Selecting Plant Species for Ecological Restoration: a Perspective for Land Managers

Ray W. Brown
Michael C. Amacher

Abstract—We recommend in this paper that land managers adopt a policy of mandatory use of native plant species for revegetation and restoration of severe disturbances on wildlands throughout the Interior West. A review of the relative advantages and disadvantages of using introduced and native species suggests that selection criteria based on ecological adaptability and suitability are more consistent with the objectives of ecosystem management than are criteria based on cost, availability, familiarity, or other nonecological considerations. We suggest that land managers initiate a policy requiring the collection and accumulation of native seral species throughout their respective regions and districts to be used in restoration activities. Further, we suggest that such a policy will foster closer ties between public land managers and public research scientists, and will enhance the implementation of science-based land management.

Recognition that ecosystem health has been in decline West-wide on public lands has motivated the adoption of ecosystem management by many public land management agencies in an effort to restore broad-scale ecosystem diversity and resiliency (Dombeck 1998; Thomas 1997). The urgency driving such measures stems, in part, from an accelerating public demand for more natural resources, a recognition that this demand cannot be met if ecosystem health continues to decline, and a realization that we may not possess the scientific or applied knowledge needed to implement appropriate solutions. Meeting the onerous challenges of an expanding public demand for natural resources while simultaneously preserving the sustainability of water quality and quantity and other renewable resources on Western wildlands should assist land managers in adopting a more science-based form of land management directed by ecological principles rather than just economic ones.

One of the direct consequences of this philosophical shift in policy is that land managers are being pressured to reexamine standard practices of reclamation and revegetation of disturbed lands. Concerns about restoration of ecosystem health center, in part, on the use of suitable plant species in revegetation that are consistent with overall ecosystem form and function. The "old saw" controversy over the relative advantages and disadvantages of using native versus exotic or introduced plant species has continued to fester for decades. The controversy is driven by strong emotions fostered by a spectrum of concerns ranging from economic issues to "environmental correctness" to physiological adaptability. In addition, the type of disturbance, its location and environmental limitations and conditions, and historic and potential future land uses also become entangled with the various arguments for or against the use of each group of species. The scientific objectives of the controversy are often lost or mixed with the passionate ones, and often are misrepresented or reformatted to fit predetermined points of view. Unfortunately, the debate has been tainted by both the political and public relations arenas. This has probably impaired progress toward adoption of a clear and unified policy favoring use of appropriate plant species by the major land management agencies.

This paper summarizes some of the concerns and constraints in selecting and using native and introduced or exotic plant species in reclamation and restoration of disturbed lands. We discuss some of the criteria used to select plant species for the active restoration of disturbed lands, including concerns about native and introduced species, and suggest alternative standards that may be useful in choosing plant species for restoration purposes. We offer principles, guidelines, and criteria for judging what plant species should be selected for restoring natural ecosystem form and function based on our more than 40 collective years of research experience in surface reclamation and restoration throughout the West.

The focus here is on the issue of selecting plant species for the restoration of severe disturbances on wildlands where the intent of management is to return the disturbed site to a self-sustaining condition supporting natural ecosystem processes. Not addressed in this paper is the issue of general reclamation techniques designed for specific land uses such as pasture development, enhancing productivity for livestock grazing, campground development and esthetics, and other uses that may differ from wildland ecosystem form and function. Where the intent of management is to sustain natural ecosystem processes with little or no continued inputs by land managers, it is our contention that management practices of reclamation and revegetation be implemented as tools to achieve full ecological restoration. Further, we support the hypothesis that use of successional native plant species is essential in order to achieve ecological restoration, that this practice is consistent with the goals of

In: Holzworth, Larry K.; Brown, Ray W., comps. 1999. Revegetation with native species: proceedings, 1997 Society for Ecological Restoration annual meeting; 1997 November 12–15; Ft. Lauderdale, FL. Proc. RMRS-P-8. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Ray W. Brown is Research Plant Physiologist and Project Leader, and Michael C. Amacher is Research Soil Chemist, USDA Forest Service, Rocky Mountain Research Station, Forestry Sciences Laboratory, Logan, UT 84321.

ecosystem management, and that the process of natural succession should be used as a guide in selecting native species. We contend that many conventional cultural practices (such as fertilizer use, organic matter incorporation, surface mulching, liming of acidic spoils), although important and ameliorative in many cases, are insufficient alone to achieve ecological restoration if suitable native species are not also included in the restoration process. Natural succession is a universal process that is ongoing throughout natural ecosystems, and should be identified, observed, and used as a guide in not only selecting species that are suited for specific sites, but in developing entire restoration plans.

We believe the interests of land managers would be advanced by developing strong ties with research entities that could assist with the identification, selection, and collection of native plant seeds suited for restoring disturbed habitats and communities throughout the West. If these agencies recognized the true value and need of additional research, and were willing to apply science-based management policies through cooperative interactions with such agencies as the Natural Resources Conservation Service and appropriate State and local offices and groups, collectively they could have the commanding privilege of finally settling the matter of restoring disturbed public wildlands to a near-natural state in the spirit of ecosystem management in virtually every Western ecosystem.

Although we contend that ecological restoration of severely disturbed wildlands should be accomplished with appropriate local native plant species, we do not support or agree with the argument that all exotic or introduced species are somehow "bad" or subversive elements to be exterminated. All plants, regardless of origin, have biological and ecological legitimacy, but not all of them are necessarily suited for restoring every habitat or ecosystem. We do not endorse emotional-based values such as political or ecological "correctness" to plants or other natural systems regardless of their origin or potential use, nor do we necessarily support the idea that human values deserve greater or less favor than natural systems. It is our contention that plants and all natural systems are intrinsic to the survival and well-being of society, but imperatively they function quite separately from human emotions, desires, and economic concerns. Therefore, we contend that in managing and using these systems we need to be careful in judging their values and functions on the basis of what they truly are, not how we would like them to be.

Definition of Terms

In order to expedite understanding of concepts, the following definitions are offered with the understanding that rarely is there universal agreement among users of just what a definition should include.

Native plant species, like any plant species, are composed of an array of ecotypes or populations whose individual genetic material is circumscribed by the forces of natural selection originating under the particular local environmental conditions of climate, agents of disturbance, and other factors that characterize the area in which they evolved. Thus, the genetic material of each ecotype within the species complex is individually imprinted with distinctive physiological or anatomical-morphological features specific to the environment in which they originated. As such, native species survived and repeatedly reproduced under the vagaries of unstable climates, predation, disease, herbivory, competition, and all the other limitations and stresses imposed by an environment over multiple generations of continuous genetic editing and sorting. We purposely avoid the contentious issue of defining scale in terms of "local" conditions, preferring to be vague here with deference to more elaborate intonations (such as "local-native" versus "indigenous native" versus other terms) in the rather extensive literature (minimally cited here: Anderson 1971; Bengson 1986; Benson 1957; Harper 1977, Society for Range Management 1974; Stebbins 1974, 1979).

Introduced or **exotic species** (we equate both terms) are considered to be species whose genetic material originally evolved and developed under different environmental conditions than those of the area in which it was introduced, often in geographically and ecologically distant locations (Bengson 1986; Society for Range Management 1974). The issue of quantifying the difference in environmental conditions between the place of origin and the current place of occurrence is always speculative, and is not pursued here. Similarly, we avoid the issue of "naturalization" by introduced species, opting to suggest that if the environment of origin of the genetic material differed from that in which it was introduced, the plant then probably remains "introduced" despite its residence time (usually measured in years or decades). Under this definition, individual plants or ecotypes of a "native species" that originated in other ecological, topographic, or climatic regimes (regardless of distance or political boundaries) are considered to be "exotic" or "introduced" relative to the local environment. Caution is warranted when working with plant populations or ecotypes bearing the same taxonomic name based on anatomical or morphological features; populations of the same species name but that originated in different environments probably do not possess identical or even similar genetic composition, physiological tolerances, or environmental requirements. Similarly, political and administrative boundaries are not barriers to the migration of plant genes, and are not considered here as legitimate separators among groups of biological entities.

Considerable confusion arises with these or any definitions in specific applications because they provide no guidelines for definitive separations among groups. Questions such as "How far away from the location of origin does a native plant fade into becoming an introduced plant?" arise, but little solace can be offered here. The fact that plants or other organisms are not sensitive to distance units is of no comfort to land managers faced with choosing one plant over another for specific sites. Alternative terms such as "near native" are often suggested to differentiate native plants originating in a given habitat from other native plants that originated in other, or different, habitats ("far natives"?). Other terms have occasionally been suggested, apparently with the intent of overcoming some of the limitations of language or historic bias, including "suitable species" or "naturalized species." Supposedly, a "suitable" species is one that is adapted (although not specifically defined so), but with desirable characteristics for a given application. This term would seem to confuse more than clarify the issue from the perspective of wildland restoration because agricultural crops may fit the definition as well as "native" or "introduced" wildland species. A "naturalized" species is apparently a formerly "introduced" species that has successfully competed with native species for numerous generations, and in recognition of its success is endowed with the estimable title of "naturalized." Although maybe such terms are conceptually more tidy and provide a more comfortable feeling among humans, they provide little guidance and have virtually no applicability to plants or other natural systems. In addition, a high degree of emotionalism has crept into these definitions, and somehow many people tend to attach anthropomorphic or teleological interpretations to them as if these terms were something more than just words to describe totally artificial entities. Contrary to some opinions, there is no actual biological entity definitively characterized by the definitions of "introduced" and "native" species. Perhaps the concepts and terminology of "native," "introduced," and other similar terms should be abandoned altogether in favor of something more precisely defined and understood, but there is little likelihood of that, and perhaps there is little benefit to be derived.

Ecological restoration is the practice of *re-establishing* natural ecosystem processes responsible for the development of ecosystem form and function, including all major biotic and abiotic components, on lands where these forces have been terminated, interrupted, or deflected as the result of disturbance. *Passive* restoration relies exclusively on management policy and other indirect options as a means of restoring the desired condition. *Active* restoration, or the "intentional intervention" by land managers to alter disturbed conditions in order to accommodate the *reinitiation* of natural processes that lead to the redevelopment of ecosystems, habitats, or communities is emphasized here. This usually involves such activities as altering management techniques combined with, on the most severe disturbances, a more operative approach that involves reclamation, revegetation, and other active options. In most cases, the primary objective of ecological restoration is the reinitiation of natural succession that will lead to the re-establishment of ecosystem form and function. Although ecosystem restoration has been the subject of numerous theoretical discussions recently (Cairns 1995; Everett 1994; Jackson and others 1995; MacMahon and Jordan 1994; Urbanska and others 1997), from a practical and applied land management posture the alleviation of limiting site conditions and the selection of ecologically fit plant species are crucial.

Reclamation is considered to be the process of mitigating physical or chemical environmental conditions perceived to be limiting to land management objectives. Reclamation is an attempt to alter or lessen the effect of environmental damage through whatever means are available. For example, reclamation may include revegetation, use of impoundments and diversion channels to control water movement, chemical treatment to improve soil and water quality, contour trenching, terracing, or other earth-shaping activities to minimize surface runoff and erosion, using chemicals or an array of surface fabrics and other treatments to stabilize steep slopes.

Revegetation is the process of establishing vegetation. The semantic issue of *re-establishment* of vegetation on sites that had previously supported vegetation versus establishment on soil materials, for the first time, is ignored.

Ecosystem management is the process of guiding and manipulating the development and use of natural resources required for society according to principles that enhance and perpetuate ecosystem diversity and sustainability. Under this definition, management options, reclamation, and revegetation are a few of the many tools employed in the implementation of ecological restoration.

Succession is defined in the broadest sense, and is intentionally not restricted to just the sequential development of vegetation or plant communities. Therefore, succession is intended to imply not only vegetation, faunal, and microorganism development, but also includes soil genesis, the establishment of hydrologic pathways both at and below the soil surface, the re-establishment of geomorphological and biogeochemical processes, and all the various natural mechanisms that are initiated following disturbance that ultimately lead to the development of ecosystem form and function. As a natural process guided by climatological, geological, and biological forces, succession is constrained by limiting factors in the environment. When these limits are exceeded, succession is retarded or repressed until they are ameliorated.

The Nature of Disturbance

Deterioration and loss of ecosystem health are often associated with the occurrences of disturbance, but it should be emphasized that not all disturbances negatively affect ecosystem health; in fact, it is widely recognized that many forms of disturbance are essential for the maintenance of natural ecosystem form and function (Kaufman and others 1994; McIntosh 1981; Rogers 1996; Vogl 1980). Ecosystems evolved under the influence of stochastic climatic conditions and variable biology and geologic activity. Forces of natural "disturbance" have been of paramount importance in the long-term development of ecosystems and include: (1) physical phenomena such as climatic change, seismic activity, landslides, and fire, and (2) biological agents such as humans, insects, diseases, grazing animals, invasion and competition from outside flora and fauna, and others. For example, some natural disturbances such as fire (Bartos and Campbell 1998), periodic insect and disease cycles (Mattson 1996; Vogl 1980), and climatic extremes (Vogl 1980) are integral components of natural systems that are responsible, in part, for healthy functioning ecosystems. Interference of such natural forces by human activities has led to significant consequences, often resulting in deterioration of ecosystem health and degeneration in the availability of natural resources and their sustainability. It is clear that such disturbances are just as compulsory for healthy ecosystem form and function as are the multitude of biotic and abiotic components and forces encompassing natural systems.

Other forms of disturbance can be detrimental to ecosystem health, and may result from either human or natural causes. In the relatively short span of only about 200 years, the impacts of an expanding human society have been introduced into the ecosystems of North America, imposing quite different forces and types of disturbance than those involved in their evolutionary development. With the encroachment of European-style society, and especially the large increases in population with accelerating economic and cultural growth in more recent decades in the West, the ranges and extent of human-caused disturbances on ecosystems are expanding as the result of two primary activities: (1) construction-development-extraction enterprises and (2) inappropriate wildland management policies and strategies. Each decade the impacts of these activities are growing in intensity and geographic extent at dramatic and sometimes accelerating rates. The traditional Western industries of logging, livestock grazing, and mining are being replaced by road and highway construction and their perpetual maintenance, recreation, development of destination resorts and associated recreational facilities, pipelines, powerlines, exploding urban expansion (compounded by changing natural resource opinions and land ethics), and associated activities as sources of pervasive disturbance in virtually every ecosystem at all elevations in the West. There are perhaps no comparative natural disturbances of the magnitudes, frequencies, and intensities of these anthropogenic activities. Even regional volcanism, seismic activity, or anomalous and extreme climatic events (such as microbursts, extreme drought, heavy precipitation events) that may have significant impacts on ecosystem form and function occur much less frequently and less pervasively than current human activity.

Disturbances are referred to as "severe" if they result in the complete loss of native soil and vegetation, if they disrupt or destroy natural surface and subsurface hydrologic pathways, and if they result in accelerated rates of erosion and sediment transport. Severe disturbances are normally most significant on the local or watershed scale, but may have far greater impacts on water quality and quantity, wildlife habitat, and other attributes than larger scale, less severe disturbances. In the context of the larger issues of "Disturbance Ecology" (Rogers 1996), understanding the distinctions between and the interactions of human-induced and natural disturbances is essential to clarifying and interpreting the significance of human impacts on natural ecosystem processes. These interactive effects can be of compounding importance over time as they relate to stream flow, erosion, vegetation, and the transport of nutrients and toxic chemicals. Although these effects are exacerbated by both human and natural disturbances, human activities are becoming increasingly commonplace throughout the West in harsh environments and steep watersheds predisposed to slope instability. Human activities have the most severe impacts when they occur in harsh environments where natural recovery occurs slowly over long periods (such as in arid regions or at high elevations), in the headwaters of drainages, near streams and rivers, or on steep unstable slopes. Under these conditions, the potential for erosion, sediment transport, and subsequent degradation of water quality are maximized. Generally, total precipitation is low in many Western ecosystems, and precipitation regimes are typified by episodic high-intensity storms that dramatically affect landscapes and streams. Intensive human activity tends to exacerbate many natural forms of disturbance because much of the mountainous terrain in the West is naturally susceptible to surface erosion and mass-wasting.

There are real dangers in altering natural ecosystems that extend well beyond concerns about aesthetic appeal or changing floristic composition. Severe disturbances can be so harsh and environmental conditions can be so austere that the reinitiation of natural succession under such conditions can be limited or even prevented altogether. As a result, succession may become deflected, suspended, or stalled (Allen 1988; Curtin 1995; Glen-Lewin and others 1992; McIntosh 1980; Schramm 1966). In such cases, severe disturbances may result in loss of the "evolutionary roadmap" that allows succession to guide a damaged ecosystem back toward restored functionality and recovery. Some of the consequences of severe disturbance include:

1. Loss of ecosystem resiliency to forces of climate change and recurring disturbance.
2. Replacement of natural floral compositions by invading weeds or other undesirable species, resulting in loss of biodiversity and sustainability.
3. Accelerated rates of erosion, loss of developed soil, altered soil development, and declines or loss of nutrient cycling.
4. Oxidation of acid-forming materials or altered weathering patterns that enhance the availability of toxic chemicals.
5. Adverse impacts on natural hydrologic pathways.
6. Sediment and chemical transport and deterioration of water quality and quantity in ecosystems.
7. Loss of fauna that may have far reaching impacts on vegetation development and vegetation-animal interactions.

Intact functioning ecosystems conserve matter and energy (Billings 1965; Whittaker 1975), hence there is little net loss of energy or components needed to sustain them. Because natural intact ecosystems are self-regulating systems, they are internally and naturally buffered against all but the most severe of outside forces. The adoption of ecosystem management by land management agencies suggests that intact ecosystem function is recognized as the primary level of natural resource management to be attained (Boyce and Haney 1997; Kohm and Franklin 1997; Perry 1994; Yaffee and others 1996). It seems clear that if ecosystem management is the goal, we need to comprehend how ecosystems function; but to achieve that, we need to also understand how ecosystems develop and form from genesis to maturity. Restoration following disturbance provides an opportunity to achieve that level of understanding. Aldo Leopold noted that "If we are serious about restoring or maintaining ecosystem health and ecological integrity, then we must first know what the land was like to begin with" (quote in Covington and Moore 1994).

Plant Species Selection: the Debate

The selection of plant species for use in the restoration of severely disturbed public wildlands in the West continues to be a controversial and encumbering issue (Bengson 1986; Brown 1980; Gutknecht 1992; Mills and others 1994; Roundy and others 1997; Shaw and Roundy 1997). The criteria used to select plant species are generally recognized as some of the most important judgments that can affect the relative success of active restoration. Concerns over adaptability, suitability, and compatibility of the selected plant species with the surrounding ecosystem often become confounded with land management objectives and even personal biases, and these often lead to species selection decisions that contradict scientific judgement and ecological concerns. Selection criteria for most wildland or rangeland applications are often based on various characteristics, including productivity, habitat attributes, palatability, cover, aesthetics, and long-term vigor in particular climatic regions. Some land managers even consider physiological adaptation and ecological compatibility with surrounding vegetation; but in selecting species for restoring wildland disturbances, the criteria most often used are ill-defined "seat-of-your-pants" concerns and should probably not be categorized as criteria at all.

The distinctions that separate native and introduced species often are not as obvious and clear cut as many would like to believe. We may even have become so enamored with the debate that we have lost sight of its substance. The true differences between native and introduced species are largely physiological, arising from entirely different evolutionary histories and genetic characteristics, but these appear to be no greater than those arising between two different native species originating in two different environments. It seems that we often forget that all natural vascular plants are (or were) native to some region and environment prior to human intervention regardless of political boundaries, and that all of them cope with the environment using surprisingly similar types of physiological mechanisms. Aside from anatomical and morphological differences of shape or size, what truly distinguishes vascular species is the efficiency with which they transfer energy and acquire resources within the range of environmental conditions that prevail. If a plant is efficient in coping with its environment, it survives; if it is not, it dies. Thus, we should probably differentiate among plants at the ecotype, population, or species level more on the basis of their adaptability and physiological tolerances than on artificial distinctions of taxonomic name alone. Too often we ascribe significance to artificial nonbiological entities, such as "native" and "introduced," and to artificial taxonomic units such as "species" (Anderson 1971; Harper 1977; Stebbins 1974, 1979).

The range of physiological tolerances of each individual plant are defined by its genetic material, and in natural plants (as opposed to taxa whose genetic material has been intentionally manipulated), genetic expression is the product of natural selection acting over long periods of evolutionary time. The total pool of genetic material among all individual plants of a population or species defines the "heritable" attributes, including the range of physiological tolerances, for that entire population or species. In its most succinct form, the term "adaptability" comes closer to defining what controls species performance and distributions than do terms like "native" or "introduced." "Adaptability" has been variously defined, but basically it is considered to include the genetically controlled process of modifying structures and physiology in response to environmental conditions (Conrad 1983). "Acclimation," however, describes the ability of plants to make physiological adjustment in response to changing or altered conditions in the environment (Conrad 1983; Nilsen and Orcutt 1996). As emphasized by Smith (1978), there is no adaptation or acclimation in response to environmental stress unless there is genetic control of physiological responses, and these are heritable.

Unfortunately, the concept of adaptability is arguably misapplied with reference to selection of plant species, frequently being confused with anthropomorphic measures of "goodness" and "badness" instead of ecological fitness and suitability. As used by land managers, the concept of adaptability probably most often includes both adaptation and acclimation, and encompasses concerns about the ability of a species to grow, survive, and reproduce in a given environment. When selecting plant species for revegetation and reclamation, the criteria used to determine adaptability are usually based on experience, and often include attributes that appear to indicate adaptability to local conditions. These may include: (1) survival over a number of generations in the same or similar areas; (2) ability of a species to reproduce and complete its entire life cycle; (3) apparent tolerance to water deficits, temperature extremes, and nutrient deficiencies; (4) relative vigor during germination and growth (often based on plant size); and (5) apparent palatability for livestock. Care should be practiced in how the concept of adaptability is applied, because to describe adaptability only in terms of competitiveness, aggressiveness, or potential above-ground productivity renders the concept meaningless for species selection. Adaptability is not a unit of measure (such as biomass, height, density, cover), nor is it a unit of forage production or quality. Adaptability is a term defined by the physiological range of tolerance of an organism as determined by its genetic material. If the environmental extremes of a habitat or ecosystem fall within the range of physiological tolerances of a species, that species can adjust to them and carry on vital physiological processes. If extreme environmental conditions ever exceed that range (just one time!), however, the plant dies.

Rationale for Selecting Introduced Species

Land managers are often pressured to select species on the basis of human-based and economic criteria, and these criteria almost always favor the selection of introduced plant species over natives because most introduced species possess attributes that more nearly favor preconceived human notions of what is "good" or ideal for achieving land management goals. These criteria encompass the ranges of tolerance of most introduced species in most environments of the West, and thus conveniently suggest that introduced species are at least equally adapted to local conditions as natives. Based on these criteria alone, and ignoring all others, it would be difficult to argue against introduced species given the geographic range over which many have been successfully used for erosion control and range improvement, and given the multiple generations that many of them (including invasive species) have endured in the West. Most of the common introduced plant species used in revegetation of Western rangelands and disturbances are the products of long-term research and "improvement" through breeding and selection. Many of these species were brought to North America from Europe and Asia decades ago because of certain beneficial traits deemed desirable at that time. It is little wonder that smooth brome (*Bromus inermis*), orchardgrass (*Dactylis glomerata*), timothy (*Phleum pratense*), crested wheatgrass (*Agropyron cristatum*), and others produce higher biomass than native species, are more favored by livestock, have higher seed production, higher seedling and mature plant vigor, green-up sooner and remain green longer, or grow to larger size than native species; they have the "advantage" of having been manipulated, selected, and sorted for specific predetermined uses and objectives over multiple generations.

Some of the reasons why introduced species may appear attractive for wildland revegetation and restoration include:

1. Familiarity: Many introduced species have been used extensively for many generations for highway revegetation, hay crops and pastures on farms and ranches, rangeland improvement by Federal and State agencies, and many other applications.
2. Availability and cost: Introduced species are more widely available commercially than natives at cheaper prices.
3. Growth and vigor: Introduced species tend to have higher germination and seedling vigor, growth and development rates, and flowering and seed productivity than most native species.
4. Plant size: For given life forms, introduced species tend to grow larger, have larger vegetative structures, and provide more cover and biomass per plant under favorable conditions than most native species.
5. Palatability: Introduced species are generally more preferred by livestock over longer periods during the growing season than natives.
6. Growth form diversity: Introduced species tend to have richer varieties of growth forms (such as rhizomatous versus bunch habit, larger spreading vegetative crowns, greater seed production) than natives.
7. Ranges of tolerance: Some introduced species tend to have broader physiological and ecological ranges of tolerance to limiting conditions, and may tolerate herbivory better than many natives.
8. Competitiveness: Introduced species tend to be more competitive for limited resources than many natives.

These features are often cited as reasons why introduced species are "better," or more advantageous, than native species (Bengson 1986; Gutknecht 1992; Thornburg 1982). Although there are no valid biological reasons to believe that one species is somehow "good" or "bad," it appears there is a surprisingly prevalent attitude that human values somehow apply to biological systems. Human measures of goodness or badness are normally based on economic and measurable quantities such as productivity, growth rate, palatability, or even availability of seed or plant stock, but rarely on biological factors such as physiological tolerance or adaptability.

The issue of seed acquisition cost is particularly bothersome because it is more often used as an excuse for substituting introduced species for native species than any other. Seed cost is almost always determined on the basis of cost per unit PLS (pure live seed) weight. Although cost concerns may be valid and are intentionally not trivialized here, there are at least two issues that tend to offset the real effects of cost:

1. Native species often have far more seeds per pound than most introduced species, and hence the cost per unit PLS seed (the potential number of live plants per unit area) may be far less than expected.
2. The "ecological cost" of *not* restoring a wildland disturbance to a self-sustaining condition by using introduced species may have "hidden" costs and negative impacts in perpetuity as has been the case in numerous documented instances throughout the West and other regions in recent history (Brown 1995; Hess 1995; Lesica and DeLuca 1996; Mills and others 1994; Roundy and others 1997; Seagrave 1976).

Another almost universal excuse for not using native species is the issue of availability of seed. There is no doubt that the *commercial* availability of native species for use in revegetation has been limiting and that real efforts to enhance that are arguably minimal and primarily local in scale (although this situation is changing rapidly). However, there is not now, nor has there ever been, a real shortage of collectible native seeds in virtually any ecosystem with which a public land management agency has been involved. The only real shortage has been in administrative imagination and commitment to implement serious steps toward ecological restoration as part of ecosystem management policy. Land managers need to be more sensitive to this need and either collect their own seed from key areas or contract with professional seed collectors or dealers as needed.

Another rationale used to legitimize the use of introduced plant species in revegetation and restoration of wildland disturbances suggests that when severe disturbances create such harsh environments that the successional "clock is reset to zero," the same successional trajectory that led to the predisturbance community may no longer be possible. This argument suggests that because climatic conditions have probably changed since the last time succession was initiated (perhaps hundreds or thousands of years ago), the identical pathway that led to the immediate predisturbance community is no longer possible or available (this argument is widely cited as a reason why the process of ecological restoration as implemented by humans is impossible). Because this pathway has been altered significantly, the native species that developed along with the community are likely no longer available or adapted. Therefore, this argument suggests, the use of new introduced genotypes may be justified.

Such reasoning also overlooks the fact that the prevailing climate, no matter how different from that of the past, is and has been the primary guide affecting succession and ecosystem form and function up to the present, and despite any recent severe surface disturbances, remains in place. The same species that were components of the predisturbance community were primarily influenced and sorted by those same variable climatic conditions. It is reasonable to extrapolate that restoration using these native species would continue to be primarily influenced by those same, albeit changing, climatic conditions. These native species have the benefit of sorting by natural selection under those variable conditions, even if of relatively recent origin, whereas introduced species do not.

Climatic and other environmental conditions are probably rarely static for significant periods of time in any ecosystem, and current conditions in most ecosystems are likely not representative of the highly variable environments that existed throughout their evolution (Covington and Moore 1994; Ellis and Galvin 1994; West and others 1994). Thus, it is almost certain that climatic instability over time has been a normal constituent of ecosystem evolution. Climatic instability or climate change in and of itself does not automatically mean that there will be measurable shifts in floral or faunal composition, soil development, nutrient cycling, or any other ecosystem attribute. We contend that it is entirely reasonable to suspect that environmental change is ubiquitous, but that it is unreasonable as an excuse to alter species compositions for the sake of economic and administrative convenience. Extending well beyond any moral or ethical considerations of public land management, such decisions require caution and far more knowledge than is currently available.

Another pretense suggests that European people have so grossly altered the original ecosystems of North America (Kay 1994) that these ecosystems no longer exist and, therefore, concerns about diversity are no longer valid. Yet another argument suggests that humans are now the prevailing force shaping ecosystem form and function and, therefore, humans should have the freedom to shape and form ecosystems in any manner deemed economically or aesthetically suitable. The latter pretense stems, in part, from archaic theological determinism based on biblical or religious belief and will not be discussed further.

Perhaps most disturbing of all is the apparently persuasive argument that if the natural soil has been destroyed through either human-caused or natural disturbances, natural ecosystem processes and function have been irreparably altered, and thus native species are no longer adapted. This argument ignores the numerous examples of natural succession on mine lands and along road cuts and fills (Brown and others 1979; Chambers and others 1984; Winterhalder 1995), on landslides and areas of volcanism (del Moral and Wood 1993; Nilsen and Orcutt 1996; White 1979), and other drastic and severe disturbances where natural soil was lost or destroyed. Many such disturbances expose raw disrupted geological materials at the surface, some of which harbor acidic constituents and high concentrations of toxic chemicals totally dissimilar to the predisturbance natural soil. Many studies have shown that natural succession occurs on such materials, and not always slowly and undramatically, composed of native seral species that are representative of adjacent undisturbed communities (Brown and others 1979; Chambers and others 1984; del Moral and Wood 1993; Munshower 1993; Winterhalder 1995). Obvious exceptions are noted on some severe disturbances where exotic annuals or perennials appear to overwhelm native species succession (cheatgrass [*Bromus tectorum*], Russian thistle [*Salsola kali*], bindweed [*Covolvulus arvensis*], dyer's woad [*Isatis tinctorum*], and other species) (Monsen and McArthur 1995; Sheley and others 1996; Young and Evans 1976).

The presence of invasive weeds, even at densities and concentrations that appear to exclude native seral species, is no indication that natives are no longer adapted; it merely suggests that the native species may be less competitive and have temporarily been suppressed. In some cases observations suggest that dominance by invasive weeds may be a temporary phenomenon lasting perhaps several years, and that eventually the perennial natives resume succession (personal observation), although notable exceptions are also common (Monsen 1994; Munshower 1993; Smith and others 1988). Unfortunately, in addition to being wrong, we believe the argument tethering species adaptability to natural soil is even more subtly dangerous than most misinterpretations of natural forces because it has implanted an unwarranted fear of failure for using native species in areas where highly competitive exotics may appear to dominate early succession. It is clear that despite infestations of aggressive introduced weeds in some areas, natural selection has sorted for highly adapted early seral native species that successfully and routinely colonize severely disturbed lands in virtually every ecosystem, and that the challenge for the land manager is to identify them and learn to capitalize on their adaptability.

Consequences of Using Introduced or Unadapted Plant Species

Historically, the most common form of reclamation or rehabilitation used by public land managers on disturbed areas has been revegetation. Typical revegetation and reclamation activities by public land managers have overwhelmingly relied upon the use of non-native plant species, and have largely been guided by the perception that such practices can improve natural conditions. Land managers revegetate and reclaim disturbances to minimize erosion and sediment movement, and to improve water quality, wildlife habitat, forage for livestock grazing, aesthetics, and other attributes, but give little thought to the long-term ecological restoration of the disturbance. It is generally accepted that re-establishing a plant cover on disturbed sites is the most effective means of achieving these goals, although engineering structures and other facilities are often utilized as well. In many cases, the land manager is less concerned about the origin or physiological tolerances of the selected species than for how quickly and inexpensively a plant cover can be established to provide forage, surface stability, and reduce erosion. By doing this, the conventional leap of faith instills the hope that surface stability will somehow allow "nature to take its course" and return the disturbed area to ecological recovery or some other desired state. This belief further extrapolates that the other attributes of wildlands, including wildlife habitat, aesthetics, and water quality, will eventually be enhanced as a result of these practices. This rests heavily on the assumption that the course of natural succession following revegetation will be directional and desirable regardless of what plant species are used, and that immediate stabilization of disturbed soil is far more important than other concerns about plant species composition of the newly created community (Hull and Holmgren 1964; Thornburg 1982; Thornburg and Fuchs 1978).

Although stabilizing the surface and minimizing surface erosion are beneficial and in some cases (such as following extensive fires) may be considered essential, we have learned in recent decades that revegetation and many other remedial land management activities are no guarantee that the course of nature and succession will progress as expected or as desired (Brown 1995; Lesica and DeLuca 1996; Mills and others 1994). Often the use of exotic plant species and various cultural treatments on severely disturbed wildland areas, although perhaps capable of providing short-term surface stability and protection from the immediate problems of surface erosion, lead to highly oversimplified and artificial plant community systems totally inconsistent with the form and function of the natural ecosystem, or with the objectives of ecosystem management. Vast areas of these unnatural systems have been created throughout the West as a result of unfortunate land management decisions and poor information, and not only have these areas not returned to a natural ecosystem state, but may in fact be retarding or delaying ecosystem re-establishment following disturbance. In many documented instances, some land management practices such as fire exclusion (Bartos and Campbell 1998) and spurious revegetation-reclamation activities (Allen 1988; Bradshaw 1995; Briggs 1996; Brown 1995; Lesica and DeLuca 1996; Mattson 1996; Monsen 1994; Munshower 1993) can result in apparently near-permanent losses in plant community biodiversity, deflected, arrested, or misdirected successional development (Glen-Lewin and others 1992), ecosystem dis-integrity (Regier 1993), and loss of ecosystem resiliency to climate change and the impacts of cyclic redisturbances caused by either natural or human agents (Briggs 1996; Mattson 1996; Rogers 1996; Williams and others 1997; Woodley and others 1993).

It has been firmly established that many introduced species are more aggressive and more competitive for limited resources such as water and nutrients than most native species. Hence, a loss of biodiversity over time (loss in species numbers; shifting proportions or loss of life forms such as grasses, forbs, shrubs, and trees; reduced pools of physiological traits; and so forth) is being widely observed. A primary consequence of loss of diversity is a concomitant loss in flexibility, resiliency, and recoverability under environmental stress (such as recurring drought, climate change, increasing levels of human influence, grazing). As emphasized by Mattson (1996), if diversity in ecosystems is in any way simplified, it likely may lead to more frequent and possibly more severe outbreaks of various insect pests. Analogous extrapolation would suggest similar responses for disease and other agents of disturbance.

For example, extensive National Forest lands impacted by phosphate mining in southeastern Idaho on and adjacent to the Caribou National Forest were revegetated with exotic species including alfalfa (*Medicago sativa*), smooth bromegrass, and orchardgrass over the last 25 to 30 years (Richardson and Farmer 1985). Most of those revegetated areas, including the oldest ones, still support extensive communities of these highly competitive high-resource-consuming plant species to the virtual exclusion of native species. As a result, vascular plant biodiversity has been depleted by more than an order of magnitude, natural succession has been arrested or deflected, soil development lags far behind expectations, insect and disease outbreaks have devastating impacts on the species-poor flora, and land managers are discovering that wildlife populations and domestic livestock congregate in these artificial communities to the detriment of soil stability, water quality, and community stability (Zufelt 1998). Other similar examples are common throughout the West, including Watersheds A and B on the Wasatch Plateau (Meeuwig 1960), some tall forb communities in the Wasatch Mountains (Monsen and McArthur 1995), and other revegetated mine lands (Zufelt 1998).

Similarly, thousands of acres of crested wheatgrass stands were established throughout the West in the name of rangeland improvement (Hull and Holmgren 1964; Johnson 1986; Vallentine 1989) over the last 60 to 80 years, and they continue to persist with analogous consequences (Lesica and DeLuca 1996). Countless miles of interstate highway seedings planted decades ago with crested wheatgrass (the so called “savior of the West”) are still dominated by that species, as are numerous rangeland seedings throughout the West. In our opinion it has become clear that, relative to restoration of native communities in wildland areas, the concept that any plant cover is equally beneficial and “good” if it provides site protection and minimizes surface erosion, regardless of what species are involved, is not only misleading and dangerous, but is probably ecologically wrong as applied to Western ecosystem restoration and should be abandoned. Equally, the concept that any form of reclamation (including revegetation) will eventually allow nature to take its course in restoring disturbed ecosystems is naive and misleading. In recent decades, numerous documented cases describe where artificial revegetation with introduced or exotic species and other reclamation activities (under the best of intentions) have arrested successional development and altered the natural composition of significant portions of numerous ecosystems throughout the United States (Bartos and Campbell 1998; Brown 1995; Covington and Moore 1994; Detwyler 1971; Kaufmann and others 1994; Kay 1994; Lesica and DeLuca 1996; Mills and others 1994).

The Alternative: Ecological Restoration

The Role of Natural Selection

A strong and compelling argument can be made that species selection should be based on physiological adaptability and on the suitability and compatibility of the species with local flora or ecosystem conditions. The criteria commonly used in selecting introduced species do not usually encompass the physiological basis for, and the process of, adaptability to changing conditions over time. Given that our total experience with revegetation, natural resource management, and environmental science in the West only spans about 100 years, in terms of evolution and genetic flexibility we really know very little about the true adaptability of any species, native or introduced, or about the full range of environmental conditions that really characterize our Western regions. In fact, we have begun to gain quantitative knowledge about the physiological tolerance ranges and adaptability characteristics of native Western wildland plant species only in the last two decades or so, and thus our ability to select from them on the basis of hard facts is still a weak link in ecological restoration. If we knew as much about the physiology of many native species as we believe we do about some introduced species, perhaps a greater variety of native species would be selected for restoration and related other uses.

Native species have at least one enormous advantage over introduced species that is often overlooked by restorationists and land managers and that provides us with an encouraging advantage in selecting species: native species have a very long history of genetic sorting and natural selection by the local environment. Throughout evolutionary time, ubiquitous adjustments in climate and other conditions have altered environments significantly. Strong sorting forces or genetic editing are continuously refining the genetic programming in plants, and as a result native flora have developed broad ranges of physiological tolerances (Bazzaz 1979). In arid and semiarid regions, natural selection is particularly strong in eliminating many genotypes in response to overwhelming sorting pressures exerted by limiting environmental conditions (Smith 1978). Selection pressure usually results in a highly resilient biome over time in balance with resource availability and biodiversity, and large numbers of relatively low-resource-requiring species can be accommodated in resource-poor environments (Smith and Nowak 1990; Trimble 1989). The ranges of physiological tolerance of each species are thus continuously being stressed as environmental conditions swing from one extreme to another under the influence of climate change, fire, insects and disease cycles, seismic events, volcanism, and other stresses (Nilsen and Orcutt 1996).

Succession under such systems is highly dynamic, with natural selection continually sorting out marginal or unadapted species, populations, and ecotypes (Bazzaz 1979; MacMahon 1983). The overall trajectory of such a system points toward complete resource allocation and eventual stability (although probably never truly achieved) defined within the limits of that particular system. These pressures create a flora and fauna that are resilient to disturbance and yet compatible with available environmental resources. Examples abound, but include such adaptations as root system stratification and above-ground architectural attributes of cover and biomass that are compatible with the availability of water, nutrients, and other resources in balance with soil development, hydrologic pathways, and other natural characteristics of the ecosystem (Bazzaz 1979; Nilsen and Orcutt 1996). Resource extraction patterns by plants are thus in balance with climatic configurations of growing season length, precipitation, dormancy, snowfall and accumulation, temperature extremes, competition, reproduction and flowering of associated species, symbiotic associations, herbivory and predation pressures, fire recurrence patterns, insects and disease cycles, and other variables. This balance is achieved over long-term exposures in an ecosystem (such as adaptability), and is likely not equivalent to those developed by introduced species that originated in a geographically and ecologically distant environment, even though these introduced species may have successfully survived in the new environment for long periods of time in human terms. The introduction of new species into a system can cause imbalance by shifting resource allocation patterns and altering successional trajectories at the cost of diversity, resiliency, recoverability, and flexibility to stress and other forces (Brown 1995; Lesica and DeLuca 1996; Mattson 1996; Mills and others 1994). Although we may have limited knowledge about some aspects of native plant adaptability and the mechanisms that contribute to it, we do know that natural selection has provided a broad spectrum of genetic material in many native species that are very well suited and ecologically fit for the seemingly harshest conditions.

Grazing in the Interior West prior to European settlement was probably largely limited to wildlife, insects, and perhaps native peoples (Kay 1994). It is doubtful that the arid portions of this region ever supported massive herds of grazing animals on a permanent basis and that herbivory was ever a major natural selection or sorting force of plant evolution, especially in comparison with mesic regions like the Great Plains. Native plant species in the Great Basin and throughout the Interior West were more likely sorted by forces of extreme climatic events like water deficits and highly variable climatic swings characteristic of this largely arid and semiarid region. Therefore, native species in the Interior West are unable to withstand repeated and frequent foliage removal. When European peoples began to populate the West in large numbers on a permanent basis, they attempted to apply standards of land use and management that had been learned in more moderate climatic regions. The unfamiliar environmental extremes encountered throughout the West doomed many agrarian attempts to failure (Cottam 1947; Hidy and Klieforth 1990; Smith and Nowak 1990; Stewart 1941; Trimble 1989).

Selecting Adapted Native Species

An essential step in implementing ecological restoration of disturbed wildlands is recognizing the significance of the compatibility required among the biological, physical, and chemical components of an ecosystem with natural succession. Equally important is the realization that although native species have had the benefit of extended periods of natural selection to shape their heritable features of adaptability, not all native species are equally well suited for initiating the early processes of succession. One attribute of a highly variable and rich native flora in Western ecosystems is a broad array of adaptations in many different species for colonizing severe disturbances. Colonizers have physiological requirements and adaptations that allow aggressive invasion and establishment in openings or other niches created by disturbances within all ecosystems. Despite their essential function, many native colonizer species, like some annuals and perennials, are often mistakenly relegated to the lowly regarded role of "weeds" (Kricher 1980), but rarely do early seral natives resemble the common image of pesky annual garden plants. A surprisingly large proportion of the native species found to be aggressive colonizers on disturbances are also major components of mid- to late-successional communities in most Western ecosystems at virtually all elevations, such as arid desert lands as in Wallace and Romney (1980); mesic steppe ecosystems as in Daubenmire (1975) and Smith and others (1988); coniferous forests of *Pseudotsuga* and *Pinus* as described by Arno and others (1985), Franklin and Hemstrom (1981), and Perry (1994); *Thuja-Tsuga* forests of northern Idaho as in Mueggler (1965); alpine and subalpine ecosystems as in Baig (1992), Chambers and others (1984), and Churchill and Hanson (1958). These examples are quite unlike the classical models of succession in which early seral species are described as aggressive and persistent annuals and biennials, especially as developed from prairie and "old-field" sites in the Midwest (Clements 1916). However, where introduced species have escaped and spread in Western ecosystems, early seral plant development may be dominated by highly competitive and aggressive exotic annuals and biennials, which probably deserve the label "weeds" (Monsen 1994).

Typically, early seral native species play an extraordinarily important role in ecosystem development because they are instrumental in initiating the interactive phases of succession among the biological, physical, and chemical components of the environment. In some cases where extreme disturbance exposes toxic geologic material, physical and chemical forces of weathering or intensive human-induced reclamation may be required before active colonization can occur. However, once early seral species colonize and become established on a disturbance, the processes leading toward soil genesis and nutrient cycling are initiated (Glenn-Lewin and van der Maarel 1992; MacMahon 1983; McIntosh 1980, 1981).

The contrasts between early and late seral species is not always absolute, but a summary of their comparative attributes is listed in table 1 in broad terms to indicate their general characteristics. These are only general characteristics of the differences between the two groups of seral species because exceptions can always be found. This illustrates the generally higher aggressiveness and colonizing ability of early seral species over later seral species. The summary also illustrates that early seral species tend to have higher seed productivity, seed germination, seed dispersal, growth rate, and tolerance to stress than late seral species. Although early seral species tend to display broad ecological amplitude and nutrient requirements, they generally have lower competitiveness and root/shoot biomass ratios than later seral species.

The land manager responsible for selecting plant species to be used in restoration efforts can take advantage of a broad range of opportunities. Typically plant species are selected either on the basis of personal experience or after consultation with local county agents or other experienced workers. Another more productive opportunity rarely exercised is the observation of existing active successional areas on disturbances such as may be found along old roads, on road cuts and fills, at abandoned mine or building and construction sites, in areas of heavy rodent activity, along powerline and pipeline corridors, on landslides, in abused campground areas, at burned sites where succession remains in its earlier stages, and at similar disturbed sites. Careful observation of these sites often reveals valuable information about a range of plant species adapted to colonization and survival on disturbed lands in the area of concern. Perhaps more than any other source, succession displays those species that are obviously adapted to local conditions of disturbance. Depending on environmental conditions and the life zone in which the disturbance occurs, a range of seral species will likely be displayed. These may include an assortment of life forms composed of graminoids, broadleaf plants, and woody species.

It is recommended that land managers and restoration specialists identify the seral native species for each of the ecosystems within their area of responsibility, and that the distribution, growth, phenological and other ecological habits of these species be carefully observed and documented. It is these species that should be selected for whatever seed mixture or planting stock is designed for the restoration of disturbed wildlands. Ideally a range of disturbance ages and community types will provide a broader spectrum of seral species to include in the pool of potential available native species to be used for restoration. We recommend that careful notes of species locations, rates of phenological development including flowering habits and seed maturity dates, growth habit (such as bunch versus rhizomatous growth, prostrate versus erect form), and other habits be maintained over time. Not all native species produce high quantities of germinable seed every year, and in some areas seed production may be highly variable with climatic conditions.

Some examples of seral native species found to be useful for restoration of severely disturbed wildlands in the Interior West are listed in table 2. This species list is only intended to be used as an indication of some of the many appropriate seral native species that can be selected; there

Table 1—Some contrasting characteristics between early- and late-seral species.

Characteristics	Early seral	Late seral
Ecological amplitude	Broad	Narrow
Nutrient requirements	Broad	Narrow
Aggressiveness	High	Low
Colonizing ability	High	Low-medium
Seed production	High	Low-medium
Seed germination	High	Low
Seed dispersal	High	Low
Stress tolerance	High	Low
Growth rate	High	Low
Competitiveness	Low-moderate	High
Root/shoot ratio	Low	High
Succession stage	Low	High

Table 2—Examples of some native seral species useful for restoration of severe wildland disturbances by major ecosystem.

Species	Common name
Arid to semiarid (such as desert shrub and pinyon-juniper communities)	
Oryzopsis hymenoides	Indian ricegrass
Sitanion hystrix	Squirreltail bottlebrush
Stipa comata	Needle-and-threadgrass
Penstemon palmeri	Palmer penstemon
Sphaeralcea coccinea	Scarlet globemallow
Artemisia tridentata	Big sagebrush
Chrysothamnus nauseosus	Rubber rabbitbrush
Foothill lower montane	
Agropyron spicatum	Bluebunch wheatgrass
Agropyron trachycaulum	Slender wheatgrass
Bromus marginatus	Mountain brome
Festuca idahoensis	Idaho fescue
Festuca ovina	Sheep fescue
Achillea millefolium	Western yarrow
Artemisia tridentata	Big sagebrush
Chrysothamnus nauseosus	Rubber rabbitbrush
Subalpine to alpine	
Agropyron trachycaulum	Slender wheatgrass
Deschampsia caespitosa	Tufted hairgrass
Phleum alpinum	Alpine timothy
Poa alpina	Alpine bluegrass
Trisetum spicatum	Spike trisetum
Sibbaldia procumbens	Sibbaldia

are many more not listed here (Thornburg 1982 for more complete species lists by ecosystem).

Although many of these species are becoming available by commercial native plant seed dealers and collectors, some species may only be available in limited quantities in any given year (see Dunne, this publication) and may require the land manager to collect seeds of key species locally. Despite concerns about cost, available time, or personnel shortages, we strongly urge land managers and restoration specialists to consider collecting seral native plant seeds in their local areas to build up a supply of appropriate species, even during years when no restoration is planned. It is not uncommon for unusually abundant seed production to occur in years when no restoration or revegetation activity is implemented, but it is recommended that land managers take advantage of these periodic high seed production years by collecting and storing the seed properly until needed (see Shaw and Roundy 1997 and Young and Young 1986 for guidelines). Although in research we collect seeds of seral species either by hand or with seed collecting equipment to acquire small research plot seed mixtures, land managers may find it more cost effective to contract with native seed suppliers to collect, clean, and store larger quantities of seed until restoration is implemented. In addition, perhaps land users such as mining companies, ski area developers, and grazing permittees may be required to pay for developing sources of native plant seeds and appropriate storage facilities.

We recommend that each local office maintain a seed herbarium consisting of small samples of native seed of different important species. These samples should be stored individually in small labeled glass vials representative of the district to be used as a reference and for verifying seed composition of mixtures supplied by contractors. Also, land management agencies should streamline and standardize the permitting process for native seed collection on public lands to facilitate collection by private contractors. This will ease time constraints for obtaining adequate native seed. Although it may be inappropriate for land managers to conduct detailed observations and studies of germination requirements, physiological tolerances, successional processes, and soil chemical and physical properties, research personnel can assist in this effort and enhance understanding of the important interactions among the physical, chemical, and biological components of a particular ecosystem. Methods of collecting, cleaning, and storing seeds are discussed elsewhere (Young and Young 1986).

A pervasive concern regards the issue of how far from the disturbed site can plant seeds be collected to ensure they are still adapted and representative of the local population of that species. Although there are no definitive guidelines to ensure adaptability (distances in feet or meters are not sensitive measures), seed should be collected from those species observed as colonizers and found growing on disturbances within the same relative environmental zone as the disturbance to be treated. We find that seed collection is often more convenient on disturbances where seral species frequently occur in highest densities. However, we have also successfully collected seeds of seral species from adjacent undisturbed communities. Slope, aspect, and elevation are quantitative guides easy to follow, but are not necessarily indicative of adaptability. Similar species composition between adjacent undisturbed communities and disturbed areas serves as an excellent indicator of environmental congruity, and is probably the best guide for assuring that the collected seed is from an adapted population of the species (see Currans and others 1997 and Young and Young 1986 for additional guidelines).

Implementing Ecological Restoration on Severe Disturbances

Ecological restoration is viewed here as the process of "operative intervention" by land managers to intentionally interfere with the disturbed state in order to initiate, enhance, and accelerate succession. Restoration is envisioned in the context of an "active" process to re-establish a self-sustaining community of interdependent chemical, physical, and biological components on severely disturbed wildlands where natural succession has been deflected, suspended, or terminated. Active intervention may be required when the soil is exposed to erosion, or when the soil is lost, and alterations have occurred to the natural surface and subsurface hydrologic pathways on a local or watershed scale. In addition, when these forces impact water quality, intervention may become critical. To re-establish natural succession, and to ensure that it be consistent and congruent with successional processes of the surrounding ecosystem, we contend it is essential that indigenous native seral plant species be selected that are evolutionary products of the full range and extent of natural selection and other forces affecting genetic sorting in that environment. It is more likely that indigenous native seral species will be genetically and physiologically better suited to survive, grow, and reproduce under the environmental extremes and fluctuations of the local area than any introduced or exotic vascular plants that originated in other environments.

More passive approaches to restoration that rely on nonaggressive and nonoperative approaches, such as excluding grazing or recreational activity, are largely ineffective for restoring severe disturbances within acceptable timeframes. Active erosion, transport of toxic-laden sediments, and surface and subsurface hydrologic deterioration on such sites likely have already arrested succession and resulted in degraded water quality and other intrinsic resource values. Unless an active operative approach of intervention is adopted, whereby land managers take physical steps to reinitiate succession, deterioration of site characteristics will likely continue. In fact, in most cases of severe disturbance restoration, a combination of active and passive management strategies are required to achieve ecological restoration.

The undisturbed state is regarded here as a relative reference condition, recognizing that forces such as climate change, industrial and social pollution, and human interference in environmental conditions may have altered the overall forces responsible for creating the original community before it was disturbed. Like the original undisturbed community, the restored community would be considered to be self-sustaining, yet dynamically seral in response to successional forces, when it requires no continued long-term inputs of resources by land management to support soil genesis and nutrient cycling. Also, the restored community will minimize erosion and sediment transport commensurate with the natural ecosystem, and will ultimately lead to the re-establishment of natural surface and subsurface hydrologic pathways.

Limiting site conditions on disturbances can be ameliorated using various reclamation practices, including saving natural soil where possible or replacing the natural soil if soil is available. On many wildland disturbances, however, the natural soil has been lost and replacement soil is not available. In these cases, a complete soil analysis is justified to characterize the chemical and physical properties that may be limiting to plant establishment and growth. Various cultural amendments may be required, including:

1. Liming to adjust acidic pH upward.
2. Incorporating organic matter to improve nutrient and water-holding characteristics and to prevent contamination by heavy metals or other toxic chemicals.
3. Nutrient enhancement with various fertilizers to provide essential nutrients for plant growth.

Although some land managers may be hesitant to apply chemical fertilizers to wildland disturbances for fear of contamination of nearby waters or because of other concerns, we have found that nutrient enhancement is essential for plant emergence and early growth following germination, especially on barren inert or sterile geologic materials such as occur on mine spoils or road cuts and fills. Leaching and surface movement of fertilizers is retarded on such areas if organic matter is incorporated to assist in retention of the nutrients, and if normal precautions of application rates are followed.

Seeding of adapted native seral species collected in adjacent areas should be performed in the fall of the year to mimic natural phenological phases of seed dispersal. In most native Western environments, seed of native plants is dispersed in the late summer or fall and lays in place over the winter in or on the soil. In the spring following snowmelt and with the onset of optimum temperature and soil water conditions, germination is able to proceed at optimum rates. If seed is not distributed until spring, access to the site may be hampered or prevented due to unpredictable spring weather and other access problems, which will prevent the seed from being in place when optimum conditions occur.

It is recommended that high seeding rates be used, especially on harsh sites where conditions are particularly limiting to assure adequate seedling densities to minimize surface erosion and sediment movement. Seedling mortality tends to be very high in most Western wildland environments, and care should be taken to compensate. On severe disturbances, we usually recommend that seeding rates be adjusted to provide about 250 to 350 PLS seeds per ft^2 (2,700 to 3,800 seeds per m^2). Although these figures are higher than that recommended for normal rangeland reseedings, this seeding rate compensates for high seedling mortality and poor seed viability and germinability on severe disturbances. Mortality due to competition that might result from heavy seeding rates will compensate for mortality due to limiting environmental factors, and will provide site cover and protection at the optimum level that the environment can support. Alternatively, underseeding to avoid competition does nothing to minimize environmental mortality, and opens up bare soil to potential erosion. We also recommend that seeds be applied on the basis of "number of seeds per unit weight per unit area" rather than on the basis of a "unit weight per unit area" alone. This provides a quantitative means of assessing plant establishment by species in followup assessments and tends to equalize competition among all the species of a seed mixture.

Following seeding, a surface mulch should be applied to: (1) aid in retarding evaporative loss of soil water; (2) minimize wind redistribution of seed, fine soil particles, and other amendments; (3) moderate soil or spoil surface temperature extremes; (4) minimize chances of erosion; and (5) maximize seed trapping of native species from surrounding plant communities. Surface mulching can be applied with straw or hay, straw tacked down with commercial tackifying agents, straw crimped into the soil, or with commercial erosion blanket netting materials. We strongly favor erosion blanket materials, especially those constructed with cotton or other natural biodegradable twines and netting, due to their ease of application and efficiency in achieving the goals of surface mulching.

We recommend refertilization of restoration plant communities with N-P-K (nitrogen-phosphorus-potassium) granular fertilizers each year for three to five growing seasons following seed application to enhance growth and rooting activity and to hasten the initiation of nutrient cycling. Repeat fertilization favors the establishment of graminoid species (grasses and sedges) to the near exclusion of forbs and other life forms, but does build up a nutrient base in the rooting zone of the new community that also accelerates the re-establishment of nutrient cycling. Following the termination of repeat applications of fertilizer, graminoid dominance declines and gradually opens niches for invasion and colonization by other species and more diverse life forms. We have observed rapid increases in species and life-form richness following the termination of refertilization schedules, characterized by dramatic increases in species and life-form numbers during subsequent growing seasons. At this stage of community development, continued repeat applications of fertilizer are not warranted or required. In short, we advocate that succession be "pushed" to create an environment where nutrient cycling can sustain these early seral communities. In this manner, an attempt is made to harness succession as a restoration tool that also guides community development.

It is unreasonable to expect that restoration will ever lead to the absolute and exact replacement of each species and other components of the original community or habitat that existed prior to disturbance. This is an unnecessary and perhaps undesirable expectation from a land management perspective because: (1) there are rarely data available to guide such an objective, and (2) there are no guarantees that the current climate and other environmental factors are consistent with conditions as they may have existed when the original community was formed. Therefore, it seems more realistic to accept current successional trajectories, which are guided by prevailing environmental conditions, and to achieve a natural self-sustaining community appropriate with the overall form and function of the total ecosystem as it currently exists. Frequently, species composition, abundance, and diversity in the "restored" community vary, sometimes significantly, from that which was perceived to have characterized the original community before disturbance. This is less concerning than is the capability of the new system to function biotically and abiotically in equilibrium with the contemporary overall environment and ecosystem of the area. With the initiation of natural succession in "new" or "restored" communities or community systems, the ultimate commencement of such seral milestones as nutrient cycling and soil genesis will occur under the influence of current vegetation and climatic forces, with the end products being entirely compatible with the overall existing natural system.

The Challenge

One of our objectives in this paper is to challenge land managers to face the issue of restoring severe disturbances in wildland areas with ecologically appropriate techniques and methods, and to act on it. It is now clear that one of the most appropriate policies that should be adopted and implemented universally is to require the use of locally adapted seral native species for restoring disturbed natural communities. We recognize the many overused and cadaverous excuses for why this may be difficult to implement, but find continued reliance on them unconvincing. There are some highly significant concerns about the continued use of introduced species on wildlands that extend well beyond economics, commercial availability, livestock palatability, vigor, and other apparent attributes. These center around ecosystem health, sustainable natural resources, biodiversity, and resiliency to disturbance. In short, these concerns involve the very principles of ecosystem management.

No administrative entity has greater opportunity, means, or incentives to favor and influence the use of native species than public land management agencies. Certainly no other organizations have comparable networks of motivated and trained personnel and facilities located in such intimate contact with so wide a diversity of ecosystems and habitats as do agencies such as the Forest Service, Bureau of Land Management, National Park Service, various State and local agencies, and private organizations. In view of the commitment made by such agencies to ecosystem management, an excellent opportunity is waiting to influence the direction of future management of disturbed wildlands. By adopting an aggressive policy of requiring the production and use of native plant species and of restoring native communities, these agencies can have a tremendously positive influence on how disturbed lands are treated and restored by other government agencies, private individuals and businesses, and perhaps foreign governments.

The challenge is straightforward, simple, and consistent with ecosystem management policy: restore severely disturbed lands only with native species that originated in the same ecosystem in which the disturbance occurs. Managers should encourage their personnel to work cooperatively among themselves and with others to identify, collect, and store the various successional native species found colonizing disturbances, or contract with private firms to do so. They could encourage research to provide indepth scientific knowledge and information about native species to ensure that disturbed lands are restored to the sustainable and ecologically diverse systems they once were. With recent commitments to ecosystem management by most land managers and with the realization that concerns over biodiversity and sustainability stem in significant measure from the prevalence of introduced and exotic species now occurring within native ecosystems (due in part at least to aberrant historical revegetation practices and other ill-advised management policies), shifts in land management philosophies can finally begin favoring a more ecologically based approach to the restoration of ecosystem health.

Conclusions

It is our thesis throughout this paper that severely disturbed wildlands can be restored to a condition commensurate with the overall ecosystem of the area, and that this can be achieved by using adapted local native seral species. We advocate this position because the scientific evidence supports the hypothesis that native seral species are ecologically and physiologically adapted and acclimated to the local vagaries of climate and other environmental conditions by virtue of local natural selection over much longer time periods than species that evolved in other environments. It is clear that a major consequence of introducing exotic plant species into Western ecosystems has been a significant loss in ecosystem diversity, resiliency, and recoverability in the face of accelerating human demands for natural resources. Historically, revegetation and reclamation of disturbances by public land managers have overwhelmingly relied upon the use of non-native plant species, and have been influenced by the perception that such practices can "improve" natural conditions. This practice has been justified on the basis that any form of vegetative cover is better than none at all, and that by using introduced species to minimize erosion and stabilize disturbed sites, "nature will take its course" and return the site to some desirable state. In view of overwhelming scientific evidence, a considerably juvenile "leap of faith" is required to believe that using introduced species will accomplish erosion control, stabilize sediment, improve water quality, provide plant biomass and livestock palatability, and achieve ecosystem restoration. Not only is this position naive and misleading, it is almost certainly wrong. The evidence is now abundant that there are no guarantees that nature will take a desirable course following the use of introduced species, or that succession will progress in expected trajectories following their establishment.

Not all objections to using introduced (or unadapted) plant species are entirely justified. For example, following large devastating wildfires, enormous quantities of inexpensive and readily available seed of rapidly growing species are often required to at least temporarily stem the impacts of accelerated erosion. Given the immediacy of such problems, especially in highly populated areas, the relative unavailability and cost of most native seed in the large quantities required for such applications almost dictate that introduced species will likely continue to be used. Until recently, native species were virtually unavailable commercially, and even though many species are now readily available, their unit cost per unit weight tends to be more variable and much higher than that of most introduced species. When balanced against the goals of ecosystem health, ecological fitness, and public demands to protect sustainable wildlands, and with continued and increasing requirements that native seed be used, we are optimistic that near-parity will eventually be reached in the availability and cost of native and introduced species.

We also advocate the position that natural succession is the driving force of restoration, either human-implemented or natural, and that it should be used as a guide in designing restoration activities on disturbed public lands. Further, we suggest that land managers learn to use succession as a tool that can be "pushed" or accelerated once it has been initiated on severe disturbances through the use of such practices as revegetation and reclamation. Part of the process of implementing ecosystem management by public agencies must be upgraded to instilling in land managers an understanding of how ecosystems function under conditions of stability and under the influence of disturbance. Disturbance either by natural or human causes is a universal state in natural ecosystems. The processes of succession and ecosystem formation are foundation blocks in how these systems function, and they must be used as guides in making management decisions.

Native successional colonizer species on disturbances often are components of mid- or late-successional communities in Western ecosystems. Therefore, the most appropriate species for use in restoration are very often the same species commonly identified as the more desirable individuals for erosion control, grazing, wildlife habitat development, and other uses. Seral native species play an extraordinarily important role in ecosystem development because they are instrumental in initiating succession and all the interactive forces among the biological, physical, and chemical components of the environment that lead to ecosystem formation. Once these species colonize and become established on a disturbance, the processes of soil genesis and nutrient cycling are initiated.

As envisioned here, ecological restoration is the fairly straightforward operative and active approach to reinitiating natural succession. It is our position that active restoration is the most ecologically efficient and sound means of reestablishing natural communities that are structurally and functionally similar to the surrounding natural "undisturbed" ecosystem, especially on severe disturbances where succession has been arrested due to limiting conditions. The process of active restoration is best achieved through the direct application of various cultural treatments to ameliorate limiting site conditions, and may include revegetation and reclamation to facilitate invasion, colonization, and establishment of plants on severely disturbed sites. However, the integral upgrading step that differentiates ecological restoration from traditional revegetation and reclamation is the added caveat of utilizing natural succession as a guide, and of selecting adapted native seral species instead of more traditional introduced species. Obviously, this additional requirement involves the use of techniques and procedures that are ecologically compatible with the physiological requirements of the selected native species (such as seed collection timing and methods, seed cleaning and storage, seeding and planting methods, seedbed preparation).

Land managers can definitively set the standard for how disturbed natural ecosystems and communities can be restored to a self-sustaining, diverse, resilient condition. It is already known that healthy natural wildlands provide balance and stability, and are resilient to the long-term managed use of sustainable natural resources. In the face of accelerating public demands, ecological restoration of severe disturbances is an unavoidable necessity. No private or public entities have such unrestricted access to, and knowledge of, natural resources and the components of natural ecosystems as do public land management agencies. We urge land managers to advocate or require the use of native adapted seral plant species for revegetation, reclamation, and restoration of disturbed public lands. We suggest that stronger ties with natural resources research scientists to help identify, select, and collect seed banks of native plants suited for restoring disturbed habitats and communities throughout the West will help advance this effort. Collectively, land managers could have the commanding privilege of finally settling the matter of restoring disturbed wildlands to a near-natural state in virtually every Western ecosystem. Central to this concept is recognition of the role of natural succession as the primary driving force of ecological restoration, not only of biological components, but also of soil genesis, nutrient cycling, hydrologic patterns, and all other biotic and abiotic components of functioning ecosystems. This represents a major departure from traditional attitudes of restoration and reclamation in land management, and its significance is being substantiated and advanced by basic and applied research within some of the very agencies in which change is slowly, but finally, occurring.

Acknowledgments

The authors express deep gratitude for review comments and other helpful suggestions made by Dr. Dale L. Bartos (Research Ecologist), Ms. Lori Bentley (Technician), Dr. Jesse A. Logan (Project Leader and Research Entomologist), Dr. Walter F. Mueggler (Scientist Emeritis), and Mr. Nick Zufelt (R-4 Regional Reclamation Specialist).

References

Allen, E. B., ed. 1988. The reconstruction of disturbed arid lands: an ecological approach. Boulder, CO: Westview Press. 261 p.

Anderson, E. 1971. Man as a maker of new plants and plant communities. In: Detwyler, T. R. Man's impact on environment. New York: McGraw Hill: 653-666.

Arno, S. F.; Simmerman, D. G.; Keane, R. E. 1985. Forest succession on four habitat types in western Montana. Gen. Tech. Rep. INT-177. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 74 p.

Baig, M. N. 1992. Natural revegetation of coal mine spoils in the Rocky Mountains of Alberta and its significance for species selection in land restoration. Mountain Research and Development. 12(3): 285-300.

Bartos, D. L.; Campbell, R. B. 1998. Decline of quaking aspen in the Interior West—examples from Utah. Rangelands. 20(1): 17-24.

Bazzaz, F. A. 1979. The physiological ecology of plant succession. Annual Review of Ecology and Systematics. 10: 351-371.

Bengson, S. A. 1986. "Native" vs. "Exotic"- the dilemma of ecological mine waste revegetation. Rangelands. 8(2): 65-67.

Benson, L. 1957. Plant classification. Boston: D. C. Heath and Co. 688 p.

Billings, W. D. 1965. Plants and the ecosystem. Belmont, CA: Wadsworth Publishing Co, Inc. 154 p.

Boyce, M. S.; Haney, A., eds. 1997. Ecosystem management: applications for sustainable forest and wildlife resources. New Haven: Yale University Press. 361 p.

Bradshaw, A. D. 1995. Alternative endpoints for reclamation. In: Cairns, J., ed. Rehabilitating damaged ecosystems. Boca Raton, FL: Lewis Publishers: 165-185.

Briggs, M. K. 1996. Riparian ecosystem recovery in arid lands: strategies and references. Tucson: University of Arizona Press. 159 p.

Brown, D. 1995. The impact of species introduced to control invasion on the vegetation of an electrical utility right-of-way. Canadian Journal of Botany. 73: 1217-1228.

Brown, R. W. 1980. Revegetation with native and introduced plant species. In: Twenty-sixth grass breeders work planning conference. June 18-20, 1980. Logan: Utah State University: 9-16.

Brown, R. W.; Johnston, R. S.; Johnson, D. A. 1979. Revegetation of disturbed alpine rangelands. In: Johnson, D. A., ed. Special management needs of alpine ecosystems. Range Science Series No. 5. Denver: Society for Range Management: 76-94.

Cairns, J., ed. 1995. Rehabilitating damaged ecosystems. Boca Raton, FL: Lewis Publishers.

Chambers, J. C.; Brown, R. W.; Johnston, R. S. 1984. Examination of plant successional stages in disturbed alpine ecosystems: a method of selecting revegetation species. In: Colbert, T. A.; Cuany, R. L., eds. Proceedings high altitude revegetation workshop No. 6. Information Series No. 53. Fort Collins: Colorado State University, Water Resources Research Institute: 215-224.

Churchill, E. D.; Hanson, H. C. 1958. The concept of climax in arctic and alpine vegetation. Botanical Review. 24: 127-191.

Clements, F. E. 1916. Plant succession: an analysis of the development of vegetation. Publ. 242. Washington, DC: Carnegie Institution of Washington. 512 p.

Conrad, M. 1983. Adaptability: the significance of variability from molecule to ecosystem. New York: Plenum Press. 383 p.

Cottam, W. P. 1947. Is Utah Sahara bound? Eleventh annual Fredrick William Reynolds lecture. Salt Lake City: University of Utah. 42 p.

Covington, W. W.; Moore, M. M. 1994. Southwestern ponderosa forest structure: changes since Euro-American settlement. Journal of Forestry. 92: 39-47.

Currans, S. P.; Kitchen, S. G.; Lambert, S. M. 1997. Ensuring identity and quality of native seeds. In: Shaw, N.; Roundy, B. A., eds. Proceedings: using seeds of native species on rangelands. Gen. Tech. Rep. INT-GTR-372. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 17-20.

Curtin, C. G. 1995. Can montane landscapes recover from human disturbance? Long-term evidence from disturbed subalpine communities. Biological Conservation. 74: 49-55.

Daubenmire, R. 1975. Plant succession on abandoned fields, and fire influences, in a steppe area in southeastern Washington. Northwest Science. 49: 36-48.

del Moral, R.; Wood, D. M. 1993. Early succession on barren volcanic plain at Mount St. Helens, Washington. American Journal of Botany. 80(9): 981-991.

Detwyler, T. R. 1971. Man's impact on environment. New York: McGraw Hill. 731 p.

Dombeck, M. P. 1998. Memorandum: Forest Service natural resource agenda. March 2, 1998. On file at: Washington, DC: U.S. Department of Agriculture, Forest Service.

Ellis, J.; Galvin, K. A. 1994. Climate patterns and land-use practices in the fry zones of Africa. BioScience. 44: 340-349.

Everett, R. L., comp. 1994. Volume IV: Restoration of stressed sites, and processes: Eastside forest ecosystem assessment. Gen. Tech. Rep. PNW-GTR-330. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 123 p.

Franklin, J. F.; Hemstrom, M. A. 1981. Aspects of succession in coniferous forests of the Pacific Northwest. In: West, D. C.; Shugart, H. H.; Botkin, D. B., eds. Forest succession: concepts and application. New York: Springer-Verlag: 212-245.

Glenn-Lewin, D. C.; van der Maarel, E. 1992. Patterns and processes of vegetation dynamics. In: Glenn-Lewin, D. C.; Peet, R. K.; Veblen, T. T., eds. Plant succession: theory and prediction. London: Chapman and Hall: 11-59

Glenn-Lewin, D. C.; Peet, R. K.; Veblen, T. T., eds. 1992. Plant succession: theory and prediction. London: Chapman and Hall. 352 p.

Gutknecht, K. W. 1992. Native vs introduced species: the new range war. Utah Science. 53(3): 69-78.

Harper, J. L. 1977. Population biology of plants. New York: Academic Press. 892 p.

Hess, K., Jr. 1995. A tale of the exotic. Utah State University Magazine. 1(3): 26-31.

Hidy, G. M.; Klieforth, H. E. 1990. Atmospheric processes affecting the climate of the Great Basin. In: Osmond, C. B.; Pitelka, L. F.; Hidy, G. M., eds. Plant biology of the basin and range. Berlin: Springer-Verlag: 17-45.

Hull, A. C.; Holmgren, R. C. 1964. Seeding southern Idaho rangelands. Res. Pap. INT-10. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 31 p.

Jackson, L. L.; Lopoukhine, N.; Hillyard, D. 1995. Commentary - ecological restoration: a definition and comments. Restoration Ecology. 3(2): 71-75.

Johnson, K. L. 1986. Crested wheatgrass: its values, problems and myths; symposium proceedings. 1983 October; Logan, UT: Utah State University. 348 p.

Kaufmann, M. R.; Graham, R. T.; Boyce, D. A., Jr.; Moir, W. H.; Perry, L.; Reynolds, R. T.; Bassett, R. L.; Mehlhop, P.; Edminster, C. B.; Block, W. M.; Corn, P. S. 1994. An ecological basis for ecosystem management. Gen. Tech. Rep. RM-246. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 22 p.

Kay, C. E. 1994. Aboriginal overkill: the role of native Americans in structuring Western ecosystems. Human Nature. 5(4): 359-398.

Kohm, K. A.; Franklin, J. F. 1997. Creating a forestry for the 21st Century: The science of ecosystem management. Washington, DC: Island Press. 475 p.

Kricher, J. C. 1980. Needs of weeds. Natural History. 89(12): 36-45.

Lesica, P.; DeLuca, T. H. 1996. Long-term harmful effects of crested wheatgrass on Great Plains grassland ecosystems. Journal of Soil and Water Conservation. 51: 408-409.

MacMahon, J. A. 1983. Nothing succeeds like succession: ecology and the human lot. 67th Faculty Honor Lecture. Logan: Utah State University Press. Logan. 31 p.

MacMahon, J. A.; Jordan, W. 1994. Ecological restoration. In: Meffe, G. K.; Carroll, C. R. Principles of conservation biology. Sunderland, MA: Sinauer Associates, Inc.: 409-438.

Mattson, W. J. 1996. Escalating anthropogenic stresses on forest ecosystems: forcing benign plant-insect interactions into new interaction trajectories. In: Korpilahti, E.; Mikkela, H.; Salonen, T., eds. Caring for the forest: research in a changing world. Congress Rep. Vol 2. Finn. IUFRO World Congress Organizing Committee: 338-342.

McIntosh, R. P. 1980. The relationship between succession and the recovery process in ecosystems. In: Cairns, J., Jr., ed. The recovery process in damaged ecosystems. Ann Arbor, MI: Ann Arbor Science Publishers: 11-62.

McIntosh, R. P. 1981. Succession and ecological theory. In: West, D. C.; Shugart, H. H.; Botkin, D. B., eds. Forest succession: concepts and application. New York: Springer-Verlag: 10-23.

Meeuwig, R. O. 1960. Watersheds A and B—a study of surface runoff and erosion in the subalpine zone of central Utah. Journal of Forestry. 58: 556-560.

Mills, E. L.; Leach, J. H.; Carlton, J. T.; Secor, C. L. 1994. Exotic species and the integrity of the Great Lakes. BioScience. 44(10): 666-676.

Monsen, S. B. 1994. The competitive influences of cheatgrass (*Bromus tectorum*) on site restoration. In: Monsen, S. B.; Kitchen, S. G., eds. Proceedings—ecology and management of annual rangelands. Gen. Tech. Rep. INT-GTR-313. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 43-50.

Monsen, S. B.; McArthur, E. D. 1995. Implications of early Intermountain range and watershed restoration practices. In: Roundy, B. A.; McArthur, E. D.; Haley, J. S.; Mann, D. K. Proceedings: wildland shrub and arid land restoration symposium; 1993 October 19-21; Las Vegas, NV. Gen. Tech. Rep. INT-GTR-315. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station:16-25.

Mueggler, W. F. 1965. Ecology of seral shrub communities in the cedar-hemlock zone of northern Idaho. Ecological Monographs. 35: 165-185.

Munshower, F. F. 1993. Practical handbook of disturbed land revegetation. Boca Raton, FL: Lewis Publishers. 265 p.

Nilsen, E. T.; Orcutt, D. M. 1996. The physiology of plants under stress: abiotic factors. New York: John Wiley and Sons, Inc. 689 p.

Perry, D. A. 1994. Forest ecosystems. Baltimore, MD: The Johns Hopkins University Press. 649 p.

Regier, H. A. 1993. The notion of natural and cultural integrity. In: Woodley, S.; Kay, J.; Francis, G., eds. Ecological integrity and the management of ecosystems. St. Lucie Press: 3-18.

Richardson, B. Z.; Farmer, E. E. 1985. Revegetation of phosphate mined lands in the Intermountain West. In: Robertson, D. J., ed. Reclamation and the phosphate industry: proceedings of the symposium; 1983 January 26-28; Clearwater Beach, FL. Florida Institute of Phosphate Research: 374-389.

Rogers, P. 1996. Disturbance ecology and forest management: a review of the literature. Gen. Tech. Rep. INT-GTR-336. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 16 p.

Roundy, B. A., Shaw, N. L.; Booth, D. T. 1997. Using native seeds on rangelands. In: Shaw, N. L. and Roundy, B. A., eds. Proceedings: Using Seeds of Native Species on Rangelands. Gen. Tech. Rep. INT-GTR-372. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 1-8.

Schramm, J. R. 1966. Plant colonization studies on black wastes from anthracite mining in Pennsylvania. Philadelphia: Transactions of the American Philosophical Society. 56 Part 1: 1-194.

Seagrave, S. 1976. Scientists learn from wild plants. BioScience. 26: 153-155.

Shaw, N. L.; Roundy, B. A., eds. 1997. Proceedings: using seeds of native species on rangelands. Gen. Tech. Rep. INT-GTR-372. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 30 p.

Sheley, R. L.; Manoukian, M.; Marks, G. 1996. Preventing noxious weed invasion. Rangelands. 18: 100-101.

Society for Range Management. 1974. A glossary of terms used in range management. Denver, CO: Society for Range Management. 36 p.

Smith, J. M. 1978. Optimization theory in evolution. Annual Review of Ecology and Systematics. 9: 31-56.

Smith, P. W.; DePuit, E. J.; Richardson, B. Z. 1988. Plant community development on petroleum drill sites in northwestern Wyoming. Journal of Range Management. 41(5): 372-377.

Smith, S. D.; Nowak, R. S. 1990. Ecophysiology of plants in the Intermountain lowlands. In: Osmond, C. B.; Pitelka, L. F.; Hidy, G. M., eds. Plant biology of the basin and range. Berlin: Springer-Verlag: 179-241.

Stebbins, G. L. 1974. Flowering plants: evolution above the species level. Cambridge, MD: Harvard University Press. 399 p.

Stebbins, G. L. 1979. Fifty years of plant evolution. In: Solbrig, O. T.; Jain, S.; Johnson, G. B.; Raven, P. H., eds. Topics in plant population biology. New York: Columbia University Press: 18-41.

Stewart, G. 1941. Historic records bearing on agricultural and grazing ecology in Utah. Journal of Forestry. 39(4): 362-375.

Thomas, J. W. 1997. Foreward. In: Kohm, K. A.; Franklin, J. F. Creating a forestry for the 21st Century: the science of ecosystem management. Washington, DC: Island Press. 475 p.

Thornburg, A. A. 1982. Plant materials for use on surface-mined lands in arid and semiarid regions. SCS-TP-157. U.S. Department of Agriculture, Soil Conservation Service. 88 p.

Thornburg, A. A.; Fuchs, S. H. 1978. Plant materials and requirements for growth in dry regions. In: Schaller, F. W.; Sutton, P., eds. Reclamation of drastically disturbed lands. Madison, WI: American Society of Agronomy: 411-423.

Trimble, S. 1989. The sagebrush ocean: a natural history of the Great Basin. Reno: University of Nevada Press. 248 p.

Urbanska, M.; Webb, N. R.; Edwards, P. J., eds. 1997. Restoration ecology and sustainable development. Cambridge, MA: Cambridge University Press. 397 p.

Vallentine, J. F. 1989. Range development and improvement. San Diego, CA: Academic Press. 524 p.

Vogl, R. J. 1980. The ecological factors that produce perturbation-dependent ecosystems. In: Cairns, J., Jr., ed. The recovery process in damaged ecosystems. Ann Arbor, MI: Ann Arbor Science Publishers Inc.: 63-94.

Wallace, A.; Romney, E. M. 1980. The role of pioneer species in revegetation of disturbed desert areas. Great Basin Naturalist Memoirs No. 4. Provo, UT: Brigham Young University: 31-32.

West, N. E.; Stark, J. M.; Johnson, D. W.; Abrams, M. M.; Wight, J. R.; Heggem, D.; Peck, S. 1994. Effects of climatic change on the edaphic features of arid and semiarid lands of Western North America. Arid Soil Research and Rehabilitation. 8: 307-351.

White, P. S. 1979. Pattern, process, and natural disturbance in vegetation. Botanical Review. 45(3): 229-299.

Whittaker, R. H. 1975. Communities and ecosystems. New York: Macmillan Publishing Co., Inc. 385 p.

Williams, J. E.; Wood, C. A.; Dombeck, M. P., eds. 1997. Watershed restoration: principles and practices. Bethesda, MD: American Fisheries Society. 561 p.

Winterhalder, K. 1995. Natural recovery of vascular plant communities on the industrial barrens of the Sudbury area. In: Gunn, J. M., ed. Restoration and recovery of an industrial region: progress in restoring the smelter-damaged landscape near Sudbury, Canada. New York: Springer-Verlag: 93-102.

Woodley, S.; Kay, J.; Francis, G., eds. 1993. Ecological integrity and the management of ecosystems. St. Lucie Press. 220 p.

Yaffee, S. L.; Phillips, A. F.; Frentz, I. C.; Hardy, P. W.; Maleki, S. M.; Thorpe, B. E. 1996. Ecosystem management in the United States: an assessment of current experience. Washington, DC: Island Press. 352 p.

Young, J. A.; Evans, R. A. 1976. Responses of weed populations to human manipulations of the natural environment. Weed Science. 24: 186-190.

Young, J. A.; Young, C. G. 1986. Collecting, processing, and germinating seeds of wildland plants. Portland, OR: Timber Press. 236 p.

Zufelt, N. 1998. [Personal communication]. Regional Reclamation Specialist. Ogden, UT: U.S. Department of Agriculture, Forest Service, Region 4.