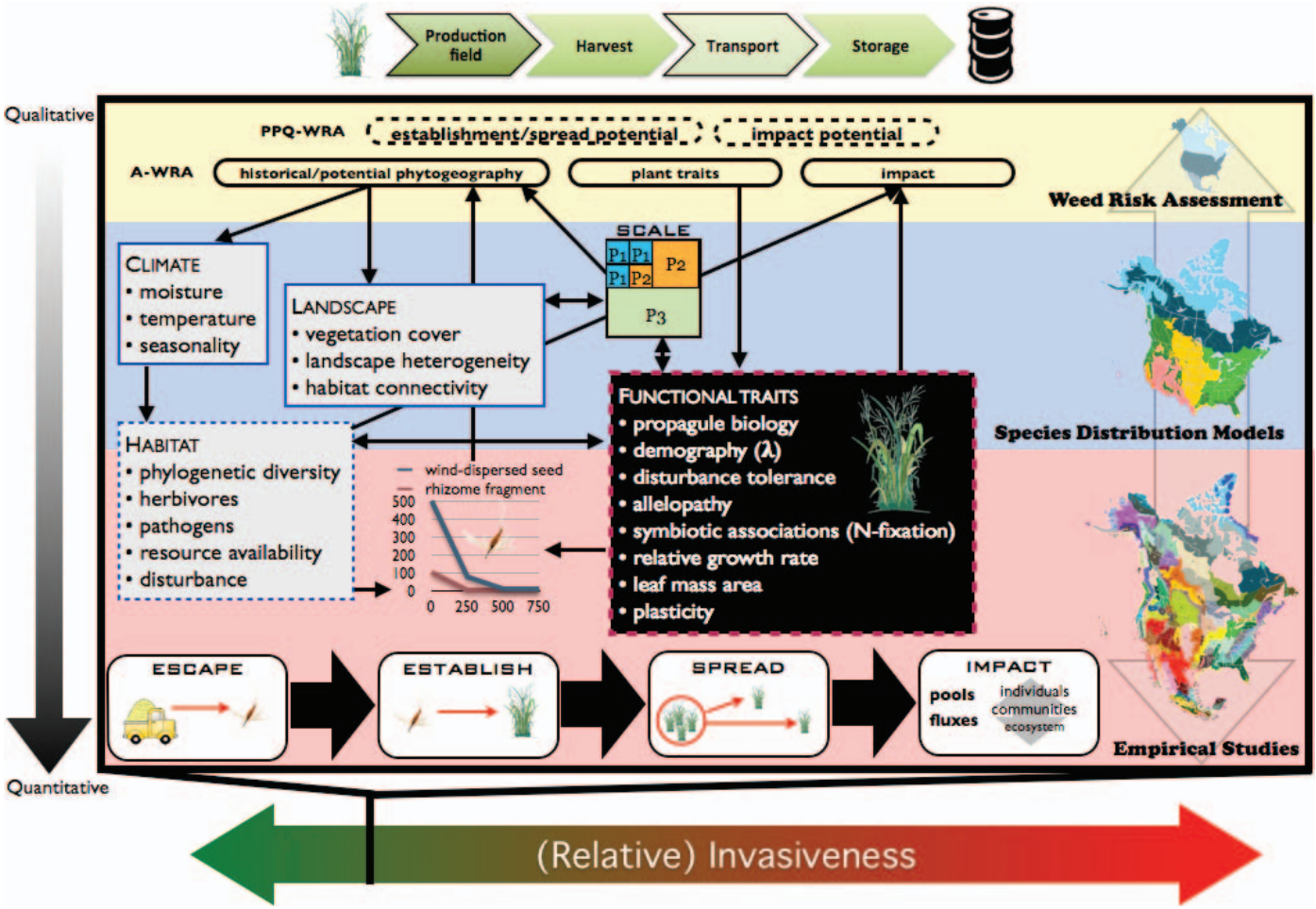


Figure 1. Nested-feedback risk assessment (NFRA) resulting in a relative invasion risk (lower double arrow), which accounts for the biofuel supply chain (upper pathway of green arrows) and invasion stages (lower pathway of white boxes) that is composed of three primary components: (1) weed risk assessment (yellow box); (2) species distribution models (blue box); and (3) empirical studies (red box). Each of the primary components occurs at different spatial resolutions (e.g., national to ecoregions), with sub-components feeding information back into other tiers.

of intentional plantings in the streams for decades) is imprudent, and could potentially lead to a necessary adoption of lower productivity crops requiring additional land. This supports the suggestion of using the NFRA in a new context; e.g., giant reed in the Southeastern United States.

## Mitigating the Inevitable

**Best Management Practices.** The scale of cultivation, number of kilometers transported, and development of more productive feedstocks will likely result in some instances of "right plant, right place, right time" (i.e., a successful escape and invasion). Therefore, we should have realistic expectations of the negative externalities of bioenergy, including escape probabilities, and design pragmatic mitigation procedures of invasion risk directly informed by NFRA analyses. Best management practices (BMPs) should be species specific and provide effective invasion mitigation without being overly onerous on growers and other bioenergy stakeholders. For example, surety bonds covering the cost of eradication are commonly recommended as an incentive for responsible management (Quinn et al. 2013a), but are impractical at commercial scales. Thus, ecologically-derived BMPs are the most appropriate defense against unintended consequences.

There are several examples of generic BMPs (Barney 2012), as well as research needs (DiTomaso et al. 2007; Flory et al. 2012) aimed at invasion mitigation. BMPs should be implemented along the bioenergy pathway, and should be supported by an eradication plan acting as the fail-safe switch. Eradication, the complete removal of all viable propagules from an area, has a lamentable track record, but is most successful on small populations



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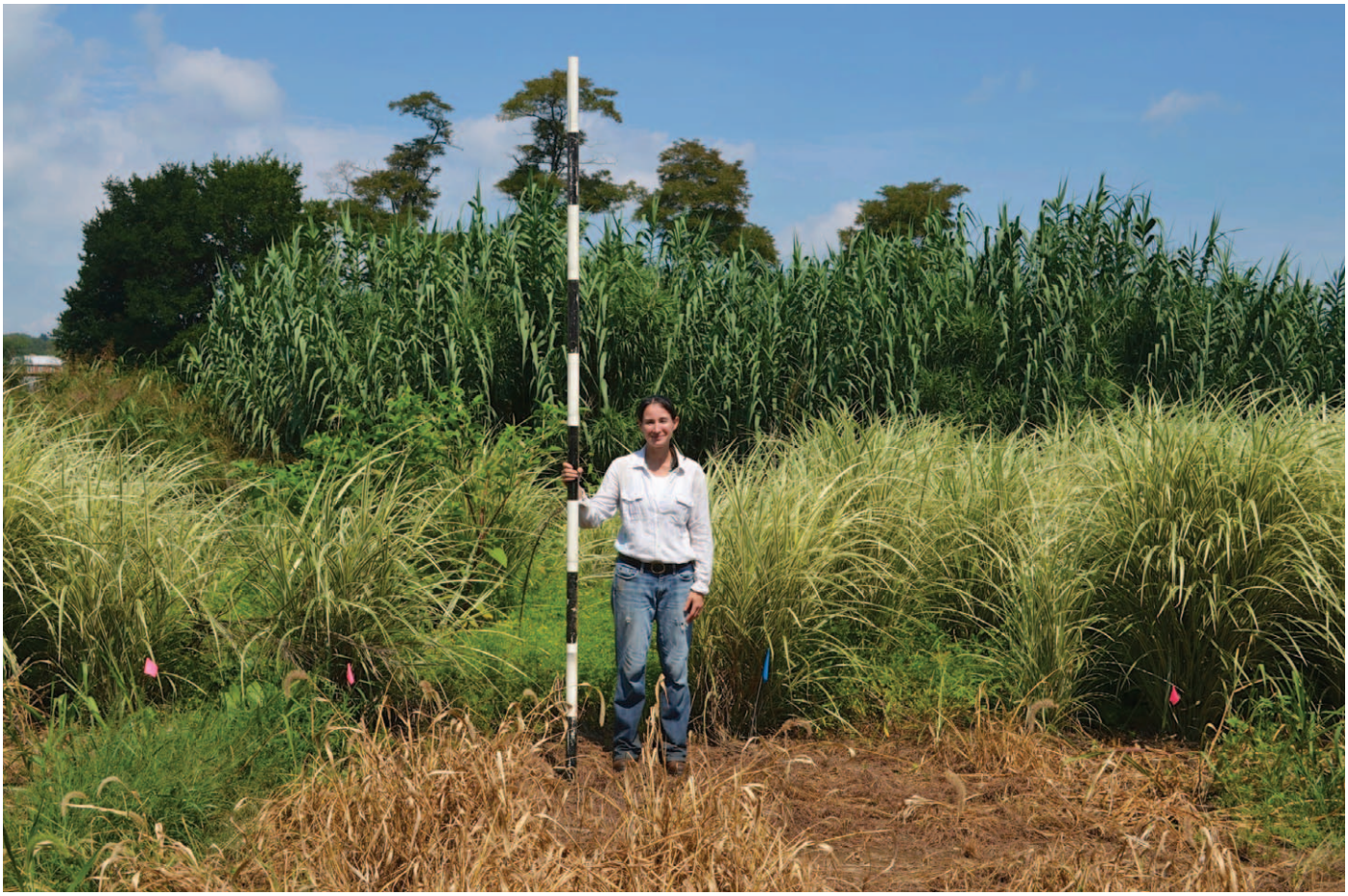

Figure 2. Third year ornamental silvergrass (foreground) and giant reed (background) in head-to-head comparison of performance, population growth, and yield under four competition and disturbance treatments.

requiring fewer resources (Myers et al. 2000). Thus, early detection via regular scouting and reporting of production field, harvest and transport equipment, transport routes, and storage sites followed by rapid response including identification and mitigation are paramount for success. A toolbox of management options must be available that include selective and nonselective control techniques for dealing with escapes and production field transitions, respectively.

**A Path Forward.** While a renewable energy portfolio built upon large exotic perennial grasses selected for traits that may facilitate their escape is risky, the fulcrum on which success will be judged is balanced between the goals and coordinated efforts of agronomists, crop developers, weed scientists, and policy makers (Simberloff 2008). This path must be tread lightly, and all efforts must be taken to prevent the next great wave of invasion. However, application of the precautionary principle is a slippery slope that may lead to unwarranted and broad rejection of high biomass species in favor of more acceptable externalities, such as using lower biomass crops on more land, or using agricultural residues with concomitant resource intensification, erosion, and carbon loss (Robertson et al. 2008). The summative balance between the competing demands of agricultural intensification, climate change mitigation, policy, and conservation must begin with a completed balance sheet, which we currently lack. An integrated cost-benefit analysis must be conducted (Yokomizo et al. 2012), recognizing the limits of scant information and uncertainty.

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