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Ecosphere

Post-landslide soil and vegetation recovery in a dry, montane system is slow and patchy

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Abstract. Landslides are common disturbances in forests around the world, and a major threat to human life and property. Landslides are likely to become more common in many areas as storms intensify. Forest vegetation can improve hillslope stability via long, deep rooting across and through failure planes. In the U.S. Rocky Mountains, landslides are infrequent but widespread when they do occur. They are also extremely understudied, with little known about the basic vegetation recovery processes and rates of establishment which restabilize hills. This study presents the first evaluation of post-landslide vegetation recovery on forested landslides in the southern Rocky Mountains. Six years after a major landslide event, the surveyed sites have very little regeneration in initiation zones, even when controlling for soil coverage. Soils are shallower and less nitrogen rich in initiation zones as well. Rooting depth was similar between functional groups regardless of position on the slide, but deep-rooting trees are much less common in initiation zones. A lack of post-disturbance tree regeneration in these lower elevation, warm/dry settings, common across a variety of disturbance types, suggests that complete tree restabilization of these hillslopes is likely to be a slow or non-existent, especially as the climate warms. Replacement by grasses would protect against shallow instabilities but not the deeper mass movement events which threaten life and property.

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INTRODUCTION

Disturbances are ubiquitous in the world's forests (Hansen et al. 2013). Agents such as wildfires, windstorms, and landslides regularly cause widespread mortality, with their relative importance depending on biome, location, topographic setting, human exposure, and site history. In steep forested terrain, mass movements (landslides, debris flows) are not uncommon over long timescales (Walker and Shiels 2012). Landslides and debris flows (hereafter denoted as "slides") occur wherever soil overlays steep topography such that triggering events, such as heavy rainstorms, can reduce factor resisting gravity, like soil cohesion and friction with the underlying bedrock. It is difficult to estimate their frequency, given the difficulty of mapping via remote sensing and the often-remote terrain in which they occur. Estimates are that >50% of terrestrial landscapes are exposed to landslide events, and 17.8% are highly susceptible, and 5% of that very highly exposed (excluding snow and ice regions; Hong et al. 2007). Within landscapes exposed to landslides, >50% can be impacted in any given event (Walker and Shiels 2012).

Landslides are a major natural hazard. Reported landslides are typically biased toward

human settlements (Kirschbaum et al. 2015) and so total rates are likely underreported. Even so, Petley (2012) found reports of 32,322 fatalities over seven years, a high rate which disproportionately impacts poorer regions of the globe (Froude and Petley 2018). Infrastructure is also frequently impacted, with upwards of one billion in damages annually in the USA alone (Dale et al. 2001).

Climate change is expected to drive increased landslide rates due to more frequent storms at higher intensity (Dale et al. 2001), which historically correlate with higher landslide activity (Borgatti and Soldati 2010, Kirschbaum et al. 2010, Zerathe et al. 2014). Local studies suggest that increased precipitation intensities will drive substantial increases in landslide rates (Jakob and Lambert 2009, Biasutti et al. 2016, Tichavský et al. 2019). Broadly, the majority of the United States is expected to see heavier rain events (Easterling et al. 2017), and landslide frequency is likely to increase at global scales (Kirschbaum and Stanley 2018).

Plant communities on hillslopes are increasingly recognized as important factors in landslide occurrence, both directly limiting their occurrence (Roering et al. 2010, Corenblit et al. 2011, Rengers et al. 2016) and indirectly via interactions with other weather phenomena like windstorms (Buma and Johnson 2015). Vegetation, especially forests, plays an important role in hillslope stabilization (Ziemer 1981, Marden 2004, O'Loughlin 2005, Rickli and Graf 2009, Pawlik 2013, Amundson et al. 2015). The mechanical reinforcement of soil by tree roots is well established (Cohen and Schwarz 2017, Vergani et al. 2017), as is the importance of regenerating vegetation to hillslope stabilization post-disturbance (e.g., logging: Sidle 1992). Long, extensive roots provide support by crossing of zones of weakness in the soil column and anchoring into cracks in stable bedrock (Roering et al. 2003, Kuriakose et al. 2011, Cohen and Schwarz 2017). Three different root reinforcement mechanisms are generally considered as follows: (1) basal and (2) lateral root reinforcement, and (3) the soil mass stiffening by roots (see Cohen and Schwarz 2017). These effects are easily observed via the increase in landslide activity after forest clear-cuts (Guthrie 2002, Glade 2003, Goetz et al. 2015) and through interactions with other disturbances (Geertsema and Pojar 2007, Buma and Johnson 2015). Other significant ways in which

trees stabilize hillslope include changes in hillslope hydrology due to interception and evapotranspiration (delaying a watershed response on heavy rainfall), thick forest litter layer and dam-like effect caused by logs limit soil erosion and sediment transfer to the fluvial system (rivers and streams), and reductions in changes in soil creep intensity (Pawlik and Šamonil 2018).

Unfortunately, while the immediate effects of forests on landslides are well established, post-landslide succession and restabilization are understudied, especially in systems where slides are sporadic and/or infrequent (but not necessarily unimportant). In many areas, slides may impact a large fraction of the region but primarily occur during infrequent weather events (e.g., 1000-yr storms); as a result, landslides do not necessarily attract attention given their seeming rarity—which may be an illusion due to their sporadic temporal nature. Yet the long-lasting implications of landslides (mass removal and translocation of soil, initiation of primarily succession; Walker and Shiels 2012) and the implications of potential increases in the future due to increases in storm/precipitation intensity (Jakob and Lambert 2009) require a strong understanding of how those ecosystems respond and recover from landslides. From a vegetation perspective, the regeneration timelines of plants and the development of roots in landslide initiation zones need to be understood, because that controls the reestablishment rate of biotic stabilization—critical when the drivers of landslides are increasing. Other disturbances may be illustrative; numerous studies in similar dry, warm environments have raised concerns that forest regeneration is becoming climatically limited in lower treeline montane systems (Rother et al. 2016, Stevens-Rumann et al. 2018). This research suggests that forest recovery from landslides—critical to restabilization of steep hillslopes and prevention of future landslides—may be limited by climate as well. But this has never been tested in an actual landslide system.

In this study, we investigated post-landslide forest recovery in a dry montane system (in particular, the U.S. southern Rocky Mountains). These are especially understudied systems for landslide ecology; the authors are not aware of any similar work in these environments. In these steep, mountainous regions, landslides coincide

with rare triggering weather events—thousands of slides can occur in a short period of time (e.g., Coe et al. 2014). Often, infrastructure and populations are not prepared for landslide events (likely due to their seeming rarity), adding to their impact (Regmi et al. 2014). Linking to the points above, we are interested in (1) types and densities of plant regeneration in both initiation zones vs. accumulation zones and both (2) biomass accumulation (aboveground and belowground) and (3) root length in those areas after a slide. The main need for information relates to the steep initiation zones and the growth of vegetation in those areas, important for stabilization and recovery (Walker et al. 2009; note that here, we are considering stabilization of the existing scar, not the broader hillslope). The accumulation zone provides an important comparison point, one that is stable and experiences the same climate and seed availability. For example, initiation zones may have lower regeneration for several reasons, such as thinner soils, slower growth, or ongoing instability. Each would manifest differently. If soil coverage is the limiting factor, then controlling for soil cover should make initiation and accumulation areas appear similar in terms of regeneration. If growth is limiting (e.g., due to biogeochemical differences; Shiels et al. 2008), than root/shoot lengths and masses should differ between sites. If stability is an issue, then differences in regeneration densities should differ even after controlling for available soil, but length and masses should be similar for those plants that do establish.

Study Site

The study area is in Boulder County, Colorado. Topography is steep, and regional climate is generally dry (~520 mm precipitation annually), though convection events can result in high amounts of precipitation in a short period of time. Ponderosa pine (*Pinus ponderosa*) dominates the forested area, with a smaller proportion of juniper (*Juniperus* spp.) and Douglas fir (*Pseudotsuga menziesii*), typically in draws and gullies. Although not directly assessed, we suspect the forest is the result of a historic wildfire, given the similar sizes of the trees and the general lack of fire scars on the mature individuals. Tree cores taken for a related study indicate the forest is at

least 150 yr old. The area was subject to an extreme rainstorm 11–13 September 2013, which triggered >1100 mass movement events (Coe et al. 2014) and resulted in the deaths of 8 people. Damage to infrastructure was immense (Godt et al. 2014). Slides were biased toward areas with less vegetation (Rengers et al. 2016) regardless of aspect, supporting the hypothesis that forests stabilize hillslopes in this area. Plots were therefore located in forested areas that did slide, to look at the initial ecological response in areas formerly stabilized by the trees. Forests on the study site were stable for at least 80 yr based on aerial photographs from 1941 till the 2013 event (data not shown). Plots were constrained to the same aspect (northwest; found to control soil moisture, Ebel et al. 2015), solar insolation, elevation, and slope angles as a means to limit covariance with other factors such as soil moisture or forest type.

Methods

Four slides were investigated. Two slides were surveyed in both their initiation zone and their accumulation zone. The third slide was investigated in its initiation zone only, and the fourth in its accumulation zone only (due to private property and safety concerns, respectively), for a total of three intensive slide initiation surveys and three accumulation zone surveys. A neighboring intact forest area with the same aspect, slope, and elevation was surveyed as a reference. The initiation zone was defined by the obvious loss of soil material and presence of exposed bedrock, the accumulation zone via the obvious presence of colluvial material and uprooted trees overlaying the original soil horizons. Only disturbed sites were considered in the accumulation zone (e.g., no undisturbed islands of soil were considered).

Ground cover is dominated by coarse rock fragments from ongoing upslope cliff erosion, with overall soil depths reaching 2 m. The substratum is mainly sandstones and conglomerates. Three main soil horizons can be distinguished (Fig. 1). The A horizon is frequently thin, up to 10 cm. Where it exists (e.g., apart from the surficial rock fragments), the O horizon is minimal. Soil material is generally well mixed with no signs of larger clast orientation (which could

Fig. 1. Soil profile along one of the landslides. Depth from the bedrock to the top of the slide is approximately 1.5 m. A thin O horizon is found between the surficial rock fragments. A steep cliff upslope provides a continual (though presumed slow) influx of rock material to the site.

suggest soil creep). Below the upper layer of coarse rock fragments the A horizon changes gradually into the B horizon. It was frequently difficult to distinguish between the B and C horizons because of lack of well-developed soil horizonation. This also indicates that the soils are relatively young. Rock fragments were found in all horizons. Tree roots grow vertically crossing all horizons and sometimes enter bedrock fractures.

Vegetation sampling

For each survey, the same general protocol was utilized. Five cross-slope transects were established in the landslide or intact forest at various locations (hereafter "treatments"); for initiation zone transects ($n = 15$), this extended from edge to edge (ranging from 11 to 19 m); for the accumulation zone ($n = 15$) and reference forest transects ($n = 10$), the length was set at 10 m. Elevation, aspect, and slope were comparable between zones: initiation zones averaged 299° aspect, 32° slope, and 1845 m elevation; accumulation zones averaged 294° aspect, 27° slope, and 1803 m elevation; and the reference forest averaged 315° aspect, 29° slope, and 1832 m elevation.

A point-intercept method was used to survey vegetation every meter, starting at the zero-point. Plant functional group (annual, perineal, graminoid, shrub, or litter), height, rooting depth (by carefully extracting the plant from the soil), substrate, and soil depth (to a max of 30 cm) were recorded. Note this was not necessarily depth to bedrock, but rather depth to the first solid rock layer encountered. This was done to assess initial root zone conditions for establishing plant. Soil was defined as unconsolidated <2 mm material. Tree species regeneration was counted along the transect, using a belt survey that extended 1 m to either side of the transect tape. Density was calculated for each transect as the sum of all seedlings tallied divided by the area of the belt transect.

Root and plant data

To look for differences in potential restabilization of initiation zones as a function of plant establishment, we destructively sampled regeneration. At ten random 1-m^2 locations within the slide initiation zone, plants were harvested to evaluate root/shoot length and root/shoot mass. All plants were taken to avoid size bias. Plants

were dried at 60°C for 48 h and then measured for length and mass of both above and belowground parts. Because of their low density, tree seedlings were opportunistically sampled outside of those areas. Plants were representative of the broader ecosystem and comprised four functional groups: tree seedlings, grasses, herbaceous forbs, and small shrubs. We compared root mass and root length as a function of overall mass/length (see below) to estimate differences in stabilization potential.

Biogeochemistry

Three soil samples were randomly collected along each transect within both the initiation and accumulation zones ($n = 15$ in each). Soil was dried at 60°C for 48 h, sifted through a 2-mm mesh to remove rocks, roots, and other large particles, and ground for 48 h on a roller mill. The soil was then analyzed on a Costech EA8020 elemental analyzer for total carbon and nitrogen concentrations (980°C combustion, 650°C reduction temperatures). Replicates were run on every 5th sample to ensure adequate homogenization of the ground soil.

Statistical analyses

Given the high variability at fine scales, individual point samples (soil presence, depth, vegetation height, rooting depth; $n = 546$) were treated as independent within each treatment. This was tested by estimating the autocorrelation function (Venables and Ripley 2002, as implemented via the acf function in R); there was no apparent pattern of autocorrelation ($P > 0.05$) within any sample type along each transect. Because regeneration was surveyed along the belt transect and not points, those data were analyzed at the transect level ($n = 40$, size varied from 20 to 38 m^2).

Soil depth measurements had a high number of zero values due to extensive bedrock and surficial colluvial rocks. To test differences between locations, we took a two-step approach recommended by Fletcher et al. (2005) similar to a conditional model for count data (Welsh et al. 1996). First, we modeled the presence of soil using logistic regression, then we compared depths in locations with soil using Tukey-Kramer HSD test (assumptions checked). This has the advantage of allowing us to check soil presence and depth

separately while avoiding the known problems of non-count zero-inflated data on both standard parametric and non-parametric models (McElduff et al. 2010).

Root mass and length percentages were analyzed via a two-way ANOVA across functional groups; if significant differences were found, Tukey HSD tests were conducted to identify between group differences. All analyses were conducted in R (R Core Team 2018).

RESULTS

The landslides varied in depth from ~1 to 2 m throughout the initiation zone. There were significant differences ($P < 0.05$, logistic regression) in soil coverage in the initiation zone (47% of points had soil, $n = 85$), the accumulation zone (63%, $n = 104$), and the reference sites (77%, $n = 127$). Where soil was found, depth to an impermeable surface (typically a large rock fragment) varied between 0.5 and >30 cm. Those depths were significantly deeper in the accumulation zone (13.30 cm) compared with either the initiation (10.62 cm, $P = 0.02$) or the reference plots (10.64 cm; $P = 0.04$), which were not significantly different from each other (Fig. 2). When only considering areas with soil, the accumulation zone was mostly densely vegetated (86%), though not significantly more than the reference sites (75%; $P = 0.23$). The initiation zone had the least amount of cover on available soil (64%), significantly less than the accumulation zone ($P = 0.0005$) but not significantly different ($P = 0.14$) from the reference sites.

Of the regenerating vegetation, plants (average for all types) were largest in the accumulation zone (mean height = 33.0 cm) significantly taller than both the initiation (mean h = 22.6 cm) and the reference plots (mean h = 23.3 cm) which were not significantly different from each other. This pattern was not reflected in the rooting depths (Table 1), which had no significant differences between vegetation in the accumulation zone (average for all types; mean length = 12.9 cm) and the initiation (mean length = 11.1 cm).

Substrate

Substrate differed between the three zones (Fig. 3), with the initiation zone having substantial amounts of exposed bedrock (19%), not
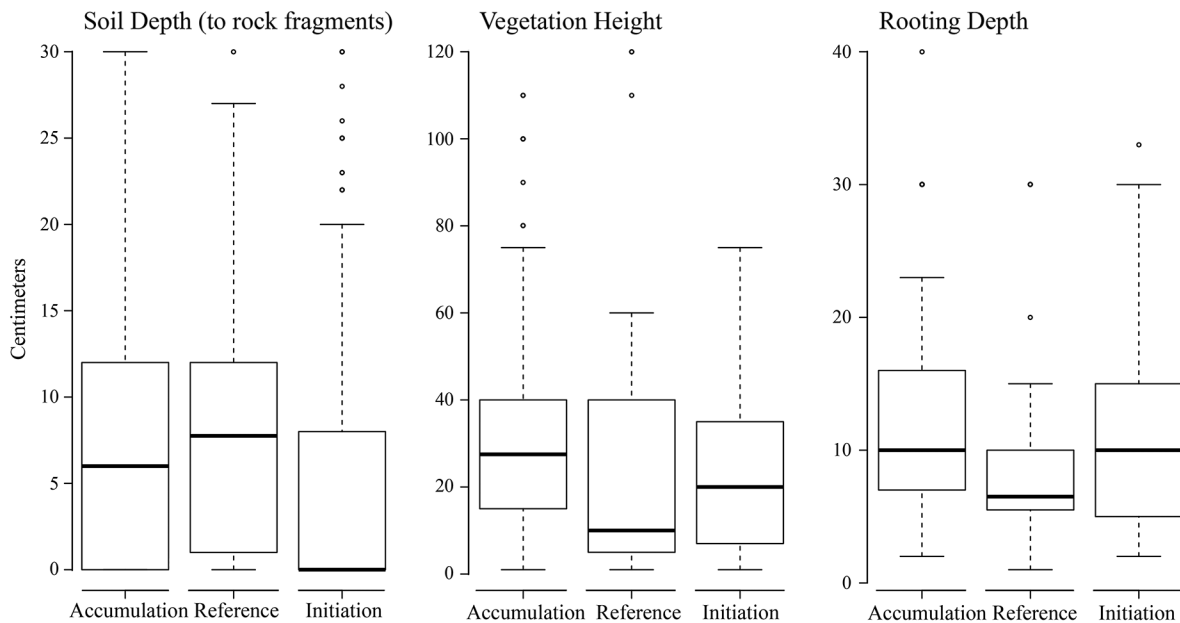
Fig. 2. Comparison of soil depth, vegetation height, and vegetation rooting depth (cm) across the initiation, accumulation, and undisturbed zones in the study area.

Table 1. Heights (cm), root lengths (cm), and ratios for functional groups on random transects across the landslides in the initiation, accumulation, and reference zones.

Zone	Functional group	Height (SD)	Root length (SD)	Height/root ratio (SD)
Accumulation	Forb	20.8 (16.3)	9.5 (4.6)	2.1 (1.1)
	Grass	49.8 (28.5)	13.5 (8.8)	4.2 (2.3)
	Shrub	20.1 (18.2)	17.2 (9.6)	1.3 (0.7)
Initiation	Forb	15.2 (15.5)	7.8 (4.3)	1.6 (1.2)
	Grass	31.0 (13.8)	9.9 (5.3)	3.9 (2.8)
	Shrub	21.9 (19.9)	15.8 (10.2)	1.5 (1.3)
Reference	Forb	27.3 (39.8)	9.6 (9.3)	2.6 (1.3)
	Grass	37.3 (20.6)	8.1 (2.6)	4.9 (2.8)
	Shrub	19.0 (20.1)	12.7 (9.5)	1.6 (1.5)

found on the other two site groups. This difference is primarily explained by lower soil cover in the initiation zone (48% vs. 62–65%). The fraction of the ground covered with large rock fragments was consistent across all cover types (31–35%). Dead wood was a minor component of the accumulation zone (6%) and not present in substantial amounts in the other site groups. Nitrogen content in both the initiation and the accumulation zone was generally below detection limits (approximately <0.03%). As a result, statistical comparisons were not made, though the accumulation zone did have more samples with

detectable N (4 out of 15). Carbon contents were ~4× higher in the accumulation zone than the initiation zone (1.608% vs. 0.454%, respectively; unpaired t-test, square root transformed: $P < 0.05$, $t = 4.59$, df = 26.3).

Comparison of regeneration densities and heights

Regeneration was significantly lower in initiation (0.02 stems/m^2) and reference zones (0.01 stems/m^2) relative to the accumulation zone (0.21 stems/m^2; $P < 0.05$). These differences were retained even after controlling for the differences in relative amounts of soil coverage (initiation
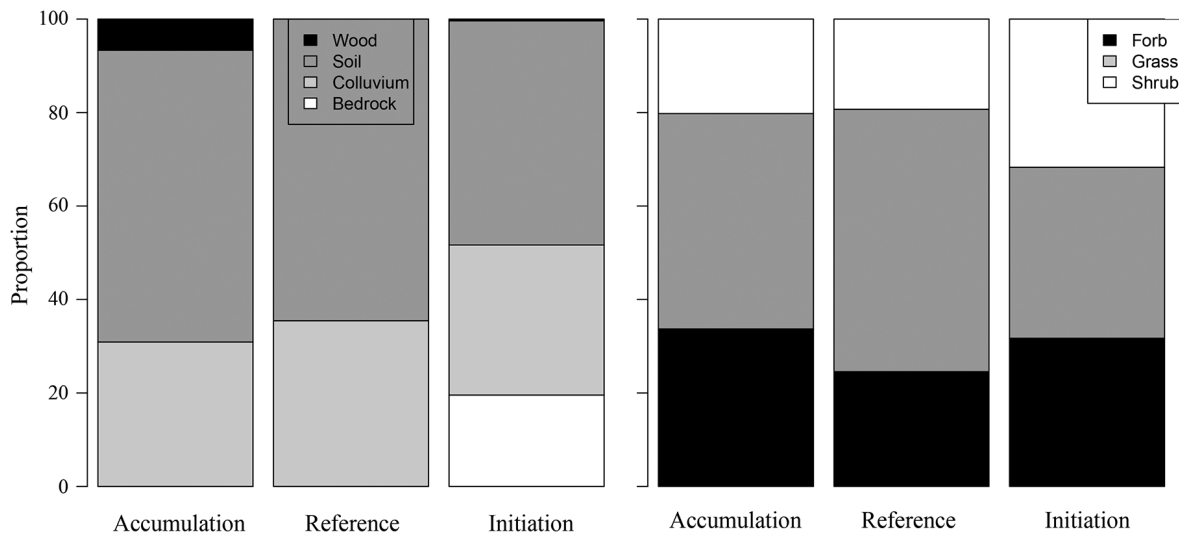
Fig. 3. Proportion of substrate coverage and functional group coverage. (Left) Substrate: Soil refers to any covering of material <2 mm in diameter, regardless if undisturbed (reference) or modified by the landslide event. (Right) Functional group: non-vegetated and moss-covered points ($n = 2$) excluded. Points with ferns ($n = 2$) included in forb coverage.

zone = 0.01 stems/m^2 of soil; reference zone = 0.004 stems/m^2 of soil; accumulation zone = 0.14 stems/m^2 of soil), highlighting that available soil substrate is not limiting establishment. In both the initiation and the reference zones, regeneration was exclusively *P. ponderosa*. Scattered *P. menziesii* was found in the accumulation zone (9% of all seedlings). Mean height between the initiation (16.2 cm) and accumulation (15.5 cm) was similar, and both were significantly less than the reference sites (47 cm). Proportions of functional groups were broadly similar across treatments, with slightly (non-significant) more shrub and less grass coverage in initiation zones (Fig. 3).

Root mass comparison

A total of 81 individual plants were excavated: 20 grass individuals, 17 conifers, 25 small woody shrubs, and 19 forbs. Heights ranged from 3.8 to 71.1 cm (mean = 22.9), and root lengths ranged from 0.75 to 29.4 cm (mean = 12.9). Aboveground dry mass per individual ranged from 0.02 to 28.1 g (mean = 3.3); belowground dry mass per individual ranged from 0.01 to 8.0 g (mean = 1.1; variation between individuals, and thus the range, is naturally high due to the various sizes of plants, especially in perennial plants

with different establishment years). Shrubs had significantly less mass as a percentage of overall mass allocated to their roots than both grasses (10% lower, $P = 0.04$) and forbs (16% less, $P < 0.001$), and less than conifers though not significantly so (6% lower, $P = 0.40$; Fig. 4). Length of roots as a percentage of total height was also different across groups. Shrubs had significantly longer roots (proportionally) than grasses (17% longer, $P < 0.001$), but were not significantly different from either conifers (shrubs 10% shorter, $P = 0.09$) or forbs (shrubs 4% longer, $P = 0.65$). Grasses were also significantly shorter than forbs (12% less, $P = 0.003$). Conifers had the longest proportional root lengths overall (Fig. 4).

DISCUSSION

The significance of landslides over broad time-scales is well established. But in locations where triggering events are infrequent, their sporadic nature means they are often understudied despite the broadscale impact when those events do occur. In particular, the early successional reestablishment of vegetation is critical to hill-slope stabilization and needs to be better understood (Walker and Shiels 2012). This study represents the first to characterize post-landslide
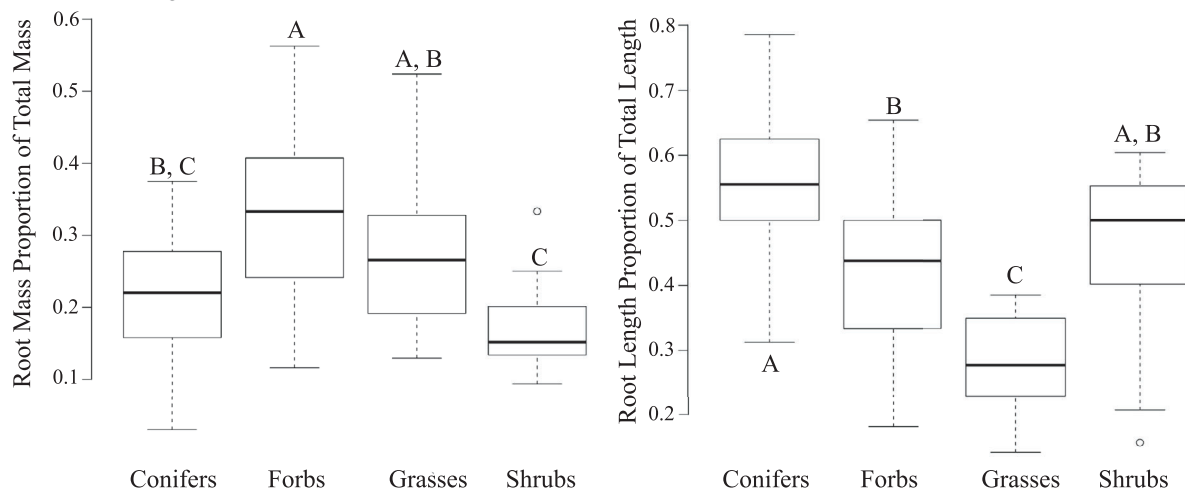
Fig. 4. Root mass and length as a percentage of overall plant mass and length. Significance groupings shown by letters.

ecological regeneration in southern Rocky Mountain foothills and demonstrates that regeneration patterns and rooting behavior are largely in line with theoretical expectations of plant functional group rooting differences. However, regeneration of deeper rooted, stabilizing trees is proceeding extremely slowly in the initiation zones.

Differences in regeneration densities between the accumulation and initiation zone are not a result of available substrate, and presumably not of available seed, given the ample seed sources in the surrounding forest and the close proximity between the initiation and accumulation sites. If the lower regeneration in initiation zones was simply due to lower available soil area, we would observe similar densities after controlling for available soil coverage. However, we saw significantly less tree regeneration (approximately 1/10 the density) and lower regeneration overall, even after controlling for the lower available soil in the steep initiation zones. We suggest this is due in part to ongoing downslope movement of the soil via sheet erosion during precipitation events rather than soil or growing conditions, as there was little difference in mass for those plants that did establish regardless of location, and N content was low across all plots. That these differences in soil coverage and vegetation are still significant six years after the event indicates that post-landslide succession is slow. Average rooting depth was not significantly different between

the initiation and accumulation zone, despite the latter having significantly deeper soils. This suggests that rooting patterns of the species involved in landslide succession are constrained more by species characteristics than available soil space, at least over this timescale.

Plants appear to be successful accessing fractures or other features to increase their rooting depth regardless of direct soil depth; nonetheless, the lower quantity of vegetation in the initiation zone even after controlling for soil coverage differences indicates that stabilization is occurring slowly. Vegetation establishment and soil accumulation exist in a positive feedback, with more vegetation stabilizing more soil, providing more opportunity for vegetation establishment and growth (Walker and Shiels 2012). A limitation of this study is the lack of temporally resolved soil moisture data (though all sites are collocated, intentionally, on moister, northwest facing slopes, and thus, differences between landslides are minimal). Soil moisture is often a strong predictor of regeneration densities in other disturbance types (Rother et al. 2016, Harvey et al. 2016, Stevens-Rumann et al. 2018). Accumulation zones were nearly as steep as initiation zones (average of 27 vs. 32 degrees, respectively), but their deeper soils and higher organic C content likely retain more moisture. This could result in higher vegetation densities, but the lack of differences in mass suggests growth rate once

established is independent of this factor. The complex feedback between stabilization, vegetation growth, and soil development/stabilization that promotes further growth in this area should be investigated to tease apart the specific factors most important in regeneration densities, and how the relative importance of these factors changes over time.

There were significant differences between plant functional groups in their belowground allocation patterns; for example, shrubs had long but relatively low mass root systems whereas grass root systems were shorter but more massive relative to plant biomass. Conifer roots did not stand out as either extensive length or substantial mass at this stage in their lifespan. Shorter but dense roots are common in monocots, whereas longer taproots are more common in dicots (Cofie 2001). This is in line with other research that found shrubs and trees to increase stability to greater depths (>0.5 m) compared with surficial grasses and rush species, which are more effective than deeper-rooted species at shallow depths (<0.2 m; Baets et al. 2008). This suggests that stabilization is proceeding from the top down, with shallow stabilization occurring before the trees stabilize deeper slides. While grasses can prevent erosion, they were not effective in stabilization against major landslides in the area (Rengers et al. 2016), further emphasizing the need to understand regeneration on landslides in the context of climate change. While the differences between functional groups are not unexpected, they have important implications for the future given the anticipated retreat of trees due to climate warming and the increase in grass coverage in dry areas along the edge of forest distributions, as is the case here (Rother et al. 2016, Stevens-Rumann et al. 2018). Regeneration densities are low in the initiation zones, despite the several years of potential establishment. Climate is increasingly expected to limit plant regeneration post-disturbance. Kemp et al. (2019) found threshold responses for *P. menziesii* at 17°C in summer even when seed source is not limiting (as typically the case of landslides), equivalent to current summer temperatures in the study area. Further warming will likely only exacerbate low recruitment of tree species. During intense rainstorm events in these semi-arid landscapes, landslide events are more common in grassy areas (Rengers et al. 2016). Though grasses are well known to limit surficial erosion, they are less effective in deep stabilization relevant to larger but more infrequent sliding, which is much more strongly limited by trees (Waldron and Dakessian 1982). These sites, by virtue of their location close to the grassland ecotone, should be the first to see such a loss of trees (accomplished by lack of recruitment post-disturbance; Stevens-Rumann et al. 2018) as a result of climate change —they are potentially good examples of shifts in slope stability driven by climate change impacts on regenerating vegetation via landslides, fires, and other disturbances.

One limitation of this aspect of the study is root tensile strength, which was not directly assessed here. Root strength varies by species (De Baets et al. 2008) and may also vary by growing conditions. Determining which specific species are likely to be favored by future climates therefore will have strong implications for expectations of future stability. Landslides themselves also alter hydrological conditions within the affected areas (Mirus et al. 2017), which can directly impact stability via alterations to pore pressures and soil cohesion but may also influence regeneration in the future. Modeling how root growth, architecture, and integrity will change under future growing conditions is also an important research need, such as increased temperatures (e.g., reductions in fine roots in maple trees: Wan et al. 2004) and atmospheric CO_2 levels (increase in root biomass in crop plants: Juozapaitienė et al. 2019). There is little known on which to base broad expectations; for example, increased CO_2 and temperature caused higher fine-root biomass in *Pinus taeda* seedlings, but no effect on *Pinus ponderosa* (King et al. 1996). Given the limited regeneration of trees, any CO_2 effect significant at this location may be a moot point, but it could be more significant in areas where regeneration is more likely.

Conclusions

Overall, plant stabilization of post-landslide hillslopes six years after the disturbance is minor. Rooting depths were shallow, and regeneration of deeper-rooted trees and shrubs in the initiation zones was minimal. This lack of current regeneration and evidence from other

disturbance types suggesting declines in tree recruitment at lower elevation, dry slopes in the Rocky Mountains, suggests that deep restabilization of the hillslope will be slow or non-existent due to constraints on trees from climate change pressures. Future work should focus on confirming if regeneration patterns at higher elevations are similarly constrained, determining the relative influence of limiting factors on regeneration (water, slope stability, nutrient availability), and elucidating the rooting characteristics (depth, mass, tensile strength) of plant species that are likely to dominate landslide scars in future climate conditions.

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