PLANT INVADERS, GLOBAL CHANGE AND LANDSCAPE RESTORATION

D.A. Pyke[1] and S.T. Knick[2]

[1]U.S. Geological Survey, Forest & Rangeland Ecosystem Science Center, Corvallis Research Group, 3200 SW Jefferson Way, Corvallis, Oregon 97331, U.S.A. E-mail: david_a_pyke@usgs.gov
[2]U.S. Geological Survey, Forest & Rangeland Ecosystem Science Center, Snake River Field Station, 970 Lusk Street, Boise, Idaho 83706. U.S.A. E-mail: steve_knick@usgs.gov

1. INTRODUCTION

Increases in human populations, advances in technology and transportation, and shifts toward global economies have created human activities that have transformed land uses, modified the earth's biogeochemistry and have influenced the distribution of biological resources on our planet (Vitousek *et al.* 1997b). Historic biogeographic barriers that formerly restricted the spread of organisms into new landscapes have been lowered, thus creating an opportunity for species to colonize and in some cases dominate new environments.

The spatial spread and influence of a species in an environment is a consequence of a combination of intrinsic and extrinsic factors that govern the species' population dynamics. Intrinsic factors include the dispersal, growth, survival and reproductive constraints dictated by the species' physiological and morphological capabilities whereas extrinsic factors include the spatial and temporal availability of suitable habitat for survival, growth and reproduction and of suitable corridors and vectors for dispersal. Spatial spread may vary from species extending their current range of distribution into new, adjacent ecosystems, to species being transported to new continents with similar environments. Those species that invade new ecosystems and dominate otherwise intact pre-existing native ecosystems are commonly known as invasive species (Invasive Species Specialist Group http://www.issg.org/ 11 December 2002).

Invasive species may respond to human-induced environmental changes or, in turn, they may initiate environmental changes through their dominance on the landscape. In addition, the spatial and temporal extent of that change may be mediated or expressed at scales ranging from local to global (Shugart 1998, Mooney and Hobbs 2000). Effective programs must recognize multiple spatial and temporal scales in their management for invasive species because bottom-up as well as top-down processes influence significantly the mechanisms of spread and dominance.

In this paper, we will provide a brief overview of the major human-induced agents of environmental change and their potential consequences on invasive species. We will also examine invasive species as agents of environmental change. For each agent of environmental change, we will discuss the potential spatial and temporal extent of the agent's influence especially as it relates to rangeland management. Using *Bromus tectorum* (cheatgrass) as an example invasive species, we will examine spatial management options for restoration of native plants in the Great Basin of the U.S.A.

Keywords: Invasive Plants, Stressors, Spatial Restoration

2. HUMAN-INDUCED AGENTS OF GLOBAL CHANGE

The global human population is projected to reach 9.3 billion people by 2050; an increase of over 50% of the 1998 world level. Most of the world's population growth is projected to occur on continents dominated by less developed countries (Asia, Latin America and the Caribbean, Near East and Africa) (McDevitt 1999). Generally, less developed countries maintain greater reliance on subsistence agriculture leading to greater overuse and degradation of natural resources than more developed countries. More developed countries, however, contribute higher amounts of atmospheric emissions than less developed countries. Human agents of global change are commonly grouped into three major categories, land and disturbance transformations, biogeochemical modifications, and biotic additions and losses (Vitousek *et al.* 1997b, Huenneke 1997). The importance of specific alterations to rangeland ecosystems will depend on the size and extent of changes and on the physical and biological resilience of the ecosystem.

2.1 Land and Disturbance Transformations

Global land transformations are estimated to affect between 39 and 50 % of the earth's surface (Vitousek *et al.* 1997b), but land transformations are difficult to accurately measure in all ecosystems (Foody 2002). They are typically assessed using remote sensing technology (Meyer & Turner 1994), thus major alterations in the configuration of ecosystems, such as deforestation and urbanization, are detected readily. Estimates of changes in arid ecosystems are more difficult to obtain because of lack of vegetative biomass relative to background influences (Knick and Rotenberry 1997) and may require additional sensors or images from specific seasons to detect changes (Lambin & Ehrlich 1997). Changes in composition of arid ecosystems, such as the conversion from perennial grasses to cheatgrass in the shrubland understory, also are difficult to detect, but may be as significant as major transformations in dictating future trajectory of vegetation in the landscape (Foody 2001). As a result, the total spatial extent of change as well as the impacts of transformations and degradations on rangelands may be vastly underestimated.

High intensity disturbances that transform landscapes often occur at local spatial scales. For example, land conversions from rangeland to human habitations, such as urban, suburban or village expansions, are concentrated near the fringe of human population centers (Hart 1991, McClaren, Romm & Bartolome 1985). In contrast, lower intensity disturbance that results in degradation often is more diffusely distributed throughout the landscape. Inappropriate livestock management that results in overuse of the resources is a common form of rangeland degradation that can lead to land transformations (Kauffman & Pyke 2001). Riparian ecosystems are sensitive to livestock trampling and herbage removal. In many arid and semiarid regions, the riparian zone is the major source of tree structure for the landscape. Physical damage of plants and soil erosion leading to stream entrenchment and lower water tables may shrink the width of the zone dominated by riparian plants (Kauffman & Krueger 1984). Approximately 84 % of upland rangelands are at least moderately desertified (productivity reduction ≥ 25 %) (UNEP 1990). Desertification conversions may spatially reflect the land ownership, political boundaries, as well as soil and climate regions. Therefore, the spatial scale of upland degradation is likely to range from local to regional levels.

Invasive plants may directly benefit from the land transformations and degradations. Transformations often result from large land disturbances. Disturbance extent, patch size, fragmentation distribution, and dispersal ability interact to influence invasion spread. With (2002) provided modeling proof for the existence of thresholds of disturbance extent across a landscape. Invasive organisms have equal probability of spread regardless of the distribution of the disturbances when disturbance extent exceeds the threshold area. Below this threshold, the proportion of the landscape disturbed, the distribution of the disturbances and the dispersal ability interact to affect the probability of spread for invasive organisms. Generally, aggregated distributions of disturbances result in higher probabilities of spread for the invasive organism than those found for random distributions of disturbances, but as dispersal ability and disturbance extent increases this trend can reverse. Restoration or rehabilitation planners might be able to use the same principles to encourage desired native plant spread while establishing barriers to reduce spread of invasive species (Whisenant 1999).

Transformations often entail increased proliferation of human transportation corridors, which serve as immigration pathways for invasive plants. For example, railroad stations and river ports were common locations of early collections of cheatgrass (Mack 1981). Recent studies indicate that vehicles are common dispersal vectors for invasive plants (Lonsdale & Lane 1994) and that former roads may provide a corridor for establishment and growth of some invasive species (Silveri, Dunwiddie & Michaels 2002). Therefore, invasive plant management may require cleaning vehicles or restricting their travel routes to reduce the spread of invasive species. It may also dictate control treatments along roads or restoration of abandoned roads to reduce the spread of invasive plants.

Alterations of natural disturbance regimes may produce opportunities for plant invasions to occur (Hobbs & Huenneke 1992) which may lead to land transformations. Disturbances include grazing, fire, and floods and incorporate not only the introductions of these disturbances, but also the elimination of them. Changing disturbance frequency or intensity may cause ecosystem modifications. Intense grazing has been implicated in the encroachment of shrubs on grasslands or trees on shrublands (Burrows *et al.* 1990, Miller, Svejcar & West 1994, Archer 1994), but in many areas this phenomena is also enhanced by the reduction in fire frequency (Madany & West 1983, Miller *et al.* 1994). Invasive plants may contribute to modifications in the natural fire cycles providing a positive feedback that enhances their spread (D'Antonio & Vitousek 1992).

2.2 Biogeochemical Modifications

Life depends on a balance of biogeochemical cycles, however, human alternations of these cycles may result in unintended consequences to natural communities. Some chemical elements are byproducts of industry, transportation, or energy production while others are synthetically produced and intentionally added to the available pool within natural cycles. As with land transformations, the spatial scale at which elements influence cycles may range from local to global. For plants, the essential resources for establishment and growth are CO_2, water, light, the macronutrients (N, P, K, S, Mg and Ca) and the micronutrients (Fe, Mn, Zn, Cu, Mo, B and Cl). Those that commonly limit growth and influence interactions among species are N, P and water and to a lesser degree, yet possibly more important in the future is CO_2 (Tilman & Lehman 2001).

Of the many chemicals released through human enterprises, CO_2 provides the best evidence for atmospheric increases of gases that may trap energy and warm the earth (Giorgi *et al.* 1998, Keeling, Chin & Whorf 1996). The primary source for CO_2 increase is combustion of fossil fuels. Although the main sources of this CO_2 are concentrated in more developed countries in the northern hemisphere (Andres *et al.* 1996 cited in Vitousek *et al.* 1997b) the increases are circumpolar (Tucker *et al.* 1986).

Plant responses to increases in CO_2 depend mainly on their photosynthetic pathway. Cool season (C_3 pathway) plants should have the advantage over warm season (C_4 pathway) and CAM plants because cool season plants are not carbon saturated. Also, increases in carbon should improve water-use efficiency of cool season plants. Although some warm season and CAM plants do show increased growth or physiologic responses with elevated CO_2, cool season plants tend to show greater responses (Poorter 1993, Poorter, Roumet & Campbell 1996).

Invasive species generally exhibit positive growth responses to elevated CO_2, however, most studies were conducted in competition-free experiments (Dukes 2000). Arid ecosystems are predicted to be the most responsive ecosystems to CO_2 increases because of water limitations. It follows that invasive species will be additionally favored in arid and semiarid ecosystems. Invasive cool season shrubs are expanding in the Chihuahua desert of North America. Recent studies on the physiology and demographics of some of these trees and shrubs lend strong evidence that rising CO_2 contributes to woody plant encroachment in this ecosystem (Polley *et al.* 1996, 1999, 2002). In cool deserts with winter-dominated precipitation (Mojave and Great Basin), moisture availability became the critical limiting resource to determine if positive growth responses were detected among species. During drought years the growth difference between ambient and elevated-CO_2 treatments were not detected, but during high precipitation years production exceeded predicted increases (Smith *et al.* 2000). Invasive annual grasses show both physiologic and demographic enhancements when grown under elevated CO_2 relative to ambient conditions (Huxman *et al.* 1998, Huxman & Smith 2001) and they experience greater growth than natives (Smith, Strain & Sharkey 1987, Smith *et al.* 2000).

Human-induced emissions of other gaseous elements, such as nitrogen and sulfur, also have increased. Spatial impacts of these chemicals are more regional than CO_2, because they are more reactive than CO_2 in the atmosphere and tend to associate with water vapor or fall to earth with precipitation. Atmospheric N contributes to the greenhouse effect via nitrous oxide, to fluxes in reactive forms such as ammonium, and to acid rain (Vitousek *et al.* 1997a). Nitrogen enrichment in ecosystems with N limitations will result in higher production. For nitrogen, humans have also intentionally fixed N_2 for fertilizers. Galloway *et al.* (1995) estimate that the human-induced N-fixation is 60% of the total.

Since N limitation is common in most arid and semiarid ecosystems, increases in plant available N should result in increases in C fixation. The form of the carbon pool (active, slow or passive) and its position within the soil profile may be important factors for ecosystem processes (Gill *et al.* 1999). Shifts in plant composition and reductions of species richness often result in communities dominated by highly competitive nitrophilous species with rapid growth rates from increased N availability (Tilman & Lehman 2001). Soil microbial communities may also shift as shifts occur in the nutrient quality of the vegetation that they feed. In southern California U.S.A., the historical increases in N deposition have lead to shifts in the mycorrhizal fungal communities (Padgett *et al.* 1999, Egerton-Warburton & Allen 2000, Egerton-Warburton *et al.* 2001).

Fast growing ruderal species respond more rapidly to increases in resources than slower-growing later successional species. Species adapted to low nutrient environments tend to have lower maximal growth rates and respond less to nutrient increases than those that thrive on fertile soils or ruderal species (Grime 1977, Chapin, Vitousek & Van Cleve 1986). Thus, invasive annual grasses tend to respond more positively to increased nutrients than native species in a variety of arid, low nitrogen rangeland ecosystems (Gutiérrez *et al.* 1988, Hobbs *et al.* 1988, Huenneke *et al.* 1990, Gutiérrez, Aguillera & Armesto 1992, Brooks 1998, Padgett & Allen 1999, Young *et al.* 1999). In fertile ecosystems, annual ruderal species dominate low nutrient environments while late successional species dominated high nutrient environments (Tilman 1987).

Greenhouse gas emissions are predicted to increase temperatures and adjust global circulation patterns, but these increases will not occur equally throughout the Earth (Aber *et al.* 2001). The Intergovernmental Panel on Climate Change recommended risk-assessments that evaluate the impacts and the adaptive management options under various climate change scenarios (Parry and Carter 1998). Sutherst (2000) proposes that the impact of climate change on invasive species can be examined by assessing invaders, competitors and their environments under various climate scenarios and management options. The invasion process should be examined spatially for the source populations, pathways of spread and at new colonization sites. Obvious changes such as shifts in dominance between plants of differing photosynthetic pathways or life forms (woody plants vs. herbaceous) are predicted using vegetation-ecosystem process-climate models such as MC1 (Daly *et al.* 2000). Empirical studies of these concepts are difficult to construct or are confounded because we lack the ability to control environmental parameters. Free Air CO_2 Enrichment (FACE) (e.g., Jordan *et al.* 1995) or precipitation manipulation studies (Weltzin & McPherson 2003) are the best sources for *in* *situ* testing of climate change impacts on ecosystem species and processes. For example, Svejcar *et al.* (2003) have shown that shifts from winter to spring dominated precipitation in the northern Great Basin will create more bare ground as native annual herbs are eliminated and as biomass of the surviving plants is reduced. These changes might lead to plant invasions or to soil erosion as greater areas become susceptible to rainfall.

2.3 Biotic Additions and Losses

Species extinction and colonization are natural processes within regional flora and fauna, but with global transport and travel becoming more commonplace, the spread of invasive species is anticipated to grow. Economic and ecological consequences of these invasions are staggering. Pimentel *et al.* (2001) estimated that over 120,000 species (plants and animals) have been introduced into new regions of six countries (Brazil, U.S.A., United Kingdom, South Africa, Australia, and India). Many of these species are important as sources of food, but the costs from damages caused by introduced species are roughly estimated at US$ 314 billion per year. Barbault & Sastrapradja (1995) estimated that 8% of the global plant species in the early 1990's were threatened with extinction. Reductions in plant diversity create voids that may enhance the spread of invasive species into ecosystems. In some cases, invasive species are a leading threat that might drive other species to extinction. A recent report in the U.S.A. estimated that 16% of the 250 plant species listed as threatened or endangered with extinction in 1991 had invasive species as a contributing cause while 6% had invasive species listed as the primary reason for their listing (U.S. Congress, Office of Technology Assessment 1995).

Large-scale modifications of biogeochemical processes due to effects of invasive plants may be rare (Vitousek 1990). The clearest examples come from invasions of N-fixing species into relatively infertile systems (see citations in Mack, D'Antonio and Ley 2001). Vitousek (1990) also reports examples of invasive plants impacting water cycles, shifting the soil levels of available nutrients, modifying fertility via salt or through low-quality acid litter accumulations. The interactions among invasive plants and fire may also impact nutrient cycles (Mack *et al.* 2001).

For many invasive species, fire and the invasive ability of the species are inter-related. D'Antonio (2000) provides an excellent review on how fire can enhance the density or distribution of invasive species, while the presence of the invasive species can enhance the spatial or temporal characteristics of fires within an ecosystem. Fire appears to promote invasive species in a number of arid and semiarid ecosystems. Mesic ecosystems have fewer examples and those with associated invaders are typically in seasonally dry woodlands. In most documented cases, the invasive plant enhances either the frequency or intensity of the fire by providing either additional fuel for fires or by providing a continuous source of fuel. Within the U.S.A. Great Basin, the continual expansion of cheatgrass (*Bromus tectorum*) throughout this region has lead to more frequent fires (Whisenant 1990, Figure 1) and may contribute to progressively larger fires (Figure 2).

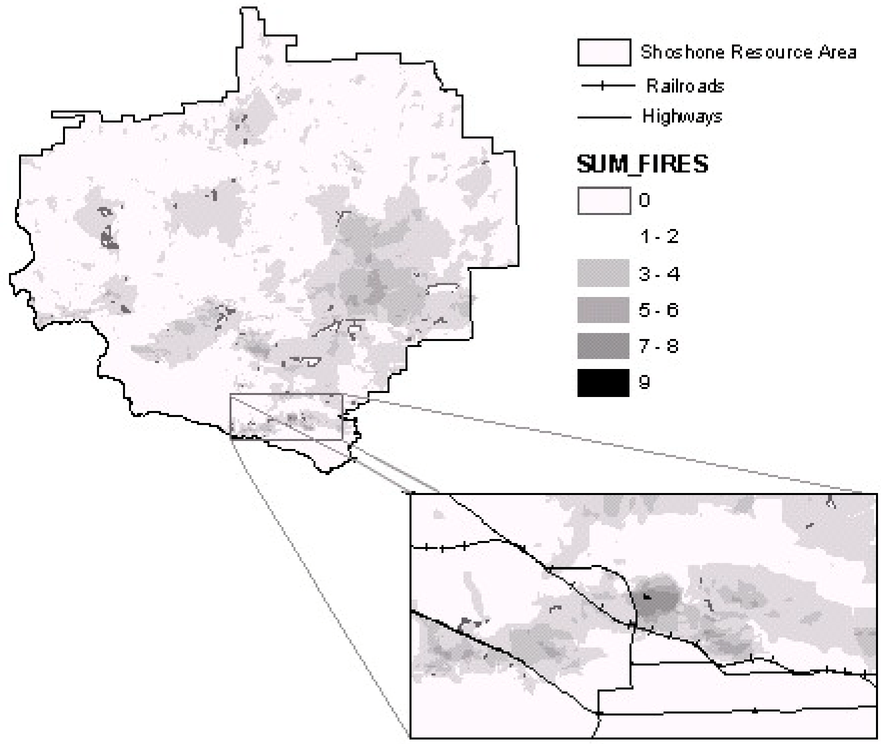

Figure 1. Fire frequency (shades of grey) between 1941 and 1993 on the Shoshone Field Office, Bureau of Land Management, Idaho, U.S.A (717,000 ha). Expanded area is 9 km north of the city of Twin Falls (DA Pyke & SJ Popovich unpublished data).

D'Antonio (2000) cites some cases of invasive plants reducing the frequency of fires. Most of these species retain moisture or are photosynthetically active during part or all of the fire season. The concept of using plants as 'fire breaks' has been attempted with mostly non-native species (Pellant 1990), but the success of this technique in slowing or stopping fires has never been documented in a study.

3. RESTORATION IN A SPATIAL AND TEMPORAL CONTEXT

Within the Great Basin region of the western U.S.A., as much as 50% of the native sagebrush steppe has been converted to annual grasslands through the invasion and dominance of cheatgrass (West 2000). Nearly 99% of the basin big sagebrush (*Artemisia tridentata* ssp. *tridentata*) grassland communities in the Snake River Plains, Idaho have been converted to croplands because of their deep and well-drained soils (Noss, LaRoe & Scott 1995); these ecosystems are among the most endangered ecosystems in North America (Noss *et al.* 1995). The widespread conversion of the major communities within this expansive ecosystem (≈ 63 million ha) has ramifications on wildlife species within the region. Greater Sage-grouse (*Centrocercus urophasianus*) depend on combinations of grasses, forbs, shrubs and their associated insects in the sagebrush ecosystem for the maintenance of viable populations. Greater Sage-grouse currently are being considered for listing as a Threatened or Endangered Species.

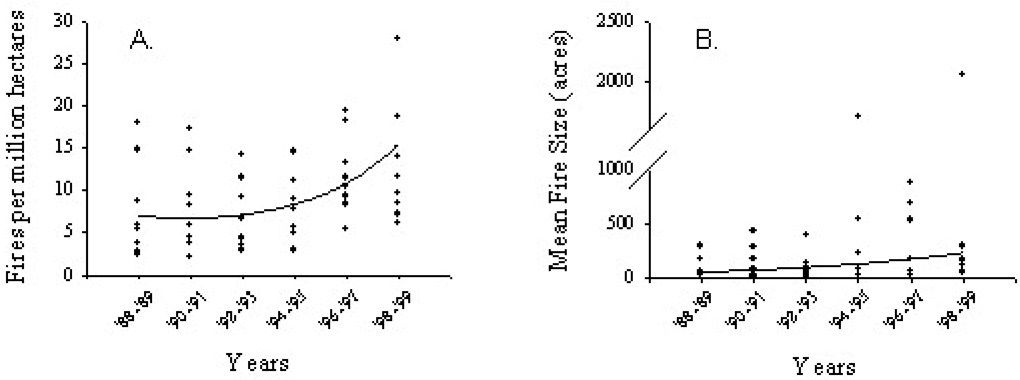

Figure 2. Number of fires per land area and mean fire size on eight Bureau of Land Management field offices in the Great Basin of Oregon, Idaho, Utah, and Nevada, U.S.A. 1999 (D. A. Pyke and T.O. McArthur, unpublished data)

The future of sagebrush ecosystems will be determined primarily by government policies for land use because federal and state agencies manage over half of the total land area covered by sagebrush (Knick, personal observation). Very little (<3%) of the sagebrush area is protected from a multitude of potential uses including livestock and feral horse grazing, mining, off-road vehicle use, and recreation (Scott et al. 2001). Although government policies cannot regulate the conversion of sagebrush ecosystems to croplands, management agencies can institute policies for restoring lands dominated by invasive species. The Bureau of Land Management has initiated the Great Basin Restoration Initiative (http://www.fire.blm.gov/gbri/ 11 January 2003) to halt the cheatgrass-wildfire cycle by attempting to control cheatgrass while proactively restoring native plants. Simultaneously, a consortium of scientists and land managers are working together to investigate new plant accessions for restoration, new techniques for controlling invasive plants and changes in ecosystem processes that may occur when invasive annual grasses dominate former sagebrush grasslands (e.g., Pyke & Pellant 2003).

3.1 Prioritizing Landscapes for Restoration

The extent of the cheatgrass dominance in the Great Basin and the limited funds available dictate that land managers prioritize when and where they conduct restoration projects. Restoration objectives may be as complex as restoring habitats for a suite of threatened and endangered species or as simple as rehabilitation of a burned area to prevent further spread of invasive species within an ecosystem. Regardless, short- and long-term objectives need to consider both the spatial and temporal context in which restoration and rehabilitation efforts take place. At the individual site level, managers should consider the probability of natural disturbances (e.g., fires, floods, etc), climatic cycles (e.g., those driven by oscillations in ocean currents), or locations of invasive plants when prioritizing when and where to attempt treatments. At the larger scale, the prioritization of treatment sites over time must include landscape variables such as fragmentation and connectivity to existing desirable habitats, while also considering these variables relative to the invasive species. The challenge is to integrate efforts so the checkerboard of treatments will form a functioning landscape (Whisenant 1999).

Achieving the objective requires knowledge of a desired future condition, but included in this condition is the range of natural variability. Natural variability is the ecological conditions and spatial variation in those conditions that are relatively unaffected by people within a period and geographic area appropriate to the expressed temporal and spatial context (Landres, Morgan & Swanson 1999). This deviates from the concept of managing for a single condition or community that is represented by an ideal successional end-point (e.g., late seral or historical climax) community or by some historic composition. In a landscape context, the natural variability is better represented by a distribution of successional stages that might be found within the reference state. The reference state, in state and transition model terminology, would represent this variety of communities that exist under natural disturbances severities and frequencies.

Federal lands in the U.S.A. are managed for multiple resources and uses. Therefore, managers may need to evaluate multiple treatment combinations to optimize the benefits for a suite of important resources. For example, habitat requirements for multiple wildlife species might be optimized through modeling approaches that consider each species' habitat form and function at the appropriate scale needed for that species. Wildlife often require a mix of vegetation communities within landscapes larger than individual restoration sites.

Location of treatments can be identified and prioritized once objectives are defined and the geographic boundaries are identified. Given the economic constraints that limit the number of projects in any year, a process of prioritization can be used to identify those locations having the highest probability of success. Predictor variables, such as topographic positions or soil units, are based on requirements of individual plant species. For example, precipitation zones between 22-28 cm are most favorable for Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*). Additional predictor variables may refine the priorities. These might include global change agents that impact invasive species as well as habitat requirements for wildlife, reductions in fragmentation of desire communities, and limitations dictated by current management requirements. Predictor variables are combined using basic functions or statistical models within a Geographical Information Systems (GIS) to delineate potential treatment areas most likely to have a combination of variables similar to an optimum set of conditions. Ultimately, maximum sizes and number of treatment units within a growing season are constrained by the annual available labor and money.

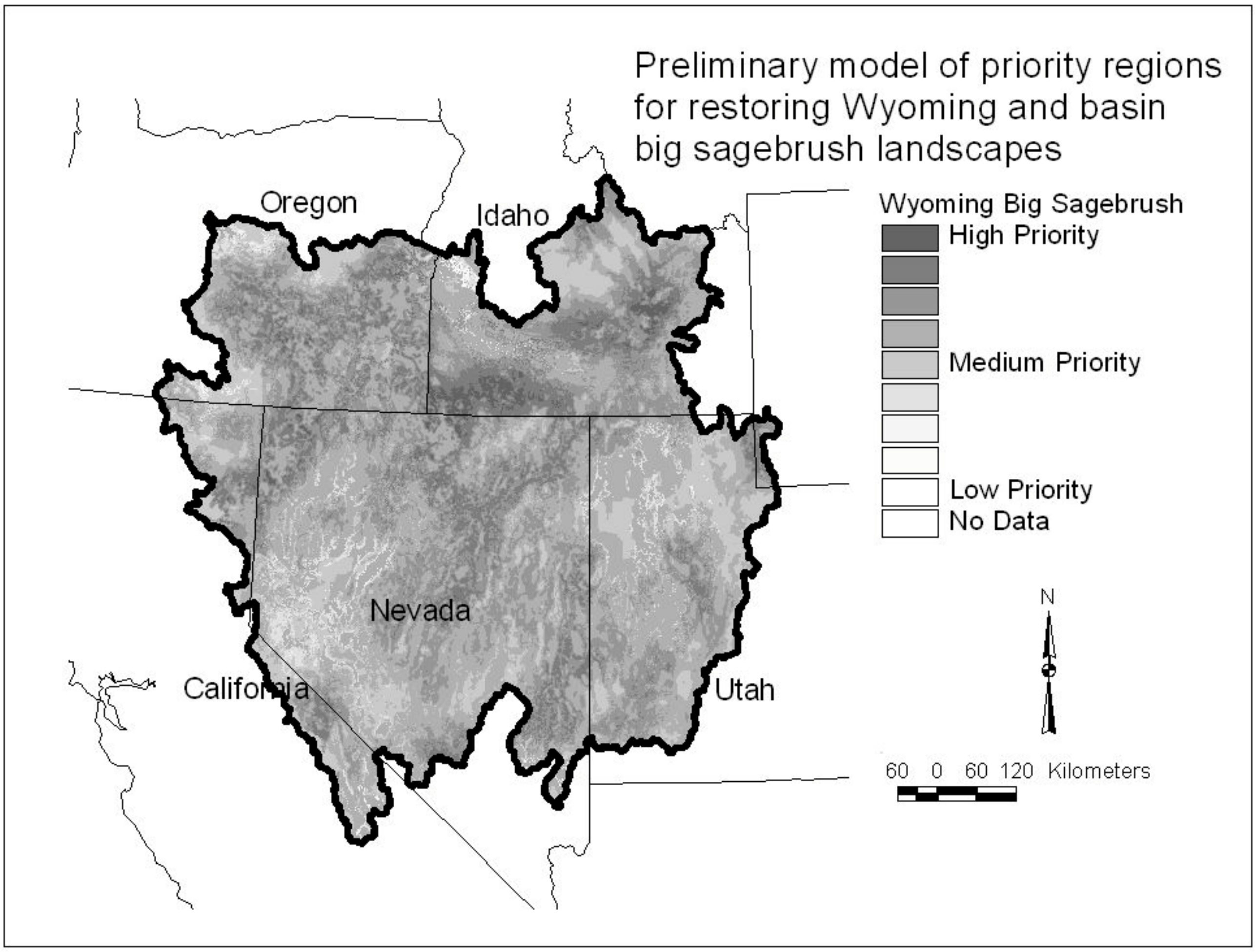

Figure 3. Preliminary model of priority regions within the Great Basin for restoring Wyoming and basin big sagebrush habitats.

This strategy to prioritize sites within a large region based on spatial modeling of landscape variables was recently demonstrated in a model for prioritization of restoration sites in the Great Basin region of the western United States (Knick presentation June 2002 to Sagebrush Restoration Conference, Elko Nevada, U.S.A.). The Great Basin contains >30 x 10^6 ha of which approximately 9 x 10^6 ha is in sagebrush habitat. The Great Basin has a long history of improper grazing practices, droughts, and plant invasions, especially cheatgrass, which have lead to widespread conversions of native shrub grasslands to near monocultures of cheatgrass (Young and Sparks 1985, Yensen 1981). Given the large geographic size, spatial modeling was used to (1) identify areas most likely to have the set of environmental and landscape conditions necessary for restoring Wyoming big sagebrush communities and (2) prioritize the Great Basin based on relative probability of restoration success. A preliminary set of 4 variables for which we had complete coverage of the entire region was used to develop the model. Variables included (1) low risk of cheatgrass invasion, (2) precipitation in the 22-28 cm range, (3) large scale dominance of sagebrush in the landscape, but (4) small-scale fragmentation within which projects could be located to benefit from available seed sources and increase the connectivity in the landscape. We used the Mahalanobis D^2, or generalized squared distance, function as a similarity index to determine the probability of the variables each cell to be similar to an optimum set of conditions for each cell in the GIS (Knick and Rotenberry 1998). The mapped distribution of the cell values then provided a regional model for prioritizing the entire region and identifying general areas in which restoration success was most likely (Figure 3).

This planning process based on spatial modeling of variables important for restoration success will provide managers with a tool for prioritizing sites. The process also permits managers to identify potential sites for rehabilitation in a proactive approach should wildfires occur within prioritized areas. Wildfire rehabilitation on these sites to control spread of invasive plants might allow another source of federal money to become available for shrubland recovery. By developing models integrating spatial criteria to select optimal conditions, managers can budget limited annual resources over a multi-year planning framework to address relative priorities for restoration locations within the region.

4. CONCLUSIONS

In many instances, invasive species and global changes are interrelated with invasive species responding to human-induced environmental changes while also providing mechanisms for ecosystem changes. Thus, positive feedbacks often occur between invasive species and global changes. Depending on the agent of global change, the spatial and temporal scale in which the agent is expressed may vary from local to global and from sporadic to constant. Land managers are faced with the difficult task of restoring landscapes and simultaneously preventing and controlling the spread of invasive species even though factors influencing the invasion process may be spatially or temporally beyond their control. However, when managers plan prevention or recovery treatments, success should improve if they consider spatial and temporal factors in their plans. Limited resources will dictate that priorities be set for treatment locations and times. The invasive species action plan for the U.S.A. calls for active prevention, detection, control and management of invasive species and restoration of ecosystems on those locations where invasive species have eliminated the native organisms (National Invasive Species Council 2001). Landscape principles should be used in restoration and control of invasive species to recreate the form and function of the original landscape. This is accomplished by integrating individual projects into interacting components of a larger mosaic. Local site-specific (bottom-up) efforts must be done in the context of the large-scale landscape (top-down) processes. Conversely, restoration success within the entire landscape can be improved by incorporating large-scale spatial and full range of temporal processes of ecosystem dynamics in local restoration plans. Understanding the effect of landscape process on both degradation and recovery processes is critical if managers hope to meet the goals for combating invasive species and restore landscapes throughout the world.

5. REFERENCES

Aber J, Neilson RP, Mcnulty S, Lenihan JM, Bachelet D & RJ Drapek 2001. Forest processes and global environmental change: predicting the effects of individual and multiple stressors. BioScience 51:735-750.

Archer S 1994. Woody plant encroachment into southwestern grasslands and savannas: rates, patterns and proximate causes. Pages 13-68 IN: Vavra M, Laycock W & Pieper R (eds) Ecological implications of livestock herbivory in the west. Denver, Colorado, U.S.A.

Barbault R & Sastrapradja SD 1995. Generation, maintenance and loss of biodiversity. Pages 193-274 IN: Heywood VH & Watson RT (eds) Global biodiversity assessment. Cambridge University Press, New York New York U.S.A.

Brooks ML 1998. Ecology of a biological invasion: alien annual plants in the Mojave Desert. Ph.D. Dissertation. University of California, Riverside California, U.S.A.

Burrows WH, Carter JO, Scanlan JC & Anderson ER 1990. Management of savannas for livestock production in north-east Australia: contrasts across the tree-grass continuum. Journal of Biogeography 17:503-512.

Chapin FS, Vitousek PM & Van Cleve K 1986. The nature of nutrient limitation in plant communities. American Naturalist 127:48-58.

Daly C, Bachelet D, Lenihan JM, Neilson RP, Parton W & Ojima D 2000. Dynamic simulation of tree-grass interactions for global change studies. Ecological Applications 10:449-469.

D'Antonio CM 2000. Fire, plant invasions, and global changes. Pages 65-94 IN: Mooney HA & Hobbs RJ (eds) Invasive species in a changing world. Island Press, Washington DC, U.S.A.

D'Antonio CM & Vitousek PM 1992. Biological invasions by exotic grasses, the grass/fire cycle, and global change. Annual Review of Ecology and Systematics 23:63-87.

Dukes JS 2000. Will the increasing atmospheric CO_2 concentration affect the success of invasive species. Pages 95-113 IN: Mooney HA & RJ Hobbs (eds) Invasive species in a changing world. Island Press, Washington DC U.S.A.

Egerton-Warburton LM & Allen EB 2000. Shifts in arbuscular mycorrhizal communities along an anthropogenic nitrogen deposition gradient. Ecological Applications 10:484-497.

Egerton-Warburton LM, Graham RC, Allen EB & Allen MF 2001. Reconstruction of the historical changes in mycorrhizal fungal communities under anthropogenic nitrogen deposition. Proceedings of the Royal Society of London Biology 268:2779-2484.

Foody GM 2001. Monitoring the magnitude of land-cover change around the southern limits of the Sahara. Photogrametric Engineering and Remote Sensing 67:841-847.

Foody GM 2002. Status of land cover classification accuracy assessment. Remote Sensing of Environment 80:185-201.

Galloway JN, Schlesinger WH, Levy H II, Michaels A & Schnoor JL 1995. Nitrogen fixation: anthropogenic enhancement-environmental response. Global Biogeochemical Cycles 9:235-252.

Gill R, Burke IC, Milchunas DG & Lauenroth WK 1999. Relationship between root biomass and soil organic matter pools in the shortgrass steppe of eastern Colorado. Ecosystems 2:226-236.

Giorgi R, Meehl GA, Kattenberg A, Grassl H, Mitchell JFB, Stouffer RJ, Tokioka T, Weaver AJ & Wigley TML 1998. Simulation of regional climate change with global coupled climate models and regional modeling techniques. Pages 427-437 In: Watson RT, Zinyowera MC & Moss RH (eds) The regional impacts of climate change: an assessment of vulnerability. Cambridge University Press, Cambridge U.K.

Grime JP 1977. Evidence for the existence of three primary strategies in plants and its relevance to ecological and evolutionary theory. American Naturalist 111:1169-1194.

Gutiérrez JR, Silva OA, Mipagani DA, Weems D & Whitford WG 1988. Effects of different patterns of supplemental water and nitrogen fertilization on productivity and composition of Chihuahuan Desert annual plants. American Midland Naturalist 119:336-343.

Gutiérrez JR, Aquillera LE & Armesto J 1992. The effects of water and macronutrient addition on aboveground biomass production of annual plants in an old field from a coastal desert site in north-central Chile. Revista Chileña de Historia Natural 65:83-90.

Hart JF 1991. The perimetropolitan bow wave. Geographic Review 66:1-17.

Hobbs RJ, Gulmon SL, Hobbs VJ & Mooney HW 1988. Effects of fertiliser addition and subsequent gopher disturbance on a serpentine annual grassland community. Oecologia 75:291-295.

Hobbs RJ & Huenneke LF 1992. Disturbance, diversity, and invasion: implications for conservation. Conservation Biology 6:324-337.

Huenneke LF 1997. Outlook for plant invasions: interactions with other agents of global change. Pages 95-103 IN: Luken JO & Thieret JW (eds.) Assessment and management of plant invasions. Springer-Verlag, New York, New York U.S.A.

Huenneke LF, Hamburg SP, Koide R, Mooney HA & Vitousek PM 1990. Effects of soil resources on plant invasion and community structure in Californian serpentine grassland. Ecology 71:478-491.

Huxman TE, Hamerlynck EP, Jordan DN, Salsman KJ & Smith SD 1998. The effects of parental CO_2 environment on seed quality and subsequent seedling performance in *Bromus rubens*. Oecologia 114:202-208.

Huxman TE & Smith SD 2001. Photosynthesis in an invasive grass and native forb at elevated CO_2 during an El Niño year in the Mojave desert. Oecologia 128:193-201.

Jordan DN, Zitzer SF, Hendrey GR, Lewin KF, Nagy J, Nowak RS, Smith SD, Coleman JS & Seeman JR 1999. Biotic, abiotic and performance aspects of the Nevada Desert Free-Air CO_2 Enrichment (FACE) Facility. Global Change Biology 5:659-668.

Kauffman JB & Krueger WC 1984. Livestock impacts on riparian ecsystems and streamside management implications - a review. Journal of Range Management 37:430-438.

Kauffman JB & Pyke DA 2001. Range livestock, global livestock influences. Pages 33-52 In: Levin S (ed.), Encyclopedia of Biodiversity, Volume 5, Academic Press, San Diego, California, U.S.A.

Keeling CD, Chin JFS & Whorf TP 1996. Increased activity of northern vegetation inferred from atmospheric CO_2 measurements. Nature 382:146-149.

Knick ST & Rotenberry JT 1997. Landscape characteristics of disturbed shrubsteppe habitats in southwestern Idaho (U.S.A.). Landscape Ecology 12:287-297.

Knick ST & Rotenberry JT 1998. Limitations to mapping habitat use areas in changing landscapes using the mahalanobis distance statistic. Journal of Agricultural, Biological, and Environmental Statistics 3:311-322.

Knick ST & Rotenberry JT 2000. Ghosts of habitats past: contribution of landscape change to current habitats used by shrubland birds. Ecology 81:220-227.

Lambin EF & Ehrlich D 1997. Land-cover changes in sub-saharan Africa (1982-1991): application of a change index based on remotely sensed surface temperature and vegetation indices at a continental scale. Remote sensing of environment 61:181-200.

Landres B, Morgan P & Swanson FJ 1999. Overview of the use of natural variability concepts in managing ecological systems. Ecological Applications 9:1179-1188.

Lonsdale WM & Lane AM 1994. Tourist vehicles as vectors of weed seeds in Kakadu Natiional Park, Northern Australia. Biological Conservation 69:277-283.

Mack MC, D'Antonio CM & Ley RE 2001. Alteration of ecosystem nitrogen dynamics by exotic plants: a case study of C_4 grasses in Hawaii. Ecological Applications 11:1323-1335.

Mack RN 1981. Invasion of *Bromus tectorum* L. into western North America: an ecological chronicle. Agro-Ecosystems 7:145-165.

Madany MH & West NE 1983. Livestock grazing-fire regime interactions within montane forests of Zion National Park, Utah. Ecology 64:661-667.

McClaran MP, Romm J & Bartolome JW 1985. Differential farmland assessment and land use planning relationships in Tulare County, California. Journal of Soil and Water Conservation 40:252-255.

McDevitt TM 1999. World population profile, 1998. U.S. Bureau of the Census, Report WP/98, U.S. Government Printing Office, Washington DC, U.S.A.

Meyer WB & Turner BL 1994. Changes in land use and land cover: a global perspective. Cambridge University Press, Cambridge, U.K.

Miller RF, Svejcar TJ & West NE 1994. Implications of livestock grazing in the intermountain sagebrush region: plant composition. Pages 101-146 IN: Vavra M, Laycock W & Pieper R (eds) Ecological implications of livestock herbivory in the west. Denver, Colorado, U.S.A.

Mooney HA & Hobbs RJ (eds) 2000. Invasive species in a changing world. Island Press, Washington, D.C., U.S.A.

National Invasive Species Council. 2001. Meeting the invasive species challenge: national invasive species management plan. National Invasive Species Council, U.S. Government Printing Office, Washington DC, http://www.invasivespecies.gov/council/mpfinal.pdf 11 January 2003.

Noss RF, LaRoe ET & Scott JM 1995. Endangered ecosystems of the United States: a preliminary assessment of loss and degradation. Biological Report 28, U.S. Department of the Interior, National Biological Survey, Washington DC U.S.A.

Padgett PE & Allen EB 1999. Differential responses to nitrogen fertilization in native shrubs and exotic annuals common to mediterranean coastal sage scrub of California. Plant Ecology 144:93-101.

Padgett PE, Allen EB, Bytnerowicz A & Minich RA 1999. Changes in soil inorganic nitrogen as related to atmospheric nitrogenous pollutants in southern California. Atmospheric Environment 33:769-781.

Parry M & Carter T 1998. Climate impact and adaptation assessment. Earthscan, London, U.K.

Pellant M. 1990. The cheatgrass-wildfire cycle – are there any solutions? Pages 11-18 IN: McArthur ED, Romney EM, Smith SD & Tueller PT (compilers). Proceedings – Symposium on cheatgrass invasion, shrub die-off, and other aspects of shrub biology and management, Las Vegas, NV, April 5-7, 1989. United States Forest Service, Intermountain Research Station, General Technical Report GTR-INT-276, Ogden, Utah, U.S.A.

Pimental D, McNair S, Janecka J, Wightman J, Simmonds C, O'Connell C, Wong E, Russel L, Zern J, Aquino T & Tsomondo T 2001. Economic and environmental threats of alien plant, animal, and microbe invasions. Agriculture, Ecosystems & Environment 84:1-20.

Polley HW, Johnson HB, Mayeux HS, Tischler CR & Brown DA 1996. Carbon dioxide enrichment improves growth, water relations and survival of droughted honey mesquite (*Prosopis glandulosa*) seedlings. Tree Physiology 16:817-823.

Polley HW, Tischler CR, Johnson HB & Pennington RE 1999. Growth, water relations and survival of drought-exposed seedlings from six maternal families of honey mesquite (*Prosopis glandulosa*): responses to CO_2 enrichment. Tree Physiology 19:359-366.

Polley HW, Tischler CR, Johnson HB & Derner JD 2002. Growth rate and survivorship of drought: CO_2 effects on the presumed tradeoff in seedlings of five woody legumes. Tree Physiology 22:383-391.

Poorter H 1993. Interspecific variation in the growth respose of plants to an elevated ambient CO_2 concentration. Vegetatio 105:77-97.

Poorter H, Roumet C & Campbell BD 1996. Interspecific variation in the growth response of plantss to elevated CO_2 a search for functional types. Pages 375-412 IN: Körner C & Bazzaz FA (eds) Carbon dioxide, populations and communities. Academic Press, San Diego, California, U.S.A.

Pyke DA & Pellant M 2003. Strategic plan for the coordinated intermountain restoration project: U.S. Geological Survey, Biological Resources Discipline, Information and Technology Report, USGS/BRD/ITR2002-0011, Washington DC U.S.A.

Scott JM, Murray M, Wright RG, Csuti B, Morgan P & Pressey RL 2001. Representation of natural vegetation in protected areas: capturing the geographic range. Biodiversity and Conservation 10:1297-1301.

Shugart HH 1998. Terrestrial ecosystems in changing environments. Cambridge University Press, Cambridge, U.K.

Silveri A, Dunwiddie PW & Michaels HJ 2002. Logging and edaphic factors in the invasion of an Asian woody vine in a mesic North American forest. Biological Invasions 3:379-389.

Smith SD, Huxman TE, Zitzer SF, Charlet TN, Housman DC, Coleman JS, Festermaker LK, Seemann JR & Nowak RS 2000. Elevated CO_2 increases productivity and invasive species success in an arid ecosystem. Nature 408:79-82.

Smith SD, Strain BR & Sharkey TD 1987. Effects of CO_2 enrichment on four Great Basin grasses. Functional Ecology 1:139-143.

Sutherst RW 2000. Climate change and invasive species: a conceptual framework. Pages 211-240 IN: Mooney HA & RJ Hobbs (eds) Invasive species in a changing world. Island Press, Washington DC U.S.A.

Svejcar T, Bates J, Angell R & Miller R 2003. The influence of precipitation timing on the sagebrush steppe ecosystem. Pages 90-106 IN: Weltzin JF & McPherson GR (eds) Changing precipitation regimes & terrestrial ecosystems: a North American perspective. University of Arizona Press, Tucson Arizona, U.S.A.

Tilman D 1987. Secondary succession and the pattern of plant dominance along experimental nitrogen gradients. Ecological Monographs 57:189-214.

Tilman D & Lehman C 2001. Human-caused environmental change: impacts on plant diversity and evolution. Proceedings of the National Academy of Sciences 98:5433-5440.

Tucker CJ, Fung IY, Keeling CD & Gammon RH 1986. Relationship between atmospheric CO_2 variations and a satellite-derived vegetation index. Nature 319:195-199.

UNEP, United Nations Environmental Programme 1990. Global assessment of soil degradation. United Nations Environmental Programme, Nairobi, Kenya.

U.S. Congress, Office of Technology Assessment 1995. Harmful non-indigenous species in the United States. OTA-F-565, U.S. Government Printing Office, Washington DC, U.S.A.

Vitousek PM 1990. Biological invasions and ecosystem processes: towards an integration of population biology and ecosystem studies. Oikos 57:7-13.

Vitousek PM, Aber JD, Howarth RW, Likens GE, Matson PA, Schindler DW, Schlesinger WH & Tilman DG 1997a. Human alteration of the global nitrogen cycle: sources and consequences. Ecological Applications 7:737-750.

Vitousek PM, Mooney HA, Lubchenco J & Melillo JM 1997b. Human domination of earth's ecosystems. Science 277:494-499.

Weltzin JF & McPherson GR 2003. Changing precipitation regimes & terrestrial ecosystems: a North American perspective. University of Arizona Press, Tucson Arizona, U.S.A.

West, NE. 2000. Synecology and disturbance regimes of sagebrush steppe ecosystems. Pages 15-26 IN: Entwistle PG, Debolt AM, Kaltenecker JH & Steenhof K (eds) Proceedings: sagebrush steppe ecosystems symposium. Bureau of Land Management Publication No. BLM/ID/PT-001001+1150, Boise Idaho, U.S.A.

Whisenant SG 1990. Changing fire frequencies on Idaho's Snake River Plains: ecological and management implications. Pages 4-10 IN: McArthur ED, Romney EM, Smith SD & Tueller PT (compilers) Proceedings – symposium on cheatgrass invasion, shrub die-off, and other aspects of shrub biology and management, Las Vegas, NV, April 5-7, 1989. U.S. Forest Service, Intermountain Research Station, General Technical Report INT-276, Ogden Utah, U.S.A.

Whisenant, SG. 1999. Repairing damaged wildlands. Cambridge University Press, Cambridge, U.K.

With KA 2002. The landscape ecology of invasive spread. Conservation Biology 16:1192-1203.

Yensen, DL 1981. The 1900 invasion of alien plants into southern Idaho. Great Basin Naturalist 41:176-183.

Young JA & Sparks BA 1985. Cattle in the cold desert. Utah State University Press, Logan, Utah U.S.A.

Young JA, Blank RR & Clements CD 1999. Nitrogen enrichment and immobilization influences on the dynamics of annual grass community. Pages 279-281 IN: Eldridge D & Freudenberger (eds) People and rangelands – building the future, Proceedings of the VI International Rangeland Congress, Aitkenvale, Queensland, Australia.