Plant Succession and Approaches to Community Restoration

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Abstract—The processes of vegetation change over time, or plant succession, are also the processes involved in plant community restoration. Restoration efforts attempt to use designed disturbance, seedbed preparation and sowing methods, and selection of adapted and compatible native plant materials to enhance ecological function. The large scale of wildfires and weed invasion requires large-scale approaches to restoration. Practices and equipment from traditional rangeland revegetation are being adapted to establish diverse, native communities. The challenge is to meet the establishment requirements of different species and to create weed-resistant plant communities.

Introduction

In the past, range scientists developed range improvement techniques directed mainly at controlling unpalatable woody species and establishing forage grasses for livestock and erosion control, but also to establish plants critical for big game habitat (Roundy 1996). Our goals now are to restore functional, diverse native plant communities. The processes of restoration are the same processes that operate in plant succession. Understanding these processes can help us develop realistic techniques and goals for large-scale restoration. I will briefly review concepts and processes of plant succession and discuss associated aspects of community restoration.

Models of Succession

Plant succession is the change in vegetation that occurs over time after fire, heavy grazing, flooding, or other natural or human-related disturbances. Secondary succession occurs when the land retains some residual soil and biological components from the plant and animal communities that existed before the disturbance (Barbour and others 1998). Primary succession occurs on new substrates, such as on a newly formed volcanic island. Two major views of this process were taught by Clements (1916) and Gleason (1926). In Clement's model, vegetation changes from pioneer species through a series of predictable communities or seres, which replace each other in order until a final stable or climax community dominates the site. This model is said to be linear (always follows the same order) and deterministic (is predictable). On the other hand, Gleason suggested that vegetation change after disturbance was a function of the kinds of plants involved and their characteristics relative to the disturbance. More recently, ecologists have recognized that features of both models may describe what actually happens. State and transition models that allow for multiple steady states of vegetation, with different probabilities of transition or change from one state to another (Westoby and others 1989), have been proposed. These models work better with the current recognition that some disturbances, such as fire, have a natural frequency and play a major role in shaping many upland plant communities. Similarly, seasonal flooding shapes riparian communities (Middleton 1999).

Clements (1928) identified the processes of succession as nudation (disturbance), migration (movement of new seeds or other plant propagules to the site), ecesis (plant establishment), interaction (sorting out of species that establish), reaction (the effects of the successful species on the environment), and stability. These processes correspond to revegetation and restoration principles and practices (table 1). Although all of these successional processes may be active in most systems, some are more controlling for some systems

Table 1—Plant successional processes that correspond to restoration and revegetation principles and practices.

Process	Principle or practice
Disturbance	Site potential after disturbance, designed disturbance to control undesirable plants
Dispersal, migration, residuals	Sowing sufficient germinable seed, preempting resources from or controlling residual propagules of undesirable species, renovation to maintain or stimulate residual desirable species
Establishment	Seedbed preparation and sowing to maximize germination and seedling establishment; selecting adapted plant materials
Interaction/reaction	Selecting ecologically functional, compatible plant materials for mixed communities that are weed resistant
Stabilization	Restoring disturbance regimes and management strategies that favor ecological function

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In: Shaw, Nancy L.; Pellant, Mike; Monsen, Stephen B., comps. 2005. Sage-grouse habitat restoration symposium proceedings; 2001 June 4–7; Boise, ID. Proceedings RMRS-P-38. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

and biomes, while others are more controlling for other biomes (Chambers and others 1992). For example, historically, fire is a major controlling disturbance in grasslands, but not arid deserts, where drought is most operative. Interaction (especially competition) and reaction are particularly operative in forest systems, but not as operative in tundra or desert systems where the harsh environment may result in fewer highly adapted species.

The state and transition model has been used to describe successional processes in the sagebrush system (Westoby and others 1989). Continued heavy spring grazing moves a sagebrush/bunchgrass system to one dominated mainly by big sagebrush (*Artemisia tridentata*). Introduction of annual weeds like cheatgrass (*Bromus tectorum*), and the attendant increased fire frequency holds this system in cheatgrass dominance unless major inputs in weed control and revegetation move it to another state. The cheatgrass-dominated system has little likelihood of transitioning to another state on its own because the rapidly maturing cheatgrass provides a fine and well-distributed fuel over a long fire season. It is adapted to establishment after fire, while other species cannot survive the 3- to 5-year frequency that can occur; so the system is stuck in a stable, but not highly desirable state. Reasons that systems can be held in a stable steady state include (1) frequent or severe disturbances, such as fire, or heavy, continuous grazing; (2) establishment inertia, or lack of establishment associated with harsh environmental conditions, such as in arid deserts where establishment occurs only in unusually wet years; or (3) competitive exclusion where shrubs or trees that are highly competitive, such as pinyon and juniper (*Pinus edulis, P. monophylla* and *Juniperus osteosperma, J. occidentalis*), eventually dominate in the absence of disturbance such as fire. Another example of the latter reason is the replacement of aspen (*Populus_tremuloides*) by conifers in the absence of fire.

Changes in the disturbance regime interact with the environmental context of a plant community to result in transitions or the lack of transitions to other states. For example, Harper (1985) provides evidence that could be used to suggest that lack of fire on acidic soils, such as those in the Uinta Mountains, may result in conifer replacement of aspen much sooner than on the calcareous limestone soils of the Wasatch Mountains, Utah. Damming of rivers and streams controls spring flooding, a process that is necessary for dispersal and establishment of cottonwood (*Populus*) and other riparian species (Middleton 1999; Stromberg and others 1991). This flooding is also essential for the erosion, deposition, and sediment transport functions of the stream that result in the natural geomorphologic features that are necessary to support riparian plant communities.

Lack of fire in communities once dominated by sagebrush and bunchgrass can result in invasion and dominance by pinyon and juniper (Tausch 1999). On deeper alluvial or more fertile soils, tree canopies expand until they touch while the understory vegetation and seed bank die out. These communities are then susceptible to catastrophic crown wildfires and subsequent weed invasion. On shallower soils invaded by pinyon and juniper, resources are too limited for tree canopies to touch, but loss of understory vegetation and subsequent wind and water erosion of interspaces may still degrade the site (Roundy and Vernon 1999).

Transitions to states of much reduced biotic and physical function are said to have passed a biotic or physical threshold, after which a transition back to the previous state is highly unlikely without major intervention (Whisenant 1999). Such thresholds are called irreversible because natural processes alone are insufficient to move them back to the prethreshold state. Susceptibility to such thresholds depends on the environmental context and past management of the site and plant community. For example, sagebrush communities that lack a good understory of perennial grasses pass a biotic threshold into cheatgrass dominance after fire more easily than those with a good perennial grass understory that survives the fire. Invasion and dominance of pinyon and juniper on a site of high erosion potential (higher slopes, finer textured soils, and more frequent occurrence of intense summer thundershowers) may result in major erosion and passing of a physical threshold, while such invasion on sites of low erosion potential may not (Davenport and others 1998).

Management to avoid passing biotic and physical thresholds is much preferred and less costly than attempting restoration after passing these thresholds. For example, use of fire or mechanical treatments to control pinyon and juniper before the understory vegetation or soil is lost, or grazing management to maintain a good perennial understory in sagebrush communities is less costly and risky than attempts to restore these communities after they have passed degrading thresholds. Nevertheless, many of our landscapes have already passed such thresholds and now require major intervention. Restoration after crossing a biotic threshold requires control of the dominating vegetation and revegetation with more ecologically functional and desirable species. Restoration to some historic, natural plant community after passing a physical threshold may not be possible at all, requiring that we set our goals as restoration of ecological function, rather than historic composition. For example, our goals on an eroded site might be to establish a persistent perennial plant cover to hold the remaining soil in place, rather than risk additional soil erosion by attempting to establish a diverse, native community that may no longer be adapted to the degraded conditions of the site. On riparian areas, biotic restoration may be very difficult without restoring the physical disturbance regime or seasonal flooding that drives the biotic responses.

Environmental Context of the Sagebrush Systems

The environmental context of both succession and restoration efforts has an overriding influence on the outcome. Every restoration project requires characterization of the site in order to determine potential for success, species selection, and seeding methods. Two major sagebrush systems are recognized across the Western United States (West 1983a,b). Sagebrush steppe is north of the drier Great Basin and Colorado Plateau sagebrush systems and has more potential for sagebrush renovation and revegetation success than those drier systems to the south. Mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*) communities with higher precipitation have more potential for revegetation than the lower Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) communities. Black sagebrush (*Artemisia nova*) communities on shallow soils are more difficult to revegetate than many Wyoming big sagebrush sites. Higher elevation pinyon-juniper communities have higher precipitation than lower elevation Wyoming big sagebrush communities, and are therefore not only easier to successfully revegetate with desirable species, but may also be more susceptible to invasion of more mesic weeds such as the knapweeds (*Centaurea* spp.). Salt-desert shrub communities are especially hard to revegetate due to low precipitation and fine-textured, saline-alkaline soils that can flow and crust after disturbance and wetting. In general, direct-seeding revegetation is risky with annual precipitation <250 mm, and much less risky at >400 mm. Between these ranges, soils, species selection, seedbed preparation and sowing methods, but especially precipitation during and following the year of seeding greatly affect success.

Designed Disturbance: Residuals

Just as natural disturbances, such as fire and flooding, free up resources for new colonization, revegetation or restoration requires a designed disturbance to reduce undesirable plant populations prior to planting (Sheley and others 1996). Methods of undesirable plant control include biological, mechanical, chemical, and fire. Biological control works best when used in a program of integrated weed control that employs other methods to greatly reduce weed populations. Concurrent or subsequent biological control can then work well to contain weeds. Herbicides have the potential for greatest control of specific undesirables. Mechanical methods have the versatility of configuring the control across the landscape in patterns to maximize wildlife benefits by providing cover, edge, and vegetation linkages where desired. Mechanical methods have less risk of treating nontarget areas than do fire, which can get away, or herbicides, which may drift or move with soil in wind or water erosion. Smooth, Ely (railroad rails attached perpendicular to the chain), or Sagar (rails attached parallel to the chain) anchor chains that produce less to more soil disturbance can be chosen to control large areas of nonsprouting pinyon and juniper trees. Chaining after fire and broadcast seeding helps cover seeds. Broadcast seeding without chaining can result in weed dominance (Ott and others 2003).

The goal of designed disturbance may be to retain components of the original community. For example, sagebrush communities may be treated to rejuvenate older shrubs and control enough of them to establish a more diverse and productive herbaceous understory. Chaining and one-way harrowing may kill about half of the sagebrush in a stand treated for renovation, while discing or two-way harrowing will kill 70 to 90 percent of the sagebrush (Summers and others 2003). This designed disturbance for renovation is equivalent to the successional importance of residuals after a disturbance. Designing plant control to remove undesirables and leave some or most of the desirable plants requires knowledge of plant characteristics such as regeneration potential in relation to the kind of disturbance and location of growing points or seed survival.

Dispersal, Migration, Establishment: Sowing Sufficient Pure Live Seed in Seedbeds Prepared to Maximize Establishment

The restoration equivalent of the successional process of dispersal is direct seeding or transplanting. Plant communities not dominated by desirable plants will require sowing of desirable plants after designed disturbance or wildfire to restore vegetation, ecological function, and value. The large-scales of our weed-dominated landscapes and burned areas require direct seeding for restoration or fire rehabilitation. Since weed seeds are often in the seed bank or are adapted to wind or mechanical dispersal to a site, sowing of desirable plants to reduce weed invasion or reestablishment is necessary. Methods of sowing also employ methods of seedbed preparation to place seeds in the seedbed where their requirements for germination and seedling establishment will be met. This can be a challenge when seeding species of different seed sizes and shapes into the highly variable seedbeds and soils of wildlands. However, if seeds are not placed where their establishment requirements are met, plants will not establish.

Seedbed Preparation and Sowing Methods

Common methods of large-scale sowing of rangelands include drilling and broadcasting seeds. With both methods, the goal is to bury seeds, but to place them at the best depth for their size. Species with smaller seeds like sagebrush or kochia (*Kochia prostrata*) may establish best when broadcast, then firmed into the surface by a rubber-tired cultipacker. The challenge for these seeds is to firm them in, but to not bury them deeper than a few millimeters. Sagebrush seeds can also be seeded through a rangeland drill using a trashy seed box with a pick-wheel inside to force the seed into the seed tubes. To avoid competition with sown grasses, sagebrush is commonly seeded separately in its own rows, and the seed tubes are pulled to let seed fall on top of the ground and avoid excessive burial. A concern for seeding sagebrush this way is that seeds may not be anchored to the surface and may blow away.

Grasses generally establish best when drilled in the fall. Larger seeds such as those of Indian rice grass (*Achnatherum hymenoides*) can be sown deep (2 to 5 cm) in sandy soils, while most grass species should be sown 1 to 2 cm deep. Depth bands on the disks of a rangeland drill are used to prevent excessive seed burial on sandy soils. Newer drills are now available that provide better control of seed placement than the standard rangeland drill. These drills should be tested for success with a range of species on different sites and soils.

Where topography or surface debris makes it impossible to pull a drill across the landscape, or where small seeds can be firmed into the soil, broadcasting is used. The best example of large-scale broadcasting is for fire rehabilitation in burned pinyon-juniper woodlands. Typically seeds are broadcast from a whirlybird seeder suspended from a helicopter or broadcast from a venturi-type seeder on a fixed wing aircraft. Sites are oneway chained to cover seed after broadcasting. Broadcasting seeds without chaining or some other form of seed coverage results in lack of revegetation success and weed invasion (Ott and others 2003). Larger seeds, such as those of four-wing saltbush (*Atriplex canescens*) or bitterbrush (*Purshia tridentata*), can be sown while chaining. These seeds are sown from a dribbler box attached above the tracks of the two caterpillar tractors pulling the chain. The seeds fall out of the box onto the top of the track and are buried in the track imprints. Smaller seeds of sagebrush or rabbitbrush (*Chrysothamnus* spp.) should not be seeded this way because they will be buried too deep.

It is a challenge to broadcast seed mixtures of grass, forb, and shrub seeds with different seed shapes and sizes. Continuous mechanical stirring of seeds mixed with trashy seeds such as those of sagebrush is necessary to provide adequate seed flow from broadcast seeders. The different seed burial requirements of seed mixtures make it difficult to maximize establishment for any one species. Chaining or other post-broadcast seed coverage techniques, such as railing, cabling harrowing, or imprinting, will probably best help establish diverse mixtures when done soon after broadcasting on soils where a wide range of micro topographically diverse safe sites will be created. Determining the advantages of different methods requires experimental comparisons for different sites, species, and methods.

Various methods of improving the seedbed environment have been developed over the years. These include furrowing, imprinting, aerating, or otherwise configuring the seedbed to create safe sites or locations for seeds that favor their germination and establishment. The idea is to bury seeds at the proper depth for emergence, but to increase the time of available water, reduce salinity, moderate temperature, or otherwise maximize favorable environmental conditions for establishment. The success of these methods depends on the soil, seeded species, and precipitation after seeding. Some methods such as drilling and imprinting can result in excessive seed burial on sandy soils, or lack of sufficient burial on heavy-textured or compacted soils. Various methods of seedbed enhancement and sowing should be compared experimentally across a range of sites and with a variety of species in order to make best recommendations for specific sites. Seedbed enhancement may increase seedling establishment on average to moderately wet years, but does not ensure establishment on dry years (Winkel and Roundy 1991).

Seeding Rates

Seeding sufficient germinable seed of adapted species requires an understanding of germination characteristics as well as adaptability of candidate species. Traditional rangeland revegetation guidelines recommend sowing 5.4 to 8.9 lbs/acre (6 to 10 kg/ha) of pure live seed of grass species known to have a fairly wide range of adaptability. These recommendations have proven successful for introduced grasses, but additional considerations are needed to successfully sow native species. Pure live seed is the amount of viable seed in a bag of seeds. It can be expressed as a percentage of the total weight of viable seeds, plus other matter such as seed parts, weed seeds, and nonviable seeds. Viability, or whether the seed is dead or alive, can be determined by a tetrazolium chloride solution or TZ test, where the active dehydrogenase enzyme in live seeds results in a red staining. This test does not determine germinability. Dormant seeds are viable but not germinable until dormancy is broken by artificial means or by specific environmental conditions. State seed testing laboratories determine germination percentages at temperature, light, and other incubation specifications generally known to maximize germination for a particular species. Some species may also be subjected to pretreatments such as seed coat scarification or chilling prior to incubation to maximize germination. When both seed viability and germination are tested, seed tags may bear germination and hard seed (viable but dormant) percentages.

Bulk seeding rates are calculated by dividing the recommended pure live seeding rate by the pure live seed percentage. For large-scale fire rehabilitation projects, the Bureau of Land Management (BLM) typically contracts for lots of seed specifying at least a 80 percent germination for grasses, or lesser percentages for some species, such as sagebrush, that are hard to clean. The BLM sends samples of their seed purchases to a State seed testing laboratory to verify the specifications. Because fire rehabilitation seeding is rushed in the late summer and early fall, seed labs may only have time to do a TZ test. If seed lots are found to have lower pure live seed percentages than was specified in the contract, the BLM may return the seed or adjust their price downward. The BLM often seeds using bulk rates for introduced grasses and legumes known to have high germination percentages (>80 percent). These rates typically run from 1.8 to 3.6 lbs/acre (2 to 4 kg/ha) of each species in a mixture of three or more species. For fire rehabilitation in the past, mainly introduced grasses have been seeded with some native grasses and a few introduced legumes such as sainfoin (*Onobrychis vicifolia*), small burnet (*Sanguisorba minor*), or alfalfa (*Medicago sativa*) (Richards and others 1998).

Successful establishment of native grasses, forbs, and shrubs may require higher seeding rates than those for simple introduced species mixtures. Thompson (2002) found successful large-scale establishment of native seed mixtures at 17.8 lbs/acre (20 kg/ha) bulk total seed drilled on burned sagebrush sites and 16 to 26.8 lbs/acre (18 to 30 kg/ha) total seed broadcast and chained on burned pinyon-juniper sites. Bulk rates required to get similar establishment from standard BLM seed mixes were generally lower and cost much less, but did not result in comparable establishment of native plants. Pyke and others (2003) found native plants in BLM fire rehabilitation projects, but they were unable to determine if those plants were residual to the sites or established by seeding. Native mixtures may require higher seeding rates than introduced species, and more careful species selection for specific sites. In Thompson's (2002) study, seeding predominately Indian ricegrass, known for its ability to emerge from deeper sandy soils, could have saved the extra expense and failure of other native grasses that were probably drilled too deep on the sandy sagebrush site tested.

Interaction/Reaction: Establishing and Facilitating Diverse, Native Communities

Plant ecologists have identified numerous combinations of plant-plant interactions (Barbour and others 1998). Clements (1928) stressed the importance of competition as a driving force in succession and what eventually dominates a site. This makes sense for classic forest succession where the dominant climax tree species are the ones that eventually develop large enough root and canopy structures to compete best for resources. Disturbance plays a vital role in opening up resources for a more diverse suite of species. Although competition evidently is a major driving force for the plants that eventually dominate a site after disturbance, other interactions may be more important in providing for long-term compatibility and diversity in a community. Plants may partition resources among themselves in time by growing during different seasons, or in space by accessing different soil depths. Rabbitbrush is evidently less competitive with grasses than Wyoming big sagebrush because its taproot uses deeper soils and avoids major competition with shallower grass roots (Frischknecht 1963). On the other hand, the two-layered surface and taproot system of Wyoming big sagebrush makes it a strong competitor with perennial bunch grasses. Scientists in Turkmenistan developed range improvement practices to improve forage quality and quantity for livestock. They selected woody species to use deeper soil moisture than the extant herbaceous communities (Nechaeva 1985). Agroforestry and intercropping practices are dependent on finding crops and trees that yield more when grown together than when grown separately. The best known example is growing nitrogen-fixing legumes with grasses. However, legume enhancement of grass growth requires long periods of available soil moisture to work best.

Although shrubs are generally considered competitive with herbaceous species, they also offer a suite of services to a diverse community, such as

- Enhance soil fertility
- Catch seeds, spores, soil, and snow
- Moderate the temperature environment
- Improve soil aggregate stability and infiltration rates
- Harbor beneficial insects

(Call and Roundy 1991; West 1989). Because there are many ecological and management benefits to mixed communities, we would like to restore them or establish them in fire rehabilitation seedings. Such a goal is much more ambitious than the single species or simple introduced species mixes of past rangeland revegetation. Use of native species in this effort requires understanding about which ecotypes are best adapted to specific regions or sites. The large-scale requirements of fire rehabilitation suggest that we should use native plant materials with a wide range of adaptation if possible.

Plant Materials Selection and Improvement

Plant adaptation and plant materials trials in the past have taken an agronomic approach. Numerous collections are planted in separate rows or blocks in common gardens and evaluated over many years. When a particular collection appears to be more vigorous than the others, it is selected for release. This approach takes a very long time to release a given plant material, and fails to address some important ecological aspects of mixed community restoration. It limits genetic diversity for out-crossing species by keeping the collections separate. Mass selection and other crossing techniques could be used to maximize genetic diversity. Very few examples of such approaches have been tested. An emerging approach is to certify seeds as “source-identified” (Young 1995). These collections are certified as originally collected from a particular site, representing a specific environment. Managers could choose “source-identified” plant materials from sites with similar regional environmental conditions as the sites they need to restore or rehabilitate. Once a large native seed industry is developed, managers could even choose physical mixes of a number of source-identified plant materials to best cover their estimated environmental conditions. Such an industry will need establishment of large seed warehouses and seed storage guidelines to allow stockpiling and a consistent market for these plant materials. Commitment of government to large restoration efforts such as the Great Basin Restoration Initiative will also support a more consistent demand for specific native plant materials. In the past, the demand for native seeds has been highly variable and subject to the severity and extent of the current fire season.

Establishing Diverse, Weed-Resistant Communities

Another aspect of plant materials evaluation, not generally tested much in the past, is that of how well plant materials work together rather than separately. The larger concern, of course, is that of establishing “stable, diverse” plant communities that are resistant to weed invasion. That is a major challenge. Not only are we not really sure how mixtures of plants will persist together, but it is a major challenge to seed diverse mixtures and have all the seeded species establish. When we have seeded aggressive, more weed-resistant introduced grasses in mixtures with native species, the introduced species eventually dominate (Pyke 1996). Approaches to more successfully establishing diverse communities include (1) seeding more aggressive species at a much lower rate than less aggressive ones, (2) seeding certain species or mixes in separate rows, strips, or patches, or (3) interseeding slower growing species such as some shrubs after scalping out established grasses. Because sowing equipment and environmental conditions favor establishment of many grass species over that of shrub and forb species, you cannot expect that just mixing species will produce a community in the same proportion as the seed mix (Newman and Redente 2001).

Weed-control strategies include designed disturbance to reduce weed populations and controlled establishment of desirable plants to preempt resources from weeds and prevent their invasion in the future (Sheley and others 1996). In that regard, we really do not know enough about what constitutes a community resistant to specific weeds on different sites. Our goal is to establish a suite of desirable plants that allows for their own coexistence, but excludes weeds. Goldberg (1990) has suggested that plant-plant interactions are often indirect through intermediate resources. To guide seeding mixture recommendations of the future, we must look at resource needs and use in time and space by desirable and weedy species. This type of research can require many years to develop recommendations, given the great range of species, weeds, and environmental conditions. In the meantime, it is very important that diverse seedings be monitored for response of both desirable and weedy species. Every fire rehabilitation or restoration project is an experiment from which something can be learned to guide future efforts.

It may be unrealistic to expect weed control and successful revegetation of native plants on sites where precipitation is low and the proximity of weed populations threaten reinvasion. On such sites, use of bridging species such as crested wheatgrass (*Agropyron desertorum*), which are more easily established and resistant to weed invasion, may be necessary. The bridging species could later be controlled and may be much easier to replace with native species than weedy species (Cox and Anderson 2004). Restoration will require innovative approaches to meet the requirements of native plants.

References

Barbour, M. G.; Burk, J. H.; Pitts, W. D.; Gilliam, F. S.; Schwartz, M. W. 1998. Terrestrial plant ecology. Menlo Park, CA: Benjamin/Cummings. 649 p.

Call, C. A.; Roundy, B. A. 1991. Perspectives and processes in revegetation of arid and semiarid rangelands. Journal of Range Management. 44: 543–549.

Chambers, J. C.; MacMahon, J. C.; Wade, G. L. 1992. Differences in successional processes among biomes: importance for obtaining and evaluating reclamation success. In: Chambers, J. C.; Wade, G. L., eds. Evaluating reclamation success: the ecological consideration—proceedings of a symposium; 1990 April 23–26; Charleston, WV. Gen. Tech. Rep. GTR-NE-164. Radnor, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station: 59–72.

Clements, F. E. 1928. Plant succession and indicators. New York: H. W. Wilson Company. 453 p.

Cox, R. D.; Anderson, V. J. 2004. Increasing native diversity of cheatgrass-dominated rangeland through assisted succession. Journal of Range Management. 57: 203–210.

Davenport, D. W.; Breshears, D. D.; Wilcox, B. P.; Allen, C. D. 1998. Viewpoint: sustainability of piñon-juniper ecosystems—a unifying perspective of soil erosion thresholds. Journal of Range Management. 51: 231–240.

Frischknecht, N. E. 1963. Contrasting effects of big sagebrush and rubber rabbitbrush on production of crested wheatgrass. Journal of Range Management. 16: 70–74.

Gleason, H. A. 1926. The individualistic concept of the plant association. Bulletin of the Torrey Botanical Club. 44: 463–481.

Goldberg, D. E. 1990. The components of resource competition in plant communities. In: Grace, J. B.; Tilman, D., eds. Perspectives on plant competition. San Diego, CA: Academic Press, Inc.: 27–49.

Harper, K. 1985. Predicting successional rates in Utah aspen forests. In: Foresters' future: leaders or followers? Proceedings Society of American Foresters national convention; 1985 July 28–31; Fort Collins, CO. [Place of publication unknown]: Society of American Foresters: 96–100.

Middleton, B. 1999. Wetland restoration, flood pulsing, and disturbance dynamics. New York: John Wiley and Sons. 388 p.

Nechaeva, N. T., ed. 1985. Improvement of desert ranges in Soviet Central Asia. New York: Harwood Academic Publishers. 327 p.

Newman, G. J.; Redente, E. F. 2001. Long-term plant community development as influenced by revegetation techniques. Journal of Range Management. 54: 717–724.

Ott, J. E.; McArthur, E. D.; Roundy, B. A. 2003. Vegetation of chained and non-chained seedings after wildfire in Utah. Journal of Range Management. 56: 81–91.

Pyke, D. A. 1996. Rangeland seedings and plantings: exotics or natives? In: Edge, W. D.; Olson-Edge, S. L., eds. Sustaining rangeland ecosystems symposium: proceedings; 1994 August 29-31; La Grande, OR: Eastern Oregon State College SR 953. Corvallis, OR: Oregon State University: 32–44.

Pyke, D. A.; McArthur, T. O.; Harrison, K. S.; Pellant, M. 2003. Coordinated Intermountain Project—fire, decomposition, and restoration. In: Palmer, A. R.; Scogings, P. F., eds. Proceedings of the VIIth International Rangeland Congress; 2003 July 26–August 1; Durban, South Africa. African Journal of Range & Forage Science. 20: 1116–1124.

Richards, R. T.; Chambers, J. C.; Ross, C. 1998. Use of native plants on Federal lands: policy and practice. Journal of Range Management. 51: 625–632.

Roundy, B. A. 1996. Revegetation of rangelands for wildlife. In: Krausman, P. R., ed. Rangeland wildlife. Denver, CO: Society for Range Management: 355–368.

Roundy, B. A.; Vernon, J. L. 1999. Watershed values and conditions associated with pinyon-juniper communities. In: Monsen, S. B.; Stevens, R., comps. Proceedings: ecology and management of pinyon-juniper communities within the Interior West; 1997 September 15–18; Provo, UT. Proc. RMRS-P-9. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 172–187.

Sheley, R. L.; Svejcar, T.; Maxwell, B. D.; Jacobs, J. S. 1996. Successional rangeland weed management. Rangelands. 18: 155–159.

Stromberg, J. C.; Patten, D. T.; Richter, B. D. 1991. Flood flows and dynamics of Sonoran riparian forests. Rivers. 2: 221–235.

Summers, D. D.; Roundy, B. A.; Walker, S. C.; Davis, J. N. 2003. Vegetation response of a Wyoming big sagebrush (*Artemisia tridentata* ssp. w*yomingensis*) community to 6 mechanical treatments in Rich County, Utah. In: Abstracts: diversity through time. Society for Range Management annual meeting; 2003 February 2–6; Casper, WY. Abstract 282.

Tausch, R. 1999. Historic pinyon and juniper woodland development. In: Monsen, S. B.; Stevens, R., comps. Proceedings: ecology and management of pinyon-juniper communities within the Interior West; 1997 September 15–18; Provo, UT. Proc. RMRS-P-9. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Experiment Station: 12–19.

Thompson, T. W. 2002. Fire rehabilitation in Tintic Valley, Utah, using native and exotic species. Provo, UT: Brigham Young University. 67 p. Thesis.

West, N. E. 1983a. Great Basin—Colorado Plateau sagebrush semi-desert. In: West, N. E., ed. Temperate deserts and semi-deserts. New York: Elsevier Scientific Publishing Company: 331–349.

West, N. E. 1983b. Western Intermountain sagebrush steppe. In: West, N. E., ed. Temperate deserts and semi-deserts. New York: Elsevier Scientific Publishing Company: 351–374.

West, N. E. 1989. Spatial pattern-functional interactions in shrub-dominated plant communities. In: McKell, C. M., ed. The biology and utilization of shrubs. New York: Academic Press: 283–305.

Westoby, M.; Walker, B.; Noy-Meir, I. 1989. Opportunistic management of rangelands not at equilibrium. Journal of Range Management. 42: 266–274.

Whisenant, S. G. 1999. Repairing damaged wildlands. Cambridge, UK: Cambridge University Press. 312 p.

Winkel, V.; Roundy, B. A. 1991. Effects of cattle trampling and mechanical seedbed preparation on grass seedling emergence. Journal of Range Management. 44: 176–180.

Young, S. A. 1995. Verification of germplasm origin and genetic status by seed certification agencies. In: Roundy, B. A.; McArthur, E. D.; Haley, J. S.; Mann, D. K., comps. Wildland shrub and aridland restoration symposium: proceedings; 1993 October 19–21; Las Vegas, NV. Gen. Tech. Report INT-GTR-315. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 293–295.