

Invasion Note

Relative importance of wetland type versus anthropogenic activities in determining site invasibility

Gary Ervin^{1,*}, Melissa Smothers², Christopher Holly¹, Cori Anderson³ & Jeffrey Linville¹

¹*Department of Biological Sciences, Mississippi State University, MS, 39762, USA;* ²*Department of Biological Sciences, Humboldt State University, Arcata, CA, USA;* ³*Department of Biological Sciences, Birmingham-Southern College, Birmingham, AL, USA;* **Author for correspondence (e-mail: gervin@biology.msstate.edu)*

Received 5 September 2005; accepted in revised form 20 January 2006

Key words: anthropogenic disturbance, aquatic plants, dispersal, invasion indices, invasive species, landscape characteristics, wetland plants

Abstract

We assessed wetland invasibility by conducting surveys of three wetlands in each of five categories (riverine, depression, lacustrine fringe, mineral flat, and seepage slope). Invasibility was measured as the number of invasive species present, percent of plant species classified as invasive, percent cover of invasive plants, and percent of total cover represented by invasive species. The working hypothesis for this study was that certain types of wetlands (e.g., lacustrine fringe and riverine) would be more prone to invasion than others (spring-seep/slope wetlands or mineral flat wetlands). No significant differences were found among wetland types in any of the invasion metrics evaluated, despite high average invasibility in the riverine and lacustrine fringe categories. However, invasion was correlated very strongly with a qualitative index of anthropogenic modification to the surrounding landscape. A probable result of the substantial influence of human activities on wetland invasion in this study was that effects potentially attributable to greater opportunity for dispersal in certain types of wetlands were obscured. Another factor that likely contributed to the lack of differences among wetland types was the high variability in human activities observed among wetlands within types. These results further highlight the overwhelming contributions of anthropogenic habitat modification and human-assisted dispersal of invasive species to the currently observed homogenization of natural ecosystems.

Introduction

Invasibility, the relative propensity of an ecosystem to acquire and retain invasive plant species, is a complex and multifaceted phenomenon. Invasion of natural systems may be influenced by biotic factors such as propagule pressure, non-native species traits, and resilience of the native

community. Establishment of invasive species may be enhanced by such small-scale factors as light availability (Foster et al. 2002; Prieur-Richard et al. 2002; Milbau et al. 2003), soil disturbance (Foster et al. 2002), and soil nitrogen availability (Prieur-Richard et al. 2002; Brewer and Cralle 2003; Brooks 2003). At larger scales, invasion can be enhanced by the very structure

of the landscape through large-scale effects on dispersal mechanisms, including habitat fragmentation or provision of dispersal corridors (With 2002). Anthropogenic landscape-scale modifications such as agricultural land use (Lopez et al. 2002) and transportation corridors, such as paved roads (Gelbard and Belnap 2003; Gelbard and Harrison 2003) and other highly traveled routes (Jules et al. 2002; Larson 2003), also serve to facilitate invasion. On the other hand, less intensively disturbed land cover, such as grasslands and forests, tends to reduce the likelihood of non-native species establishment (Lopez et al. 2002). Such also may be the case in areas with high densities of natural wetlands.

An additional dimension to the invasibility of aquatic and wetland systems is the potential movement of propagules by surface waters, especially streams and larger rivers. Such transport is inherent in aquatic ecosystems and may serve as a natural dispersal mechanism for invasive species. Dispersal of exotic aquatic plants was discussed by Madsen (2004) in particular reference to the state of Mississippi (USA), with both natural and human-assisted dispersal assisting in dissemination of invaders along and among water bodies. Owens et al. (2001) found that more than half the biomass of aquatic plant fragments transported on the San Marcos River (Texas, USA) belonged to exotic species, with the highest biomass contributed by *Hydrilla verticillata* (L.f.) Royle and *Hygrophila polysperma* (Nees. T. Anderson). In Australia, spread of *Egeria densa* Planchon through the Hawkesbury-Nepean River basin further illustrated natural dispersal of invasive species in aquatic ecosystems (Roberts et al. 1999). Dispersal of *Egeria* was enhanced by flood events that dislodged vegetative fragments of the plant and transported them downstream. Additional accounts of the potential opportunities for invasive spread in aquatic and wetland ecosystems, such as sedimentation, influx of debris, and hydraulic-induced sediment disturbance, were discussed by Zedler and Kercher (2004).

The present study was conducted to evaluate the hypothesis that certain types of wetlands (e.g., lacustrine fringe and riverine) would be more prone to invasion than others (spring-seep/slope wetlands or mineral flat wetlands). This hypothesis was based on the premise that natural

dispersal routes (i.e., flowing surface waters) may contribute to the spread of invasive aquatic plant species. To test our hypothesis, we assessed wetland plant invasibility in each of five types of naturally occurring wetlands across the state of Mississippi.

Methods

Study sites

We assessed invasibility of different wetland types by conducting vegetation and habitat surveys in riverine, depressionnal, lake fringe, mineral flat, and spring-seep slope wetlands (Clarain 2002). Invasibility was measured as the number of invasive species present, percent of plant species classified as invasive, percent cover of invasive plants, and percent of total cover represented by invasive species. A total of 15 sites were chosen, 3 from each type of wetland (Table 1, Figure 1). Sites within a given wetland type were selected based strictly on accessibility and sampling logistics, with an effort to sample across a fairly large geographic extent; no site characteristics other than wetland type were used in choosing sites for inclusion in the study. Study sites were located on a mixture of public and private lands, across a range of anthropogenic impact.

Sampling

At each site, 21 nested 0.25 m² plots were systematically placed along the land-water interface (where water was present at the time of sampling) beginning at a randomly selected point. Each set of plots consisted of three subsets of seven 0.25 m² plots that were spaced at the corners of nested 1 m² and 9 m² quadrats; the three nested sets of plots were spaced at 10 m intervals, thus the plots in total covered 107 m² at each site.

Data collected at each site included: soil pH, water depth, litter mass, light interception by the plant canopy, and vascular plant species present and percent ground coverage by species. Above-ground litter mass was collected from the central 0.04 m² of the 0.25 m² subplots. Data were

Table 1. Characteristics of the wetlands study sites used in this research.

Site	Type	Native species	Invasive species ^a	Total cover	Invasive % cover	Litter (g)	pH	% Light Red. ^b	Light available ^c	Water depth (cm)	Dist rank
1. DeSoto	Flat	11	0	39	0	6.4	5.8	54	1390	0.0	3
2. SH Crane	Flat	21	1	62	1	6.8	5.4	43	1598	13.7	3
3. Crosby	Flat	19	0	82	0	9.9	5.0	66	934	0.2	3
4. Pigeon R	Slope	16	0	23	0	14.0	5.8	34	145	0.7	1
5. Snake Spr	Slope	16	0	12	0	13.6	5.8	18	21	1.9	2
6. Shaw Spr	Slope	9	1	29	0	18.7	5.3	52	63	1.3	2
7. Church P	Depr	9	2	74	11	7.4	5.8	75	853	5.6	10
8. Tupelo	Depr	12	0	85	0	1.6	7.6	25	1748	3.4	18
9. Holly Spr	Depr	7	0	76	0	14.5	5.2	52	1294	8.1	4
10. N Bay	Fringe	8	3	74	53	12.0	5.3	73	924	26.0	22
11. Choctaw L	Fringe	18	0	61	0	5.6	6.3	83	1548	6.3	3
12. NNWR	Fringe	22	0	26	0	12.5	5.6	29	811	22.4	4
13. Larry P	Riv	10	0	70	0	6.8	6.2	81	465	1.6	3
14. Yazoo	Riv	13	1	75	22	2.7	6.1	61	1643	4.6	18
15. TTWW	Riv	22	5	89	45	39.7	5.4	81	411	24.4	22

^aSpecies encountered were: *Alternanthera philoxeroides* (Mart.) Griseb.; *Centella asiatica* (L.) Urban; *Commelina diffusa* Burm. f.; *Eichhornia crassipes* (Mart.) Solms; *Ligustrum sinense* Lour.; *Lonicera japonica* Thunb.; *Myriophyllum aquaticum* (Vell.) Verde.; *Paspalum notatum* Fluegge; *Spirodela punctata* (G.F.W. Mey) C.H. Thompson; and *Verbena brasiliensis* Vell.

^bCalculated as a percent of above-canopy PAR.

^cMeasured as $\mu\text{mol m}^{-2} \text{s}^{-1}$.

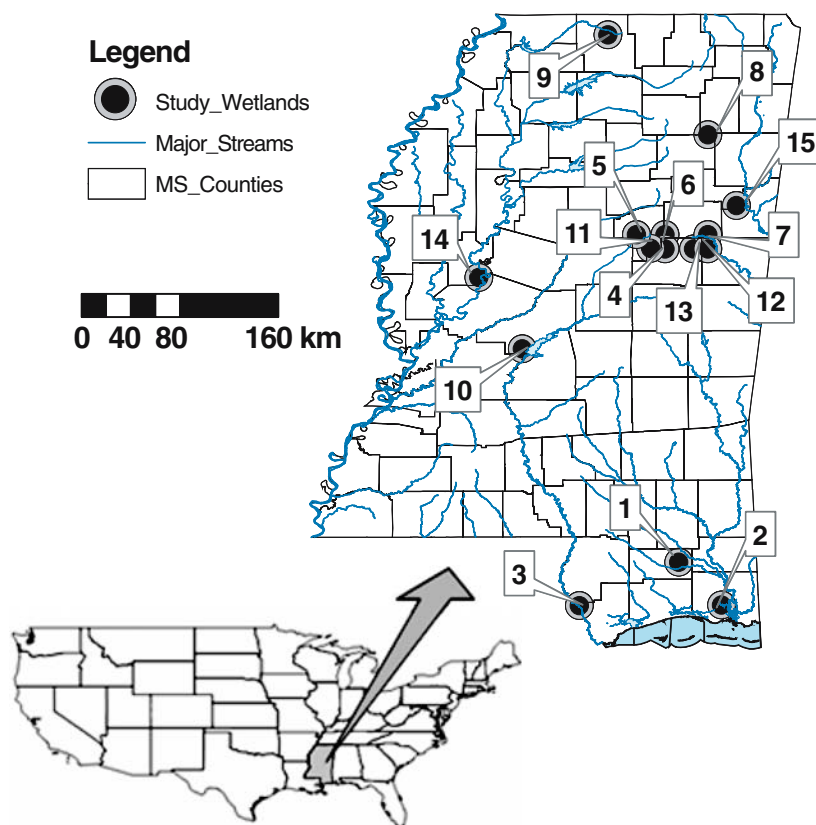


Figure 1. Locations of study sites. Numbers refer to those assigned to each site in Table 1.

collected for litter, water depth, and plant species in each subplot; light and soil pH data were collected only at the corners of the 9 m² plots (12 sub-samples per site).

Soil pH was measured for soil samples that were returned to the laboratory and mixed into a 1:1 slurry with distilled water. The pH of these slurries then was measured with an Ex-Tech ExStik[®] pH meter, calibrated against known pH buffers (FisherBrand BufferPac buffers). Water depth was measured with a 1-m metal ruler at the center of each subplot. Litter mass was measured by collecting all above-ground dead plant material within a 0.04 m² round sub-sample at the center of each subplot, returning the material to the laboratory and drying at 105°C for ≥24 h prior to weighing. Light availability above the canopy and at the soil or water's surface was measured with a Li-Cor Environmental LI 190SA quantum sensor connected to a LI 250A meter.

Using the methodology presented by Lopez and Fennessy (2002) and US EPA (2002), we evaluated study sites based upon the qualitative intensity of human activities within the immediately surrounding landscape. Each site was evaluated hierarchically as to whether it (1) was surrounded within the landscape by (a) forest or grassland, (b) fallow agriculture, (c) active agriculture, or (d) urban areas, (2) was surrounded by immediately adjacent (a) forest, (b) grassland, or (c) no buffer zone, and finally (3) possessed obvious signs of hydrologic alteration. Possible disturbance rankings resulting from this approach ranged from 1 to 24, where a rank of 1 indicated a wetland surrounded by natural forest or grassland, with a forest buffer and no obvious signs of hydrologic alteration, and a rank of 24 represented an urban wetland with no vegetative buffer and having undergone apparent hydrologic alteration.

Analyses of plant species data initially were planned both for exotic species only and for total invasive species defined as those species introduced to the US (Flora of North America Editorial Committee 1993; USDA, NRCS 2004), plus those native species included in Miller et al. (2004). Since no native invasive species were recorded in the surveys, analyses were performed simply on invasive species defined as all non-

native/introduced species, according to the above references.

Analyses

Plot-level environmental data were averaged or otherwise compiled across subplots at each site to derive a single value for each site; this was done to ameliorate the problem of pseudoreplication in data analyses, as the main factor of interest, wetland type, was "imposed" at the level of the whole wetland in each case. For species inventories, data were accumulated over all subplots for native and invasive richness, and cover data were averaged across the 21 subplots. Other parameters (litter, pH, light, water depth) were simply averaged across all available subplots at each site to obtain the single measure for each wetland.

To evaluate the effect of wetland type on invasibility, analyses of variance were performed in Systat v 11.0 (Systat Software, Inc.), using average or composite values for each parameter calculated for each site as described above (total of 15 replicate values of each). Also, bivariate correlations (Pearson's) were performed for all variables, and regression analyses conducted among significantly correlated variables whose relationships appeared ecologically informative. Correlation analyses, including regressions, were carried out in SPSS v 12.0 (SPSS, Inc.).

Results and discussion

Data on floristic and physical characteristics of the study sites are given in Table 1, including the 10 invasive species encountered during these surveys. Only 2 of those 10 species were absent from the Miller et al. (2004) listing of invasive plants in the southeastern United States: *Centella asiatica* and *Commelina diffusa*. However, *Commelina diffusa* is considered a weed by the Southern Weed Science Society (1998) and is recognized as having been introduced into the US, north of tropical Florida (Faden 2000). *Centella asiatica* is not recognized as an economically important weed, but is a frequent component of disturbed habitats in areas along the US Gulf Coast (Ervin et al. personal observations).

Contrary to our principal hypothesis, invasibility did not vary significantly among wetland types (Figure 2). Despite the rejection of our initial hypothesis, the study revealed significant relationships between invasibility and both water depth and disturbance rank, with water depth and disturbance rank themselves also being significantly correlated (correlation coefficient = +0.55; $P=0.03$). We presumed human activity to be of more importance than water depth in dispersal of invasive species; thus, data are presented graphically only for the correlations between invasibility and disturbance rank (Figure 3). Additional regressions were conducted on natural log-transformed data for invasive species percent cover and relative cover of invasives. Results were virtually identical to those using non-transformed data, thus the non-transformed data are depicted for ease of interpretation.

Failure to find a significant effect of wetland type on degree of invasibility may have resulted from low levels of replication within wetland types ($n=3$). However, power analyses indicated that 60–100 replicate wetlands of each type would have been needed in order to discern statistically (with $1 - \beta = 0.8$) the largest differences in invasibility among wetland types, given the amount of variability observed among the wetlands we surveyed. Thus, no realistic increases in sample size would have provided the ability to distinguish degree of invasibility among wetland types in this study.

Other potential factors that may have obscured the effects of natural dispersal mechanisms among wetland types were the state and relative timing of initial system conditions. Differential periods of post-disturbance recovery and variation in species pools certainly could have contributed to the observed high variance

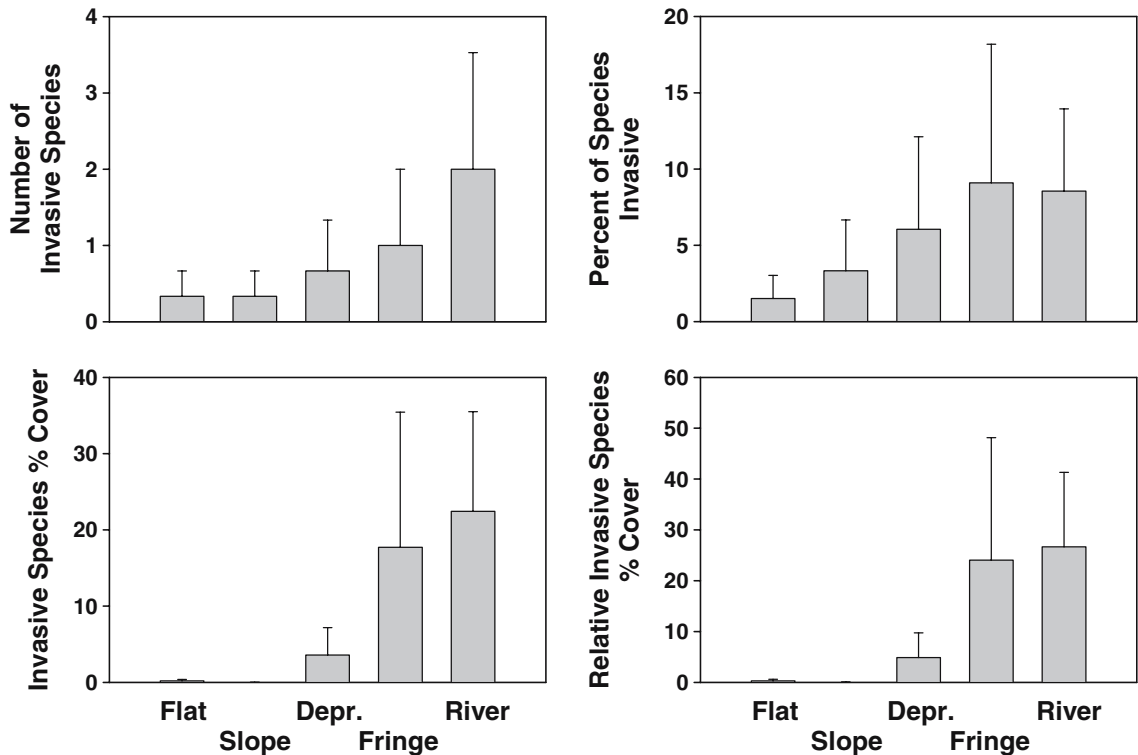


Figure 2. Comparison of invasibility metrics across wetland types. No measure of invasibility was found to differ significantly among types of wetlands, despite the marked differences in means, especially for percent cover of invasives. Relative invasive % cover is invasive species cover divided by mean total cover per subplot at each site.

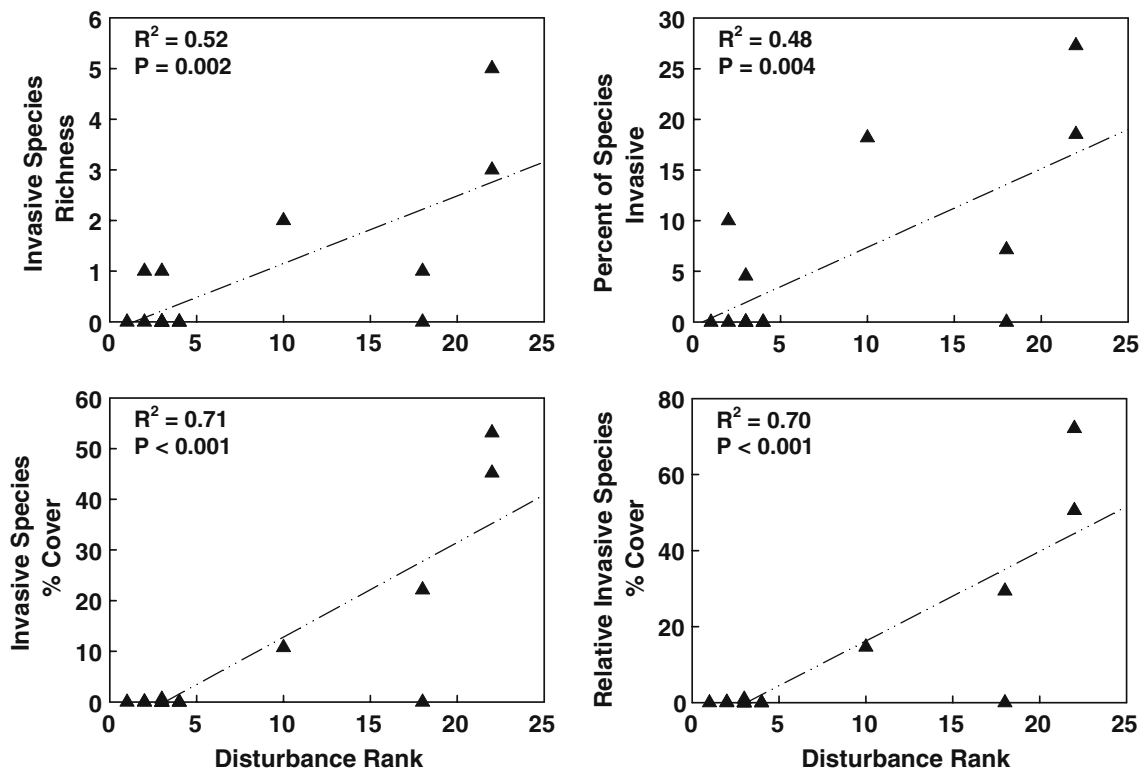


Figure 3. Correlations between invasibility metrics and site disturbance rank. Relative invasive % cover is invasive species cover divided by mean total cover per subplot at each site.

in invasibility among wetland sites within types, but these factors were not quantified in the present work. Temporal dynamics in complex ecological systems have the potential to create considerable variance among seemingly similar systems, and this phenomenon has been credited with some major disconnects between the typically equilibrium-based ecological theory and the typically short-term studies used to evaluate theory (Hastings and Higgins 1994; Brown et al. 2001; Hastings 2001).

It is also possible that the lack of a statistically discernible relationship between wetland type and invasibility resulted from a masking effect of variation in anthropogenic influence within types. Human activities such as agriculture, boat traffic, and recreational land use certainly contribute to the dispersal of invasive species and provide suitably disturbed habitats for establishment. In fact, numerous landscape-scale factors, including those used in assigning disturbance ranks via the

method of Lopez and Fennessy (2002), have been linked to invasive plant response. Whereas grasslands and forests tend to reduce the likelihood of non-native species establishment (Lopez et al. 2002), invasion can be facilitated by agricultural land use (Lopez et al. 2002) and transportation corridors (Gelbard and Belnap 2003; Gelbard and Harrison 2003). More direct effects of human use such as trails in national parks and other semi-wild lands (Lonsdale 1999; Jules et al. 2002; Larson 2003) and even direct attachment of propagules to automobiles have been well established (Hodkinson and Thompson 1997).

Deeper wetland habitats may be more likely to experience human visitation associated with recreation, and a number of pieces of evidence here suggested that the correlations between invasibility and disturbance rank or water depth were driven largely by human activity, rather than water depth itself. For one, the three sites with deepest water measurements were exposed to high levels

of recreational activities (North Bay on the Ross-Barnett Reservoir in the state's capital city, fringe areas of Bluff Lake at the USFWS Noxubee National Wildlife Refuge, and a backwater area of the USACE Tennessee-Tombigbee Waterway). Bivariate scatterplots of the data also suggested a more clear-cut correlation between disturbance and invasibility than between water depth and invasibility. Finally, the patterns observed in bivariate plots were supported by data from correlation analyses (e.g., Figure 3). Thus, although we set out to evaluate the importance of natural mechanisms of invasive plant dispersal, the available data suggested instead that pervasive anthropogenic landscape-scale factors masked any potential differences in invasibility resulting from natural properties of the wetland ecosystems.

Acknowledgements

This work was supported in part by a Research Experiences for Undergraduates grant from the National Science Foundation (DBI0244113 to Giselle Thibaudeau) and by USGS grants 01HQGR0088 (WRI) and 04HQAG0135 (USGS BRD) to G N Ervin. Mark Fishbein, Brook Herman, and Jason Bried assisted in plant identification. Provision of access to field sites by the Crosby Arboretum, Sandhill Crane and Noxubee National Wildlife Refuges, De Soto and Tombigbee National Forests, and the National Audubon Society is greatly appreciated. This paper has benefited from comments provided by Leslie Adams, one anonymous reviewer, and James Drake. The views and conclusions contained in this document are those of the authors and should not be interpreted as necessarily representing the official policies, either expressed or implied, of the U.S. Government.

References

- Brewer JS and Cralle SP (2003) Phosphorus addition reduces invasion of a longleaf pine savanna (Southeastern USA) by a non-indigenous grass (*Imperata cylindrica*). *Plant Ecology* 167: 237–245
- Brooks ML (2003) Effects of increased soil nitrogen on the dominance of alien annual plants in the Mojave Desert. *Journal of Applied Ecology* 40: 344–353
- Brown JH, Whitham TG, Morgan Ernest SK and Gehring CA (2001) Complex species interactions and the dynamics of ecological systems: long-term experiments. *Science* 293: 643–650
- Clarain EJ Jr (2002) Hydrogeomorphic approach to assessing wetland functions: guidelines for developing regional guidebooks; Chapter 1, Introduction and Overview of the Hydrogeomorphic Approach. ERDC/EL TR-02-3. U.S. Army Engineer Research and Development Center, Vicksburg, MS
- Faden RB (2000) Commelinaceae R. Brown. In: Flora of North America Editorial Committee (eds) *Flora of North America north of Mexico*, p 195. Oxford University Press, New York
- Flora of North America Editorial Committee, (eds) (1993) *Flora of North America North of Mexico*, 7 vols. New York and Oxford, Retrieved from <http://www.efloras.org> on 05 July 2005
- Foster BL, Smith VH, Dickson TL and Hildebrand T (2002) Invasibility and compositional stability in a grassland community: relationships to diversity and extrinsic factors. *Oikos* 99: 300–307
- Gelbard JL and Belnap J (2003) Roads as conduits for exotic plant invasions in a semiarid landscape. *Conservation Biology* 17: 420–432
- Gelbard JL and Harrison S (2003) Roadless habitats as refuges for native grasslands: interactions with soil, aspect, and grazing. *Ecological Applications* 13: 404–415
- Hastings A (2001) Transient dynamics and persistence of ecological systems. *Ecology Letters* 4: 215–220
- Hastings A and Higgins K (1994) Persistence of transients in spatially structured ecological models. *Science* 263: 1133–1136
- Hodkinson DJ and Thompson K (1997) Plant dispersal: the role of man. *Journal of Applied Ecology* 34: 1484–1496
- Jules ES, Kauffman MJ, Ritts WD and Carroll AL (2002) Spread of an invasive pathogen over a variable landscape: a nonnative root rot on Port Orford cedar. *Ecology* 83: 3167–3181
- Larson DL (2003) Native weeds and exotic plants: relationships to disturbance in mixed-grass prairie. *Plant Ecology* 169: 317–333
- Lonsdale WM (1999) Global patterns of plant invasions and the concept of invasibility. *Ecology* 80: 1522–1536
- Lopez RD and Fennessy MS (2002) Testing the floristic quality assessment index as an indicator of wetland condition. *Ecological Applications* 12: 487–497
- Lopez RD, Davis CB and Fennessy MS (2002) Ecological relationships between landscape change and plant guilds in depressional wetlands. *Landscape Ecology* 7: 43–56
- Madsen J (2004) Invasive Aquatic Plants: A Threat to Mississippi Water Resources. In: McBride D (ed) *Proceedings of the 34th Mississippi Water Resources Conference*, April 2004. Mississippi State University, Mississippi
- Milbau A, Nijs I, Van Peer L, Reheul D and DeCauwer B (2003) Disentangling invasiveness and invasibility during invasion in synthesized grassland communities. *New Phytologist* 159: 657–667
- Miller JH, Chambliss EB and Barger CT (2004) Invasive Plants of the Thirteen Southern States. *Invasive.org: Invasive and Exotic Species of North America*. Retrieved from <http://www.invasive.org/> on 04 July 2005

- Owens CH, Madsen JD, Smart RM and Stewart RM (2001) Dispersal of native and nonnative aquatic plant species in the San Marcos River, Texas. *Journal of Aquatic Plant Management* 39: 75–79
- Prieur-Richard AH, Lavorel S, Dos Santos A and Grigulis K (2002) Mechanisms of resistance of Mediterranean annual communities to invasion by *Coryza bonariensis*: effects of native functional composition. *Oikos* 99: 338–346
- Roberts DE, Church AG and Cummins SP (1999) Invasion of *Egeria* into the Hawkesbury-Nepean River, Australia. *Journal of Aquatic Plant Management* 37: 31–34
- Southern Weed Science Society (1998) Weeds of the United States and Canada. Southern Weed Science Society, Champaign, Illinois, CD-ROM
- US EPA (2002) Methods for evaluating wetland condition: using vegetation to assess environmental conditions in wetlands. US EPA, Washington, DC, EPA-822-R-02-020, Office of water
- USDA NRCS (2004) The PLANTS Database, Version 3.5. National Plant Data Center, Baton Rouge, LA 70874-4490 USA. Retrieved from <http://plants.usda.gov> on 05 July 2005
- With KA (2002) The landscape ecology of invasive spread. *Conservation Biology* 16: 1192–1203
- Zedler JB and Kercher S (2004) Causes and consequences of invasive plants in wetlands: opportunities, opportunists, and outcomes. *Critical Reviews in Plant Sciences* 23: 431–452