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Passive Reestablishment of Riparian Vegetation Following Removal of Invasive Knotweed (*Polygonum*)

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Japanese knotweed and congeners are invasive to North America and Europe and spread aggressively along rivers establishing dense monotypic stands, thereby reducing native riparian plant diversity, structure, and function. Noxious weed control programs attempt to eradicate the knotweed with repeated herbicide applications under the assumption that the system will recover to a native assemblage which will inhibit future invasions. However, eradication efficacy studies typically only measure the amount of knotweed reduced, not the reestablished species diversity or plant origins. For a community scale efficacy study, we measured vascular plant species diversity and cover in riparian areas along five rivers in Washington State, 3 to 6 years after Bohemian knotweed was initially treated with herbicide. Plant species composition was compared between riparian sites treated to remove knotweed and reference sites where knotweed was absent. Sites where knotweed had been removed had significantly more exotic species and vegetative cover than reference sites; however, native species richness and cover were greater in reference sites and areas with more overstory vegetation. The native plants observed were primarily shade tolerant and perennial, as opposed to many of the exotics, which were shade-intolerant annuals. In general, reestablishment of native and exotic vegetation was not related to pretreatment knotweed stem count, size of the invaded area, or timing of herbicide application. However, residual native tree cover was negatively correlated with initial knotweed stem count. Monitoring the success of restoration objectives (e.g., native plant reestablishment or increased species diversity) and characterizing associated habitat features following knotweed eradication will help in the development of site-specific protocols for successful plant community scale restoration.

Nomenclature: Glyphosate; imazapyr; Bohemian knotweed, *Polygonum* x *bohemicum* Zika & Jacobson; Japanese knotweed, *Polygonum cuspidatum* L. Sieb. & Zucc. POLCU.

Key words: Community composition, exotic, invasive, native, nonnative.

Riparian areas are transitional environments between aquatic and terrestrial ecosystems and are ecologically important for their diversity, productivity, and habitat complexity (Gregory et al. 1991; Naiman et al. 2005). Riparian vegetation communities are different from adjacent terrestrial communities and are affected by fluvial processes such as flooding, sediment deposition, and erosion, depending on location in the drainage network (Hupp and Osterkamp 1996; Naiman et al. 2000; Swanson et al. 1987). Because these fluvial processes act as important disturbance agents, riparian areas are especially susceptible to invasion by exotic plants. Hydrologic disturbances efficiently propagate and disperse plants along river corridors (Hood and Naiman 2000; Planty-Tabacchi et al. 1996).

Invasive knotweeds (*Polygonum*, syn. *Fallopia*) are considered a threat to riparian habitats because they establish dense monotypic stands that spread aggressively along streams. Invasive knotweed species, native to eastern Asia, include Japanese (*P. cuspidatum* Sieb. & Zucc.), giant (*P. sachalinense* F. Schmidt ex Maxim.), their hybrid Bohemian (*P.* x *bohemicum* Zika & Jacobson), and Himalayan (*P. polystachyum* Wallich ex Meisn.). The World Conservation Union lists Japanese knotweed and cogeners within the world's worst 100 invasive alien species (Lowe et al. 2000). Japanese knotweed was first introduced as an ornamental in Europe and the British Isles in approximately1825 and North America in the late 1800's. Today, invasive knotweed species can be found in many European and North American countries, up to at least 68°N latitude.

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Management Implications

Foliar herbicide applications successfully reduced the cover of invasive knotweed; however, treatment effects on riparian plant assemblages from passive reestablishment were mixed and depended on site habitat conditions and stream size. Along smaller streams, 2nd to 3rd order, the post-treatment plant assemblages were primarily native in composition, although exotic species were more abundant in the knotweed-treated areas compared to reference (no knotweed) areas. The removal of knotweed cleared sites for colonization by native plants, but also released nontarget exotic invaders. In contrast, riparian areas along the largest stream (4^{th} order) contained more exotic forbs and shrubs than native plants in both the knotweed-treated and reference sites. The larger stream had riparian areas with little canopy cover and extensive gravel bars and wide floodplains that are subject to higher levels of seasonal flooding. These conditions promote exotic propagule movement and survival.

Successful reestablishment of native vegetation following knotweed removal along large, complex rivers may require active restoration, such as the control of secondary invasions by other exotic plants or replanting native species as conditions change over time. Noxious weed control programs can benefit from including post-treatment plant community surveys, in addition to measuring the amount of the target weed species reduced, to determine if continued management is needed to achieve restoration objectives.

These knotweeds are herbaceous perennials with robust erect stems up to 4.5 m (15 ft) tall and large leaves that reproduce rapidly from seed, extensive rhizomes, or stem fragments (McHugh 2006). Dispersal by vegetative means occurs when rhizome or stem fragments are washed downstream by water and deposited on banks or by humans when transported in soil or fill dirt. Knotweeds prefer partial to full sun and are found primarily in moist habitats (e.g., waterways, wetlands, coastlands), but also grow in urban areas, waste places, along roadways and other disturbed areas. Once established, knotweed forms a dense stand that shades and crowds out all other vegetation, displacing native flora and fauna (Gerber et al. 2008; Urgenson et al. 2009). Knotweed is capable of growing in a wide variety of soil types, can alter soil chemistry (Dassonville et al. 2011; Vanderhoeven et al. 2005), and release allelochemicals (Vrchotová and Šerá 2008). Relatively little is known about the effects of invasive knotweed on the ecological processes within the streams they border. By displacing native plant assemblages, dense knotweed colonies could disrupt or alter the flow of terrestrial organic materials to streams (Lecerf et al. 2007), as well as prevent the recruitment of riparian trees that provide shade, organic matter, and ultimately serve as aquatic habitat when they fall into stream channels (Urgenson et al. 2012).

Knotweed invasions are costly to control primarily because removing knotweed usually requires many years of repeated herbicide applications (Bashtanova et al. 2009; McHugh 2006). Noxious weed control programs focus on removing the target species under the assumption that the impact of the exotic plant will dissipate and the system will recover a native assemblage and inhibit future invasions (Pearson and Ortega 2009). However, weed control programs typically only measure the amount of the target species reduced and not the subsequently reestablished plant community (e.g., Howell 2012; McHugh 2006). One published study has examined the response of understory species to knotweed removal and found that both native and exotic plants colonized recently treated knotweed sites, but this study did not assess vegetation beyond two years post-treatment (Urgenson 2011). It is important to assess the long-term post-treatment recovery of areas with knotweed removal efforts in order to determine whether continued management is needed to achieve restoration objectives (Holmes et al. 2005; Richardson et al. 2007). Without successful native vegetation recovery, these ecosystems are prone to re-invasion by the same exotic or secondary exotic species (Pearson and Ortega 2009). Furthermore, the efficacy of achieving knotweed control objectives on streams of different characteristics (width, gradient, and valley topography) is poorly known.

The success of invasive plant removal as a restoration strategy is rarely documented in terms of the recovery of native communities. Results appear to be mixed and likely depend on habitat characteristics, invader species traits, and control methods. For example, in western U.S. grasslands, removal of spotted knapweed (*Centaurea stoebe* L.) by broadcast herbicide (picloram) application resulted in the intended effects of reduced knapweed and increased native grass cover, but herbicide treatment also had the unintended consequences of reducing native forb cover while increasing abundance of another invasive grass (Ortega and Pearson 2010). This study found treatment effects to be complex and dependent on initial levels of spotted knapweed invasion. Biological control by flea beetles of leafy spurge (*Euphorbia esula* L.) did not result in the recovery of native vegetation after nine years (Butler and Wacker 2010). In fact, leafy spurge was replaced by grasses (*Poa* spp.) in both beetle-controlled and noncontrolled areas. Native species did recover, though, following the felling of invasive pine (*Pinus halepensis* Mill.) trees in montane grasslands of Argentina (Cuevas and Zalba 2010). The proximity of cleared areas to patches of natural vegetation and the density of invasive trees before control influenced natural regeneration of native plants.

Recovery of native plant assemblages after invasive species removal may be influenced by stream size and associated habitat or disturbance characteristics. Floodplain width generally increases along with stream size, and the transition between the channel and the riparian zone becomes less defined because of gravel bars and secondary channels (Naiman et al. 2000). Hydraulic disturbances are more common in large rivers and, because many invasive species perform best in recently disturbed habitats with abundant sunlight (Hood and Naiman 2000), reestablished plant communities may differ than those along smaller streams.

The objectives of this study were to determine if Bohemian knotweed removal with herbicide followed by passive reestablishment of vegetation resulted in native plant assemblages, and/or communities similar to those in riparian areas without knotweed. Additionally, we sought to determine if postcontrol vegetation composition was influenced by knotweed precontrol demographic data (i.e., stem density, patch size) or temporal patterns of treatment (time, number of applications). We examined the vegetation communities in riparian areas 3 to 6 years after initial herbicide application along five streams of varying sizes in southwest Washington State. The cover and taxa richness of native and exotic vascular plants in riparian areas controlled for knotweed was compared to riparian areas without knotweed. Habitat attributes of overstory cover, distance from and height above channel, drainage area, slope, and aspect were also measured for their influence on vegetation composition. Based on our findings we offer some guidelines for establishing invasive plant removal restoration strategies and post-treatment monitoring.

Materials and Methods

Site Selection. The study took place in the Chehalis River basin of western Washington State (Figure 1), characterized by a temperate, maritime climate (mean annual temperature of 10.6 C [51 F] and precipitation of 163.3 cm [64 in]). Study streams ranged in size from 2nd order streams to a 4th order river, none of which were influenced by dams (Table 1). The streams had similar channel characteristics (pool–riffle sequences with gravel–cobble substrate) and surrounding valley morphology (glaciated). Soil type and texture were determined from the Soil Survey Geographic Database (SSURGO 2012). Because the streams supported spawning salmon populations, they were buffered by an unmanaged riparian zone that prohibited development and timber harvest. Sites along Cook Creek, Porter Creek, Stony Creek, and Satsop River were surrounded by rural housing and small-scale agriculture. Elk River sites, located in a Washington State Natural Resources Conservation Area, were surrounded by managed conifer forests. Red alder (*Alnus rubra* Bong.) was the riparian dominate overstory tree species at all streams. Big leaf maple (*Acer macrophyllum* Pursh) was also common along Cook, Porter, and Stony creeks. Sitka spruce [*Picea sitchensis* (Bong.) Carr.] was common along the Elk River. Black cottonwood (*Populus trichocarpa* Torr. & A. Gray) and Sitka willow (*Salix sitchensis* Sanson ex Bong.) were abundant along the Satsop River.

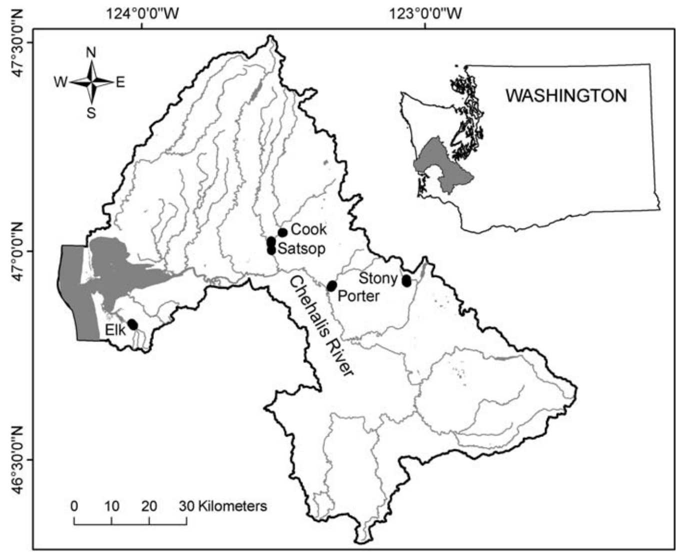

Figure 1. Location of the study streams and sites (black circles) within the Chehalis Basin of southwest Washington State, USA.

The Center for Natural Lands Management (CNLM, Olympia, WA) has been controlling invasive knotweeds, primarily the Bohemian variety, in riparian areas along streams in the Chehalis River basin of southwest Washington State since 2004. This study followed their knotweed-removal efforts by surveying all knotweed-controlled sites that had an initial knotweed patch size greater than 20 m^2 (215.28 ft^2) and were located in riparian areas of low terrace and floodplain forests adjacent to the active channel. The resulting 47 knotweed sites had been treated with herbicide to remove knotweed 3 to 6 years prior to this study in 2010 to 2011 (Table 1). At the time of initial treatment, the knotweed patches ranged in size from 20 to 1500 m^2 with 10 to over 1,000 stems. Specific herbicide mixtures and application methods changed over time as CNLM adapted their methods to use the least amount of herbicide in the most effective manner. In 2004 all knotweed was treated via stem injection with glyphosate (100% AquaMaster™, Monsanto Canada Inc, 900 One Research Road, Winnipeg, Manitoba, Canada, R3T 6E3). In 2005 and 2006 knotweed stems greater than 2 cm were injected with glyphosate, but smaller stems were spot-sprayed (foliar application) with a mixture of glyphosate (2% AquaMaster™), imazapyr (1% Habitat®, BASF Corporation, 26 Davis Drive, Research Triangle Park, NC, 27709), modified vegetable oil (1% Competitor®, Wilbur-Ellis, PO Box 16458, Fresno CA 93755), and water (96%). From 2007 to 2011 all knotweed was treated with foliar application of imazapyr (1% Habitat®), oil (1% Competitor®), and water (98%). After the first two years of herbicide treatments, very little knotweed remained, although a few small stems may have emerged and were spot-treated with close-range foliar spray annually. The

Table 1. Study streams physical characteristics and number of vegetation survey sites. All of the streams are free-flowing. Soil textures are abbreviated such that "Gr" = gravel, "Lm" = loam, "Si" = silt, "Sa" = sand. Treatment refers to herbicide application at knotweed sites only.

Characteristic	Stony	Cook	Elk	Porter	Satsop
Stream order	2nd	2nd	2nd–3rd	3rd	4th
Drainage area (km^2)	8	13	44	95	762
Bankfull width (m)	5.9	6.7	15.1	17.4	74.4
Elevation (m)	70.9	31.4	3.9	16.5	25.1
Direction of flow	SE	W	NW	SW	S
Soil type	Tenino	Cloquato	Grehalem	Humptulip	Juno
Soil texture	Gr–Lm	Si–Lm	Si–Lm	Si–Lm	Sa–Lm
Reference sites (#)	12	7	5	11	12
Knotweed sites (#)	12	7	5	11	12
Years since initial treatment (#)	3–6	3	6	3	3–4
Years of active treatment (#)	2–6	2–3	5–6	2–3	3–4

foliar spot-spray technique was extremely accurate and insured that very little herbicide was used, such that effects on nontarget plants were thought to be negligible.

Along each stream, an equal number of reference sites and knotweed sites were surveyed. Vegetation growing in the reference sites would likely have approximated the vegetation of the knotweed sites had they not been invaded. The reference sites were not pristine sites, but they had never undergone invasive control efforts and possessed 0 to 1% cover by knotweed (knotweed observed during our vegetation surveys had not been treated with herbicide). Reference sites were chosen by randomly selecting non-knotweed treated riparian areas, at least 20 m from a knotweed treated site. Since CNLM began treating knotweed along these streams, every knotweed site had been flagged and a GPS coordinate recorded. Therefore, we were able to use GIS to randomly select reference sites similar in aspect, soil type, distance from and height above the channel to the knotweed sites along each stream.

Vegetation Sampling. At each knotweed site, 20 quadrats of 1 m^2 were systematically located within the treatment area such that 10 quadrats formed a line parallel to the stream channel with two lines of 5 quadrats perpendicular to the stream. Twenty quadrats were similarly distributed in the reference sites. The % cover of each plant species was estimated per quadrat and then averaged per site. Bryophytes (mosses and liverworts) were not identified further. Habitat attributes of overstory (i.e., tree) % cover, distance from channel (m), height above channel (m), % slope, and aspect (degrees from north) were also measured at each site. The drainage area (km^2) and elevation (m) of each site was determined from a 10-m Digital Elevation Model using ArcGIS. Sites along Cook, Elk, Porter, and Stony creeks were surveyed July to September 2010 and Satsop River sites were surveyed August 2011.

Individual plant species were grouped by growth form (forb, grass, sedge, rush, shrub, or tree) and origin (native or exotic) to western Washington State based on regional flora guidebooks (Hitchcock and Cronquist 1973; Pojar and MacKinnon 2004), the national USDA PLANTS database (USDA 2011), and regional native plant lists (WNPS 2011). Species were also classified by life history (annual/biennial, perennial, or mixed), shade tolerance (tolerant, intolerant, or mixed), and for those species known, their toxicity and ability to fix nitrogen (USDA 2011). For plants not identified to species, these traits were recorded as unknown. Within each growth form or trait group the percent cover occasionally summed to > 100 caused by overlap of multiple species.

Statistical Analysis. To answer our questions concerning the reestablishment of vegetation communities after knotweed removal, we conducted statistical analyses on two different datasets. The complete dataset contained vegetation measures and habitat attributes from all 94 sites and was used to test for post-treatment vegetation differences between knotweed-treated and reference sites. The reduced dataset contained vegetation measures and habitat attributes from the 47 knotweed-treated sites only and was used to test for post-treatment vegetation differences because of pretreatment knotweed levels and herbicide application timing.

Analyses of covariance (ANCOVA) mixed-models (SAS 9.3), with streams as random blocking factors, were used to test six response metrics: species richness of native plants, richness of exotic plants, % cover of trees (all native), % cover of native forbs and shrubs, % cover of exotic forbs and shrubs, and % cover of knotweed. In the complete dataset, predictor variables were treatment (knotweed-treated vs. reference), stream by treatment interaction, and the habitat attributes of overstory cover, distance from channel, height above channel, slope, aspect, and % cover of reed canarygrass (*Phalaris arundinacea* L.). Because of the observed dominance of reed canarygrass, when present at a site, we wanted to test whether it was a significant predictor of other species responses. In the reduced dataset with knotweed-treated sites only, predictor variables were the knotweed patch area (m^2) and number of stems prior to herbicide application, the number of years since the initial application, and the number of years herbicide was applied (e.g., some years no knotweed was present so no herbicide was applied). Nonsignificant ($p > 0.1$) predictor variables were removed via step-wise model simplification. To normalize the model residuals, the % covers of native and exotic forbs and shrubs, knotweed, and reed canarygrass, the distance from and height above channel, drainage area and slope were all natural-log transformed prior to analysis (Warton and Hui 2011). Site aspect was transformed to represent north–south and east–west orientation using cosine and sine transformations, respectively. Soil type was not included in the analyses because it did not differ among sites within drainages, but varied between streams only.

Nonmetric multidimensional scaling (NMS) compared plant species community compositions between sites, streams and habitat attributes described above. NMS with Sorenson's distance measure was performed in PC-ORD 6.4. Five hundred runs were performed for each ordination, and the run with the lowest stress and a stability criterion of 0.00001 was selected. Two ordinations were created, one from the complete dataset to test for treatment differences between sites (knotweed-treated vs. reference), and the other from the reduced dataset to test for pretreatment knotweed invasion levels and herbicide timing. The ordination from the complete dataset was comprised of 94 sites (half knotweed-treated, half reference) and 135 species (rare species, those present in only two or fewer sites, were removed). The final 3-dimensional solution had a stress of 12.70 and represented 88% of the total variance. The ordination from the reduced dataset was comprised of 47 sites (knotweed-treated only) and 104 species (rare species removed). The final 2-dimensional solution had a stress of 15.18 and represented 86% of the total variance. Attributes with strong Pearson correlations ($r > 0.5$) to the ordination axes are shown as vectors. Sites were coded by stream or treatment for graphical display.

Indicator species analysis (ISA, PC-ORD 6.4) was used to detect plant species positively associated with knotweed-treated or reference sites from the complete dataset (Dufrene and Legendre 1997). Each species was given an indicator value based on its abundance and frequency of occurrence in each of the two treatment groups. Indicator values were tested for statistical significance using Monte Carlo randomizations (4999 permutations). Only species with indicator values > 25 and *p*-values < 0.10 are listed for each treatment type.

Results

Vegetation Observed. In post-knotweed removal surveys of treated and reference sites we observed a total of 209 plant species: 126 native, 56 exotic, and 27 of unknown origin because they were not identified to species. The vegetation included 138 forb, 20 grass, 4 sedge, 2 rush, 24 shrub, and 21 tree species. All identified trees, rushes, and sedges were native. Red alder was the most abundant tree species along the smaller streams, while sitka willow was more abundant along the Satsop River. Salmonberry (*Rubus spectabilis* Pursh) was the most abundant native shrub, and swordfern [*Polystichum munitum* (Kaulf.) C. Presl] and field horsetail (*Equisetum arvense* L.) were the most abundant native forbs. Native plants were more likely to be shade tolerant (48% of species) and perennial (87% of species) compared to the exotic species (21% shade tolerant, 57% perennial). Only a few native species fixed nitrogen (2%) or were toxic to humans or livestock (8%), whereas more of the exotic species were nitrogen fixers (18%) or toxic (23%). Reed canarygrass, especially dense along Cook and Porter creeks, displayed the most cover of any exotic species, and is itself often the target of eradication efforts. Other abundant exotic species were Himalaya blackberry (*Rubus armeniacus* Focke), creeping buttercup (*Ranunculus repens* L.) along the smaller streams, and Scotch broom [*Cytisus scoparius* (L.) Link] along the Satsop River.

Treatment Effects. The knotweed-treated sites contained primarily native plants along the smaller streams, but exotic species were much more prevalent in the Satsop River sites (Figure 2). Knotweed-treated sites from the smaller streams had proportionately more native forb and shrub species (61 to 78%) than exotic species (21 to 37%), and more native cover (57 to 87%) than exotic cover (12 to 42%) after knotweed removal. Their reference sites also had more native forb and shrub species (73 to 88%) than exotic species (12 to 25%), and more native plant cover (84 to 98%) than exotic cover (2 to 16%). However, the Satsop River knotweed-treated sites possessed proportionately fewer native forb and shrub species (30%) than exotic species (70%), and less native cover (11%) than exotic cover (89%). Even the Satsop River reference sites had fewer native forb and shrub species (39%) than exotic species (60%), and less native cover (38%) than exotic cover (62%). The Satsop River had a much larger channel and riparian zone, with extensive gravel bars and a wide floodplain. Its riparian areas were more open and subject to higher levels of seasonal flooding.

Vegetation assemblages in riparian areas with knotweed treatment were similar to reference sites in terms of native

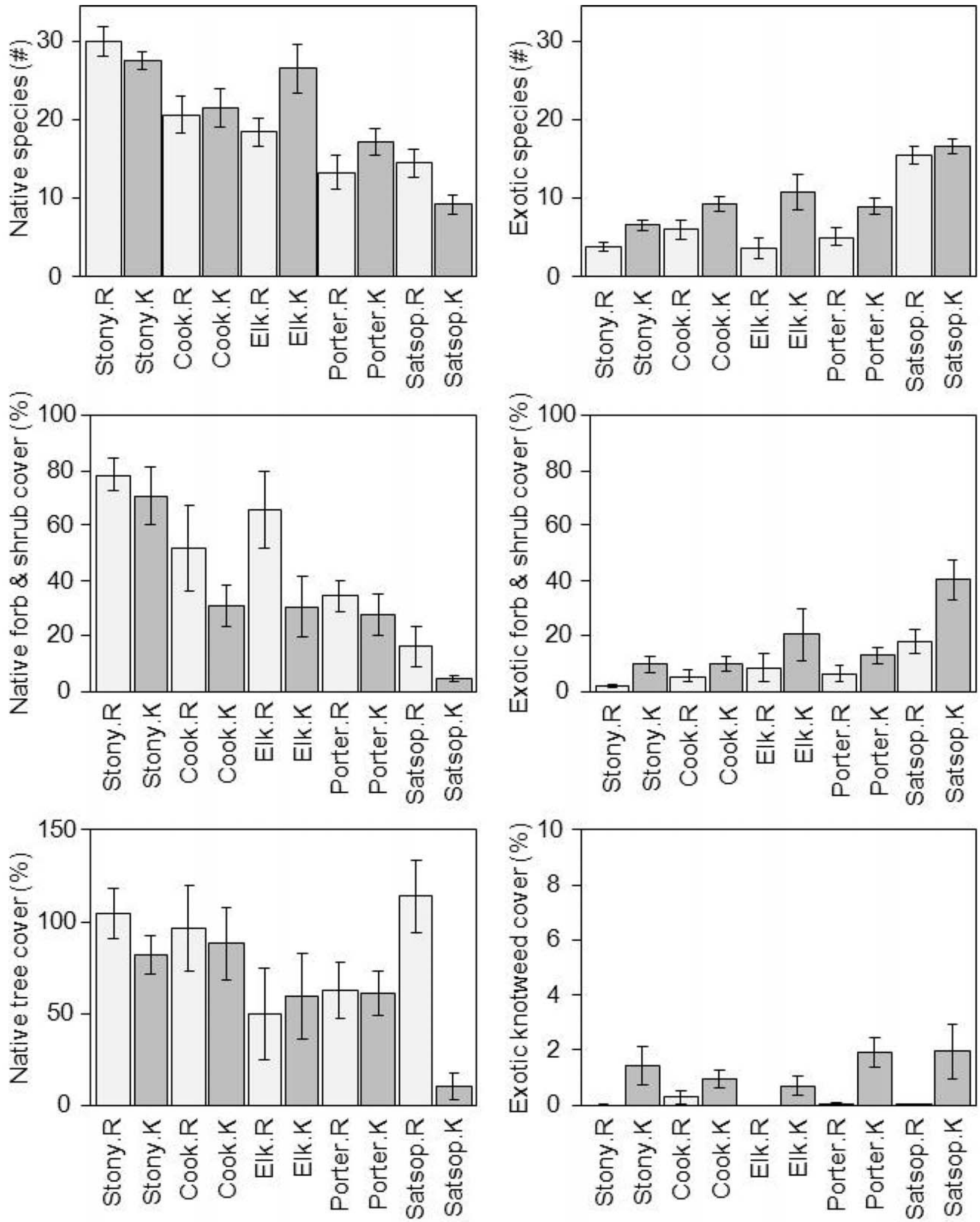

Figure 2. Native and exotic species richness and cover measures from reference (R, white bars) and knotweed-treated (K, grey bars) sites by stream (mean ± 1 SE). Streams are ordered from small (left) to large (right).

Table 2. Effects of (A) knotweed-treatment and (B) knotweed invasion levels and time on native and exotic species richness and cover measures, based on mixed model analysis from riparian sites along five streams. Coefficient values are given for the treatment (Tmt) effect and then only for those variables that are significant (see methods for complete list of habitat attributes tested). Positive values indicate related increases, negative values indicate related decreases. Significant p-values are indicated by asterisks (*$p < 0.10$, **$p \leq 0.05$, ***$p < 0.01$, ns = not significant). Certain variables were natural-log (ln) transformed prior to analysis to normalize distributions.

Response	Tmt	Stream × Tmt	Habitat attributes
	—— K ——		
A. Knotweed-treated sites (K, n = 47) versus reference sites (R, n = 47):			
Native spp. richness (#)	+0.63	**A	Overstory (%) +0.06***; Drainage area (km^2, ln) −1.67**
Exotic spp. richness (#)	+3.34**	ns	Overstory (%) −0.02*
Native forb & shrub cover (%, ln)	−0.46**	ns	Overstory (%) +0.01**
Exotic forb & shrub cover (%, ln)	+1.03***	ns	none
Native tree cover (%)	−25.71	***B	Overstory (%) +1.26***
Knotweed cover (%, ln)	+0.62***	ns	Reed canarygrass (%, ln) +0.07**

Response	Stems	Knotweed Area	Years since Tmt	Years of Tmt
	—— # ——	—— m^2 ——	—— # ——	—— # ——
B. Knotweed-treated sites (n = 47):				
Native spp. richness (#)	ns	ns	ns	ns
Exotic spp. richness (#)	ns	ns	ns	ns
Native forb & shrub cover (%, ln)	ns	ns	ns	ns
Exotic forb & shrub cover (%, ln)	ns	ns	ns	ns
Native tree cover (%)	−11.06*	ns	ns	ns
Knotweed cover (%, ln)	+0.15*	ns	ns	ns

[A] Native taxa richness: R > K at Satsop, R < K at Elk, R = K at the other streams.

[B] Native tree cover: R > K at Satsop, R = K at the other streams.

species richness and tree cover. However, native richness and tree cover were greater at the reference sites than at knotweed sites along the Satsop River, whereas native richness was lower at reference sites along the smaller Elk Creek (Table 2, Figure 2). Native forb and shrub cover was significantly greater at reference sites than at knotweed-treated sites. Exotic vegetation measures of species richness, forb and shrub cover, and knotweed cover were significantly greater at the knotweed-treated sites than at reference sites. Overstory cover was positively associated with greater levels of native species and cover (Table 2). Native species richness was negatively associated with stream size (as represented by drainage area). Reed canarygrass cover was positively associated with live knotweed cover, even though knotweed was extremely reduced after treatments (Table 2, Figure 2). Results thus indicated that sites in which knotweed control had occurred had a greater abundance of exotic vegetation than reference sites.

The NMS ordination of plant species primarily highlighted the compositional differences between the knotweed-treated and reference sites along a gradient of stream size (Figure 3A). Sites on larger streams, like Satsop River and Porter Creek, had greater site distance from and height above the channel, in addition to greater drainage area. Plant communities from Satsop River and Porter Creek sites were positively correlated with axis 1. Sites on smaller streams, like Stony, Cook and Elk Creek, had greater overstory cover and their plant communities were negatively correlated with axis 1. Knotweed-treated versus reference site differences were only slightly correlated with axis 3 (Figure 3B). The habitat attributes measured in this study were not correlated with axes 2 or 3 (Figure 3).

Indicator plant species at the knotweed-treated sites were most often exotic forbs, such as tansy ragwort (*Senecio jacobaea* L.) and bull thistle (*Cirsium vulgare* (Savi) Ten.), but included one exotic shrub (Himalayan blackberry) and one native forb fringed willowherb (*Epilobium ciliatum* Raf.) (Table 3). Indicator plant species at the reference sites were all native and composed of a mix of forbs, shrubs, and trees.

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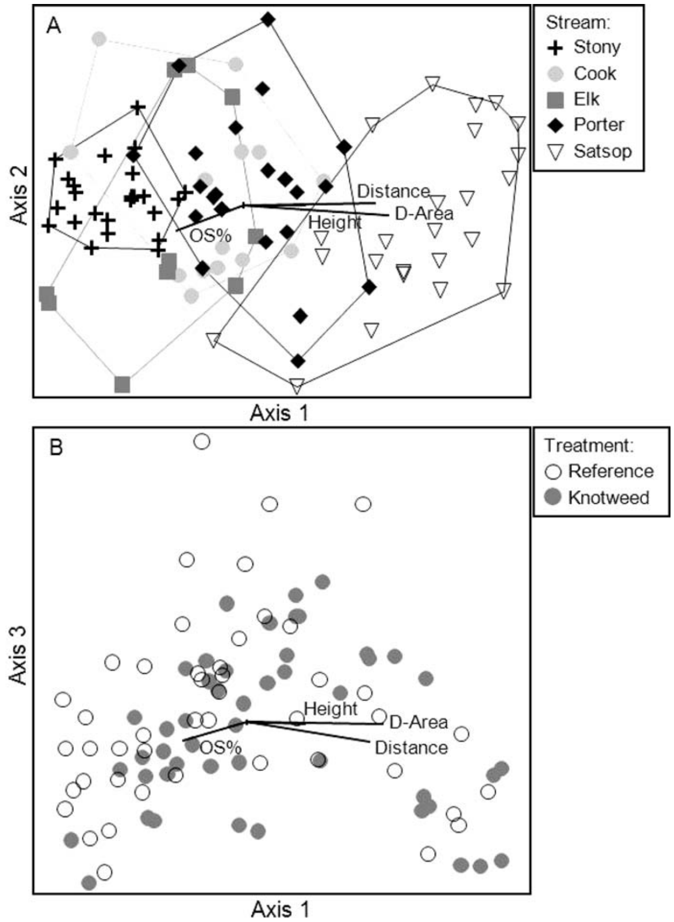

Figure 3. NMS ordination of vegetation community composition (% cover of 135 taxa) from both knotweed-treated and reference sites (n = 94) along five streams. The 3-dimensional ordination explains 88% of the variation among sites: 56% axis 1, 13% axis 2, and 19% axis 3. Vectors (black lines) are habitat attributes with correlation's greater than 0.50: overstory cover (OS%), drainage area (D-Area), distance from channel, and height above channel.

Invasion Strength and Herbicide Timing. Vegetation abundance in the knotweed-treated sites was generally not influenced by the size of the pretreatment knotweed patch (area or stem count), nor by herbicide application timing (number of years since the first treatment or number of years of treatment). Notable exceptions involved the number of knotweed stems prior to the first herbicide application, which was negatively correlated with native tree cover and positively correlated with remaining knotweed cover (Table 2). However, these correlations were not strong enough to be observed in the 2-dimensional ordination of plant communities from the knotweed-treated sites (Figure 4). The ordination of plant species from the knotweed-treated sites only is similar to the previous ordination in terms of correlations with stream size (i.e., drainage area), site distance from and height above channel, and overstory cover. In this case though, reed canarygrass cover was positively correlated with the Porter Creek sites specifically (Figure 4).

Table 3. Indicator plant species associated with knotweed-treated or reference sites from the five study streams (indicator value in parentheses, $p < 0.1$). Species are noted as being either exotic (E) or native (N) to western Washington State, and a forb (F), shrub (S), or tree (T).

Knotweed-treated sites	Reference sites
Cirsium vulgare $^{E, F}$ (30)	*Cornus nuttallii* $^{N, T}$ (27)
Digitalis purpurea $^{E, F}$ (43)	*Dicentra formosa* $^{N, F}$ (25)
Epilobium ciliatum $^{N, F}$ (57)	*Oemleria cerasiformis* $^{N, T}$ (30)
Polygonum × *bohemicum* $^{E, F}$ (67)	*Polystichum munitum* $^{N, F}$ (37)
Rubus armeniacus $^{E, S}$ (39)	*Rubus spectabilis* $^{N, S}$ (48)
Senecio jacobaea $^{E, F}$ (33)	

Discussion

Passive Reestablishment. There was no *a priori* reason to think that eradicating knotweed would prompt full-scale recovery of native vegetation when other exotic species were present in the same stream system. Eradicating knotweed removes a dominant competitor, thus providing resource and niche opportunities that favor the recruitment of rapidly-growing species (Davis et al. 2000; Shea and Chesson 2002), and fluvial transport of propagules aids colonization by new species in recently disturbed riparian habitats. Removing knotweed does not alter the underlying mechanisms that govern vegetative reestablishment.

In western Washington, we found that knotweed removal accompanied by unmanaged (passive) riparian vegetation reestablishment resulted in predominantly native plant assemblages along 2nd and 3rd order streams, but not along a larger 4th order river. Small forested streams in the Pacific Northwest tend to have high overstory cover, dense streamside vegetation, and more topographic shading (Gregory et al. 1991) that may limit shade-intolerant exotic plants and their dispersal (Magee et al. 2008). Large rivers generally have low gradients, wide sinuous channels, and large active floodplains (Naiman et al. 2005) that are more exposed to sunlight, favoring the spread of shade-intolerant species from adjacent forests and disturbed areas. Many invasive plants are early successional species that perform best in recently disturbed habitats with abundant sunlight (Davis et al. 2000). Anthropogenic disturbances, such as agriculture and grazing, are also more likely to occur in these lowland areas, further increasing the potential for colonization of herbicide treated sites by exotic species (Planty-Tabacchi et al. 1996).

Removing knotweed opens riparian areas for colonization by both native and exotic species. Along some stream channels in this study, reed canarygrass, Scotch broom,

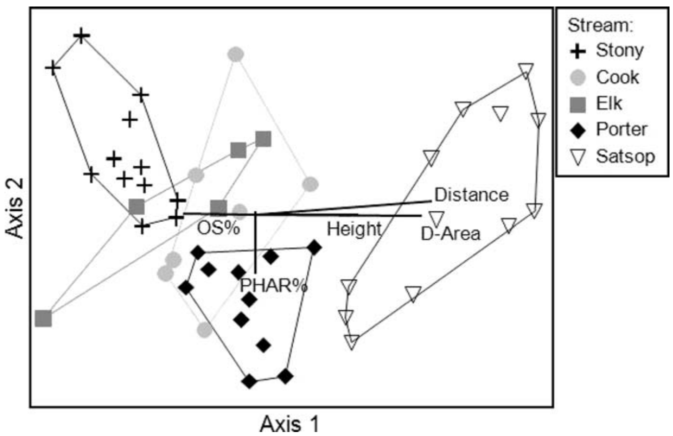

Figure 4. NMS ordination of vegetation community composition (% cover of 104 taxa) from knotweed-treated sites ($n = 47$) along five streams. The 2-dimensional ordination explains 86% of the variation among sites: 70% axis 1 and 16% axis 2. Vectors (black lines) are habitat attributes with correlation's greater than 0.50: overstory cover (OS%), drainage area (D-Area), distance from channel, height above channel, and reed canarygrass cover (PHAR%). Initial knotweed patch size, number of stems, number of years since initial treatment, and the number of years of active treatment were not correlated with the ordination axes.

Himalayan blackberry, and creeping buttercup were abundant, in addition to other exotic species. Knotweed-treated sites typically contained more exotic species than reference sites, as represented by their respective indicator species. The indicator species were selected because of their relatively high abundance and frequency of occurrence in each of the two treatment groups. Nearly all of the indicator species for the knotweed-treated sites were exotic forbs, whereas all of the indicator species for the reference sites were native forbs, shrubs, or trees, indicating a more established and complex vegetation structure compared to the forb-dominated knotweed-treated sites. In general, we found that knotweed was often replaced by other exotic plants 3 to 6 years after the initial herbicide treatment, especially along the largest channel.

A similar study along the Dickey River (3rd order) in northwest Washington State, 200 km (125 miles) north of the Chehalis basin, observed similar results after giant knotweed removal (Urgenson 2011). There, knotweed treated with herbicide resulted in rapid reestablishment of native species one and two years post-treatment, but exotic species ground cover increased significantly between the first year and second year after treatment, "suggesting further potential for increase" (Urgenson 2011). In our study, exotic plant cover was more developed 3 to 6 years after knotweed removal; however, neither study demonstrated when exotic species cover would peak. Many of the exotic species invading riparian areas along the Dickey River were the same as those in the Chehalis basin, including Himalayan blackberry and tansy ragwort.

Management. The broadleaf herbicides glyphosate and imazapyr are commonly used to control invasive plants near aquatic ecosystems. Both herbicides are nonselective, with imazapyr potentially more persistent in the soil, and thus they have the potential to influence vegetation composition and structure over time. Knotweed plants along the streams of Cook Creek, Porter Creek, and Satsop River were treated with foliar applications of imazapyr, whereas the oldest knotweed patches along the streams of Stony and Elk were treated with both glyphosate and imazapyr over several years. The chance of affecting nontarget plants increases with broad-scale foliar applications, necessary when treating very large, dense patches of knotweed. After the first year or two of relatively broad foliar application, knotweed stems were mostly gone (90 to 95% reduced in the first two years of treatments: A. Boe, personal communication). Knotweed regrowth was persistent, but drastically reduced, and foliage was directly spot-sprayed, such that effects on nontarget plants were believed to be negligible.

Prior to the first herbicide application, 10 of the knotweed-treated sites in this study had 10 to 35 knotweed stems (< 1 stem m^{-2}), while the other sites had more than 50 stems (≥ 1 stem m^{-2}; note that 1 full grown stem with leaves can cover 0.5 m^{2}). Stem count is a common measure of knotweed establishment, with low counts representing newly established colonies, and high counts representing full infestations. Herbicide effects on nontarget vegetation may be different in recent, sparse infestations versus well established, dense growths. However, we did not find pretreatment knotweed invasion levels (stem count or patch size) or herbicide timing (years since initial treatment or number of years of active treatment) to have a significant effect on most vegetation responses, nor did these parameters influence overall community composition. The exceptions were lower tree cover and higher current knotweed cover with greater pretreatment knotweed stem counts. Knotweed was able to become denser in areas where overhead tree cover was already low. Knotweed regrowth was more abundant and persistent (i.e., harder to kill) in areas where knotweed was originally quite dense. Our vegetation surveys occurred at least three years after the first herbicide treatment, while Urgenson (2011) observed native plant reestablishment in knotweed sites one year after foliar application with glyphosate. It appeared that foliar applications of glyphosate and imazapyr did not have noticeable effects on nontarget species, although these studies were not designed to test that hypothesis directly.

Once the majority of the knotweed was removed, passive reestablishment of native plant communities required several years and was influenced by the presence of exotic species. Exotic plants were able to quickly colonize, dominate succession and alter site conditions, potentially thwarting long-term restoration objectives. In the years immediately after knotweed removal, monitoring and controlling secondary invasive species while their abundance is low may be critical for the recovery of native plants. If an area has been invaded by other exotics prior to knotweed removal, controlling multiple species at the same time may be necessary for restoration success. Information on the abundance and composition of other invasive plants may help managers prioritize areas for knotweed or multiple species removal and additional post-treatment restoration actions.

In highly disturbed riparian areas, especially along large streams, active re-introduction of native plant species may help promote recovery and prevent exotic plant re-invasions. Replanting native species may inhibit invasion by exotic plants through increased direct competition or propagule pressure (Seabloom et al. 2003; Stevens and Fehmi 2011; Sweeney et al. 2002). In addition, planting larger native plants appropriate to floodplain riparian settings can provide shade, potentially weakening shade-intolerant exotic seedlings while supporting shade-tolerant native species (Funk and McDaniel 2010). McClain et al. (2011) found that disturbed sites with replanted understory species had more native species than unplanted disturbed sites, but fewer native species than undisturbed forests. They further suggested that passive reestablishment or initial replanting serves only to restore a few common native understory species, but that continued intervention over time as site conditions become appropriate (e.g., planting shade-adapted species as the canopy develops) may be needed to restore an entire native assemblage. However, management actions can sometimes result in unwanted consequences (e.g., planting native trees may attract ungulate browsers that consume plantings and promote dispersal of exotic plants). Therefore, we suggest that managers carefully weigh the advantages and potential disadvantages of management intervention options before committing to long-term restoration strategies.

Riparian areas are dynamic, but successional processes in native plant assemblages provide predictable energy and material inputs to streams that are essential for maintaining healthy, productive aquatic communities (Naiman et al. 2000). Exotic plant species can disrupt these processes, setting riparian systems on a new trajectory. It is rarely, if ever, possible to return riparian zones to their pre-invasion species assemblage with management, partly because management tools have some peripheral effects (Pearson and Ortega 2011). We found that post–knotweed re-establishment of native plant communities varied with stream size and riparian habitat attributes, and this finding has implications for how riparian areas can be managed after knotweed removal. Understanding when and where to employ passive reestablishment of vegetation, or employ active intervention to accelerate native vegetation recovery, will enhance the ability of natural resource managers to achieve restoration goals.

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Pesticide Precautionary Statement: This publication reports research involving pesticides. It does not contain recommendations for their use. CAUTION: Pesticides can be injurious to humans, domestic animals, desirable plants, and fish or other wildlife—if they are not handled or applied properly. Use all pesticides selectively and carefully. Follow recommended practices for the disposal of surplus pesticides and pesticide containers.

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