Up Smith Creek without a Paddle: A Case Study on the Barriers to Stream Restoration Assessment

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Riparian zones are uniquely diverse and play vital roles in landscape planning and aquatic systems restoration (Naiman et al. 1993, Naiman et al. 1997). In this case study, we focus on the Shenandoah Valley of Virginia, an area with a history of agricultural land use which has resulted in the degradation of the Chesapeake Bay Watershed. Stream impairment is a widespread issue and as a result the United States Department of Agriculture (USDA) Farm Service Agency's (FSA) Conservation Reserve Program (CRP) began incentivizing the protection or restoration of natural habitat on agricultural land in 1985 (USDA FSA 2019, USDA 2020, USDA FSA 2021a, USDA FSA 2021b, VA DCR 2021). In 1998, the FSA created the Conservation Reserve Enhancement Program (CREP) within CRP to serve as a partnership between federal and state governments to support restoration projects around the country (USDA 2020, USDA FSA 2021a, VA DCR 2021). Through CRP, millions of acres have undergone restoration since 1998 (USDA 2020). Our case study focuses on Virginia's Chesapeake Bay CREP (part of the larger FSA program) in which over 1,500 landowners have enrolled and more than 18,000 acres have undergone restoration since 2000 (VA DCR 2020; VA DCR 2021).

Restoration through CREP consists of excluding cattle with fencing and establishing forested buffers along streams via planting trees with tree shelters to reduce fecal bacteria and sediment, control flow rates, control microclimates within the stream, and provide food and habitat for macroinvertebrates as well as larger wildlife (Daniels et al. 1996, Naiman et al. 2000, Tabacchi et al. 2000, Lee et al. 2003, Sweeney et al. 2004, Line 2015, O'Toole et al. 2017, Zeckoski et al. 2017, VA DCR 2021). As of June 2020, Chesapeake Bay CREP reported that riparian buffers have been established along 1,000 miles of stream bank throughout the watershed (VA DCR 2020).

Current success of a CREP planting means that roughly 50% of trees survive after one year. Although attempts are made to plant species likely to survive, plantings are limited by cost, time, and seedling availability. For similar reasons, rigorous data collection during initial restoration (tagging trees, documenting distribution and composition of species, tree size measurements, etc.) and follow-up surveys are also rare.

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The success of restoration, in practice, is affected by many factors from the composition and distribution of planted trees to natural barriers such as deer or girdling by rodents or tree shelters. It is well known that deer browse increases mortality of many plant species, which creates a significant barrier to restoration (Opperman and Merenlender 2001, Horsley et al. 2003, Abrams and Johnson 2012). Since failure is determined by survival, follow up surveys to assess the growth and survival of planted trees may be necessary to inform future restoration. However, assessment typically begins and ends with the initial determination of success.

Here we explore issues inherent with restoration projects like those within CREP, identify and discuss barriers to and suggestions for assessment of restoration projects. We focus on a single property that was part of a 2005 CREP restoration project along Smith Creek in Rockingham County, Virginia. Due to historical land use this 1.15 kilometer (km) section streambank has been devoid of a riparian buffer for 100–150 years. Beginning in 2005, 12,561 bare-root seedlings (~45 centimeters (cm) tall and ~1 cm in diameter) were planted randomly across ~65 acres and several properties. This planting included *Fraxinus americana* (white ash), *Quercus rubra* (Northern red oak), *Celtis occidentalis* (common hackberry), *Acer rubrum* (red maple), *Alnus serrulata* (smooth alder), and *Platanus occidentalis* (American sycamore) with four-foot (ft) tree shelters. In 2009, several hundred larger saplings (~2–5 cm in diameter, with root-balls) with tree shelters were planted. Both plantings were considered successful with more than half of species surviving after one year.

We revisited the property in the fall of 2018 to assess survival and condition of trees from these two plantings. Data regarding initial height and diameter at the root collar were not available, as they were not documented at the time. Undergraduate students from James Madison University tagged trees soon after planting occurred—which is unique and not required for CREP. Data corresponding to tag numbers were no longer available, but the presence of tags was key to identifying planted individuals and the tag shape allowed us to distinguish between those planted in 2005 and 2009 (hereafter referred to as cohort).

We documented tree mortality and size, species composition, deer-browse, and stream proximity. Using these data, we propose several measures to improve future stream-restoration and their evaluation—with emphasis on monitoring restoration sites beyond the first two years. We established nine random transects along a 500-meter (m) section of stream, covering 1.55 acres of the 35.9 acre planting. Transects were 10 m wide, varied in length per

Table 1. Living trees documented from sampling transects across Rainbow Hill Farm. Includes species names (and USDA FIA code), count of living individuals, proportion of dead individuals documented for each species, median diameter at breast height (dbh) of living adult trees for each species, proportion of individuals with evidence of deer browse (living or dead), and sample size (n; number of individuals large enough to measure dbh).

Species (USDA FIA code)	Live Count	Proportion Dead	Median dbh (cm) ± sd	Proportion Browsed	n
Acer rubrum (ACRU)	17	0.00	15.1 ± 2.4	0.29	12
Alnus serrulata (ALSE2)	16	0.27	14.05 ± 4.1	0.50	10
Cercis canadensis (CECA4)	9	0.00	16.4 ± 5.7	0.11	6
Cercis occidentalis (CEOC)	6	0.25	9.6 ± 3.6	0.38	5
Fraxinus americana (FRAM)	49	0.17	8.3 ± 3.1	0.59	28
Juniperus virginiana (JUVI)	2	0.00	8.1 ± 1.8	0.00	2
Liquidambar styraciflua (LIST2)	7	0.00	13.5 ± 1.6	0.00	7
Pinus strobus (PIST)	3	0.00	19.2 ± 0.7	0.00	3
Platanus occidentalis (PLOC)	113	0.05	10.8 ± 5.1	0.05	109
Quercus rubra (QURU)	10	0.17	12.5 ± 0.9	0.83	2
Unknown	43	0.69	11.4 ± 5.6	0.39	NA

the distance from the streambank to the property boundary (40–100 m), and followed an azimuth of ~270°. We identified planted individuals to species and cohort (evidenced by tags) while noting the presence of tree shelters, diameter-at-breast-height (dbh; measured ~1.37 m above ground) in centimeters (cm), evidence of deer browse (via examining twigs' breakage or tearing), and mortality of planted trees. The distance from the stream was later calculated from the stream's center. Only individuals (dead or alive) with either a tag or tree shelter were documented.

We documented 275 living trees along our transects, with 131 individuals appearing to be from the 2005 cohort and 124 from the 2009 cohort (Table 1, Figure 1, Figure 2). We estimated the density of surviving trees from the initial 2005 planting to be ~84 trees per acre, while the original density was ~195 trees per acre. This is nearing 50% mortality; however, the 2009 planting improved the overall survival, with a surviving tree density of ~80 trees per acre.

Trees we found with tree shelters but without tags we presumed planted but could not be assigned to a cohort

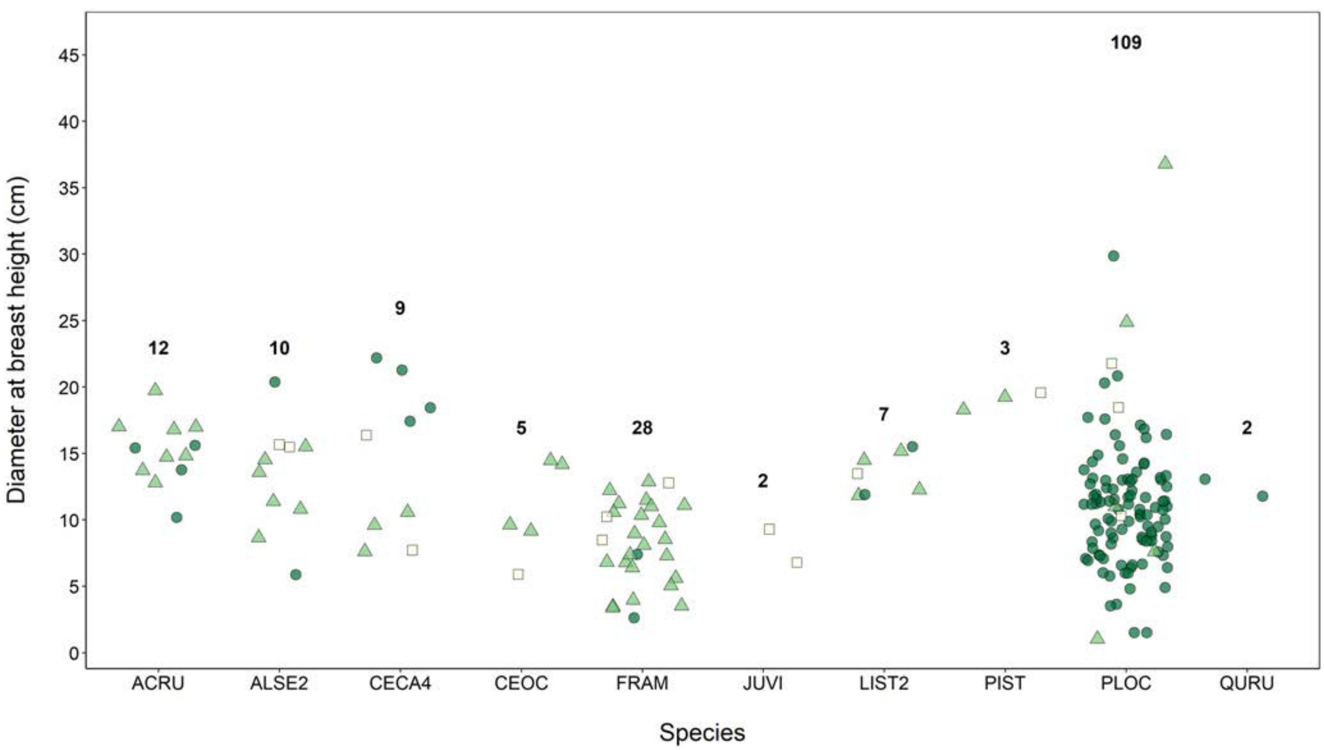

Figure 1. Diameter at breast height (dbh) of adult tree species at Smith Creek. Numbers on the chart denote the total count of surviving individuals of the corresponding species. Cohorts are denoted by symbol shape; circle = 2005, triangle = 2009, square = NA. Species codes are as follows: ACRU (*Acer rubrum*), ALSE2 (*Alnus serrulata*), CECA4 (*Cercis canadensis*), CEOC (*Celtis occidentalis*), FRAM (*Fraxinus americana*), JUVI (*Juniperus virginiana*), LIST2 (*Liquidambar styraciflua*), PIST (*Pinus strobus*), PLOC (*Platanus occidentalis*), QURU (*Quercus rubra*).

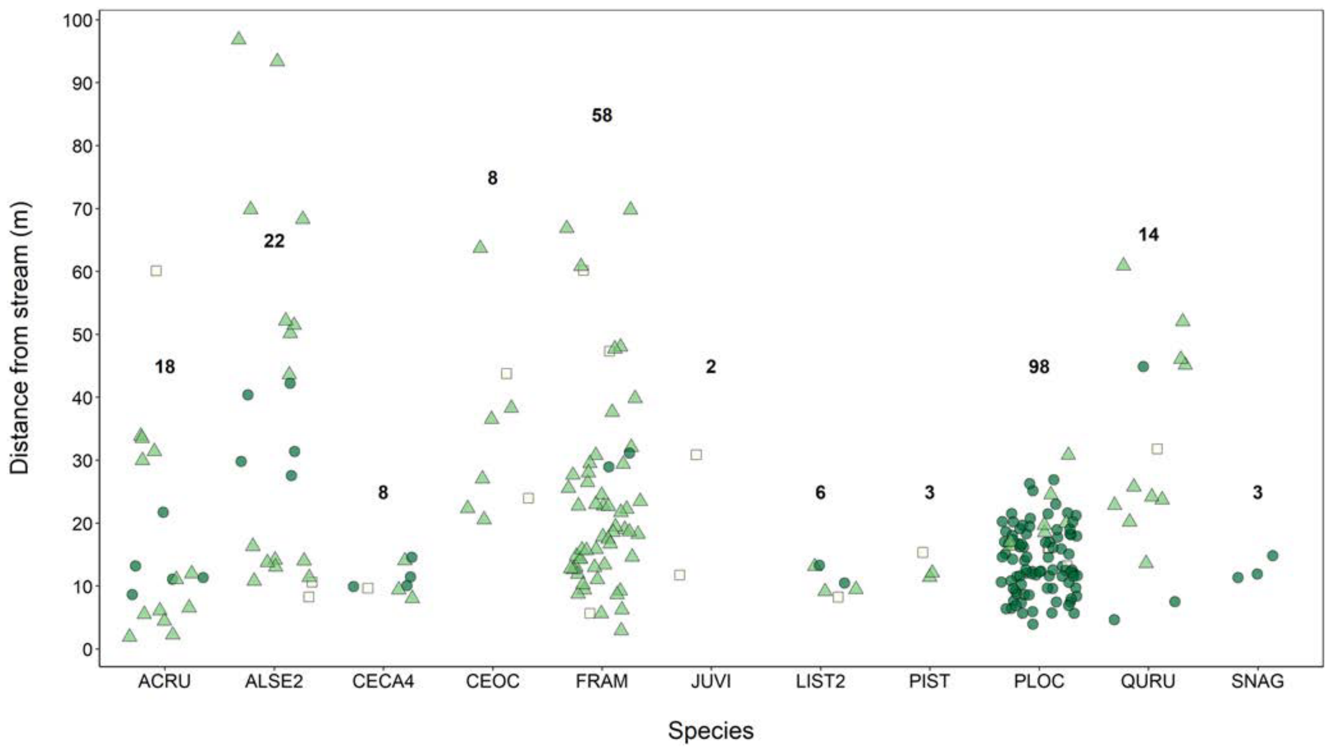

Figure 2. Proximity to stream (m) for all cohorts across species at Smith Creek. Numbers on the chart denote the total count of surviving individuals of the corresponding species. Cohorts are denoted by symbol shape; circle = 2005, triangle = 2009, square = NA. Species codes are as follows: ACRU (*Acer rubrum*), ALSE2 (*Alnus serrulata*), CECA4 (*Cercis canadensis*), CEOC (*Celtis occidentalis*), FRAM (*Fraxinus americana*), JUVI (*Juniperus virginiana*), LIST2 (*Liquidambar styraciflua*), PIST (*Pinus strobus*), PLOC (*Platanus occidentalis*), QURU (*Quercus rubra*), SNAG (snag).

with high certainty and therefore were not used in statistical analyses. We documented 128 dead individuals; however, this number is a low estimate as many dead individuals were either washed away or unrecognizable (Table 1). We did not attempt to estimate how many trees were lost in previous years as no initial planting data were obtainable. Assessments of mortality were only attempted if tags, shelters, or stakes (that held the shelter) were present, and mortality was only noted for individuals which could be confirmed dead (i.e., no leaves and all twigs appear dead).

The five most frequent species were *P. occidentalis*, *F. americana*, *A. rubrum*, *A. serrulata*, and *Q. rubra* (Table 1, Figure 1, Figure 2). There were no significant differences in dbh between species from the initial 2005 planting, although *A. rubrum* planted as saplings in 2009 were significantly larger than *F. americana* of the same cohort (pairwise Wilcoxon rank-sum test; W = 144, $p < 0.05$, Figure 1). Notably, the largest individuals documented were *P. occidentalis*, near the stream (max dbh = 36.8 cm; Figure 1, Figure 2). Anecdotally, we did not observe many tree seedlings, indicating that little to no natural regeneration is taking place. We could not compare growth or survival rates between species due to the low sample size of surviving individuals confirmed to be from the 2005 planting (n = 82 for *P. occidentalis*, n < 5 for all other species, Figure 1) and lack of data regarding initial tree sizes, species composition, and distribution. This underlines the inherent issue with assessing restoration success without initial measurements and tagging.

Fraxinus americana exhibited the most deer browse (n = 35, 78%), while *Q. rubra* had the highest percentage of browsed individuals (n = 10, 83%, Table 1). Many dead *Q. rubra* had evidence of browse, and all browsed individuals were unable to grow above the height of their four foot shelter (Table 1). Likewise, we observed that many trees were unable to grow above their shelters after 9–13 years due heavy deer browse. These individuals were exactly four feet tall with all branches clipped at the opening of the tree shelter. We propose that these shelters are not serving their purpose of negating impacts of browse: those trees are technically alive, but they are not facilitating restoration.

In a ten-year post-restoration assessment, Drayer et al. (2017) concluded that tree shelters may not always provide long-term benefits for tree success. Similarly, and contrary to the requirement that all trees planted within CREP must have tree shelters, we observed trees that had outgrown their shelters but whose shelters did not break, resulting in girdling. To ensure this does not impact restoration success, we suggest checking all tree shelters after five years and removing them if necessary. Alternatively, rejecting their use altogether in favor of other options may be appropriate.

Taller fencing or wire cages could be used, with increased cost and maintenance. We suggest planting trees within deer cribs (~10 × 1 m deer exclosure) which may improve survival with less effort and lower costs.

In close proximity to the stream, *P. occidentalis* were significantly more abundant than *Q. rubra, F. americana, C. occidentalis,* and *A. serrulata* (pairwise Wilcoxon rank-sum test; $W = 2384$, $p < 0.005$). Within 20 m of the stream, the density of *P. occidentalis* is currently ~188 trees per acre (Figure 2). Therefore, the planting of *P. occidentalis* could be considered a great success but only within 20 m of the stream. We found that the distribution of surviving trees does not match that of the original random planting. This may be relevant for future restoration projects, as success could be significantly improved by planting trees in areas chosen per species requirements (e.g., moisture, soil characteristics, stream proximity).

These plantings were considered a success early on and the 2009 planting has undoubtedly improved this restoration. However, we do not feel confident in classifying it as such when considering both the conservative yet high number of dead individuals documented here (~30% of individuals surveyed were standing dead), the inevitable mortality of *F. americana* due to emerald ash borer (the second most abundant species, comprising ~18% of living trees), the clear impact of deer browse, and the lack of natural regeneration. With adequate documentation and management of data (i.e., file types robust to time such as .csv or .txt files with readme files to describe data and their organization and maps all placed in a secure repository), evaluating restoration projects could become both easier and more informative.

Restoration projects occur with the intent to mitigate erosion, create wildlife habitat, and improve stream quality but thorough assessment is rare (Daniels et al. 1996, Naiman et al. 2000, Tabacchi et al. 2000, Lee et al. 2003, Sweeney et al. 2004, Line 2015, O'Toole et al. 2017, Zeckoski et al. 2017, VA DCR 2021). Without long-term evaluation, we are hindering the improvement of future restoration and may not recognize easily avoided failures until decades later. We recommend several practices to aid in monitoring and perhaps ensuring the success of these projects: 1) set a data management plan prior to restoration stating how data will be managed and by whom, file types to be used, and data storage locations; 2) store data in logically organized folders with informational readme files and in formats that are more robust to time and software changes (.csv, .txt, .pdf); 3) consider planting trees in clusters of ~20 within deer cribs narrow enough such that deer will not jump into them (~1 m wide), negating the need for individual tree shelters as a measure against deer browse and potentially allowing landowners to achieve tree survival goals while planting much fewer trees initially; and 4) conduct assessments every five years for at least a decade to monitor tree survival and restoration success whenever possible.

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Observations of Nocturnal Upland Habitat Use by the *Rana Draytonii* (California Red-Legged Frog), and Implications for Restoration and Other Activities

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Rana draytonii (California red-legged frog), listed as a threatened species in 1996 by the U.S. Fish and Wildlife Service (USFWS), is an example of a declining wildlife species for which many pieces of its natural history are still being assembled (Jennings and Hayes 1994, USFWS 2002, Lannoo 2005). This paucity of information is surprising given the considerable attention the species has received from researchers. In particular, upland habitat use has been studied using radio-telemetry (Rathbun and Murphey 1996, Bulger et al. 2003, Fellers and Kleeman 2007, Tatarian 2008) and through direct observation (Alvarez 2004, Surber 2019, Alvarez et al. 2021). Nevertheless, numerous aspects of the use of upland habitat by *R. draytonii* remain enigmatic, including the effect of restoration and enhancement projects within occupied habitat. Alvarez et al. (2002) included information on the positive response of *R. draytonii* to a habitat restoration project that included non-native fish removal, but the response of *R. draytonii* to such ecological restoration activities as *Lithobates catesbeianus* (American bullfrog) control, silt and vegetation reduction in stock ponds, or riparian zone enhancement has yet to be reported by ecologists involved in such projects (pers. obs.). To contribute to the natural history literature on the species, below we report a dramatic shift in nocturnal activity in upland habitat following a large-scale habitat restoration project that was installed in occupied habitat.

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We surveyed a 2.1 km (1.3 mile) perennial section of Kellogg Creek (Contra Costa County, CA) in the eastern San Francisco Bay Area in 2013, during both daylight (approximately 1500 hrs to 1800 hrs) and nighttime (approximately 2000 hrs to 2330 hrs), while conducting *L. catesbeianus* control. Our surveys were conducted once per month for two years, excluding the peak of the breeding season (i.e., December–February) for *R. draytonii*.

Surveys were conducted by walking the upper edge of the bank, outside of the stream channel, and scanning the creek channel from the open water to the top of the bank for both *R. draytonii* and *L. catesbeianus*. The dominant upland habitat type at the time of the surveys was heavily grazed annual grasslands with little to no riparian vegetation lining Kellogg Creek. We collected data on size cohort, location, and position for all anurans observed in or along the creek. Every *L. catesbeianus* encountered was collected, when possible; individual *R. draytonii* were left in place undisturbed. Water temperature, air temperature, and relative humidity were collected at the beginning of both daytime and nighttime surveys efforts.

During surveys conducted in the summer of 2013 our daytime observations averaged ≤ 5 individuals of each frog species (*R. draytonii* and *L. catesbeianus*) per visit, and we noted that all individuals were in or within 2.5 cm (1 in) of the water's edge. Data we collected during nighttime surveys on the same dates as daytime surveys were similar for *L. catesbeianus*, never exceeding seven individuals, all of which were in the water. Nighttime observations of *R. draytonii*, however, were different; each visit included ≥ 100 *R. draytonii* (max. = 141) observed, with 10% or fewer using water as their preferred habitat when observed during nighttime hours. The majority of *R. draytonii* (approximately 90% or more) were found on the top of the bank of the creek, outside of the stream channel, as much as 3 m (9.8 ft) from, and approximately 0.6 m (2 ft) to 4 m (13.1 ft) above, the water's surface.

The observations reported above preceded an extensive creek restoration project, lasting seven months, and designed to decrease bank slopes, increase vegetation adjacent to the creek, and improve habitat for native wildlife, in particular *R. draytonii* and *Actinemys pallida* (southwestern pond turtle). Heavy equipment was used as part of the restoration work to draw back slopes and contour creek banks. Although the site was originally comprised of heavily grazed annual grassland, the restoration project included hand-planted *Populus fremontii* (Fremont cottonwood), *Aesculus californica* (California buckeye), *Sambucus cerulea* (blue elderberry), *Frangula californica* (coffeeberry), *Rosa californica* (California rose), *Muhlenbergia rigens* (deergrass), and associated weed cloth, mulch, and irrigation piping. Vegetation was planted at a density and spacing that ranged from 2 to 3 m between plantings, and plants ranged from 0.5 m to 3 m tall at the time of planting.