Restoration Notes

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Passive Soil Manipulation Influences the Successional Trajectories of Forest Communities at a Denuded Former Campsite

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Passive site manipulation is an ecological restoration approach that strives to improve heavily degraded sites by altering physical conditions to encourage the establishment of autogenic ecosystem processes. Passive restoration approaches have been successfully used in numerous ecosystems to alter soil nutrition, moisture retention, and propagule retention, which can drive both the establishment of vegetation and plant successional trajectories (Ewing et al. 2002, Cole 2007, Hough-Snee et al. 2011). In some ecosystems, increasing plant recruitment from seed rain and/or intentional seeding through microsite enhancement may provide an effective low-investment, low-intensity revegetation strategy to land managers when unmediated, 'natural' revegetation is unfeasible due to poor site conditions (Cole 2007). One application of passive restoration techniques is in heavily impacted sections of natural areas where the surrounding vegetation matrix is strongly intact but ecosystem processes may act too slowly to reclaim the disturbed site or prevent further degradation. Active planting approaches may be logistically difficult or prohibitively expensive in some natural

areas, leaving passive site manipulation as the only feasible option. In 2006, we used low-investment surface soil treatments to retire a denuded former campsite and restore the resulting disturbed area to local forest vegetation. This project was an opportunity to test the effectiveness of 3 seedbed treatments in recruiting tree species from seed rain and to concurrently assess forest ecosystem development over time.

Our project site is adjacent to the North Cascades Environmental Learning Center (ELC) on the northern shore of Diablo Lake in the North Cascades National Park Complex (Marblemount, Washington, USA). The ELC was developed on the site of a former fishing camp where after camp removal in 2004, the site was used for construction material storage, resulting in highly compacted soils, denuded entirely of vegetation and soil organic matter. The surrounding forest occurs in the northern reach of Washington's western hemlock (*Tsuga heterophylla*) zone (Franklin and Dyrness 1988) and is dominated by western hemlock, Douglas-fir (*Pseudotsuga menziesii*), lodgepole pine (*Pinus contorta* var. *latifolia*), red alder (*Alnus rubra*), bigleaf maple (*Acer macrophyllum*), and paper birch (*Betula papyrifera*). Forest canopy near the ELC ranges from 70–90% cover in most areas, with the largest canopy gaps occurring adjacent to roads. The ELC is situated at an elevation of ≈ 370 m with a southwest aspect and slopes ranging from 3–5%. Outside of the immediate 1-ha restored area, surrounding forest hydrology, soils and vegetation have remained largely undisturbed.

We anticipated that the compacted ELC site would benefit from soil treatments of different source materials and particle size classes that promoted heterogeneous seedling establishment across the site. More specifically, we hypothesized that the dominant tree species establishing within each treatment would correspond to the soil treatments most conducive to their seed size and type. We used a completely randomized block design to evenly allocate 3 soil treatments, scarified mineral soil, forest litter, and woodchip mulch, to 120 1m $\times$ 1m plots across 3 blocks. The scarified soil treatment (S) consisted of post-denudation mineral soils (particle size <0.5cm) free of organic matter that were raked and spread evenly across each plot. The forest litter treatment (FL) contained needles, seed cones, lichens, and small twigs collected from accumulations on nearby maintained campsites (particle size $\approx$ 1–3cm). We

Table 1. Combined 2007 and 2010 indicator species analyses for woody species community composition at the Environmental Learning Center restoration site within North Cascades National Park Complex. Species not included for a given year were not found within plots in that year. * indicates significant differences.

Species (<i>Taxonomic name</i>)	2007 Indicator Species			2010 Indicator Species		
	Treatment	Indicator Value	<i>p</i> - value	Treatment	Indicator Value	<i>p</i> - value
Red alder (<i>Alnus rubra</i>)	S	35.62	0.004*	S	37.96	0.015*
Black cottonwood (<i>Populus trichocarpa</i>)	S	5.76	0.976	S	3.00	0.690
Big-leaf maple (<i>Acer macrophyllum</i>)	FL	12.85	0.025*	S	21.81	0.177
Western hemlock (<i>Tsuga heterophylla</i>)	FL	36.29	0.001*	FL	24.17	0.002*
Paper birch (<i>Betula papyrifera</i>)	FL	14.22	0.157	FL	10.44	0.795
Western redcedar (<i>Thuja plicata</i>)	FL	9.68	0.102	FL	11.11	0.034*
Redstem ceanothus (<i>Ceanothus sanguineus</i>)	—	—	—	FL	9.89	0.198
Douglas maple (<i>Acer glabrum</i>)	—	—	—	FL	9.13	0.933
Serviceberry (<i>Amelanchier alnifolia</i>)	—	—	—	FL	6.42	0.692
Vine maple (<i>Acer circinatum</i>)	—	—	—	FL	2.81	0.889
Dwarf rose (<i>Rosa gymnocarpa</i>)	FL	3.23	1.000	—	—	—
Oceanspray (<i>Holodiscus discolor</i>)	FL	3.23	1.000	—	—	—
Douglas-fir (<i>Pseudotsuga menziesii</i>)	FL	7.95	0.942	WM	39.93	0.017*
Low Oregon grape (<i>Mahonia nervosa</i>)	FL	20.54	0.125	WM	15.12	0.161
Lodgepole pine (<i>Pinus contorta</i>)	WM	20.33	0.036*	WM	11.79	0.158

spread litter evenly over scarified denuded mineral soil to a depth of ≈ 2 cm. The woodchip mulch treatment (WM) was comprised of shredded red alder wood chips (particle size ≈ 2 –5 cm) that were applied directly to existing scarified mineral soil at a depth of 2 cm (Table 1).

We monitored natural vegetation recruitment into these sites in 2007 and 2010 by collecting tree and shrub abundance data, counting individual stems of all woody species within each plot, measuring woody species stem height, and estimating percent cover of herbaceous vegetation and bare ground. In 2007, total sample size was reduced from 120 plots to 101 plots when a fallen tree destroyed 19 plots. Because our objective was to identify patterns in potential canopy tree species, and herbaceous cover commonly totaled $< 5\%$, we excluded herbaceous vegetation from our analyses. We used PERMANOVA, a permutation-based ANOVA test that uses distance matrices to assess differences between community groups (Anderson 2001), to compare

the vegetation communities between treatments and to perform between-group pairwise comparisons. To illustrate what species drove differences in community composition, we used indicator species analysis (ISA; Bakker 2008) to define species indicative of each treatment. ISA calculates the product of each species' relative frequency and relative abundance in a given treatment and returns an indicator value between 0 and 100. As a species' indicator value approaches 100, it is thought to be a more perfect indicator of a given treatment. Each species' indicator value is statistically assessed using Monte Carlo randomizations to return a *p*-value. We used 10,000 permutations, constraining permutations within each block, to derive *p*-values for both PERMANOVA and ISA analyses. We performed analyses independently for years 2007 and 2010 using R statistical software (version 2.11.1) and considered a species' indicator value (IV) to be statistically significant and PERMANOVA groups to significantly differ if $p < 0.05$.

Figure 1. a) Mean stem density per plot for the 4 statistically significant indicator species found using indicator species analysis in 2010. b) Mean stem heights of the 4 indicator species from 2010. Mean heights only differed between groups for red alder (*Alnus rubra*) (ANOVA and Tukey HSD $p < 0.05$) and are denoted by grouping letters. Note that western hemlock (*Tsuga heterophylla*) was absent from the S treatment while redcedar (*Thuja plicata*) was only found in the FL treatment in 2010.

Soil treatments effectively drove regenerating forest composition in both 2007 ($F = 5.6$, $p < 0.01$) and 2010 ($F = 3.3$, $p < 0.01$), while block did not have a significant effect. Community composition in the S treatment significantly differed from both the FL and WM treatments in both 2007 (S—FL: $F = 2.65$, $p < 0.01$; S—WM: $F = 5.56$, $p < 0.05$) and 2010 (S—FL: $F = 7.51$, $p = 0.03$; S—WM: $F = 7.86$, $p < 0.01$) while the FL and WM treatments' community composition did not significantly differ from one another in either year (2007: $F = 0.27$, $p = 0.65$; 2010: $F = 1.13$, $p = 0.26$). ISA revealed that the community differences between the S and the WM and FL treatments were driven by strong red alder abundance within S in both years (2007: $IV = 35.62$, $p = 0.004$; 2010: $IV = 37.96$, $p = 0.015$). In 2007, 1 year after restoration, bigleaf maple ($IV = 12.85$, $p = 0.025$) and western hemlock ($IV = 36.29$, $p = 0.001$) were the strongest indicator species for the FL treatment while lodgepole pine ($IV = 20.33$, $p = 0.036$) was the strongest indicator of the WM treatment. The indicator species for each group changed over time, but only FL and WM exhibited different indicator species in 2010 than in 2007. In 2010, bigleaf maple shifted to become a weak indicator in S group while western redcedar (*Thuja plicata*; $IV = 11.11$, $p = 0.034$) joined western hemlock ($IV = 24.17$, $p = 0.002$) as a significant indicator species. In 2010, the WM group showed higher abundances of Douglas-fir ($IV = 39.93$, $p = 0.017$), a weak FL indicator in 2007, and replaced lodgepole pine as the treatment's only significant indicator species.

Seedbed manipulation to promote seed recruitment and seedling establishment has historically served as a primary tool to overcome thresholds for reinitiating ecosystem processes on damaged sites (Luken 1990, Whisenant 1999). At ELC we observed that when intentional seeding proves to

be impractical and the damaged site occurs within an intact matrix of reproductively mature species, applying specific seedbed treatments to select for and promote desired species offers a viable passive restoration approach that can reinitiate ecosystem processes and set early-successional forest trajectories. All 3 treatments clearly demonstrated that enhancing soil surface roughness successfully recruits canopy and woody understory species from seed rain (Table 1). Simple soil scarification, either as a single treatment or when combined with light applications of either woodchip mulch or forest litter, retained seed, promoted germination, and allowed for plant establishment and survival 3 yr after application. Furthermore, varying soil surface treatments proved to be strongly selective and significantly drove species composition both 1 and 4 yr following restoration.

The change in indicator species over a brief 3-yr period suggests that seedbed conditions exert a strong selective pressure during colonization but interact with environmental filters (light availability, herbivory, soil moisture and temperature, etc.) and species performance to determine which species persist over time. We expect that highly shade intolerant red alder (mean 2010 height = 81.92 cm), though the most common recruit in the S treatment and overall, will not be recruited into the overstory canopy because the experimental blocks occupy small ($< 500 \text{ m}^2$) canopy gaps. Similarly, shade intolerant lodgepole pine (mean 2010 height = 19.06 cm) and Douglas-fir (mean 2010 height = 11.04 cm) may be suppressed by the taller, more densely colonizing red alder (Figure 1). Given the existing canopy gaps in the experimental blocks, we expect that shade tolerant western redcedar and western hemlock are the most likely species to persist long enough to reach the canopy. Subsequently, other factors such as ungulate herbivory, local climate and fire cycles, and pathogen

outbreaks may further determine which recruited species reach the forest canopy. We conclude that selecting a low-investment seedbed preparation to recruit target species from seed rain must be done with both the knowledge of target species seedbed preferences and acknowledgement that initial treatments may not allow recruited species to persist to maturity. Additional treatments may need to be undertaken to shape selective forces that enhance the likelihood that target species will persist to maturity and continue to drive forest vegetation communities.

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Ibity Mountain, Madagascar: Background and Perspectives for Ecological Restoration

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For decades, Madagascar has drawn scientific attention for its unique biodiversity and high levels of endemism. This island nation has long been recognized for its remarkable biota, which has evolved during nearly 100 million yr of tectonic isolation, with a diverse range of climates and substrates, and a varied topography (Kull 2000). Current estimates suggest that the Malagasy flora consists of approximately 13–14,000 species, more than 90% endemic (Phillipson et al. 2006), which occur in a wide range of native vegetation types, many of which have been highly degraded or replaced by secondary formations resulting from human activities (Lowry et al. 1997).

Tapia woodland is a sclerophyllous formation restricted to Madagascar's central highlands, dominated by the endemic tapia tree (*Uapaca bojeri*). This formation is adapted and resilient to the region's natural fire regime and occurs today in scattered, isolated stands that total no more than 132,255 ha. Most other native woody vegetation types of the central highlands have been replaced by grasslands formed during the last 1,500–2,000 yr primarily by a human-caused increase in fire frequency well above that of the natural regime. These grasslands are burned annually to renew livestock pasture, clean fields, or control pests (Kull 2000), and fires encroach well into the interior of the tapia woodland. Today the remaining tapia woodland has an extensive herbaceous layer and an open canopy with many woody representatives of the endemic family Sarcocaulaceae (Lowry et al. 1997, Figure 1). Traditional management practices appear to have spared remnants of tapia woodland because this vegetation plays an important role in the local economy as a source of fuel wood, edible fruits, mushrooms and medicinal plants, and as habitat for wild silk worms (*Borocera madagascariensis*), whose cocoons are collected by local residents to produce cloth (Gade 1985, Kull 2002). However, the remaining stands are fragmented and structurally-modified by a changing fire regime that hinders woody species regeneration. Tapia woodland was once managed using traditional methods, which allowed sustainable extraction of forest resources for local consumption. Today it is still partly managed using traditional methods, but newer practices have caused degradation through increased cultivation and grazing, illegal exploitation and marketing of natural resources, reforestation with exotic tree species such as swamp mahogany (*Eucalyptus robusta*), Baguio pine (*Pinus*