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70-foot waterfall 130-feet underground 0-9 Well cave north of Ozona. ©2023 Erich Ross Schlegel.

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Establishment of Native Vegetation Following Dredging of Torpedograss *(Panicum repens)* Above a Run-of-River Dam in a Central Texas Spring Run

Jeffrey T. Hutchinson[1]*, Scott Gardner[2], Analisa Saenz[3]

Abstract: Torpedograss (*Panicum repens* L.) is a highly invasive grass that is difficult to control in aquatic environments. Dredging was used to remove a torpedograss invasion above a run-of-river dam in a Central Texas spring run. Following dredging, the sediment was disturbed along a gradient from aquatic to upland habitats, and no vegetation was present. This study evaluated the success of replanting native aquatic, riparian, and terrestrial plants following the dredging of sediment accumulation to remove torpedograss above the run-of-the-river dam. Planting of native plants occurred in the disturbed areas on four occasions between September 2018 and November 2019. Planting of native graminoids resulted in the most successful establishment and increase of plant coverage at 5 years post-planting from the upland-mesic to the riparian zone. The aquatic habitat was dominated by the spontaneously occurring *Chara* spp. pre- and post-treatment. Planting of apical tips was not successful in the reestablishment of aquatic plants post-restoration. The upland to transitional zone was the most susceptible to spontaneous non-native plant invasions in this study, especially by yellow bluestem (*Bothriochloa ischaemum* [L.] Keng). At 5 years post-planting, no torpedograss was observed within the restoration area, indicating that excavation of sediment removed the torpedograss rhizomes resprouting.

Key words: spring run, invasive grass, run-of-river dam, restoration, dredging, native plants, Central Texas

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Terms used in paper

Acronym/Initialism	Descriptive Name
cm	Centimeter
cm^3	cubic centimeters
FAC	facultative plant
FACU	facultative upland plant
FACW	facultative wetland plant
ha	Hectares
K	Potassium
km	Kilometers
L	Liter
m	meter
m^2	square meters
m^3	cubic meters
N	nitrogen
NRCS	Natural Resources Conservation Service
OBL	obligate wetland plant
P	phosphorous
PCA	principal component analysis
UPL	obligate upland plant
USDA	U.S. Department of Agriculture
UTSA	University of Texas at San Antonio

INTRODUCTION

Dams are documented to alter a lotic system to a lentic system by blocking the flow of water, sediment, and fish movement (Nilsson et al., 2005). Above the dam, plant species richness, diversity, and functional traits are different compared to the former riparian zone pre-dam construction (Wang et al., 2014). Over time, the area above the dam creates suitable habitat for invasive vegetation due to siltation (Wang et al., 2022). Run-of-river dams are constructed across the channel at bank full level, allowing continuous flow, unlike larger dams constructed above the bank full level that contain water. Research indicates that run-of-river dams function similarly to undammed rivers and have minimal impact on channel morphology and sediment movement (Csiki & Rhoads, 2014; Magilligan et al., 2021).

Torpedograss (*Panicum repens* L.) is a C_4 perennial grass native to Europe, Asia, and Africa (Hossain et al., 1999). The species is considered one of the world's most aggressive grass weeds in agriculture (Holm et al., 1977) and natural areas (Langeland et al., 2008). It is widely naturalized throughout the New World tropics and subtropics, displacing native riparian and macrophytes, and it is currently distributed throughout the southeastern United States, California, and Hawaii (Langeland et al., 2008). In Texas, the species is documented from herbarium records in Trinity, Jefferson, Chambers, and Galveston counties in the eastern part of the state (Shaw, 2012). Torpedograss forms mats in water, spreading from an extensive network of rhizomes and stolons, with extensive mats documented in water ≤ 1.2 meters (m) deep, displacing native aquatic plants (Tarver, 1979; Sartain, 2003). The species invades a wide variety of habitats and can be found growing in aquatic, riparian, wetland, and terrestrial habitats in the southeastern United States, but it especially thrives in wetland and riparian habitats. Its invasive potential is due to its ability to establish in new areas from short stem fragments, rhizomes, and stolons.

C.L. Browning Ranch is a private conservation area (395 hectares [ha]) in Johnson City, Texas. Honeycutt Spring (30°16'05.19"N, 98°20'00.29"W) is one of the centerpieces of the ranch that supports a diverse community of aquatic flora

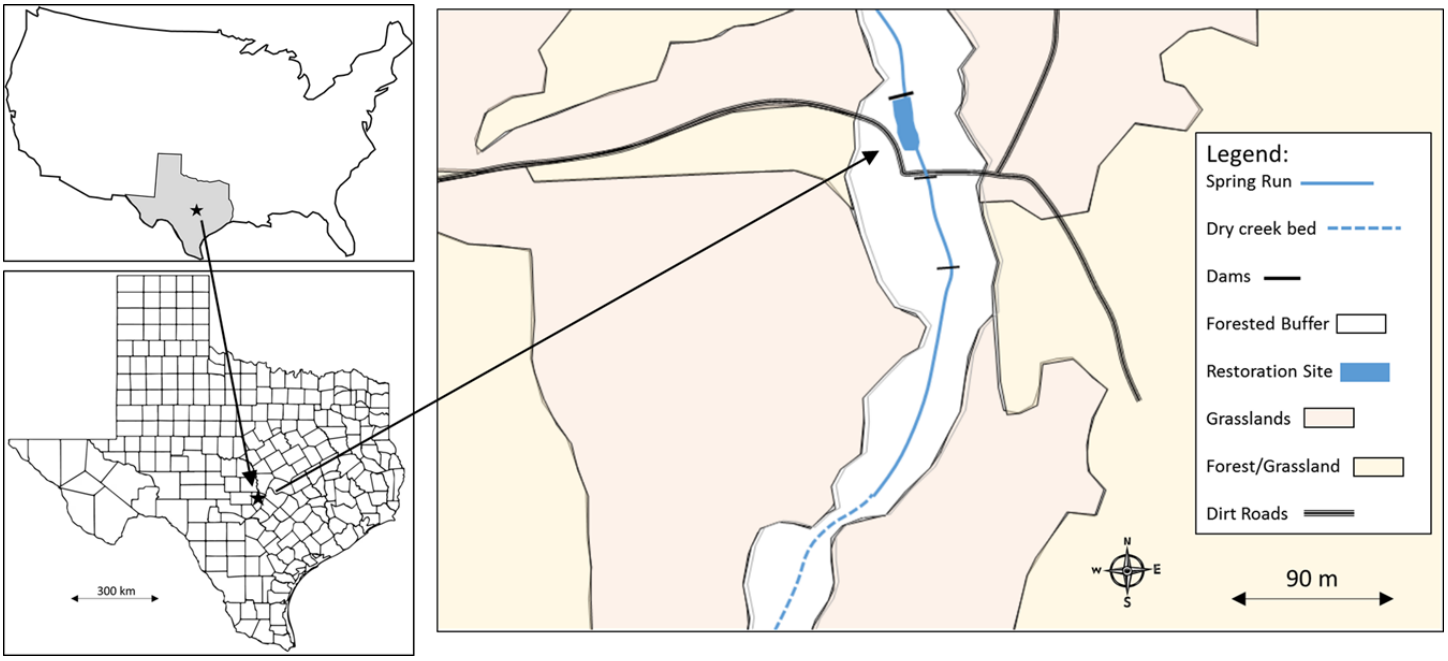


Figure 1. Location of restoration site (30°16'05.19"N, 98°20'00.29"W) at C.L. Browning Ranch, Johnson City, Texas, depicting the location in Texas relative to the United States (top left), the location in Texas (bottom left), and a vegetation map of the study site (right).

and fauna (Figure 1). A run-of-river concrete dam was constructed 600 m below the springhead, and the area above the dam was dredged out to create a small pool in the 1950s. Over 60+ years, silt has accumulated above the dam at depths of 0.9 to 1.10 m. The silted area above the dam became dominated by torpedograss (*Panicum repens* L.), and native sycamore trees (*Platanus occidentalis* L.) were encroaching, indicating succession from aquatic to mesic habitat.

C.L. Browning Ranch is an undisturbed natural area except for three run-of-river dams and limestone roadways that occur through the ranch. Development at C.L. Browning Ranch is limited to < 5% of the property and consists of housing and a maintenance facility. The spring run is buffered by a forest strip composed primarily of Texas live oak (*Quercus fusiformis* Small). The remainder of the habitat surrounding the spring run is tracts of grasslands and forest-grassland mixtures. There have been no known over-the-bank floods along the upper spring for 24 years (S. Gardner, personal communication, October 22, 2022). The lack of impervious structure and limited development within the C.L. Browning Ranch watershed likely allows percolation of precipitation into the soil and reduces runoff and discharge.

Restoration of altered habitats often requires the use of heavy machinery, resulting in significant soil disturbance. Invasive plant species can quickly invade the disturbed sites and create additional management problems, hindering the restoration goals. The objectives of this project were to evaluate the success of replanting native aquatic, riparian, and terrestrial plants following the dredging of sediment accumulation to remove torpedograss above the run-of-the-river dam and complete disturbance to aquatic, riparian, and upland habitat.

METHODS

Sediment and torpedograss removal

A site evaluation in 2017 indicated that 221 square meters (m^2) above the dam was silted and dominated by torpedograss. The main channel was reduced from 395 m^2 to 174 m^2 due to accumulated silt. The silted area upstream of the dam was dredged with an excavator during July 2018 to remove accumulated silt and torpedograss biomass, including roots and rhizomes. The estimated amount of sediment removed was 300 cubic meters (m^3; 30 m long x 10.6 m wide x 0.94 m deep). Sediment was removed where torpedograss occurred and included the main channel. The slope along the eastern bank was excavated at a 1:1 ratio at a 75–80° angle. Following dredging, the eastern side of the disturbance lacked vegetation in an area of 289 m^2 due to damage from the excavator and temporary storage of sediment removed from the eastern riparian area and channel. The western bank contained native vegetation and was left intact.

Replanting of native vegetation

All plants and seeds used for plant propagation were collected within 100 kilometers (km) of C.L. Browning Ranch and were native to Central Texas. Plants were propagated from seeds except for all aquatic plants, Emory's sedge (*Carex emoryi* Dewey), and beaked spikerush (*Eleocharis rostellata* [Torr.] Torr.), which were grown from fragments. Floating fragments of aquatic plants and the sedges were collected from the spring run at C.L. Browning Ranch and two locations in the Guadalupe River in Central Texas. Following collection, fragments of each species were planted within 4 hours in a 6-liter (L) mixture of 70% clayey soil of the Tarrant-Brackett association, commercially purchased sand (15%), and pea gravel (15%) in 21 L mesocosms. Potted fragments were exposed to full sunlight and a natural photoperiod of a 13–14-hour day in the greenhouse at the University of Texas at San Antonio (UTSA). Plants were watered daily and maintained for 2–3 months prior to the start of the study. For each species, 20–25-centimeter (cm) emergent apical tips were cut from greenhouse plants and placed in a plastic container with water to prevent desiccation during transport for replanting.

The seeds of native grasses were collected locally from the study site and from grasslands and ephemeral streams on UTSA's campus. Species selected for the study were based on native grasses common to Central Texas. The native grass species used in this study included *Bothriochloa laguroides* (DC.) Herter (silver beardgrass), *Panicum virgatum* L. (switchgrass), *Tridens albescens* (Vasey) Wooton & Standl. (white tridens), *Bouteloua curtipendula* (Michx.) Torr. (sideoats grama), *Tripsacum dactyloides* (L.) L. (eastern gamagrass), and *Bouteloua dactyloides* (Nutt.) J.T. Columbus (buffalograss). Seeds (n = 500 to 750) of each species were potted in 5.7 L rectangular plastic containers filled with a commercial potting soil (Miracle-Gro Premium Potting Mix, 0.21% Nitrogen [N], 0.11% Phosphorous [P], and 0.16% Potassium [K]). Seeds were watered every 2 days to maintain saturated soils and promote germination. After germination occurred, individual seedlings were removed once they reached 6–8 cm in height. Seedlings were potted in round plastic pots (763 cubic centimeters [cm³]) and filled with 740 cm³ of Miracle-Gro Premium Potting Mix. Seedlings were maintained under ambient conditions in UTSA's greenhouse for approximately 2–3 months until being planted at C.L. Browning Ranch.

Native aquatic, riparian, and terrestrial vegetation were planted at C.L. Browning Ranch on four occasions during 2018 and 2019 (Table 1). Aquatic plants were planted by inserting 8–10 cm of the cut apical tip into the sediment. For riparian and terrestrial plants, the soil was excavated with a shovel to a depth of 20 cm, and the soil was broken up to homogenize it. Native grasses were planted randomly within the disturbed area to promote equal distribution of species within the disturbed site.

Vegetation analysis

Vegetation was evaluated prior to treatment in June 2018 with line transects (Canfield, 1941). Aquatic vegetation in the spring run above the dam was analyzed for cover along five perpendicular transect lines that ranged in length from 9.9 to 11.8 m from the eastern and western banks of the water's edge. Upland-dry, upland-mesic, transitional, riparian, and aquatic habitats, based on moisture gradient from xeric to hydric, were analyzed along five 30 m transect lines parallel to the stream on the western side. Vegetation transects began 0.2 m from the water's edge and then every 2.5 m upslope. All vegetation ≤ 1 m in height was recorded along the transect line. The distance, to the nearest 0.5 cm, of each plant species along the transect line was recorded. Percent coverage of each species was determined by dividing the distance of each plant species along the line by the distance of the line. Plant species were identified based on taxonomic keys by Diggs et al. (1999) and Shaw (2012). Plant characteristics (native and non-native, growth form, life cycle, and wetland classification) were based on classification described by the U.S. Department of Agriculture (USDA) Natural Resources Conservation Service (NRCS) PLANTS Database (NRCS, n.d.). For plants that exhibited a mixed annual/perennial or biennial/perennial life cycle, depending on environmental characteristics such as precipitation and temperature, life cycle was classified as other. The USDA Wetland Classification was further used to designate plants as upland (UPL), facultative upland (FACU), facultative (FAC), facultative wetland (FACW), and obligate wetland (OBL; NRCS, n.d.). Plants were additionally grouped as native-spontaneous, non-native-spontaneous, and native planted plants. Spontaneous native and non-native plants were plant propagules that arrived at the site by wind or animal dispersal prior to dredging during the 2018 evaluation and following post-dredging monitoring in 2023 and became established without human intervention. Native planted plants were plants that were present at the site prior to restoration or not present but were planted and present in nearby upland to riparian habitat.

Statistical analysis

Due to the small study area and lack of replication, basic descriptive statistics (means and percent change) were calculated for all variables and graphed. Habitat type was divided into upland-dry, upland-mesic, transitional, riparian, and aquatic based on moisture gradient from xeric to hydric habitat. Species richness, Shannon's Equitability Index, and Shannon's Diversity index were calculated for each habitat type and the total restored area (Zar, 1999). Regression analysis was used to analyze trends in vegetation cover from upland-dry to aquatic habitat for each diversity index. Species cover for each habitat type was compared for plant type (forbs/herbs, graminoids,

Table 1. Number of terrestrial, transition zone, riparian, and aquatic plants planted, along with dates and number planted from September 2018 to November 2019 at C.L. Browning Ranch.

Species and site planted	Common name	Date and number planted				
		9/15/2018	11/4/2018	9/20/2019	11/9/2019	Total
Terrestrial/transition zone:						
Bothriochloa laguroides (DC.) Herter	Silver beardgrass			30	50	80
Chasmanthium latifolium (Michx.) Yates	Inland sea oats			16		16
Eriochloa sericea (Scheele) Munro ex Vasey	Texas cupgrass			30		30
Panicum virgatum L.	Switchgrass			70	80	150
Tridens albescens (Vasey) Wooton & Standl.	White tridens				8	8
Tripsacum dactyloides (L.) L.	Eastern gammagrass			55	80	135
					Total	*419*
Transition/riparian zone:						
Carex emoryi Dewey	Emory's sedge	25	16	50	35	126
Cephalanthus occidentalis L.	Common buttonbush			10	22	32
Eleocharis rostellata (Torr.) Torr.	Beaked spikerush	10	12	30	25	77
Juncus texanus (Engelm.) Coville	Texas rush	25		30		55
Justicia americana (L.) Vahl	American water-willow		15	47		62
Panicum virgatum L.	Switchgrass	5				5
Paspalum distichum L.	Knotgrass	3	12			15
Tripsacum dactyloides (L.) L.	Eastern gammagrass	8				8
					Total	*380*
Riparian/aquatic zone:						
Bacopa monnieri (L.) Pennell	Herb of grace	20	30			50
Equisetum hyemale L.	Scouringrush horsetail			5		5
Heteranthera dubia (Jacq.) MacMill.	Grassleaf mudplantain	20		200		220
Hydrocotyle umbellata L.	Manyflower marshpennywort	10	15			25
Ludwigia repens J.R. Forst.	Creeping primrose-willow	50		250		300
Marsilea macropoda Engelm. ex A. Braun	Bigfoot waterclover				10	10
Pontederia cordata L.	Pickerelweed			2		2
Potamogeton illinoensis Morong	Illinois pondweed	50				50
Sagittaria platyphylla (Engelm.) J.G. Sm.	Delta arrowhead	5	12	5		22
					Total	*684*

Table 2. Plant coverage (%) by species and common name, and richness, Shannon's Diversity and Equitability indices pre-treatment (June 11, 2018) and post-treatment (June 15, 2023) combining aquatic, riparian, transitional, and terrestrial areas.

Species	Common name	Cover (%)		
		Pre-treatment 6/11/2018	Post-treatment 6/15/2023	Percent change
Adiantum capillus-veneris L.	Common maidenhair	1.0	0.6	–40
Allium canadense L.	Meadow garlic	23.0	1.8	–92
Andropogon glomeratus (Walter) Britton, Sterns & Poggenb.	Bushy bluestem	1.4	2.3	64
Aristida oligantha Michx.	Prairie threeawn	2.2	0.3	–92
Asclepias asperula (Decne.) Woodson	Spider milkweed	0.0	1.4	100
Baccharis neglecta Britton	Rooseveltweed	3.8	3.9	3
Bacopa monnieri (L.) Pennell +	Herb of grace	0.0	0.2	100
Bothriochloa ischaemum (L.) Keng *	Yellow bluestem	0.0	16.6	100
Bothriochloa laguroides (DC.) Herter +	Silver beardgrass	0.0	13.1	100
Bouteloua curtipendula (Michx.) Torr. +	Sideoats grama	0.0	4.8	100
Calyptocarpus vialis Less. *	Straggler daisy	0.0	0.9	100
Carex emoryi Dewey +	Emory's sedge	0.0	58.5	100
Carex planostachys Kunze	Cedar sedge	0.5	0.9	80
Cephalanthus occidentalis L. +	Common buttonbush	1.0	0.0	–100
Chara spp.	Muckgrass (algae)	89.0	98.0	10.1
Chasmanthium latifolium (Michx.) Yates +	Inland sea oats	0.0	0.7	100
Conoclinium coelestinum (L.) DC.	Blue mistflower	2.4	3.7	54
Croton monanthogynus Michx.	Prairie tea	0.0	2.3	100
Cynodon dactylon (L.) Pers. *	Bermudagrass	5.1	0.5	–90.2
Dichondra carolinensis Michx.	Carolina ponyfoot	0.0	1.5	100
Eleocharis rostellata (Torr.) Torr. +	Beaked spikerush	0.0	9.4	100
Eriochloa sericea (Scheele) Munro ex Vasey +	Texas cupgrass	0.0	3.8	100
Euphorbia marginata Pursh	Snow on the mountain	0.0	0.5	100
Gaillardia pulchella Foug.	Indian blanket	0.0	1.3	100
Justicia americana (L.) Vahl +	American water-willow	0.0	1.5	100
Ludwigia repens J.R. Forst. +	Creeping primrose-willow	7.5	0.7	–90.7
Medicago lupulina L. *	Black medick	0.0	0.7	100
Monarda citriodora Cerv. ex Lag.	Lemon beebalm	0.0	0.2	100
Panicum repens L. *	Torpedograss	68.8	0.0	–100
Panicum virgatum L. +	Switchgrass	0.0	37.5	100
Paspalum dilatatum Poir. *	Dallisgrass	0.0	2.2	100
Paspalum urvillei Steud. *	Vasey's grass	10.3	0.0	–100
Phyla nodiflora (L.) Greene	Turkey tangle frogfruit	0.0	2.1	100
Platanus occidentalis L.	American sycamore	4.5	1.5	–66.7
Potamogeton illinoensis Morong +	Illinois pondweed	5.5	2.8	–49.1
Quercus fusiformis Small	Texas live oak	0	2.5	100
Ratibida columnifera (Nutt.) Wooton & Standl.	Upright prairie coneflower	0.0	3.4	100

Species	Common name	Cover (%)		
		Pre-treatment 6/11/2018	Post-treatment 6/15/2023	Percent change
Rhus lanceolata (A. Gray) Britton	Prairie sumac	1.0	1.6	60
Smilax bona-nox L.	Saw greenbrier	2.1	2.8	33.3
Stellaria media (L.) Vill. *	Common chickweed	0.0	0.6	100
Tragia betonicifolia Nutt.	Betonyleaf noseburn	0.0	2.8	100
Tripsacum dactyloides (L.) L. +	Eastern gamagrass	11.6	27.3	57.5
Utricularia gibba L.	Humped bladderwort	3.1	0.0	–100
Verbena halei Small	Texas vervain	0.0	1.6	100
Verbesina virginica L.	White crownbeard	2.1	0.8	–62
Diversity indices	**Richness**	20	41	105
	Shannon's Equitability Index	1.42	1.74	22.5
	Shannon's Index Diversity	1.97	2.42	22.8

* Indicates non-native species
+ Indicates planted species

and algae), life cycle (annual, perennial, or other), wetland classification, and for planted vs. spontaneously occurring vegetation for each habitat type.

Linear and non-linear regression analysis was used to compare total cover, species richness, equitability, and diversity among upland-dry, upland-mesic, transitional, riparian, and aquatic habitat. A 2 x 3 contingency table was used to test for significant differences ($P < 0.05$) at pre- and post-treatment for native and non-native spontaneous plant and planted plant coverage in all habitat zones except aquatic. Because there was no plant species overlap between aquatic habitat and other zones, aquatic plants were analyzed separately. They were analyzed with a 2 x 2 contingency table to test for differences ($P < 0.05$) at pre- and post-treatment for native spontaneous plant and planted plant coverage. If significant differences were found, the contingency table was subdivided to test for differences ($P < 0.05$) among native and non-native spontaneous plant and planted plant coverage at pre- and post-treatment (Zar, 1999).

Principal component analysis (PCA; McCune & Grace, 2002) was used to explore which native species may be most suitable for planting and which non-native species may be most susceptible to invading zones within the study site from upland-dry, upland-mesic, transitional, riparian, and aquatic habitat. PCA was used to analyze species coverage by habitat type, status (native or non-native), plant type, life cycle, wetland classification, and planted or spontaneously occurring plants. The 21 plants with highest coverage in each habitat were used in the PCA analysis and plant coverage was Log_{10} transformed to improve normality, linearity, and equality of variance (McCune & Grace, 2002).

RESULTS

Dredging of torpedograss was successful within the restoration site at 5 years post-treatment with no torpedograss documented in July 2023. The total species richness, Shannon's Equitability Index, and Shannon's Diversity increased by 105.0% (20 to 41), 22.5% (1.42 to 1.74), and 22.8% (1.97 to 2.42), respectively, from 2018 to 2023 for all plants (planted and spontaneous) documented in the restoration area post-treatment (Table 2). Total plant cover increased from 0% on disturbed soil in 2018 to coverage that ranged from 68% (upland habitat) to 124% (transitional habitat) coverage in 2023 (Figures 2a, b, and c). Regression analysis indicated a peak in plant coverage within transitional and riparian habitats (Figure 3a). Species richness was highest in the zone from upland-dry to transitional but declined exponentially from the riparian to aquatic habitat (Figure 3b). Both Shannon's Equitability Index and Shannon's Diversity Index exhibited a linear decline from upland-dry to aquatic habitat (Figures 3c and d).

Comparison of total plant coverage from upland-dry to riparian habitat pre- and post-treatment indicated that the coverage of native plants planted was significantly greater ($x = 150.4$, $df = 2$, $P < 0.001$) than coverage of non-native and native spontaneously occurring plants (Table 3). Planted native plants increased by 1192% compared to spontaneous native plants present at pre-treatment. Spontaneous non-native plants were significantly greater ($P < 0.05$) at pre-treatment compared to post-treatment but decreased by 73% between 2018 and 2023. Spontaneous native plants coverage was not significantly different ($P < 0.05$) from pre- and post-treatment, with a 16%

Table 3. Plant coverage at pre-treatment (2018) and post-treatment (2023) by non-native spontaneous, native spontaneous, and native planted plants based on a 3 x 2 contingency table (x = 150.4, df = 2, P < 0.001). Aquatic plants were analyzed separately and not included in this analysis.

	Plant coverage (%) *		
	Non-native spontaneous	**Native spontaneous**	**Native planted**
Pre-treatment	84 [b]	44 [c]	12 [d,+]
Post-treatment	22 [cd]	37 [c]	155 [a]

* Different letters among columns and rows indicates significant differences (P < 0.05) based on subdividing the contingency table (Zar, 1999).
+ Indicates native plants present at the restoration site pre-treatment that were also planted. Spontaneous plants are plants that were not planted.

(2a)



(2b)

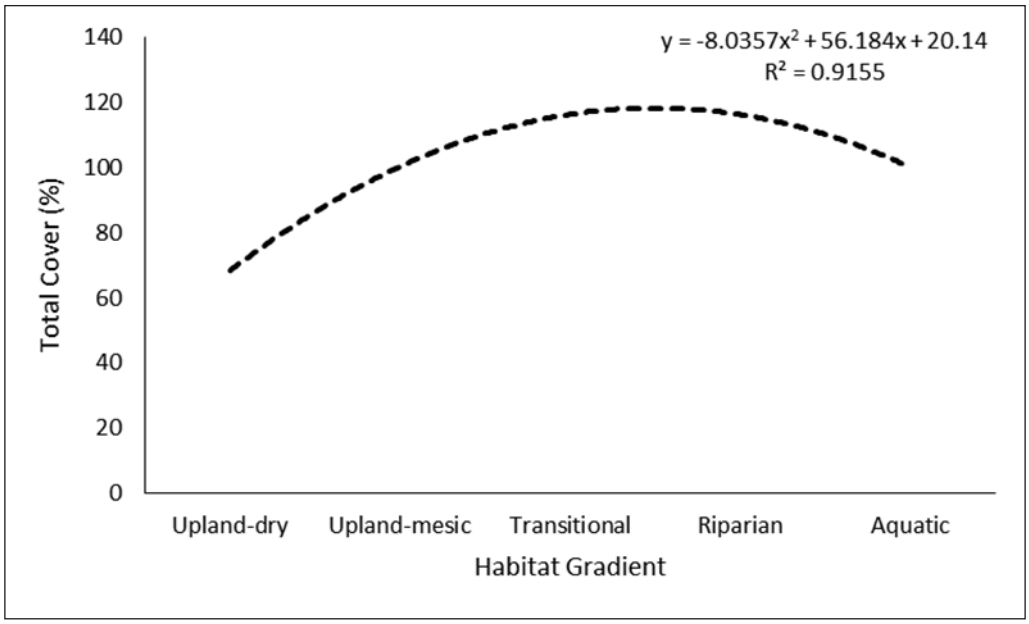


(2c)

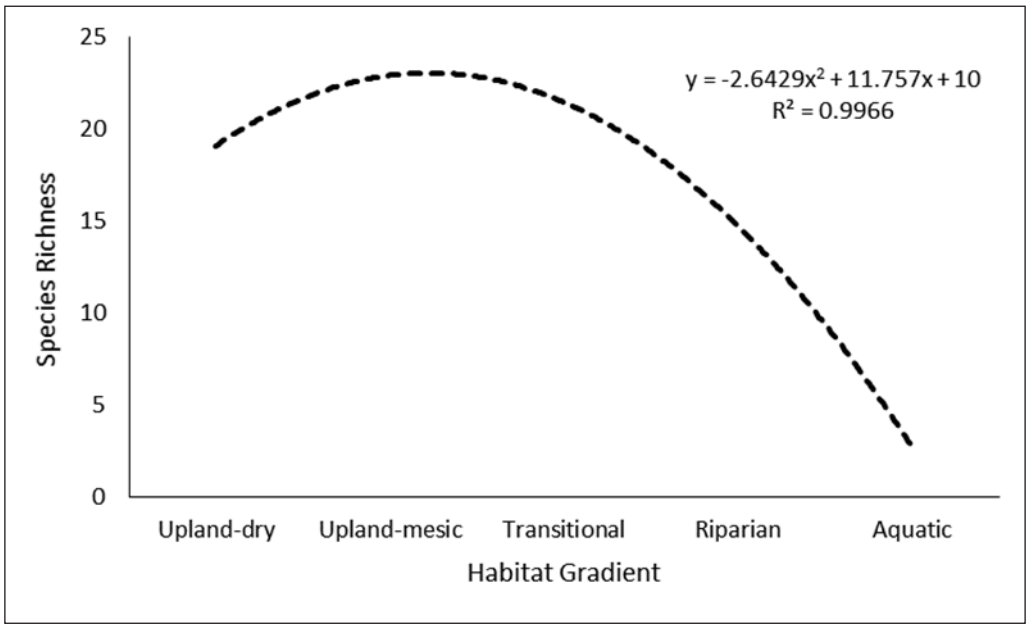


Figure 2. Photos of restoration site above the run-of-river dam at C.L. Browning Ranch in Johnson City, Texas, at: a) pre-dredging (June, 2018) with *Panicum repens* L. invasion and encroaching *Platanus occidentalis* L. trees on left side of channel; b) post-dredging (July, 2018); and c) 5 years post-planting with *Panicum virgatum* L. and *Carex emoryi* Dewey cover on the left side of channel (June, 2023).

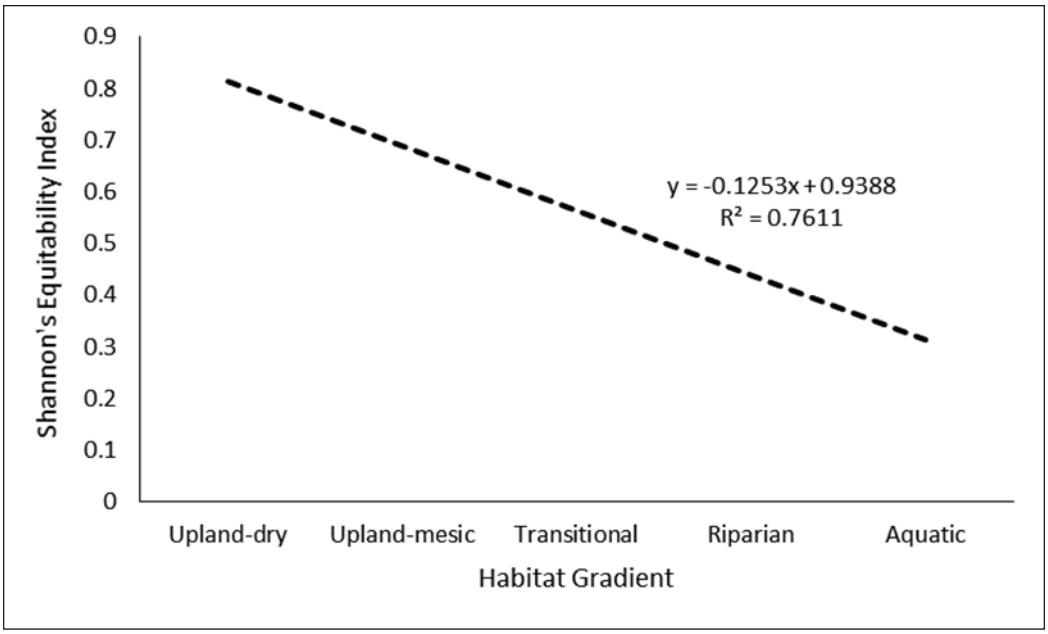
(3a)


y = -8.0357x² + 56.184x + 20.14
R² = 0.9155
Total Cover (%)
140
120
100
80
60
40
20
0
Upland-dry
Upland-mesic
Transitional
Riparian
Aquatic
Habitat Gradient

(3b)


y = -2.6429x² + 11.757x + 10
R² = 0.9966
Species Richness
25
20
15
10
5
0
Upland-dry
Upland-mesic
Transitional
Riparian
Aquatic
Habitat Gradient

(3c)



(3d)

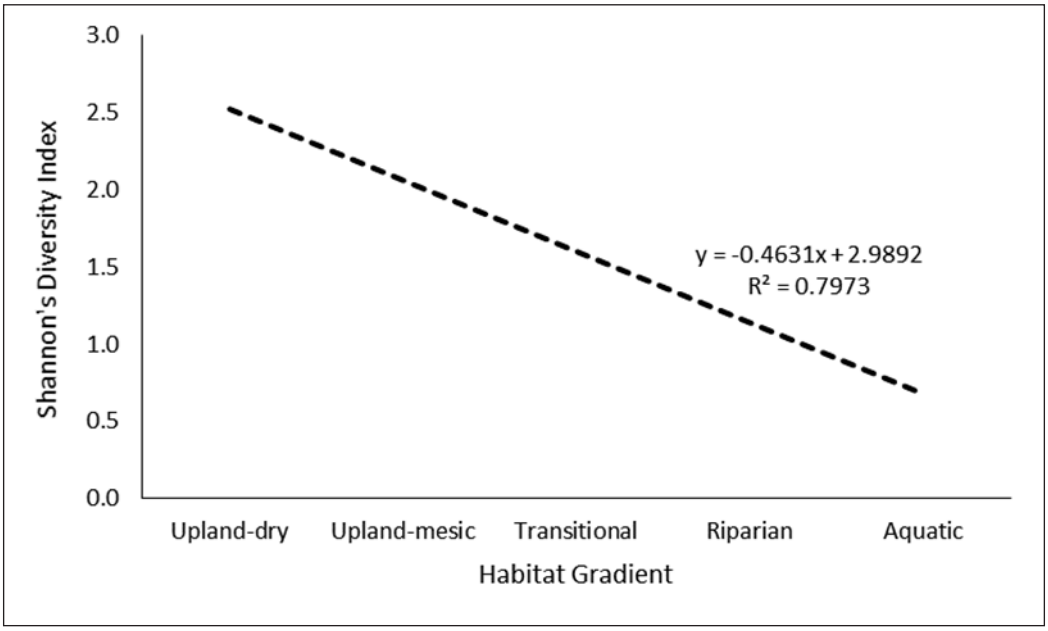
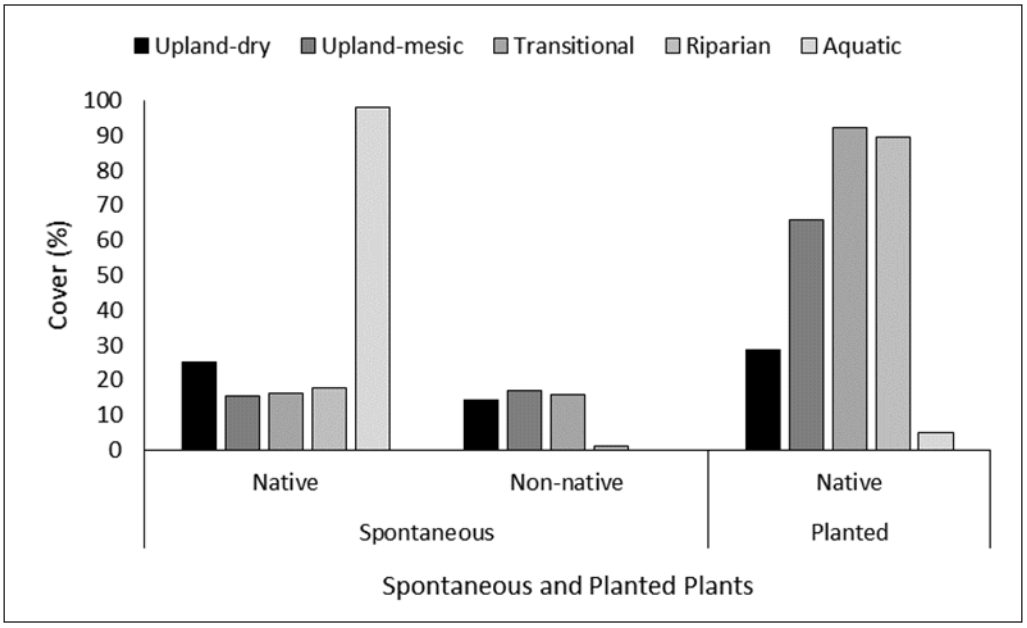
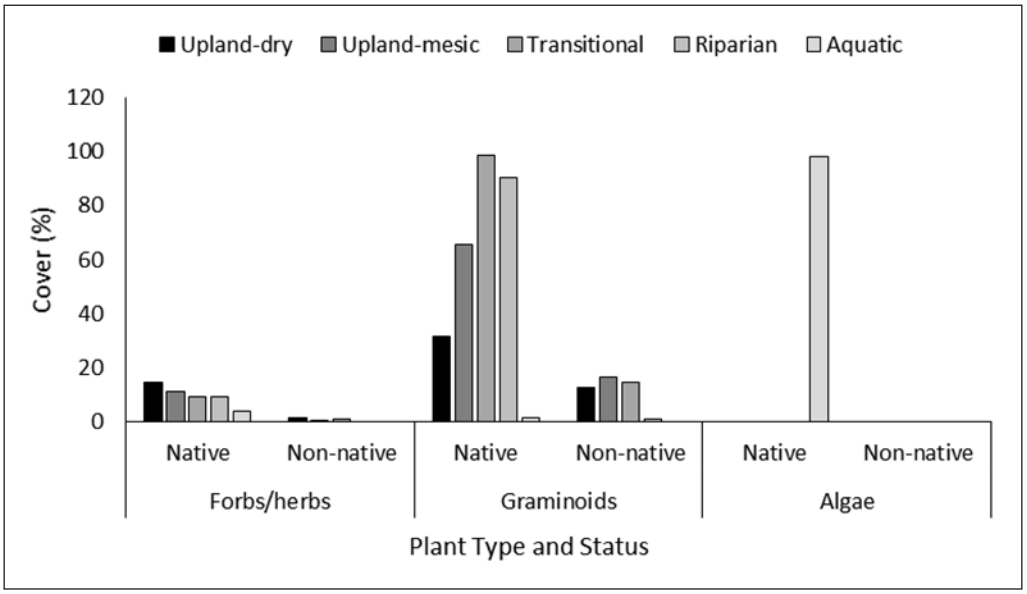


Figure 3. General trends based on linear regression for changes in a) percent total cover, b) species richness, c) Shannon's Equitability Index, and d) Shannon's Diversity Index over a habitat gradient based on linear regression from upland to aquatic habitat at 5 years post-treatment following replanting on bare soil.

(4a)



(4b)



reduction between 2018 and 2023. In aquatic habitat, *Chara* spp., a spontaneously occurring algae, was significantly greater (x = 4.1, df = 1, P < 0.043) in cover in both 2018 and 2023 compared to planted aquatic plants.

Native plant establishment from plantings was most successful in the zones from upland-mesic and riparian habitat and lowest in upland-dry and aquatic habitat (Figure 4a). Spontaneous native plants became equally established in the zones from upland-dry to riparian habitat (16–25% cover). Spontaneously occurring *Chara* spp. accounted for > 98% cover in aquatic habitat in 2023. Spontaneous non-native plants became equally established in the zones from upland-dry to transitional but accounted for only 0–2% cover in riparian and aquatic habitat, respectively.

Native graminoids were the dominant plant type from upland-dry to riparian habitat, while *Chara* spp. dominated coverage in aquatic habitat at 5 years post-planting (Figure 4b). Non-native graminoids accounted for < 18% cover in any habitat type. Spontaneously occurring native forbs and herbs accounted for < 15% in any habitat and exhibited a declining trend from upland-dry to aquatic habitat. Non-native forbs and herbs accounted for < 1% in any habitat, and none were documented in riparian and aquatic habitat.

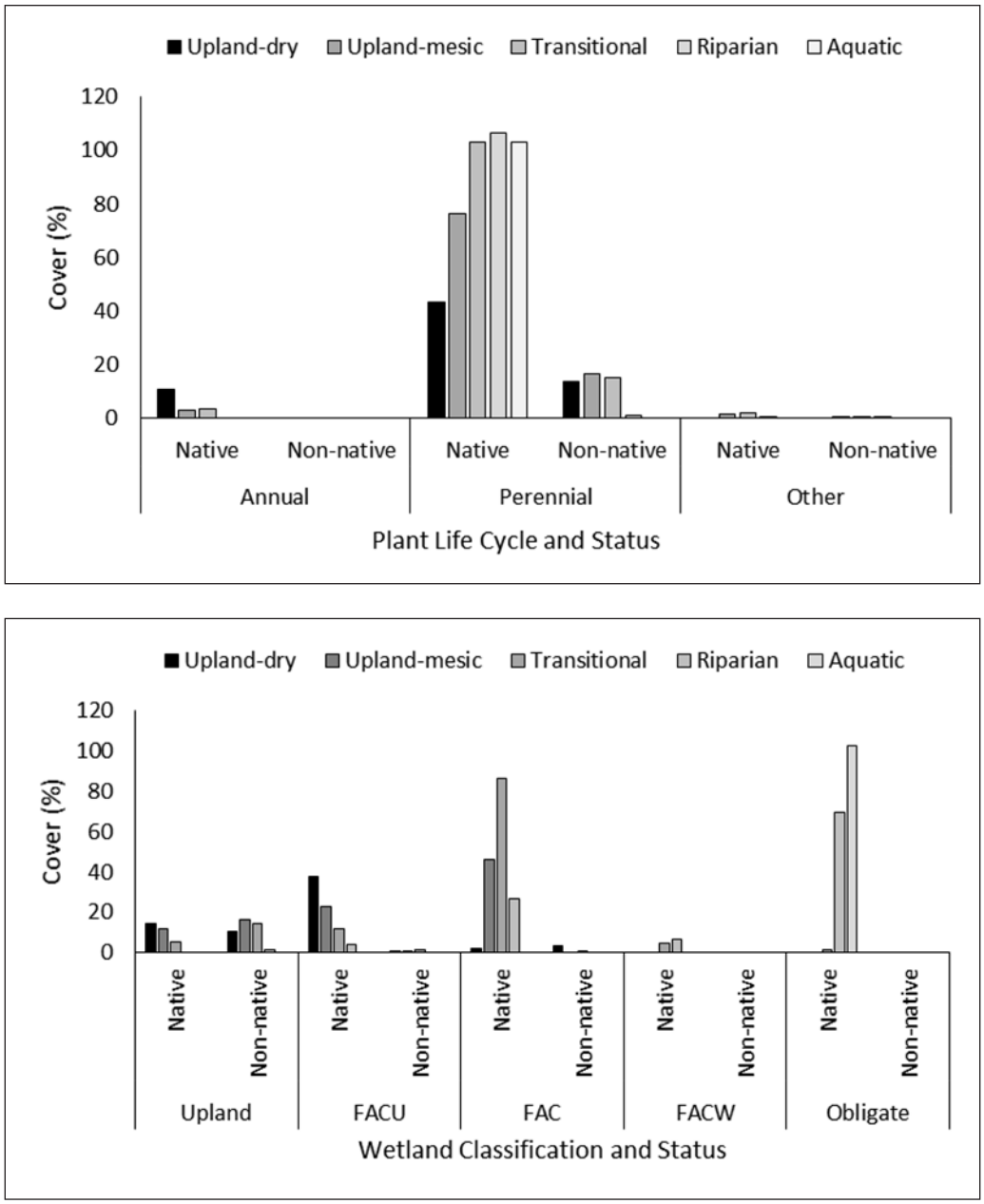


Figure 4. Percent coverage by a) planted and spontaneous vegetation, b) forbs, graminoids, and algae, c) plant life cycle, and d) wetland classification at 5 years post-planting.

Perennial plants were the dominant plant type at 5 years post-planting, with increasing tends from upland-dry to aquatic habitat (Figure 4c). Non-native perennial plant coverage was estimated to be 15–19% in upland-dry and transitional habitat but declined in riparian and aquatic habitat. Coverage of spontaneous native annual plants was highest in upland-dry habitat (10%) and less than 3% in other habitats, with no non-native annuals documented in any habitat.

USDA classification of plants by wetland status indicated native species were the dominant coverage by FACU, FAC, and OBL plants (Figure 4d). The highest native coverage of OBL wetland plants was *Chara* spp. with limited planted OBL plants surviving at 5 years post-planting. Non-native plant coverage was highest in upland habitat and dominated by the invasive grass, yellow bluestem. FACW plants accounted for < 5% coverage and were only found in transitional and riparian habitat in 2023.

Based on PCA analysis, axes 1 (P = 0.029) and 2 (P = 0.019) were both significant with Eigenvalues of 35.3 and 19.1, respectively. The percent variance explained was 42.4% and 30.8% on axes 1 and 2, respectively. Exploratory analysis with PCA indicated a negative moderate to strong correlation (–0.79)

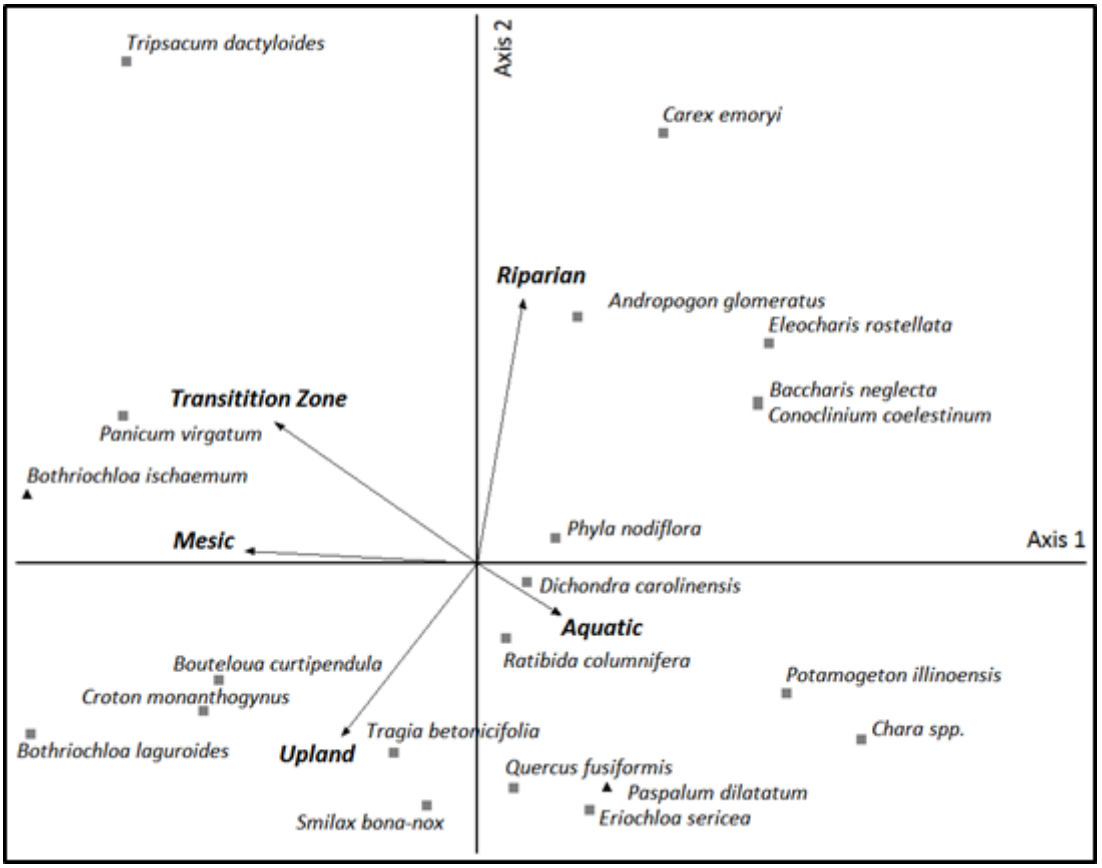


Figure 5. Principal component analysis of plant relationship along habitat gradients on axes 1 and 2.

with upland-mesic habitat on axis 1 and an association with the invasive yellow bluestem (Figure 5). For transition habitat, a moderate negative correlation (–0.589) existed on axis 1 with switchgrass. Aquatic habitat exhibited a positive moderate correlation (0.431) relationship with axis 1 and an association with the spontaneously occurring *Chara* spp. Riparian habitat exhibited a positive moderate to strong correlation (0.697) with axis 2 and an association with Emory's sedge and bushy bluestem. For upland-dry habitat, a moderate negative correlation (–0.478) existed on axis 2 with *Bouteloua curtipendula* (Michx.) Torr. (sideoats grama) and ruderal species that included *Smilax bona-nox* L. (saw greenbrier), *Tragia betonicifolia* Nutt. (betonyleaf noseburn), and *Croton monanthogynus* Michx. (prairie tea).

DISCUSSION

The results of this study indicate that dredging of torpedograss in a silted area above a dam followed by replanting native plants was successful in the removal of torpedograss and possibly other invasive species from riparian and aquatic areas, as no torpedograss was documented 5 years post-treatment. The dredging of the silted area to depths of ≤ 1 m apparently removed all torpedograss biomass and rhizomes and prevented resprouting, seed germination, and establishment from stem fragments. A single torpedograss stem fragment with one to three nodes can result in roots sprouting at rates of 26%, 50%, and 62%, allowing for the possible reestablishment of torpedograss (Hutchinson, 2021).

The benefits of upland and riparian vegetation in headwater springs and the upper reaches of streams include stabilization of the stream bank, reduced erosion, temperature regulation, and plant uptake of nutrients, all of which improved water quality (Menninger & Palmer, 2007; McMillan et al., 2014). Riparian and terrestrial vegetation is important in the longitudinal connectivity of headwater streams providing the transfer of allochthonous energy, sediment, and organisms downstream (Ferreira et al., 2023).

To our knowledge, no studies have evaluated riparian restoration in the upper reach of an undisturbed spring run or riparian restoration following removal of silt above a dam. Riparian restoration in two creeks in Pennsylvania was successful and resulted in decreased suspended sediment in the streams by 47–87% (Carline & Walsh, 2007). Yang et al. (2021) found that in riparian sites dominated by grasses, infiltration and percolation into groundwater were improved with grasses with

greater root density. Switchgrass roots can extend to 3.3 m below the soil surface (Ma et al., 2000), and eastern gamagrass roots can grow through clay (Gilker et al., 2002), resulting in reduced runoff from the restoration area. Collins et al. (2012) reported that riparian buffers < 10 m were successful in increasing dissolved oxygen levels and reducing turbidity but not specific conductivity in mixed-use farmland habitat. The vegetative buffer from the upland-dry section to the riparian zone in this study was 7–8 m with > 60% cover in all habitat.

While we did not survey macroinvertebrates or fish in this study, supporting literature indicates that restoration results in improved habitat for these taxa. Chen et al. (2022) found that riparian restoration in rural rivers improved water quality and increased the diversity of aquatic macroinvertebrates. Restoration of a 21 m buried spring run in Barton Springs located in Austin, Texas resulted in higher densities of the endemic salamander *(Eurycea sosorum)* within 1.5 years post-restoration (Bendik et al., 2025). In an urban watershed, Selvakumar et al. (2010) found that stream bank and channel restoration resulted in reduced stream bank erosion but did not improve water quality or macroinvertebrate habitat. Based on canonical correspondence analysis in urban freshwater ecosystems in Louisiana, odonates exhibited a negative relationship in abundance in riparian areas dominated by torpedograss (Thomas, 2019). A shoreline restoration project in Lake Kissimmee, Florida—in which organic sediment was dredged—improved fish diversity and water quality in the restored sites (Tugend & Allen, 2004). In another lake in Florida, 165,000 m^3 of sediment was removed, resulting in greater densities of largemouth bass, bluegill species, and forage fish compared to sites that were not restored (Moyer et al., 1995).

The upland to transitional zones were the most susceptible to spontaneous non-native plant invasions in this study. Spontaneous non-native vegetation was expected to re-invade upland-dry, upland-mesic, and transitional, and riparian habitat from adjacent habitat due to the high soil disturbance caused by dredging and heavy machinery. At 5 years post-planting, yellow bluestem, along with Bermudagrass, dallisgrass, and straggler daisy, were the most common non-native species present in coverage in upland-dry, upland-mesic, and transitional habitat. Yellow bluestem, an invasive grass, accounted for 10–14% cover in the upland-dry, upland-mesic, and transitional zone, indicating the potential of this non-native species to outcompete native species over time. Yellow bluestem has proved difficult to control in Texas (Wied et al., 2020) and has been documented in most counties in south and south-central Texas (Shaw, 2012).

Succession along the restored riparian area may shift from a graminoid and herbaceous community to shrubs and trees over time (Dodds et al., 2023). The restored site in the upper reach of the spring run represents a novel habitat from the removal of silt accumulation at the dam. Above and below the restoration site, the riparian and immediate upland is comprised of ground cover of 30–50% and shrubs and trees that provide approximately 40–80% canopy cover. It is expected that shrubs and trees will invade the restored site over time from propagules during periods of high flow. Over time, restoration efforts may become dominated by non-native plants reverting to pre-restoration conditions (Reinecke et al., 2008; Bauer et al., 2018; Duxbury et al., 2025). Yellow bluestem invasion in the upland section will require further intervention and the site needs to be monitored for other invasive plants.

Restoration of the riparian and littoral zones in this study could have been improved with the construction of a gradual bank slope, which could provide more area for native plant recruitment instead of an abrupt slope (Hasselquist et al., 2015). The abrupt drop-off of approximately 0.6–0.8 m between the riparian and aquatic zones resulted in the elimination of a gradual shallow littoral zone along the eastern edge. This abrupt change in slope may have prevented the establishment of transitional amphibious plants between riparian and aquatic habitat and resulted in reduced solar radiation reaching aquatic plants. The planting of 32 buttonbush seedlings (25–30 cm in height) at the edge of the riparian and aquatic zones resulted in no survival at 5 years post-treatment. This was unexpected because buttonbush is common along the spring run. It is probable that the rapid graminoid growth resulted in the buttonbush being shaded out, or the abrupt drop-off between riparian and aquatic zones resulted in unsuitable habitat.

Prior to restoration, the silted area above the dam resulted in torpedograss forming a monoculture on the east side of the pool that was also colonized by native shrubs and trees. The dense monoculture of torpedograss likely contributed to accelerated siltation and resulted in a decrease of 44% aquatic habitat for fish and aquatic macroinvertebrates. Restoration resulted in a 127% increase in aquatic habitat and a 223% volume increase of aquatic habitat dominated primarily by *Chara* spp. *Chara* spp. in general are good indicator of water quality (Barker et al., 2008) and are one of the first species to colonize disturbed habitat (Wade, 1990). As primary producers, *Chara* spp. provide food, habitat, and refuge for plankton, invertebrates, fish, and birds (Rey-Boissezon & Joye, 2015). As a group of green algae, *Chara* spp. are primarily associated with clear alkaline oligotrophic to slightly mesotrophic conditions in shallow water (Janse et al., 2008). In the dredged area of the spring run, the rapid expansion of *Chara* spp. on bare sediment was likely due to fragments and oospores from upstream (Casanova & Brock, 1996).

In urban streams, aquatic habitats stabilize in about 10 years in an urban environment (Gillmann et al., 2023), but it is unknown how aquatic plant succession will proceed in this study. *Potamogeton illinoensis* and *Ludwigia repens* coverage was

reduced post-treatment, but small colonies are present in the pool above the dam and may increase their coverage over time.

Knowledge of plant species in relation to their water requirements is a major aspect of restoration efforts (Hammersmark et al., 2009). Based on the native species planted, *Carex emoryi* Dewey, *Panicum virgatum* L., *Tripsacum dactyloides* (L.) L., *Bothriochloa laguroides* (DC.) Herter, and *Eleocharis rostellata* (Torr.) Torr. resulted in the most successful establishment, with coverage increases of 58.5%, 37.5%, 27.3%, 13.1%, and 9.4%, respectively, at 5 years post-planting. *Tripsacum dactyloides* was the only plant present at the study site prior to restoration. By planting native upland and riparian plants that were not present prior to dredging and restoration, non-native plants in the upland-mesic and riparian zones can be suppressed.

The collaborative efforts among C.L. Browning Ranch, UTSA, and multiple volunteers resulted in a successful small-scale restoration project. While private landowners can provide the equipment for small restoration efforts, and universities can provide expertise, space to propagate and produce plants, and other resources, the most important part of this project was performed by volunteers. No accrued cost was estimated for this project, but we estimate that volunteers accounted for > 80% of the total cost via their time, effort, and travel spent collecting seeds and propagules, propagating and maintaining plants, planting, and monitoring over the 5-year study. At 5 years post-dredging, the project has been successful, but more work is needed to control spontaneous non-native plants, such as yellow bluestem, from invading upland to transitional habitat. Additional planting of upland and upland-mesic native plants may result in suppression and control of yellow bluestem. With limited funds, the results of this study are most applicable to restoration projects in areas < 0.2 ha, because the effort and cost would increase greatly for larger areas.

CONCLUSIONS

Dredging torpedograss to 1 m below sediment depth was successful on a small scale for removing aboveground and belowground biomass at 5 years post-treatment. The restoration project increased aquatic habitat, with Chara spp. establishing spontaneously. Successful establishment of native plants in zones between upland-dry (xeric) and aquatic habitats requires selecting plants adapted to different soil moisture levels. Greater establishment of terrestrial and riparian plants was documented from native plants propagated from seed in the upland to riparian zone. Planting native aquatic plants from apical tips resulted in < 5% establishment, indicating that established plants, rather than plant fragments, may be necessary for successful establishment. Water quality can be improved locally and along the longitudinal gradient from the upper reaches of spring runs by maintaining vegetation buffers in upland and riparian zones. Vegetation buffers trap sediment, absorb nutrients, stabilize the stream bank, reduce erosion, and regulate temperature.

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