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Restoration of degraded lands in the interior Columbia River basin: passive vs. active approaches

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Abstract

Evidence for success of passive and active restoration is presented for interior conifer forest, sagebrush steppe, and riparian ecosystems, with a focus on the Columbia River basin. Passive restoration, defined as removal of the stresses that cause degradation, may be most appropriate for higher elevation forests, low-order riparian ecosystems, and for sagebrush steppe communities that are only slightly impaired. More active approaches, in which management techniques such as planting, weeding, burning, and thinning are applied, have been successful in forests with excessive fuels and in some riparian systems, and may be necessary in highly degraded sagebrush steppe communities. There is general agreement that true restoration requires not only reestablishment of more desirable structure or composition, but of the processes needed to sustain these for the long term. The challenge for the restorationist is to find a way to restore more desirable conditions within the context of social constraints that limit how processes are allowed to operate, and economic constraints that determine how much effort will be invested in restoration. Published by Elsevier Science B.V.

Keywords: Disturbance; Process; Resilience; Degradation; State-transition; Riparian; Sagebrush; Interior forest; Restoration management

1. Introduction

Ecological restoration can be defined as "the process of assisting the recovery and management of ecological integrity," including a "critical range of variability in biodiversity, ecological processes and structures, regional and historical context, and sustainable cultural practices" (Society of Ecological Restoration, http://www.ser.org/definitions.html). Restoration can be 'passive,' in which the degrading agent(s) is identified and removed (Allen, 1995), or 'active,' in which management techniques such as planting, weeding, burning, and thinning are undertaken with a particular image of desired structure, composition, or pattern in mind. The need for restoration assumes some level of impairment in an ecosystem, and indeed, the scientific assessment of the Columbia River basin concluded that much of the area is in a state of low ecological integrity (USDA and USDI, 1996). Although natural disturbances can cause impaired conditions, often human activities are responsible, and thus a change in management can theoretically bring about improved conditions. Several alternatives of the supplemental draft environmental impact statement (SDEIS) for the interior Columbia River basin emphasize active management to attain restoration objectives in the basin. The purpose of this paper is to examine the proposition that active restoration methods will be effective in improving the condition of degraded lands in the Columbia River basin by reviewing that portion of the scientific literature that allows consideration of the relative merits of active vs. passive approaches.

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2. The literature review

This paper discusses some of the evidence of restoration in three common ecosystem types of western North America — interior conifer forest, sagebrush steppe, and interior riparian — within the context of the active vs. passive restoration debate. We collected information by searching the primary literature, by examining recent issues of *Restoration Ecology* and *Restoration Management Notes* and by consulting specialists in both the management and scientific communities (listed in acknowledgments). We do not provide a comprehensive review of this complex subject, but provide examples to help frame the debate about restoration on public lands within the Columbia River basin. Nor do we provide an analysis of costs and benefits, although we recognize that economics will constrain restoration activities that are actually undertaken.

For each ecosystem type, our approach is to first describe the current state of impairment, including what is known of the activities that caused the perceived need for restoration. Then we describe structures and processes of desired restored condition and what would indicate restoration success. Next we discuss evidence that management activities intended for restoration have succeeded, and compare that with the evidence of restoration resulting from a passive approach. In conclusion, we discuss what the scientific literature implies for future restoration management of ecosystems within the Columbia River basin.

2.1. Mixed-conifer forests

For the past 12,000 years, wildfire has played a major role in determining the structure of mixed-conifer forests in the interior of western North America (Agee, 1993). Since the 1930s, organized fire suppression and selective harvest of ponderosa pine has interrupted the historical disturbance regime that produced structure in these forests — especially in the drier forests with frequent low-intensity fires — resulting in higher fuel loads, higher stem densities, and more extensive outbreaks of pest insects and disease (Everett et al., 1994; Arno and Harrington, 1999). Because wildfire intensity is determined by the combination of weather, topography, and fuel loads (Agee, 1993), wildfires tend to be uncharacteristically severe in these altered forests, often resulting in crown, rather than surface fires (Arno et al., 1995).

Passive restoration, including primarily the cessation of fire suppression activities, has not been attempted in fuel-laden forests of the western United States, owing primarily to long-standing public policy. However, in ecosystems where wildfires have been allowed to burn repeatedly over the years, fire regimes appear to approach historical standards. For example, the mosaic of wildfire patches at Yosemite National Park over the past 25 years show that more recent fires tend to extinguish at the boundaries of past fires (van Wagtendonk, 1995). Working on the Teanaway River drainage in Washington, Wright (1996) has shown a pattern of fires extinguishing at the edges of areas previously burned 1–2 years before. In the chaparral of northern Baja, uncontrolled wildfires have resulted in an ecosystem where fire size is limited compared to the fire-suppressed chaparral of the United States north of the border, despite similar fire incidence in both systems (Minnich and Chou, 1997). The explanation for these cases is that fire removes biomass that builds up through primary production, thus leaving little fuel upon which subsequent fires can feed. Similarly, in dry interior forests, historical wildfire has tended to remove biomass on the forest floor and in the smaller standing stems (Hall, 1976; Sackett et al., 1996; Arno and Harrington, 1999), thus reducing the severity of subsequent fires. In the above cases, wildfire behavior is clearly linked to previous wildfire history. Thus passive restoration, in which fires are allowed to burn at will, would be likely to eventually create forest conditions more similar to those initially perceived by European settlers.

Widespread public support for fire suppression and the recognition that fire plays a critical role in western interior forests has stimulated hundreds of fuel reduction projects on federal lands in the last 40 years featuring thinning, thinning and removal, mechanical fuel treatment, or prescribed fire. While active restoration of historical fire regime or historical structure is often the long-term restoration goal, fuel reduction has typically been employed to create structure that can accommodate disturbances such as fire, pest insects, and disease with less disruption to human culture and economy. Thus an important step toward restoration would be indicated by evidence that fuel reduction tempers wildfire severity in treated forests.

2.1.1. Retrospective studies

In pine forests of the eastern United States, changes in the occurrence, behavior, and intensity of wildfires after recent fuel treatment were observed: in Georgia loblolly pine, underburning reduced fire intensity (Helms, 1979); in Florida and Georgia slash pine, underburning reduced fire occurrence (Davis and Cooper, 1963); in New Jersey shortleaf pine, underburning reduced residual tree mortality (Cumming, 1964; Moore et al., 1955). Similarly, in pine forests of the western United States, fuel treatment or prescribed fire changed subsequent fire behavior in Montana ponderosa pine stands (Arno and Brown, 1989), in Arizona ponderosa pine (Wagle and Eakle, 1979), and in mixed-conifer stands in Idaho, Montana, and Washington (Cron, 1969). Working in ponderosa pine stands that had experienced fuel reduction within 10 years of four major fires, Pollet (1999) concluded that fire severity was significantly higher in untreated vs. treated stands, based on the following crown scorch estimates: 1994 Webb Fire, Montana (67% untreated vs. 26% treated); 1994 Tyee Fire, Washington (100% vs. 74%); 1994 Cottonwood Fire, California (78% vs. 26%); and 1996 Hochderffer Fire, Arizona (99% vs. 29%). Treated stands within the Webb Fire had been underburned 5 years previous, stands within the Tyee and Hochderffer Fires had been thinned and burned within 10 years prior to wildfire, while stands within the Cottonwood Fire had been whole-tree thinned 5 years prior to wildfire. With the use of aerial photography to assess damage and existing records to assess stand characteristics and management history, Weatherspoon and Skinner (1995) conducted a retrospective study of wildfire effects (crown scorch) on part of the Shasta-Trinity National Forest burned by the 1987 Stanislaus Complex wildfires. For tree plantations, site preparation and damage in adjacent stands were the most important variables explaining crown scorch — plantations that were cut but had slash left untreated and that were adjacent to stands with high fuel loads had the highest probability of experiencing severe wildfire effects. For both uncut and partial cut stands, fuel treatment was the variable that explained most of the variation in crown scorch. Interestingly, partial cut stands with no accompanying fuel treatment experienced the greatest degree of crown scorch, with even a nominal level of fuel treatment substantially reducing fire severity. In fact, available evidence suggests that removing large stems from a stand, leaving most of the understory of small stems, and doing little slash treatment, will substantially *increase* the probability that a wildfire will have severe stand effects (Alexander and Yancik, 1977; Weatherspoon and Skinner, 1995). The general explanation for these results is that fuel treatment changed stand structure, species composition, and biomass, thus decreasing subsequent fire severity.

The relation between forest fuel loads and fire severity found in many prescribed fire and thinning studies is even more evident upon examination of wildfire encounters with fuel breaks (Pierovich et al., 1975). Green (1977) defines a fuel break as "a strategically located wide block, or strip, on which a cover of dense, heavy, or flammable vegetation has been permanently changed to one of the lower fuel volume or reduced flammability." Fuel breaks are designed to change fire behavior, and when used in conjunction with fire suppression efforts, may increase the probability of stopping an advancing wildfire. In eight wildfire encounters with forested fuel breaks, Pierovich et al. (1975) documented seven that resulted in successful fire control, six of which were accompanied by fire suppression crews. Of two cases in which crews were not present, one succeeded and one failed. There is little question that aggressive fuel treatment in the form of fuel breaks alters fire behavior and therefore can allow fire suppression crews to stop an advancing wildfire. Yet because they are designed to function with fire suppression crews, fuel breaks alone are generally not effective at stopping wildfires. In their review of the efficacy of shaded fuel breaks, Agee et al. (2000) emphasize this point, in part by providing an example of a recent crown fire (Tyee) on the Wenatchee National Forest that dropped to the ground upon encountering a fuel break, but regained crown form after passing through. Such examples demonstrate the limited utility of the fuel break concept and suggest that landscape-level fuel reduction would be required to significantly reduce the extent and probability of independent crown fires.

Finally, it should be noted that simply documenting successful examples of fuel reduction, from the perspective of subsequent wildfire behavior, is insufficient for truly understanding the efficacy of fuel reduction as a management policy. What we really want to know is the probability that a given effort will produce the desired result, lessening the negative effects of intense crown fires and returning to a low-intensity fire regime. Retrospective studies that interpret successes and failures in light of what is known about stand structure, fuel loads, and weather conditions, would add greatly to the information on fuel reduction as a viable objective for restoring degraded forests.

2.1.2. Modeling studies

Wilson and Baker (1998) used a fire hazard model to predict postfire stand conditions under several landscape-scale silvicultural regimes, from no treatment to intensive fuel reduction. They found that while thinning or prescribed fire within stands substantially reduced predicted fire intensities, fuel reduction had little effect on adjacent stands. This conclusion supports the findings of retrospective studies. Wilson and Baker (1998) also found that stands located at valley bottoms had significantly lower fire probabilities of crown fire, simply due to the lack of adjacent stands burning down slope.

Stephens (1998) modeled fire growth and behavior in Sierran mixed-conifer forests that received simulated treatments including various combinations of prescribed fire and mechanical methods. He found that prescribed fire and mechanical methods that included slash treatment and landscape-level fuel treatment were the most effective techniques for reducing fire intensities, heat per unit area, rate of spread, area burned, and scorch heights. He concluded that restoration of mixed-conifer forests required a consideration for how proposed treatments affected fuel structure.

Landscape-level consequences of fuel treatment design were modeled by Finney (2001), who considered how a variety of fuel treatment configurations affected the spread of wildfire. He found that if 20% of the landscape were treated as fuel treatment strips, and that if these were staggered uniformly, a wildfire would consist mostly of “flanking” rather than “head fire” behavior, and would thus advance at a slower rate. His work also suggests, however, that in the absence of accompanying fire suppression efforts, fairly aggressive fuel treatment is required to temper landscape-level fire behavior, and that untreated interspersed stands would receive little or no benefit from adjacent stands treated as fuel breaks.

The relative roles of weather and fuels in predicting fire intensity was modeled by Bessie and Johnson (1995), who estimated surface fire intensity and crown fire initiation in subalpine fir stands of varying ages over a 35-year span of weather conditions. They found that weather was a significantly better predictor of crown fire initiation than was fuel, primarily because of the much greater variation in weather conditions over time. They suggested that for forests that have a low-frequency, high-intensity fire regime (stand replacement), fuel reduction does little to mitigate fire effects, because most fires occur during extreme weather conditions, become independent crown fires, and cause significant mortality regardless of fuel loadings. Their results argue for a focus of fuel reduction efforts on low-elevation forests having fire regimes featuring high-frequency, low-intensity fires.

Available evidence suggests that both passive and active restoration can play important roles in management of western interior forests. Passive restoration in which wildfires are allowed to burn may be more effective in forests that have a high-severity fire regime (Baker, 1994) in which weather plays a more dominant role in determining fire intensity (Bessie and Johnson, 1995). Active restoration involving fuel reduction by fire or mechanical means can decrease severity of subsequent wildfires that encounter fuel-reduced stands (Wilson and Baker, 1998) and may be particularly useful in forests with historical low-severity fire regimes. Landscape-scale restoration of historical structure or fire regime is likely to be achieved, however, only if treatments are widely or strategically dispersed; tactical fuel reduction projects will only serve to influence fire severity locally.

2.2. Sagebrush steppe

Most evidence suggests that sagebrush steppe communities of the intermountain West that existed prior to Euro-American settlement were not resilient to intense grazing pressure (Mack and Thompson, 1982). Between 1870 and 1900, grazing pressure from livestock introduced by Euro-American settlers increased an estimated 28-fold relative to native ungulates (Monsen and McArthur, 1995). In less than 50 years livestock grazing capacity fell overall from an estimated 0.83 animal unit months per hectare in 1880 to only 0.27 by 1930 (McArdle, 1936). The key feature of this decline was the reduction or elimination of native perennial grasses and forbs through intense livestock grazing, coupled with the introduction of exotic annual grasses such as cheatgrass (*Bromus tectorum*). Due to high colonization potential (Hull and Pechanec, 1947) and competitive ability (Evans, 1961; Melgoza et al., 1990) cheatgrass has proven to be an aggressive dominant of degraded sagebrush steppe (Young et al., 1972), covering an estimated 40 million hectares of shrub steppe ecosystem by 1991 (D'Antonio and Vitousek, 1992). The extensiveness and flammability of cheatgrass has resulted in more frequent and more extensive fires (Young et al., 1987; Whisenant, 1990), which has converted many sites from a dominance of native perennial to introduced annual species. The overall result of these processes has been the simplification of sagebrush steppe communities, in terms of structure and species composition (Billings, 1990), and a decrease in average forage quantity and quality (Whisenant, 1990). Certain species of native vertebrates, such as the pygmy rabbit and the sage grouse, have experienced significant habitat decline (Maser et al., 1984), and in most cases these declines have continued to the present time (Wisdom et al., in press).

The restoration of degraded sagebrush steppe communities to more native species compositions is a complex and challenging problem, for several reasons. At least three disturbance agents (fire, livestock, invasive plants) and a suite of biotic and abiotic factors interact in complex and unpredictable ways to produce the structure we see at the present time. For example, the introduction of livestock, exclusion of fire, and a wetter than normal period has contributed to the expansion of juniper into sagebrush steppe in southeastern Oregon (Miller and Rose, 1999). Because sagebrush species do not respond identically to disturbance (McArthur, 1994), especially when growing at different elevations (Alma Winward, Forest Service, Ogden, UT, pers. comm.), it is generally difficult to predict the outcome of changes in the operation of disturbance agents at any given site (Tausch et al., 1995). In some cases sagebrush steppe communities have degraded to the point where an alternative "steady state" has been reached, separated by steep transition boundaries from the historical state (Laycock, 1995). Restoration toward historical condition may require not only reestablishment of native vegetation but of keystone animal species as well because of their function in seed dispersal (Longland, 1995) and pollination, among other things (Whisenant, 1995). Finally, we may not be prepared to pay for the cost of restoration, given the time it may take for successful restoration to occur (Tyson, 1995), especially for those sites that have moved far from their pre-Euro-American structure and composition (Rimbey, 1994).

Because of this complexity in ecological, social, and economic realms (Allen, 1995), the answer to the active vs. passive restoration question is more likely to depend on particular site conditions. We discuss the challenges of sagebrush steppe restoration with the use of a state and transition model developed by Westoby et al. (1989), and refined by Laycock (1995) and West (1999) (Fig. 1). Evidence for restoration success would be indicated by any transition from a lower to a higher state in the model.

West (1999) estimates that less than 1% of the original area of historical pre-Euro-American sagebrush steppe has remained unaffected by livestock grazing (State I) (Fig. 1). Even these communities, however, are historical only relative to other extant communities, because they lack indigenous human disturbance (e.g., fire), and contain many of the invasive plant species common to other sagebrush steppe communities. Under the influence of moderate grazing, this pristine state succeeds quickly to late seral sagebrush steppe (State II), which occupies a much greater land area (estimated 20%). Presumably, livestock exclusion in State II communities could effect return to the 'pristine' state, although there appears to be little evidence to support that claim. Heavier grazing accompanied by fire suppression tends to cause degradation of State II to a depauperate late seral steppe community (State III), because of livestock preference for palatable perennial grasses over shrub species. Restoration from State III to State II has been shown to be possible through a number of means, all of which are designed to favor grasses over woody vegetation. For example, if sufficient populations of native herbs are still present, prescribed fire can drive a transition from States III to II (Laycock, 1991). A similar transition can sometimes be obtained by fall grazing of sheep on more edible sagebrush species

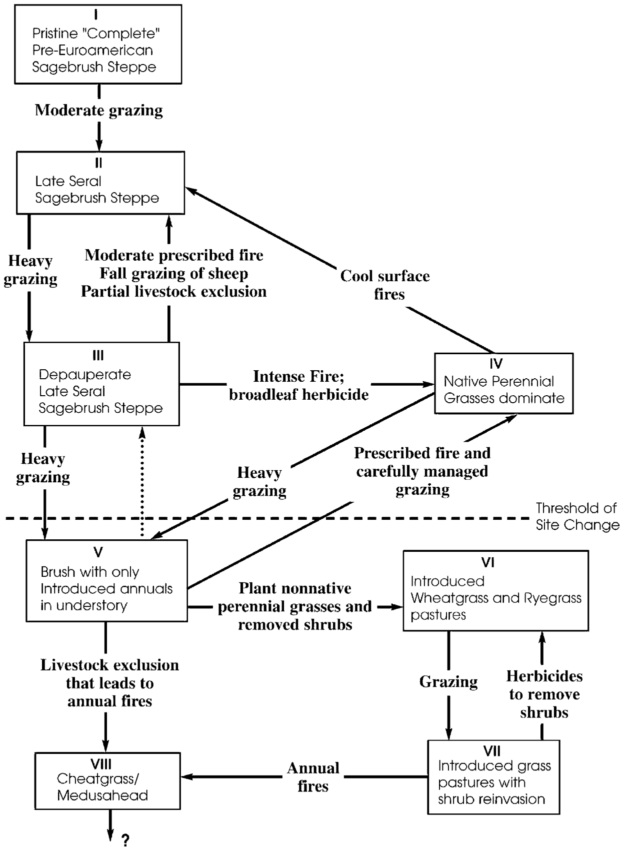

Fig. 1. State and transition model of successional change in sagebrush steppe with examples of activities expected to result in state transitions (adapted from West (1999)).

such as *Artemisia tripartita* (Laycock, 1967; Bork et al., 1998). Passive restoration through livestock exclusion can also cause a return to State II, although depauperate late seral communities can also stagnate in the absence of livestock grazing (West et al., 1984; Havstad, 1994). Intense fire, with or without the use of broadleaf herbicides, can cause the transition of State III to State IV, dominated by native perennial grasses. The effect of fire, however, will depend to some extent on site conditions. More shrubby lower elevation sites may be less likely to return to perennial plants following fire than moist higher elevation sites that have a significant component of native perennial grasses (Alma Winward, Forest Service, Ogden, UT, pers. comm.; Tausch et al., 1994). Natural succession from State IV to II will tend to occur slowly, especially if accompanied by relatively cool surface fires. Heavy grazing of perennial grasses, however, can change States III or IV into an even more degraded State V, which consists of woody shrubs with an understory of introduced annual grasses. Once State V has been reached, return to less degraded states becomes problematical, primarily because of the difficulty and cost of change from an understory of introduced annuals to one of native perennials. According to West (1999), simple removal of livestock will not only fail to return State V to State III, but may hasten degradation to State VIII (introduced annuals), because livestock exclusion will allow fuel buildup that will lead to an increase in the chance that wildfire will convert the community from a shrub perennial to an annual system (Whisenant, 1990; Davison, 1996). Managers can defer conversion of States V to VIII by reducing fuel through livestock grazing (Davison, 1996) by physical removal of woody shrubs and planting of introduced perennials (State V to State VI), or removal of shrubs by use of herbicides (States VII to VI), but these activities can hardly be expected to restore native communities in the long run. Although some success in replacing exotic annuals with native perennial grasses (State V to State IV) has been achieved through prescribed fire, carefully managed livestock grazing, or both (Stewart and Hull, 1949; Young et al., 1987; Dyer, 1993; Vallentine and Stevens, 1994) for such highly degraded sites only aggressive methods (such as seeding perennials) have any chance of restoring more desirable sagebrush steppe communities (Laycock, 1991; Sanders, 1994). A site may remain in State VI for decades, but ultimately will likely proceed to States VII and VIII under the influence of grazing and annual fire (West, 1999). Some authors have even inferred that it may be time to reclassify some sagebrush/bunchgrass communities as annual grasslands (dominated by cheatgrass), and manage them accordingly (Young et al., 1987).

In summary, for relatively healthy sagebrush steppe communities (States I–IV), the passive approach can often accomplish restoration of degraded (States III and IV) to more pristine communities (States I and II). Faster transition to at least State II from States III and IV can occur through judicious use of prescribed fire and fall grazing. Passive restoration alone will generally be ineffective for restoring communities that have degraded to State V and beyond. For these systems, relatively heroic methods are needed for successful restoration (Laycock, 1991; Davison, 1996). In general, a reliance on passive restoration amounts to a confidence that restoring natural disturbance processes will bring about a change toward more desirable community types. This confidence may be justified if degradation is relatively minor. But if profound changes in community composition or structure have occurred, restoration of natural disturbance processes may not be sufficient to return the system to a desirable or historical state. In fact, natural disturbances such as fire may further degrade plant communities, due to interaction with other disturbances such as weeds. In particular, because cheatgrass carries fire so well, restoration success may depend on a strategic consideration of its distribution across the landscape. Thus protection of a high-quality site will require not only site-specific considerations, in which both passive and active approaches are weighed, but a recognition of how that site fits into a neighboring landscape potentially dominated by cheatgrass.

2.3. Riparian ecosystems

There is widespread recognition that the majority of riparian and stream ecosystems in the western United States are degraded (Elmore and Beschta, 1987; McIntosh et al., 1994). Features of degradation include loss of native riparian vegetation, loss of native fish species, altered channel morphology, changes in the magnitude and timing of seasonal flows, increased summer water temperatures (Armour et al., 1991), and lowered water table and water storage capacity (Barrett, 1994). The causes of degradation include changes in climate and precipitation patterns, heavy streamside grazing, introduction of exotic fish species, inappropriate use of uplands, road construction, timber harvest, mining, and water diversions (Elmore and Beschta, 1987; Wissmar et al., 1994). The history of degradation has been well documented in the Columbia River basin (Wissmar et al., 1994), and can be linked to the continuous decline in the condition of salmonid fisheries (Rieman et al., in press). Although many of the more egregious human-induced stresses (placer mining, excessive livestock grazing, splash damming) are no longer practiced, their effects are still with us, and combined with current stresses create a considerable challenge to successful restoration.

The most important feature of a proper functioning riparian system is the dynamic between streamflow and landforms, both continuously and in episodic floods (Gregory et al., 1991). For example, in the dry interior of the western United States, cottonwoods, willows and other water-tolerant trees and shrubs often dominate the near-stream portion of riparian ecosystems, supplying food resource and shade to the stream. Riparian vegetation also provides coarse woody debris to the stream, and when this is incorporated into streambanks tends to produce deep and narrow channels, and a characteristic pool-to-riffle ratio (McIntosh et al., 1994). The sustained function of this riparian vegetation depends in turn on streamflow, because flooding is often required for seedling recruitment (Mahoney and Rood, 1998), and for incorporation of coarse woody debris into streambanks. Thus complete restoration of a degraded riparian system requires both the establishment of periodic variations in flow, and characteristic vegetation and geomorphology. While complete restoration has rarely been achieved, there are many examples of partial restoration, involving vegetation, geomorphology, and streamflow. We will discuss a sample of these, in the context of the debate on passive vs. active restoration.

Riparian ecosystems in the interior West tend to be highly resilient, due to the presence of water for growing vegetation and for moving sediment, and because their characteristic species are adapted to high levels of natural disturbance (Kauffman et al., 1995). Thus passive efforts, in the form of livestock exclusion or other management, are often all that are required to achieve restoration success (Briggs, 1995). Smith (1989) documented a significant increase in sprouting of cottonwoods and willows in fenced riparian areas in central California after 2 years. Fencing of a riparian area on the Starkey Experimental Forest also allowed recovery of cottonwood, alder, and willows in 2 years time (Kauffman et al., 1995). Platts et al. (1983) recorded significant stream bank improvement in a Nevada riparian area that had been rested from grazing for 5 years. Platts and Rinne (1985) discussed 16 studies that examined the consequences of riparian fencing, all of which measured significant improvement in riparian condition, including decreased stream width-to-depth ratio, increased bank stability and cover, more aquatic plants, and less ungulate damage. Briggs (1992) found that of 27 riparian re-vegetation projects, 19 achieved their objectives, largely due to the exclusion or management of livestock. He argues that active restoration approaches such as planting of vegetation should be carefully evaluated within the context of the whole riparian system, including the identification of the degradation sources and the potential for natural regeneration.

Passive restoration efforts are less likely to succeed if there are numerous factors contributing to riparian degradation, such as the occurrence of invasive plants, the presence of roads in valley bottoms, significant channel incision, and upstream interruption of natural flow. Baird (1989) identified noxious weeds as the greatest challenge in riparian restoration efforts on two rivers in southern California, and discussed the efficacy of several methods to discourage their dominance. Her work demonstrates that active methods to control weeds may have to accompany passive efforts to control other degradation sources, in order for a fully functioning riparian system to be restored. A similar argument can be made for the presence of roads in valley bottoms that serve to increase sediment delivery, and interrupt hydrology by concentrating and diverting flow. Thus passive restoration, such as closing existing roads, may not result in improved condition, because unused roads can still contribute sediment or can fail entirely (Gucinski et al., 2001 also see www.fs.fed.us/pnw/current.htm). On the other hand, recontouring roads by importing material may result in landslides, and obliterating roads may be ineffective in reducing the compaction that results in significant hydrologic interruption (Elliot et al., 1996; Java-Sharpe et al., 1996). Another concern is that streams with severe downcutting may cross a threshold in which the surrounding water table is lowered, and a passive approach will not result in much improvement. A long period of downcutting and widening may ensue before a new floodplain and vegetation are restored (Swanson, 1989). Check dams may help refill small gullies if properly installed, but work in deeply-cut gullies may be futile (Swanson, 1989). Finally, in larger river systems, flood control may constrain reestablishment of natural flow regimes, and thus passive approaches to riparian revegetation may not succeed. Alpert et al. (1999 also see www.fs.fed.us/pnw/current.htm) describe a project on the Sacramento River, in which planting, irrigating, and weed control were required for successful restoration of riparian vegetation. While this aggressive approach has achieved success in reestablishment of vegetation, and in some places improved habitat for birds (King and Geupel, 1997), it remains to be seen whether vegetation will persist for the long term, without the soil-moving force of a dynamic flow regime.

In systems where degradation is severe, active practices have in some cases succeeded in restoring riparian condition. Aggressive restructuring of the channelized Blanco River in southwest Colorado produced new meanders, deeper pools, new flood terraces, and more stable banks within 5 years (Berger, 1992). A case study on McCoy Creek in northeastern Oregon has shown a decrease in surface water temperature shortly after the creek was diverted from a recent ditch to its former channel (Whitney, 2000). The probable cause of this decrease was that cool subsurface waters began to contribute once again to the surface flows. A recent case study on the Rio Grande suggests that managed flooding offers promise in ecosystem restoration, with soil organisms and arthropods responding positively within a single year (Molles et al., 1998). Other components, such as reorganization of forest floor litter, may take many years for significant restoration. Reeves et al. (1997), observed a significant increase in pool–riffle ratio within a few years after boulder and large woody debris placement in a western Oregon stream. Response in numbers of coho and steelhead was mixed, however, although body sizes of juveniles tended to increase after restoration. Road obliteration can be successful in reducing erosion and thus sediment delivery to streams. For example, after heavy rain during the winter of 1995–1996, there were 14 landslides in the Clearwater National Forest but none on the 10 miles of road that had previously been treated (Ann Connor, Clearwater National Forest, Orofino, ID, pers. comm., 28 October 1999). Perhaps the most extensive work has been done in Redwood National Park, where treated road segments contributed about one-fourth as much sediment per mile as untreated roads (Madej, 2001). There is, however, little research to indicate whether benefits of road closures and removals exceed the associated problems (William Elliot, Rocky Mt. Research Station, Moscow, ID, pers. comm., 28 October 1999).

In summary, there is abundant evidence that passive approaches, in which degrading agents are removed or managed, are often successful in restoring riparian condition. Passive restoration may be the first and most important step in riparian restoration, especially in smaller order streams where flow regimes are still intact. Active restoration may be necessary, however, where invasive plants have become established, where channels have been severely incised, where sources of large woody debris have been eliminated, or in larger river systems where flow regimes have been permanently altered. Several authors suggest, however, that watershed-level strategic thinking should be a prerequisite to active restoration, with the degrading agents clearly identified, and the risks of restoration activities carefully explored (National Research Council, 1992; Kauffman et al., 1995; Frissell, 1997).

3. Conclusion

There is no simple answer on how to restore degraded forest, sage steppe, and riparian ecosystems. Enough is known to suggest that practitioners keep both passive and active strategies in the toolbox (Allen, 1995). Thus the proposition that active management approaches are more likely to restore degraded ecosystems in the Columbia River basin is true only under certain conditions. Specifically, when causes of degradation are well understood, the elimination or management of these may be all that is necessary to restore a more desirable condition. This passive approach may be our best option for restoring resilient riparian systems (Kauffman et al., 1995) for those sage steppe communities that are still relatively pristine (West, 1999), and for conifer forests with longer fire-return intervals (Bessie and Johnson, 1995).

In general, because active approaches have inherent costs and risks, the scientific literature tends to recommend a philosophy that passive approaches should be examined first when considering how to restore a degraded system. But because passive approaches have been shown to be ineffective for restoring highly degraded systems (Laycock, 1995), active restoration methods will often be necessary. Examples include riparian systems in which significant channel incision has occurred (Swanson, 1989), stream crossings where fill material was used (Weaver et al., 1987), sage steppe communities that have lost the native perennial grass component (Friedel, 1991; Sanders, 1994), and forests in which fuels have built to uncharacteristically high levels (Arno and Harrington, 1999). In many of these cases, active strategies will require a long-term commitment to be successful. Although the Interior Columbia River Basin SDEIS implies that passive restoration could be an important option in some cases, there is no mention of a process in which passive and active approaches would be evaluated at a given site on the basis of their relative costs and risks.

Throughout the literature there is general agreement that true restoration requires not only the reestablishment of "historical" structures and species compositions, but of the processes needed to sustain these into the future (Whisenant, 1995; Beechie and Bolton, 1999). This amounts to a call for strategic thinking, in which the restoration problem is considered within its full ecological context. Thus Alpert et al. (1999)

describe their work on riparian forest restoration of the Sacramento River, within the context of the constrained flow regime mandated by society, in order to discover whether or not the structure (riparian forest) can be sustained without the process (periodic flooding). With emphasis on restoration of dynamic processes in aquatic ecosystems, the SDEIS recognizes the link between process and structure, especially with respect to the maintenance and connectivity of fish strongholds. Similarly, fuel reduction in western dry forests is often discussed as a precursor to the reintroduction of fire. The concept is that the modified structure can better accommodate the processes that are thought to produce and maintain it (Arno et al., 1996). Forest restoration objectives in the SDEIS alternatives S2 and S3 emphasize conservation of source habitat in the short term, and active repatterning of habitat in the long term to recruit additional source habitat, and to make structure more compatible with the historical disturbance regimes. Thus while active modification of structure is the focus, language in the SDEIS implicitly recognizes the linkage between process and structure in forest ecosystems. The interaction between structure and process is even more evident in the sagebrush steppe ecosystem, where cheatgrass has not only pushed aside native vegetation, but has altered the fire regime as well. Reintroduction of the process of fire itself will clearly not restore historical structure or composition in sagebrush steppe, without first eliminating or reducing cheatgrass (Stewart and Hull, 1949; Monsen, 1994). While the SDEIS emphasizes the importance of native species and of functional components (e.g. cryptobiotic crusts), there is no clear recognition that restoration of sagebrush steppe systems will require a careful consideration of how the interaction between process and structure has been altered in recent times.

While society must ultimately decide whether or not an active strategy is worth the cost, research is needed to predict the likelihood of success and the degree of risk. Yet, while research has been very effective at identifying degraded systems and the causes of the degradation, it has also shown that there is a high degree of variability in response to restoration activities, largely because of site-specific idiosyncrasies (Tausch et al., 1995). Thus research in its traditional form may not be as useful in predicting success or risk in local situations as adaptive management, in which alternative restoration tactics are applied, and their results measured, compared, and documented. The SDEIS alternatives do emphasize monitoring, evaluation, and adaptive management as part of an outcome-based, site-specific process that would ostensibly be used to develop and improve restoration strategies.

It could be argued that for the most part, degradation has been brought about by the alteration of disturbance regimes. The conflict between short-term, local human needs and large-scale and occasionally catastrophic disturbance events will always be difficult to resolve. The challenge for the restorationist is to find a way to restore more desirable conditions within the context of societal constraints that limit how processes are allowed to operate. Landscape thinking, in which humans are a central component in a shifting mosaic through time, will be increasingly necessary in the future, if true ecological restoration is to be achieved (Allen, 1995; Whisenant, 1995).

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