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Short communication

Riparian vegetation communities change rapidly following passive restoration at a northern Utah stream☆

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ABSTRACT

Riparian vegetation may recover quickly from disturbance when the disturbance vector is removed or reduced. Grazing is a disturbance that removes plant biomass through herbivory, while overgrazing is a more severe disturbance that can deplete plant propagule pools and inhibit plant community recovery. We tested the hypothesis that riparian vegetation communities can shift quickly from ruderal grasslands to hydrophytic shrubs and graminoids when grazing is largely eliminated from riparian areas. We used a before-after-control study design to collect vegetation community data at six restored reaches and two grazed control reaches prior to and immediately following the construction of a cattle exclosure. We identified trends in *Carex* and *Salix* species abundance and quantified shifts in riparian vegetation community composition across time at each reach using PERMANOVA, multi-level pattern analysis and non-metric multidimensional scaling. Vegetation composition changed rapidly in the four years following removal of grazing disturbance. Indicator species for all impact reaches shifted away from grazing tolerant graminoids and forbs, and toward hydrophytic graminoid and shrub species. Over the same timespan control reach indicator species remained grazing-tolerant graminoids and forbs. There was little change in *Salix* abundance over time at control or impact reaches but *Carex* abundance increased at restored reaches. We conclude that herbaceous plant communities may recover rapidly following the removal of grazing disturbance, but that woody species may lag in recovery without active vegetation manipulation. We postulate that low woody-species recruitment may affect the potential of the riparian zone to quickly shade stream channels and facilitate undercut bank formation, common riparian restoration objectives. To prevent halted riparian succession, designers should proactively identify potential limitations to woody vegetation colonization. We close discussing active approaches to overcome stalled riparian ecosystem development and suggest metrics for assessing woody species recovery.

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1. Introduction

Passive restoration may be appropriate for the improvement of degraded ecosystems in which primary processes such as hydrology, soils, plant propagule dispersal, etc. remain intact (Whisenant, 1999). Livestock grazing in riparian zones is an example of a disturbance that can result in either chronic (e.g. Beever et al., 2003) or acute (e.g. Walker, 1993) ecosystem impairment, depending on the intensity, timing and duration of grazing (McInnis and McIver, 2009; Sternberg et al., 2001). In small streams and rivers, grazing can destroy natural bank structure and deplete riparian vegetation (Beschta et al., 2012; Chambers et al., 2004), increasing instream turbidity, reducing stream shade and increasing stream temperatures, altering patterns of substrate deposition and erosion and exerting a strong influence on stream channel forms (Myers and Swanson, 1996a). These impacts to the riparian zone can negatively affect instream biota and physical processes that create fish habitat (Magilligan and McDowell, 2007). By reducing or removing grazing disturbance from streams with some existing level of bank stability and riparian vegetation, autogenic primary processes may

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allow some level of recovery to in-stream habitats (Magilligan and McDowell, 2007; Myers and Swanson, 1996b) and riparian vegetation. Riparian vegetation community composition may shift from grazing-tolerant species to grazing intolerant hydrophytic species when released from livestock grazing disturbance (Chambers et al., 2004; Sarr, 2002). These changes in vegetation composition may occur in parallel with or drive instream habitat improvements such as the recovery of bank structure, stream shading from trees and shrubs, and instream wood contributions that perpetuate geomorphic change over time. We assess changes in riparian vegetation following grazing exclusion, asking the question: do riparian vegetation communities respond rapidly to release from cattle grazing?

2. Site description and restoration

Spawn Creek is a spring-fed, 2nd-order tributary to Temple Fork, which is a tributary to northern Utah's Logan River (USA, N41.82835, W-111.57795). The Logan River and specifically Spawn Creek are primary habitat for native Bonneville cutthroat trout (*Oncorhynhcus clarki utah*; herein cutthroat trout) and have historically suffered from instream and riparian degradation due to livestock grazing (Budy et al., 2007). During the twentieth century livestock grazed Spawn Creek and the surrounding Cache National Forest intensively, leading to widespread bank destabilization and high instream phosphorus levels (Budy et al., 2007). As many as 95,000 sheep and 22,500 cattle and horses were grazed annually in the Cache National Forest for periods of 48–82 days between 1935 and 1972 (Budy et al., 2007). In 1991 grazing density within the allotment containing Spawn Creek was 1488 head of cattle for a 105-day season. In response to drought, stocking was reduced by 10% annually from 1999 onward to 622 cows in 2005, the final year of permitted grazing.

Because Spawn Creek is important cutthroat trout spawning habitat (Bernard and Israelsen, 1982), passive riparian restoration was initially undertaken to increase vegetation density and abundance to meet instream habitat and fishery restoration goals (Hansen and Budy, 2011; Budy et al., 2007). The primary project goal was to shade the stream with recolonizing vegetation and reduce whirling disease prevalence by reducing stream temperature. As woody vegetation recovered from grazing, it was thought that shrubs and tall graminoids would shade the channel and reduce stream temperatures, facilitate undercut bank formation and reduce the abundance of the *Tubifex tubifex* host of the parasite that causes whirling disease in salmonids, *Myxobolus cerebralis* (Hansen and Budy, 2011). In 2006, prior to summer grazing, 6 km of double split rail fence was installed, excluding 67-ha surrounding Spawn Creek from livestock grazing (Fig. 1). The fence is raised at several points (<3 m each) each fall following cattle trailing to allow for native ungulate migration and winter foraging. Full descriptions of Spawn Creek and initial stream responses to restoration are available within Budy et al. (2007) and Hansen and Budy (2011).

3. Methods

Vegetation monitoring data was used to detect changes in plant community composition and in the abundance of species within the genera *Salix* (willows) and *Carex* (sedges) prior to and following the construction of the cattle grazing exclosure at Spawn Creek. *Carex* and *Salix* species were measured because both genera are generally good indicators of hydrologic connectivity between stream channels and streambanks (Winward, 2000), and have been shown to respond rapidly to release from grazing disturbance (Schulz and Leininger, 1990). Six 160–200 m reaches across the restored impact area at Spawn Creek were repeatedly measured between

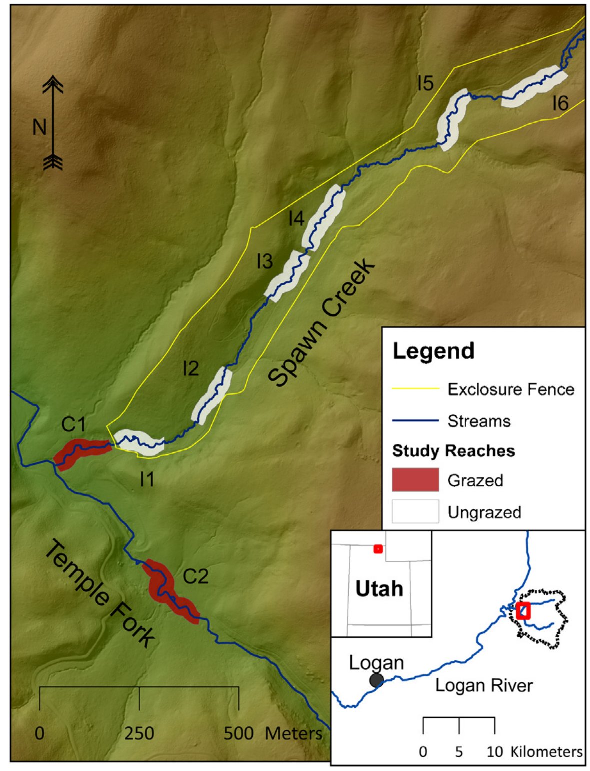

Fig. 1. Map of the Spawn Creek restored impact (I) and Spawn Creek and Temple Fork grazed control (C) reaches within the Logan River Watershed in northern Utah, USA.

2004 and 2009. Two grazed control reaches (~180 m) were monitored prior to and following restoration, one below the grazing exclosure on Spawn Creek and a second just upstream of Spawn Creek's confluence with Temple Fork (Fig. 1). As the entire lower Spawn Creek watershed was fenced, it was not possible to have an upstream control (Fig. 1). Vascular species cover was sampled across the greenline at each reach within 50 cm × 20 cm Daubenmire quadrats (Winward, 2000). The greenline is the first point of rooted perennial vegetation at channel bankfull width or on a depositional feature (Winward, 2000). At the reaches sampled within Spawn Creek and Temple Fork, the greenline occurred at stream bankfull width. There were 36–44 evenly spaced quadrats sampled at each reach depending on reach length. Physical habitat parameters, including bank stability, instream wood volume and frequency, and percent undercut banks were also measured and averaged across each reach (Appendix 1; Table A.2). Vegetation size was not measured as the methods of the PACFISH/INFISH Biological Opinion were used for vegetation sampling in all years (PIBO EM, 2012). All impact and control reaches were sampled in 2004, 2006 and 2008, and most were also sampled in 2005 and 2009.

We tested the preliminary hypothesis that species pools differed between the eight reaches in 2004 prior to grazing exclusion using PERMANOVA (Anderson, 2001). PERMANOVA is a non-parametric multivariate test for compositional dissimilarity between groups (Anderson, 2001). This initial model identified unique vegetation composition between all reaches prior to the restoration treatment, ruling out direct comparisons of restored and unrestored vegetation across all reaches and over time. Accordingly, analyses were performed on each sampled reach as individual case studies for the years in which they were sampled. PERMANOVA models were used to assess differences in vegetation community composition within each reach between the 2004 and 2005 pre-restoration communities and each post-treatment year. All PERMANOVA models used

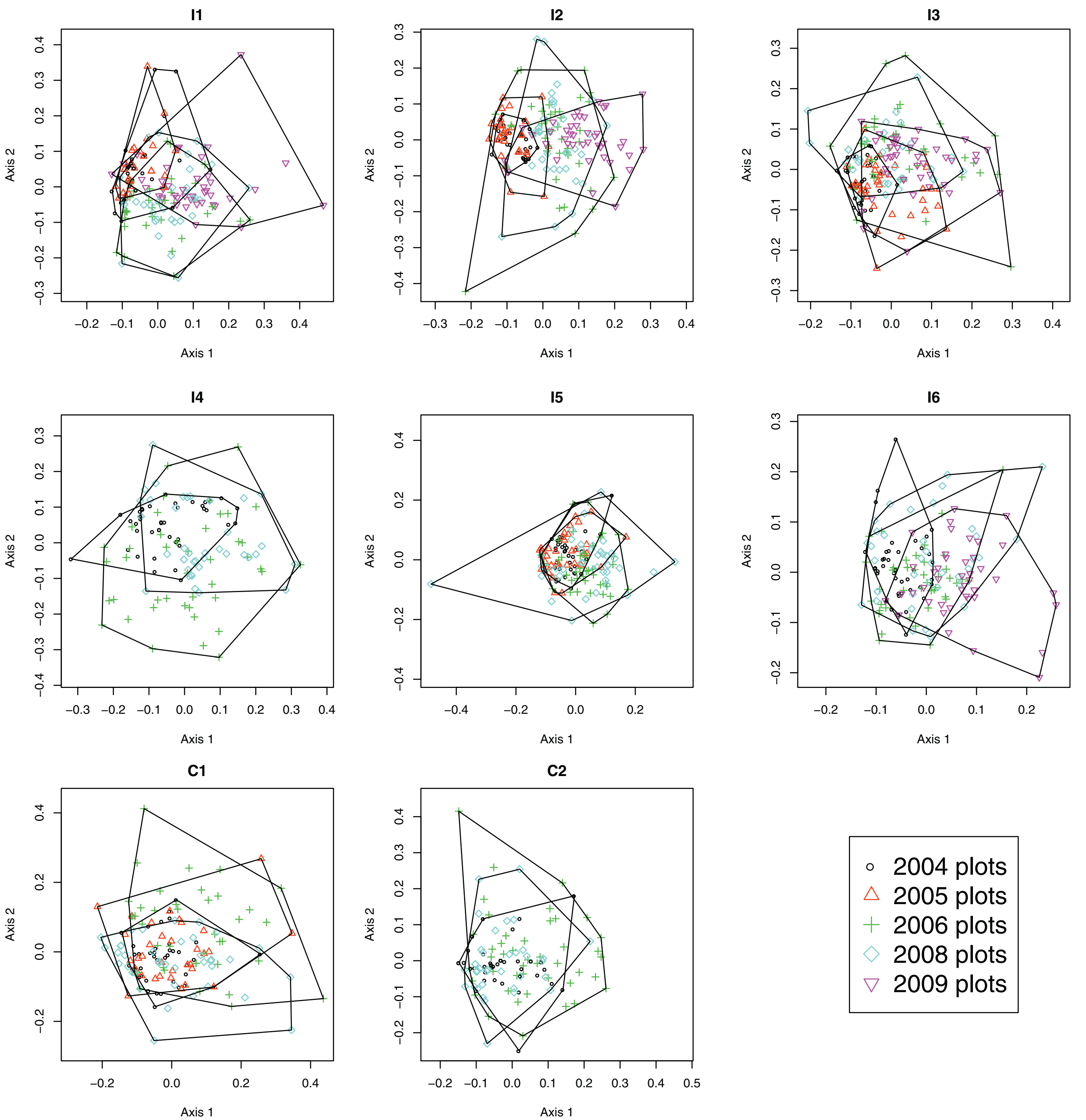

Fig. 2. NMDS ordination plots for the six impact reaches (I1–I6) within Spawn Creek and control reaches at Spawn Creek (C1) and Temple Fork (C2). Three-dimensional NMDS solutions converged within 1000 iterations and had stress values ranging between 15.0 and 19.1. Monte Carlo simulation generated *p*-values (999 randomizations) were <0.05 for ordination final stress values at all reaches.

Bray–Curtis distance matrices of the untransformed vegetation data. Monte Carlo randomization (9999 unconstrained permutations) was used to calculate probability (p) values for the resulting F-statistic as recommended by Legendre and Legendre (2012). This results in p-values based on 9999 random samples of the data plus the actual experimental data (10,000 total samples).

Non-metric multidimensional scaling (NMDS; Kruskal, 1964) ordination plots were created to visualize shifts in community composition at each reach between years. NMDS solutions were calculated from a random starting configuration using Bray–Curtis distance. NMDS was used as a visualization tool to examine the between-year compositional differences at each reach identified by PERMANOVA models using the same distance measure. To identify which species were responsible for compositional differences between years, indicator species analysis was performed for all year combinations at each reach using multi-level pattern analysis (De Caceres et al., 2010). Multi-level pattern analysis is an extension of indicator species analysis and is based on the product of the relative abundance and relative frequency of each species within a given set of years and is tested for statistical significance using Monte Carlo randomizations (1000 permutations; Dufrêne and Legendre, 1997). Multi-level pattern analysis identifies species with fidelity

Table 1
Multi-level pattern analysis results for all impact reaches at Spawn Creek and the control reaches at Spawn Creek (C1) and Temple Fork (C2). The indicator value is calculated as the product of a species' relative abundance and relative frequency within a given year. All indicator species presented below were significant at the $p < 0.05$ level. Species are presented top-to-bottom within each reach from single year indicator species to multiple year indicator species.

Reach	Species	Indicator value				
		2004	2005	2006	2008	2009
Impact 1	*Epilobium ciliatum*			43.6		
	Carex pellita					37.8
	Salix drummondiana					37.8
	Cirsium arvense					26.7
	Equisetum hyemale					26.7
	Juncus ensifolius	41.1		41.1		
	Glyceria striata			39.8		39.8
	Poa pratensis	64.3	64.3		64.3	
	Medicago lupulina	34.8	34.8			34.8
	Symphyotrichum eatonii			77.7	77.7	77.7
	Agrostis stolonifera			64.2	64.2	64.2
	Carex nebrascensis	67.3	67.3	67.3	67.3	
Impact 2	*Epilobium ciliatum*			36.7		
	Trifolium repens			30.9		
	Equisetum laevigatum				26.7	
	Poa trivialis				26.7	
	Carex praegracilis					30.9
	Salix lemonii	49.1	49.1			
	Glyceria striata			41.2		41.2
	Poa pratensis	65.4	65.4		65.4	
	Agrostis stolonifera	50.9		50.9		50.9
	Carex utriculata	41.8			41.8	41.8
	Carex pellita		45.4	45.4		45.4
	Cardamine cordifolia		37.8		37.8	37.8
	Symphyotrichum eatonii		73.3	73.3	73.3	73.3
	Salix geyeriana		59.8	59.8	59.8	59.8
Impact 3	*Medicago lupulina*	34.9				
	Salix monochroma	27.1				
	Symphyotricum foliaceum				51.3	
	Poa trivialis				28.1	
	Carex microptera				22.9	
	Carex utriculata					56.7
	Cardamine cordifolia					42.2
	Juncus ensifolius	43.9		43.9		
	Carex pellita			43.3		
	Epilobium cilliatum			40.7		
	Poa pratensis	63.1		63.1	63.1	
	Agrostis stolonifera	60.2			60.2	60.2
	Populus tremuloides	44.8			44.8	44.8
	Symphyotrichum eatonii			75.9	75.9	75.9
Impact 4	*Agrostis stolonifera*	61.3		61.3	61.3	61.3
	Poa pratensis	64.9	64.9	64.9		
	Medicago lupulina	40.0		40.0		
	Symphyotrichum eatonii			88.6	88.6	88.6
	Carex utriculata			47.2	47.2	47.2
	Glyceria striata			35.5		35.5
	Juncus balticus					66.8
	Carex pellita				31.0	
	Epilobium ciliatum			30.9		
	Juncus ensifolius			32.5		
	Trifolium repens			29.1		
Impact 5	*Juncus ensifolius*	51.7		51.7		
	Epilobium ciliatum			50.0		
	Trifolium repens			30.2		
	Poa trivialis					53.5
	Symphyotrichum eatonii			69.3		69.3
Impact 6	*Glyceria striata*	46.2		46.2	46.2	
	Muhlenbergia filiformis		37.6		37.6	
	Juncus ensifolius			53.1		
	Epilobium ciliatum			37.3		
	Symphyotrichum eatonii			70.5	70.5	
	Cardamine cordifolia				41.5	
	Carex utriculata				33.5	
	Salix melanopsis				26.7	

Table 1 (*Continued*)

Reach	Species	Indicator value				
		2004	2005	2006	2008	2009
Control 1 Spawn Creek	*Medicago lupulina*	32.7	32.7			
	Juncus ensifolius	58.0		58.0		
	Muhlenbergia filiformis	51.3	51.3		51.3	
	Symphyotrichum eatonii		66.7	66.7	66.7	
	Trifolium repens				38.5	
Control 2 Temple Fork	*Carex pellita*			33.7		
	Mentha arvensis			29.5		
	Poa trivialis				60.7	
	Trifolium repens				32.9	
	Taraxacum officinale				30.3	

to multiple treatment groups (years). Using this approach, species that were indicators of both pre-restoration and post-restoration condition at restored reaches could be identified. To examine differences in sedge and willow abundance in the years following grazing retirement, non-parametric Kruskal–Wallis tests were used and pairwise comparisons were made between years for each reach using Bonferroni corrected *p*-values (Cabin and Mitchell, 2000).

4. Results

PERMANOVA results for the six impact reaches showed that vegetation communities within each reach diverged over time (Appendix 1; Fig. 2). Within the impact reaches' PERMANOVA models, R^2 values increased with each additional year since grazing had last occurred (e.g. the R^2 for the models comparing the years 2004 and 2009 was greater than the R^2 for the model comparing years 2004 and 2006). These results contrasted with the Spawn Creek and Temple Fork control reaches (Controls 1 and 2), where R^2 values remained stable across all combinations of years and lower than those in the impact reach models. Multi-level pattern analysis yielded indicator species sets for impact reaches that shifted over time from grazing tolerant species such as *Poa pratensis, Glyceria striata*, *Agrostis stolonifera* and *Trifolium repens* to less disturbance tolerant forbs, graminoids and shrubs (Table 1). In 2009 and 2008, *Carex* and *Salix* indicator species occurred within all impact reaches (Table 1). However, at control reaches there were very few indicator species and little change in their composition over time as indicated by PERMANOVA results and NMDS biplots (Fig. 2). Common indicator species within control reaches were introduced forbs or grass species that persist under grazed conditions (Table 1).

Carex abundance increased significantly between 2004–2005 and 2009 at five of the six impact reaches and did not change over time at the grazed control reaches (Fig. 3). Dominant *Carex* species included *Carex utriculata* and *Carex nebrascensis* (Table 1), both rhizomatous wet meadow sedges, and *Carex pellita*, an obligate wetland sedge. Based on multi-level pattern analysis, *Carex* species were more frequent and abundant following restoration at impact reaches (Table 1). *Salix* species abundance increased over time at one impact reach (I5), and did not change at either control reach (Fig. 3). *Salix* species that occurred at Spawn Creek included *Salix melanopsis, Salix boothii, Salix drummondiana, Salix geyeriana*, and *Salix exigua* as well as hybrid individuals of these species.

5. Discussion and conclusions

Vegetation communities at impact reaches developed rapidly after grazing pressure was removed. Plant communities at impact reaches changed incrementally over time, shifting away from

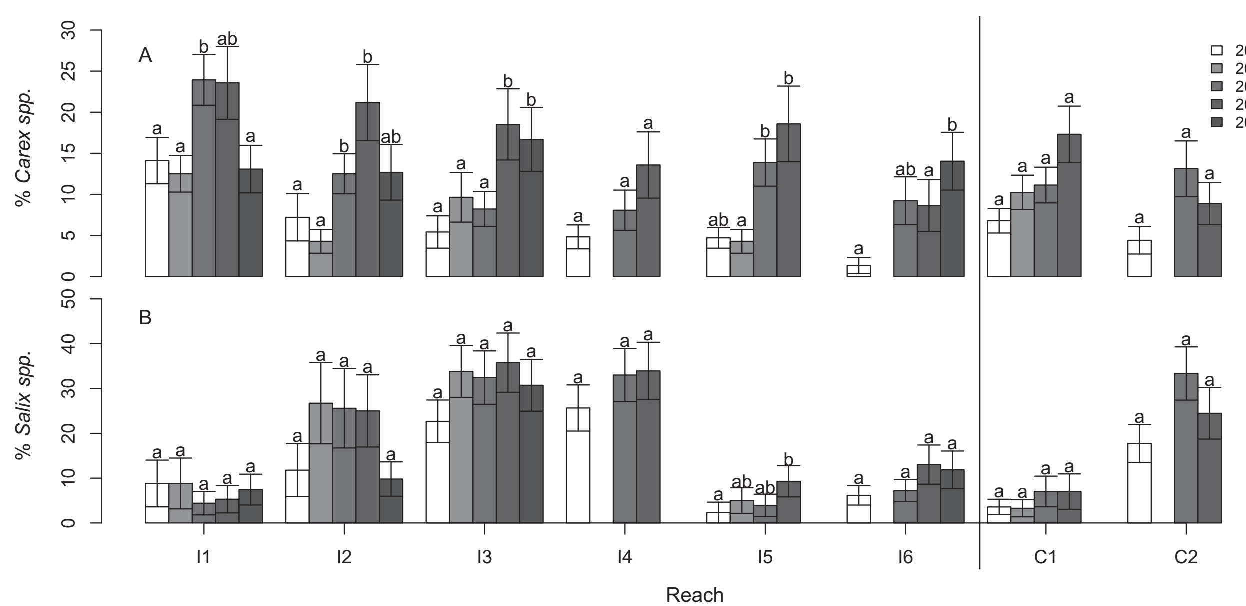

Fig. 3. Average abundance of all *Carex* species (A) and *Salix* species (B) at each reach between 2004 and 2009. I1–I6 are impact reaches while C1 and C2 are the Spawn Creek and Temple Fork control reaches. Error bars are standard error of the mean while letters above bars indicate group membership from Kruskal–Wallis pairwise comparisons. Missing bars occur where reaches were not sampled within a given year.

disturbance-tolerant pasture species as the time since grazing removal increased. There was little change in the riparian vegetation communities at grazed control reaches (Table 1 and Fig. 2). As the time since last grazing activity increased during favorable, wet years (Fig. A.1), indicator species shifted to hydrophytic plant species that may have been suppressed by prolonged cattle grazing. Specifically, we saw dramatic increases in *Carex* abundance at impact reaches and reduced pasture grass abundance. 2009 post-restoration *Carex* abundance at Spawn Creek is comparable to the greenline vegetation of riparian meadows at northern Oregon streams (Dwire et al., 2006, 2004). In headwaters of the Columbia Basin, Hough-Snee et al. (2013) showed that high bank stability and bank undercutting are correlated to riparian sedge-willow communities. These sedge-willow reaches were in better physical condition than heavily grazed, semi-arid reaches elsewhere in the Columbia Basin that largely lacked *Carex* species (Hough-Snee et al., 2013). In the future, Spawn Creek's bank condition may converge with conditions of other meadows (high stability; Table A.2) as deep-rooted *Carex* species expand.

The observed trajectory of passive riparian restoration at Spawn Creek supports two related concepts in stream restoration: (1) removing disturbance from riparian systems allows herbaceous plant communities to recover rapidly (Dobkin et al., 2008). (2) Vegetation recovery may eventually correspond to improvements in instream physical habitat quality (Herbst et al., 2012). At Spawn Creek, bank stability increased with time after grazing removal (Table A.2; Budy et al., 2007), illustrating how rapidly habitat can change as riparian vegetation recovers from disturbance. Hansen and Budy (2011) also found passive restoration at Spawn Creek to reduce the prevalence of *Myxobolus cerebralis*, the parasite that causes whirling disease, although they could not directly decouple restoration effects (e.g. stream shading from *Salix* recovery) from interannual climatic variability.

While herbaceous riparian vegetation recovered quickly using a passive restoration approach, further ecosystem recovery may not proceed as rapidly. At Spawn Creek, limited *Salix* species recruitment may preclude successful fishery and habitat restoration that requires stream shade and contributions of wood to the stream to shape habitat (Hansen and Budy, 2011; Table A.2: Wood volume and frequency). Grazing retirement effectively allows annual plants to spread by seed and perennial herbaceous plants to expand vegetatively, but woody species may be more difficult to restore using passive restoration approaches alone. While some studies show rapid willow recovery following livestock grazing retirement (Booth et al., 2012), historic grazing has been shown to reduce sexual reproduction in willows (Brookshire et al., 2002), and there may be a reproductive lag in willows at Spawn Creek preventing new individuals from establishing. Willow growth and establishment can be constrained by low water tables and soil moisture availability, as well as native ungulate grazing (Bilyeu et al., 2008; Chambers et al., 2004; Pezeshki et al., 2007; Wolf et al., 2007). In North American ecosystems that lack their historic carnivores (e.g. wolves [*Canis lupus*]), elk (*Cervus canadensis*) grazing pressure may prevent willow recruitment in riparian areas (Ripple and Beschta, 2006). The combination of low precipitation, elk grazing, and historic cattle overgrazing appears to provide enough inertia against autogenic ecosystem recovery that active restoration may be required to move Spawn Creek and comparable low-order, grazed systems toward sufficient wood production and stream shade to meet instream restoration objectives (McIver and Starr, 2001). Whether caused by biotic or abiotic filters, this lag in riparian woody species expansion directly affects sites' potential to reach instream habitat restoration objectives quickly and without active management (e.g. individual tree planting and protection).

Restoration designers must anticipate the potential for a site to respond to disturbance and identify what biotic and abiotic processes may interact to limit sites' recovery potential (Bilyeu et al., 2008; Goodwin et al., 1997). By identifying limitations to the self-design (Mitsch and Jørgensen, 2004) of the riparian ecosystem at Spawn Creek, such as willow species recruitment (*sensu* Bergen et al., 2001), limitations to instream restoration may also be identified. Once thresholds in autogenic recovery have been identified, restoration can continue passively or be assisted through adaptive management. In the case of Spawn Creek, community composition measurements fail to identify what environmental factors may limit individual willow establishment and growth. Measuring woody species flowering, growth and physiological performance (*sensu* Cooper and Merritt, 2012) may better forecast the outcomes of Spawn Creek's riparian willows and their potential to grow wood that can affect geomorphic change or shade the temperature-impaired stream (Ghermandi et al., 2009). When the objective of riparian restoration is to increase stream shade and reduce temperatures to improve biological conditions (Bernhardt, 2005; Hansen and Budy, 2011; Roper et al., 1997), shifts in vegetation toward hydrophytic herbaceous species may not lead to full fish habitat restoration (Hansen and Budy, 2011). For example, McBride et al. (2010) found that stream channels in afforested temperate forests widen at rates of only a few centimeters per year. Watanabe et al. (2005) suggests that active restoration is more effective when trying to reach time-sensitive instream restoration objectives or when design parameter success can fluctuate with environmental variability. Based on the identified limiting factors to willow recolonization at Spawn Creek, supplementing riparian areas by planting willows into the recovered, stable, hydrologically reconnected banks may expedite riparian forest development and instream temperature reduction.

Based on our findings at Spawn Creek, we encourage riparian restoration practitioners to identify the likely trajectories of initial change following passive restoration and shift project monitoring efforts to environmental factors likely to impede further recovery from passive restoration. This monitoring may include measuring riparian plant properties that correspond directly to stream habitat change or using adaptive management frameworks (Bergen et al., 2001) to plan later active restoration stages that would otherwise stall due to climatic fluctuation, trophic interactions or external disturbance.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.ecoleng.2013.07.042.

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Supplementary Materials

Table A.1 PERMANOVA model results testing the difference in vegetation composition between 2004 and 2005 and each following year within each reach. Bold years are those following grazing retirement at impact sites.

Reach	Year	2004			2005		
		R^2	F	P	R^2	F	P
Impact 1	2005	0.002	0.190	0.972	-		
	2006	0.115	10.601	<0.001***	0.111	12.934	<0.001***
	2008	0.143	13.633	<0.001***	0.144	13.783	<0.001***
	2009	0.242	26.171	<0.001***	0.230	24.451	<0.001***
Impact 2	Year	R^2	F	P	R^2	F	P
	2005	0.068	5.953	<0.001***	-		
	2006	0.118	10.976	<0.001***	0.079	7.081	<0.001***
	2008	0.111	10.261	<0.001***	0.048	4.173	<0.001***
	2009	0.182	18.184	<0.001***	0.136	12.934	<0.001***
Impact 3	Year	R^2	F	P	R^2	F	P
	2006	0.063	5.421	<0.001***	No 2005 data collected		
	2008	0.078	6.501	<0.001***			
	2009	0.118	10.84	<0.001***			
Impact 4	Year	R^2	F	P	R^2	F	P
	2005	0.011	0.926	0.454			
	2006	0.107	9.860	<0.001***	0.110	10.115	<0.001***
	2008	0.161	15.776	<0.001***	0.151	14.630	<0.001***
	2009	0.292	33.768	<0.001***	0.282	32.267	<0.001***
Impact 5	Year	R^2	F	P	R^2	F	P
	2006	0.046	4.011	0.002**	No 2005 data collected		
	2008	0.085	7.591	<0.001***			
Impact 6	Year	R^2	F	P	R^2	F	P
	2005	0.165	1.372	0.203	-		
	2006	0.105	9.620	<0.001***	0.071	6.304	<0.001***
	2008	0.120	11.193	<0.001***	0.089	7.974	<0.001***
Control 1	Year	R^2	F	P	R^2	F	P
	2005	0.018	1.509	0.170	-		
	2006	0.067	5.877	<0.001***	0.053	4.560	0.001**
	2008	0.081	7.041	<0.001***	0.054	4.543	<0.001***
Control 2	Year	R^2	F	P	R^2	F	P
	2006	0.054	4.539	<0.001***	No 2005 data collected		
	2008	0.057	4.729	0.003**			

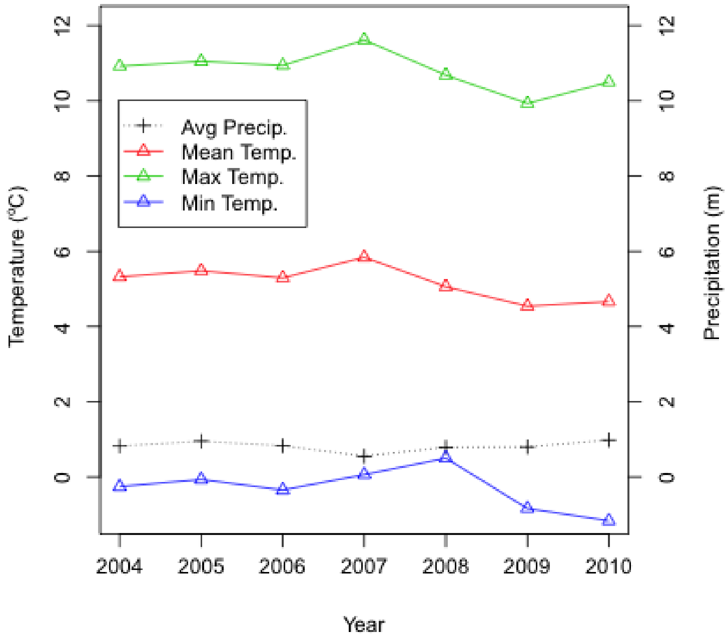

Fig A.1 Mean precipitation, mean, minimum and maximum temperatures at the confluence of Spawn Creek and Temple Fork for years 2004 – 2010.

Table A.2: Average stream morphological parameters by reach and year. Stream parameters were measured at scales different than vegetation quadrats and are presented here for context. Inference could not be made between morphological parameters and vegetation composition data. Blacked out cells indicate unavailable data.

Stream	Year	Sinuosity (%)	Bankfull Width (m)	Pool Percent	D50 (m)	Bank Angle (°)	Bank Stability (%)	Undercut Banks (%)	Large Wood Frequency	Large Wood Volume
Impact 1	2004	1.31	3.26	50.42	0.025	110	66.67	35.71	30.3	1.589
	2006	1.313	3.56	20.81	0.03	98	85.71	38.1	107.33	2.697
	2008	1.363	2.95	11.72	0.044	95	92.86	40.48	59.21	5.868
	2009	1.367	2.53	12.56	0.031	80	100	54.76	41.27	3.964
Impact 2	2004	1.14	1.99	70.12	0.029	85	66.67	61.9	30.12	3.348
	2006	1.159	2.1	23.32	0.042	79	95.24	47.62	29.52	1.405
	2008	1.173	2.08	32.83	0.033	74	95.24	61.9	11.52	0.252
	2009	1.153	2.02	30.34	0.026	75	100	61.9		
Impact 3	2004	1.356	2.02	67.41	0.021	82	80.95	64.1	30.86	3.453
	2006	1.576	2.2	31.47	0.025	91	97.83	34.78	16.62	2.174
	2008	1.462	2.14	29.01	0.041	97	95.45	34.09	11.7	1.382
	2009	1.392	2.14	35	0.031	75	100	61.9	11.88	1.26
Impact 4	2004	1.176	2.2	44.38	0.031	98	64.29	42.86	6.17	0.153
	2006	1.219	2.37	25.65	0.036	100	97.73	18.18		
	2008	1.2	2.47	19.07	0.031	95	95.24	41.46		
Impact 5	2004	1.302	1.99	42.06	0.036	94	76.19	51.22	6.06	0.137
	2006	1.319	2.02	20.45	0.043	88	97.62	47.62	5.81	0.059
	2008	1.348	1.89	17.14	0.047	91	90.48	50	5.83	0.112
Impact 6	2004	1.168	2.55	28.05	0.044	84	76.19	54.76	121.95	13.371
	2006	1.147	3.34	6.47	0.035	103	85.71	26.19	147.06	12.09
	2008	1.088	2.71	19.38	0.06	97	86.84	36.84		
	2009	1.158	2.68	23.14	0.035	96	95.45	40.91	137.46	17.214
Control 1	2004	1.26	2.3	30.13	0.042	95	71.43	33.33	6.25	0.243
	2006	1.245	2.57	32.94	0.023	84	83.33	54.76	6.25	0.196
	2008	1.239	2.63	17.03	0.034	94	90	40	25.24	0.533
Control 2	2004	1.387	4.67	52.96	0.038	8.07	12.22	100	78.57	35.9
	2006	1.393	4.78	48.88	0.036	6.33	11.83	100	90.91	25
	2008	1.294	4.7	30.46	0.042	1.61	8.61	100	78.95	36.84