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Before, during and after: the need for long-term monitoring in invasive plant species management

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Abstract

The invasion of non-indigenous plants is considered one of the primary threats to rare and endangered species as well as to the integrity and function of North American ecosystems. However, many of the suspected negative ecosystem impacts are based on anecdotal evidence. For example, there is almost unanimous agreement among natural resource managers of the detrimental ecological impacts of species such as *Lythrum salicaria* (purple loosestrife), *Phragmites australis* (common reed) and *Alliaria petiolata* (garlic mustard) but convincing documentation is scarce. Experimental and theoretical ecology predicts large ecosystem impacts of the most widespread invasive species. However, it is difficult to prioritize control of species that occur at intermediate densities. Long-term monitoring before and during the invasion as well as before, during and after any control attempts can provide valuable ecological information. In particular, it is important to understand how changes in the abundance of species influence ecosystem properties and processes which, in turn, will help guide management decisions. Ideally, this monitoring has to go beyond 'simple' impacts on plant communities, involve cross-disciplinary teams of scientists and should incorporate many different taxa and their interactions. Monitoring design and data collection should be sophisticated enough to allow statistically sound data analysis. The available information will be paramount in (1) developing new political and scientific guidelines in invasive species management, (2) helping resolve potential conflicts of interest and (3) helping change public attitudes regarding growth, sale, and control of non-indigenous species.

Introduction

The invasion of non-indigenous plant species (NIS) is considered one of the primary threats to rare and endangered species (Usher 1988; Macdonald et al. 1989; US Congress 1993; Randall 1996) as well as to the integrity and function of North American ecosystems (Drake et al. 1989; Randall 1996; Westbrooks 1998). Among the many impacts of NIS are changes of fire regimes (*Bromus tectorum*), alterations of biogeochemical cycling (*Tamarix* spp.), alterations of geomorphological processes (*Ammophila arenaria*), changes in hydrological cycles (*Melaleuca quinquenervia*), prevention of recruitment or reproduction of native species (*Lonicera japonica*, *Casuarina equisetifolia*), hybridization with native species (*Spartina alterniflora*) and concerns over human health effects (see Table 1 for details). All of the above examples involve NIS in natural areas, systems managed for the preservation of their native fauna and flora and natural processes.

While we have accumulating evidence for negative impacts of NIS (Table 1), we are faced with a checkerboard of data from many different systems and disciplines. Data collection is driven by interest and expertise of individual researchers, as well as shifting priorities of management and funding agencies and lacks coordination (US Congress 1993). Considering

Table 1. Impacts of selected invasive plants in North America.

Impact	Species	Area of concern	References
Increased soil salinity	*Tamarix ramosissima*	Riparian areas in West	Macdonald et al. 1989; di Tomaso 1998; Sala et al. 1996
	Mesembryanthemum crystallinum	Coastal areas in CA	Vivrette and Muller 1977
Increased sedimentation	*Spartina alterniflora*	Mudflats of Pacific coast	Daehler and Strong 1996
	Tamarix ramosissima	Riparian areas	Westbrooks 1998; Macdonald et al. 1989
Increased evapotranspiration	*Tamarix ramosissima*	Riparian areas	Westbrooks 1998; Macdonald et al. 1989
Increased nitrogen fixing	*Myrica faya*	Hawaii Volcanoes National Park	Vitousek and Walker 1989
Threat in turtle nesting habitat	*Casuarina equisetifolia*	Florida	Austin 1978
Increased sand-fixation	*Ammophila arenaria,*	Coastal dunes of Oregon and northern California	Schwendiman 1977; Macdonald et al. 1989
Hybridization with native species	*Spartina alterniflora*	Mudflats of Pacific coast	Anttila et al. 1998; Daehler and Strong 1996
Earlier and more frequent fires	*Bromus tectorum*	Intermountain West	Macdonald et al. 1988; Yensen et al. 1992; Knapp 1992
Preventing recruitment of native species	*Lonicera japonica* *Hedera helix*	Theodore Roosevelt Island, DC	Thomas 1980
Photodermatitis	*Heracleum mantegazzianum*	NY, PA, WA	Jaspersen et al. 1996; Westbrooks 1998

that about 5000 plant species are naturalized in the United States of which at least 10% are seriously invasive (US Congress 1993), it may not be surprising that quantitative evidence for potential changes in ecosystem processes and species diversity is missing for most of these species. Most of the available information involves 'more easily obtainable' data on ecosystem processes (salinity, sedimentation, evapotranspiration) while quantitative data on the impact on biotic communities are missing and often anecdotal. The important questions, however, are how much evidence for negative impacts of NIS do we need to begin control programs (particularly biocontrol), and can we start preventive management (eradication of newcomers)? In the following I will examine the available evidence for the justification of control programs against three widespread invasive plants *A. petiolata*, *P. australis*, and *L. salicaria*.

How much evidence do we have?

To collect data on the evidence for impacts of invasive plant species, literature databases (biosis, agricola) were searched for published papers using appropriate keywords. Additional information was obtained from submitted manuscripts, published reports and, occasionally, through personal communications.

Alliaria petiolata

First recorded in 1869, garlic mustard now occurs in the Northeast, Southeast and Midwest (Nuzzo 1993) and is suspected to displace the indigenous understory flora in invaded woods (Cavers et al. 1979; Nuzzo 1993; McCarthy 1997). Selective herbivory by increasing numbers of white-tailed deer (*Odocoileus virginianus*) stressing native plants may contribute to the decline of native species (Nuzzo, unpublished). Aggressive and repeated control attempts using hand pulling, herbicides, and fire have not resulted in acceptable reductions in spread or abundance of garlic mustard (Nuzzo 1991). Although long-term monitoring studies were initiated in 1989 (V.A. Nuzzo, pers. comm.), there is currently little (published) quantitative documentation for the negative impacts of garlic mustard on forest understory communities (Table 2). It is likely that changes in herbaceous ground layer composition and cover will have negative impacts on ground-nesting birds, reptiles, amphibians, rodents, insect communities and other ecosystem properties, but

Table 2. Ecosystem impacts of *Alliaria petiolata*.

Pattern	References
Suspected replacement of spring ephemerals	Nuzzo 1993; Cavers et al. 1979; McCarthy 1997
Population sink for native butterfly *Pieris napi oleraceae*	Chew 1981

this awaits confirmation. Surprisingly, the anticipated replacement of spring ephemerals has not been quantitatively documented, although a seven-year study by V.A. Nuzzo (unpublished data) in high quality woods in northern Illinois shows reduced cover and species richness as a result of garlic mustard invasion. This example also highlights the importance of long-term planning since changes in species abundance may not be detectable over the short term or may be difficult to link to the spread of a non-indigenous species (considering typical annual population fluctuations). The best documented evidence of a negative impact of garlic mustard is the interference with oviposition of the rare native butterfly *Pieris napi oleraceae*. Females of this butterfly lay eggs on garlic mustard; however, since larvae are unable to complete development, garlic mustard is a population sink for this species (Chew 1981).

Phragmites australis

Work on this species can be considered a special case since it is not yet clear whether the species (or at least some genotypes) are native to North America (Marks et al. 1994). Animal use of *P. australis* in North America has been summarized (Meyerson et al. 1999) and although a wide variety of species do occasionally use common reed, these are mostly generalists or species introduced from Europe. The ambivalence in the data (Table 3) is demonstrated by the fact that although the bullfrog (*Rana catesbeiana*) in the lower Colorado prefers *P. australis* over other habitat types (Clarkson and Devos 1986), the species is non-indigenous west of the Rocky Mountains and has a negative impact on some native frogs (Stebbins 1985). Two native butterflies, *Ochlodes yuma* and *Poanes viator*, are known to feed on *P. australis* in North America (Opler et al. 1995). *O. yuma* is a Western species and *P. australis* is the only known host plant. *P. viator*, an uncommon species, has recently increased its range by including *P. australis* in its diet (Gochfeld and Burger 1997). In a survey of Connecticut marshes, bird species composition differed between *Phragmites* dominated and short-grass habitats (Benoit 1997). Endangered, rare or threatened species were exclusively found in the short-grass habitats (Benoit 1997). *P. australis* hosts a depauperate insect fauna in North America; 19 species are known from *P. australis* and only 4 are native (the status of 5 species is unclear). Insect herbivores are abundant in Europe where 128 species have been reported (L. Tewksbury, R. Casagrande and B. Blossey, unpublished manuscript). Invasion of *P. australis* is also associated with changes in sediment chemistry (Meyerson et al. 1999) which will have (yet unknown) impacts on other ecosystem processes.

Table 3. Ecosystem impact of *Phragmites australis*.

Pattern	References
High bullfrog abundance	Clarkson and Devos 1986
Host plant of *Ochlodes yuma, Poanes viator*	Tewksbury et al. 1999; Opler et al. 1995
Lower plant diversity	Meyerson et al. 1999
Different N-cycling pattern	Meyerson et al. 1999
Changes in porewater chemistry	Meyerson et al. 1999
Replacement of native vegetation	Marks et al. 1994; Carroll et al. 1984
Blackbirds prefer cattails over *P. australis*	Bernstein and McLean 1980
Muskrats prefer *Typha* over *P. australis*	Clark 1994

Lythrum salicaria

The purple loosestrife control program highlights a variety of attitudes and conflicts in invasive species management. The species, introduced to North America almost 200 years ago, slowly spread through the Northeast, became a valued ornamental, and was widely distributed by beekeepers and horticulturists (Thompson et al. 1987). *L. salicaria* now occurs in all lower 48 States of the US and it was declared a noxious weed in at least 15 states prohibiting sale and growth. The species is one of the ‘dirty dozen’ invasive species identified by The Nature Conservancy because of its detrimental impacts on wetland flora and fauna (Malecki et al. 1993). A number of generalist bird (Anderson 1995; Whitt et al. 1999) and insect (Hight 1990; Diehl et al. 1997) species were found on purple loosestrife, leopard frogs may breed in flooded *L. salicaria* (Gilbert et al. 1994), and white-tailed deer or muskrats occasionally feed on shoots and shoot tips (Table 4; Thompson et al. 1987). However, specialized marsh birds (black terns, least bittern, American bittern, long billed marsh wren), of special management concern to the US Fish and Wildlife Service because of declining populations, do not nest in purple loosestrife (Hickey 1997; Lor 1999). In addition, encroachment by *L. salicaria* is suspected to reduce the available habitat and recruitment of a number of duck species (Canvasback, Wood Duck, Bluewinged Teal), Canada goose and Sandhill Cranes (Thompson et al. 1987; Table 4). Furthermore, negative impacts are suspected for 2 rare plant species, the bog turtle, muskrat and mink as a result of loss of once favorable habitat to purple loosestrife (Coddington and Field 1978; Rawinski 1982; Thompson et al. 1987; Kiviat 1989).

Experiments confirm field observations (Thompson et al. 1987) of local extinction of cattails during competition with *L. salicaria*, regardless of initial densities (Weihe and Neely 1997; Weiher et al. 1996). A

Table 4. Ecosystem impacts of *Lythrum salicaria*.

Reason	Suspected	Documented	References
Reduction in bird and wildlife habitat			
Black tern (*Chlidonias niger*)	X	X	Hickey 1997
Marsh birds (bitterns, grebes, and rails)	X	X	Lor 1999
Canvasback (*Aythya valisineria*)	X		Thompson et al. 1987
Sandhill crane (*Grus canadensis*)	X		Thompson et al. 1987
Wood duck (*Aix sponsa*)	X		Thompson et al. 1987
Bluewinged Teal (*Ana discors*)	X		Thompson et al. 1987
Canada goose (*Branta canadensis*)	X		Thompson et al. 1987
Long billed marsh-wren (*Cistothorus palustris*)	X	X	Rawinski and Malecki 1984; Whitt et al. 1999
Muskrat (*Ondatra zibethicus*)	X		Thompson et al. 1987
Mink (*Mustela vison*) due to shortage of prey	X		Thompson et al. 1987
Bog turtle (*Clemmys muhlenbergi*)	X		Kiviat 1989
L. salicaria used by			
Insects		X	Hight 1990; Anderson 1995; Diehl et al. 1995; Kiviat 1989
Leopard frog		X	Gilbert et al. 1994
Birds		X	Anderson 1995; Whitt et al. 1999; Kiviat 1989
Mammals		X	Thompson et al. 1987; Kiviat 1989; Anderson 1995
Reduction in plant biodiversity	X	X	Weiher et al. 1996; Mal et al. 1997
Threat to *Scirpus longii*	X		Coddington and Field 1978
Threat to *Eleocharis parvula*	X		Rawinski 1982
Replacement of cattail (*Typha* spp.)	X	X	Weihe and Neely 1997; Mal et al. 1997; Weiher et al. 1996
Alteration of wetland function	X		
Decomposition processes		X	Emery and Perry 1996; Barlocher and Biddiscombe 1996; Grout et al. 1997
Sediment chemistry		X	Templer et al. 1998
Increased evapotranspiration		(X)	J. Yavitt (pers. comm.)
Reduction of flow in irrigation canals	X		Bureau of Reclamation personnel (pers. comm.)

possible mechanism for the success of *L. salicaria* is an increase in leaf area resulting in increased evapotranspiration rates in mixed stands during early stages of an invasion of a cattail stand, with the resulting drier conditions favoring purple loosestrife (J. Yavitt, pers. comm). Indications that such a scenario can be reversed comes from observations during the successful control of purple loosestrife by leaf beetles in a wetland in upstate New York. As *L. salicaria* was suppressed, the remaining cattails increased in height and density, the site became flooded again and muskrats returned (B. Blossey, pers. obs.). In addition, as a result of the replacement of cattails by purple loosestrife, changes in decomposition rates and sediment chemistry occur which may have important impacts to the invaded system or to wetlands downstream (Table 4).

The above review of evidence covering three major invasive species demonstrates that we are facing either lack of (conclusive) evidence or ambivalent results. However, the inability to control *L. salicaria* using mechanical, physical, or chemical means resulted in the initiation of a biocontrol program (the use of host specific natural enemies [usually insects or pathogens] from the native range of the target plant) in 1986. Control agents were introduced in 1992 and 1994, and several million insects were released in over 1200 wetlands in at least 33 states and in Canada (Malecki et al. 1993; Blossey et al. 1996; Blossey, unpublished data). Similarly, the inability to control *A. petiolata* or *P. australis* has resulted in the initiation of biological control programs in 1996 and the first field season searching for potential natural enemies successfully completed in Europe in 1998 (Hinz and Gerber 1998; Schwarzländer and Häfliger 1998). Although there is (currently) no opposition to the biocontrol program targeting *A. petiolata*, introductions of biological control agents for *P. australis* will, at least in part, depend upon a genetic comparison of North American and European genotypes (K. Saltonstall, in prep.; pers. comm.).

The (bio)control program targeting purple loosestrife has been severely criticized (Anderson 1995; Hager and McCoy 1998) claiming a (1) lack of evidence demonstrating the negative ecological impacts of this species and (2) its use by native species. A closer examination of these claims shows, however, that suspected impacts on bird and wildlife habitat, reductions in wetland plant diversity, and alteration of wetland functions (Thompson et al. 1997; Malecki et al. 1993) have since been documented in the field and in experiments (Table 4). Regardless of its merit, the criticisms of Anderson (1995) and Hager and McCoy (1998) illustrate that conflicts of interests are to be expected and conflict resolution has to be an integral component of non-indigenous species management. Conflict resolution, however, has to be based on factual, not anecdotal evidence and valuable data can be obtained from long-term monitoring programs. Nevertheless, it is useful to examine the types of objections raised in invasive species management since not all conflicts can be solved by the collection of evidence.

Conflicts in invasive species management

Objections to controlling non-indigenous species can be grouped into 5 categories, (1) economic, (2) ecological, (3) aesthetic, (4) ethical, and (5) risks associated with the development of biological weed control (see Table 5). Many invasive species were intentionally introduced, often for their aesthetic value and sale as garden ornamentals (Randall and Marinelli 1996). The Soil Conservation Service (now the Natural Resource Conservation Service, NRCS) has introduced and distributed species such as kudzu (*Pueria lobata*) to prevent soil erosion and multiflora rose (*Rosa multiflora*) as windbreaks. More esoteric uses involve reeds in musical instruments made from *Arundo donax*, antidepressant effects of *Hypericum perforatum*, or garlic mustard leaves as salad ingredient (Table 5). Although the latter uses may represent a true (albeit small) economic value, they usually are of minor importance and alternatives are available. More persistent have been claims by beekeepers to safeguard prolific honey producers such as saltcedar or purple loosestrife.

Other arguments involve the potential value of introduced plants for game species (Segelquist and Rogers 1975; Stafford and Dimmick 1979), native insects or birds (Anderson 1995; Whitt et al. 1999). The use of introduced plants by indigenous herbivores is well documented and may lead to serious agricultural problems (Dennill and Moran 1989) or actually contribute to the control of an invasive species. This is well illustrated by the biological control of multiflora rose (*Rosa multiflora*) through an as yet unidentified disease, rose rosette disease (RRD), transmitted through an eriophyid mite (Amrine and Stasny 1992). Multiflora rose was widely planted through much of the eastern and midwestern states. Mite and vector are native to the western states and began spreading eastward after the distribution of multiflora rose reached the Rocky Mountains – providing control of multiflora rose on

Table 5. Objections to invasive plant control.

Reason	Species	Reference
Economic		
Horticultural, Beekeepers	*Lythrum salicaria*	Thompson et al. 1987
Prevents soil erosion, Cow forage	*Pueria lobata*	Lynd and Ansman 1990
Medicinal (antidepressant)	*Hypericum perforatum*	Ernst et al. 1998
Musical instruments	*Arundo donax*	Kolesik et al. 1998
Salad, high vitamin C	*Alliaria petiolata*	Guil and Torija 1997
Sewage treatment	*Phragmites australis*	Dunbabin and Bowmer 1992
Stabilizing of dunes	*Ammophila arenaria*	Schwendiman 1977; MacDonald et al. 1989
Hunting (white winged dove), Beekeepers	*Tamarix* spp.	DeLoach 1990
Ecological		
Bird use	*Lythrum salicaria*	Anderson 1995
Use by native insects		Hager and McCoy 1998
Wildlife food	*Lonicera japonica*	Stafford and Dimmick 1979; Segelquist and Rogers 1975
Willow flycatcher nests in salt cedar	*Tamarix* spp.	Ellis et al. 1997;
High rodent diversity		Brown and Trosset 1989
Aesthetic	Ornamentals	Randall and Marinelli 1996
Ethical	No further introductions of non-indigenous species	Hager and McCoy 1998
Risks associated with biological control		
Attack of native species		Knutson and Coulson 1997; Louda et al. 1997
Evolution towards reduced specificity		McEvoy 1996

the way (Amrine and Stasny 1992). While this phenomenon may explain why only a minor fraction of species introduced and established actually become seriously invasive (Blossey and Nötzold 1995; Mack 1996) we are concerned with those that are not being controlled by natural enemies and continue to expand their range, as is the case with purple loosestrife. Hight (1990) found 94 different insect herbivore species (excluding nectar feeders) feeding on *L. salicaria* in North America. None of these species is able to control purple loosestrife and most can be found on a variety of plant hosts, i.e. they are generalists.

Although potential negative impacts of herbicides on applicators, ecosystem function, and on non-target species are widely recognized, traditional techniques (mechanical, physical, chemical) used to control invasive plants usually meet little resistance. Many people assume that local control efforts have localized impacts and can be discontinued if unwanted side effects occur. Control at one site does not affect populations of the target at other sites. Biological control is irreversible after control agents are established. The release at one location has local, regional, and potential continental implications when control agents spread beyond their initial release sites. Recognizing the possibility of potential conflicts of interest, procedures have been implemented for decades to ask for scientific input, scientific review and societal input and permission before any releases of control agents occur (see review in Knutson and Coulson 1997). An integral part of any petition to introduce biological control agents is a review of evidence for the negative impacts of the target weed. In the absence of data on ecosystem impacts of an invasive plant, the potential risks associated with the introduction of biocontrol agents may appear high compared to the 'no action' scenario. Long-term monitoring will allow a better assessment of benefits and risks of any action considered during this review process. However, until such evidence becomes available, we need to base our management decisions on other guiding principles.

Decision making in invasive plant control: how much evidence is enough?

Natural areas are managed for the preservation of native fauna and flora and natural processes and management practices should favor the long-term sustainability/health of these systems. Any test whether changes in species composition and ecosystem processes (Table 1) are a result of invasion by NIS should test the following null hypothesis: 'The invasion of a non-indigenous plant does not alter native species composition nor ecosystem processes'. Similar to statistical errors when probabilistic interpretations of data are made, the possibility for Type I and Type II errors exists when testing the null hypothesis. Type I, rejecting the null hypothesis when it is actually true, and Type II, retaining a null hypothesis when it is wrong (Zar 1984; Underwood 1997). In the case of invasive plants, a Type I error would occur if advocates of control of a certain plant species claim that there is an impact when in fact there is none. A Type II error would represent failure to detect an impact even when one has occurred. By convention, statistical analyses (Zar 1984) and much environmental work are designed to keep a Type I error small, with less concern over Type II errors (Underwood 1997). For the management of invasive plants, however, not detecting impacts is contradictory to the precautionary principle (Underwood 1997) and management goals and it is particularly dangerous to assume that there are no impacts when nobody has bothered to look.

The initiation of a national program to control purple loosestrife was justified stating:

> Although we need quantitative measurements of the effects of various stages of *L. salicaria* invasion on the structure, function, and productivity of North American wetland habitats, the replacement of a native wetland plant community by a monospecific stand of an exotic weed does not need a refined assessment to demonstrate that a local ecological disaster has occurred (Thompson et al. 1987).

Although this assumption, published in 1987, was strongly criticized by Anderson (1995) and Hager and McCoy (1998), several of the predicted impacts have now been confirmed (see Table 4). The predictions are firmly grounded in experimental and theoretical ecology since it has become increasingly clear that various ecosystem properties are strongly correlated with the functional characteristics of the dominant contributors to the biomass (applicable only to autotrophs, not to symbionts, herbivores or predators; reviewed by Grime 1998). Although observations cannot be a substitute for quantitative data collections, the examples from the purple loosestrife program demonstrate the validity and predictive power of long-term experience.

If ecosystem properties change with variable contributions of different plant species to the biomass, invading species should be of special concern. Although they may have none or minor impacts when rare (as long as they are transients, *sensu* Grime 1998) their importance and impact changes as their abundance and contribution to the biomass increases. Ecosystem properties, fauna and flora, and higher order interactions may be drastically altered. This is illustrated by the replacement of *Typha* dominated wetlands by purple loosestrife. A shift in species composition causes a shift in decomposition rates and alters phosphorus input rates and timing (Emery and Perry 1996; Barlocher and Biddiscombe 1996; Grout et al. 1997). Thus, management of those invasive species that are able to dominate communities may not need further evidence to justify control: the invasion and displacement of native vegetation **is** the ecological disaster. Priority for control should be given to the most damaging species but we also have to balance potential non-target effects by widespread application of herbicides or other management practices. For most of these species, developing successful biological control is the only hope to achieve landscape level control. The availability of convincing data on ecosystem impacts of different species will help during conflict resolution or risk assessment (see above), and it is also clear that the large numbers of invasive species require prioritization of control efforts.

Hiebert (1997) introduced a ranking system to prioritize invasive plant control in National Parks and natural areas based on (1) significance of impact of NIS and (2) feasibility to achieve management objectives. Although this system ranks species according to impact and ease of control, it is immediately evident that much of the needed information is anecdotal or missing for many if not most invasive plants. For example, information is requested on the effect of NIS in relation to disturbance, threat to rare and endangered species, invasion or modification of native communities, and level of impact in other natural areas (Hiebert 1997). Ranking systems help justify and prioritize control efforts but as long as we are left with well educated guesses, at best, about many of the potential impacts of NIS, currently they only provide the illusion of 'objective' fact based assessments. Without additional data from long-term monitoring we are largely left with 'expert opinions', and can only hope that by involving natural resource managers and scientists from different agencies and disciplines in decision making processes we can avoid most conflicts of interest.

Opportunities for research: development of long-term monitoring programs

Understanding how ecosystem function is altered by changes in abundance of individual species in itself is an extremely fruitful and interesting field of ecology. Not only will it help us in managing invading species, a demonstration of widespread ecosystem-level consequences would constitute an explicit demonstration that species make a difference on the ecosystem level, and could become a milestone in integrating often disparate approaches of population biology and ecosystem-level ecology (Ramakrishnan and Vitousek 1989). Ideally, such investigations have to go beyond 'simple' impacts on plant communities, must involve cross-disciplinary teams of scientists, and should incorporate many different taxa and different trophic levels. In guiding our investigations, it is probably safe to assume that all levels and processes within an ecosystem can be affected by the invasion of a plant species. This includes net primary production and its components, microorganisms, plant diversity, invertebrate communities, mammals, reptiles, amphibians, different trophic levels, predators, diseases, symbionts, direct and indirect effects and higher order interactions. At present, our knowledge, particularly about higher order interactions and indirect effects is extremely limited. Of particular value will be standardized, well replicated and sophisticated monitoring studies that hold up to rigorous statistical analysis and interpretation. Developing such programs for a number of different species may reveal important generalization about the invasion process.

It will be particularly interesting to investigate whether we have certain threshold levels of abundance where any further increase of the invasive plant would result in unacceptable disruptions or whether changes are spread out along a continuum. Such information could be of particular use in guiding management decisions. We may even find 'indicator species' or groups that are most likely affected by invasions and could function as early warning signals. Most likely these will not be obvious reductions in plant species but more subtle effects at higher trophic levels and involve specialized species.

All investigations have to be planned long-term to be able to document temporal changes that may take decades. The power of such studies has been demonstrated for various organisms such as migratory songbirds (Berthold et al. 1993), pollinators (Frankie et al. 1998) or vegetation dynamics (Dunnett et al. 1998). The development of control programs, particularly biocontrol, offers some exciting opportunities to follow changes in plant communities and herbivores through time. Investigations should begin before control agents are released to collect baseline data. The development of standardized monitoring protocols will allow building partnerships and cooperation across disciplines and agencies and providing for the necessary replication and a more powerful analysis. Following the changes of floral and faunal communities through time will allow us to assess the 'quality' of the replacement communities, and may potentially result in management recommendations for the suppression of target plants to specific acceptable abundance levels. We usually do not know how communities or ecosystem processes respond as invasive plants spread through an area. Most often evidence for changes is reported by comparing invaded and uninvaded habitats. More powerful information, however, could come from studies that investigate species composition, interactions and ecosystem processes as a habitat is invaded.

Many of the above outlined study scenarios will become more powerful by increasing the cooperation among different scientific disciplines. Logistical difficulties (need for coordination of long-term, multiple investigator projects, resistance of funding agencies to sponsor long-term, non-experimental work) makes careful planning, funding and partnerships essential. The information we would be able to generate, however, would seem to make this one of the most exciting emerging fields in ecology. It will offer an immediate application in invasive species management and make important contributions to maintain the integrity of National Parks and other natural areas set aside for the protection of native fauna and flora.

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