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Plant Community Richness and Composition in Formerly Dammed Reservoirs of Central Massachusetts

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ABSTRACT

Small dam removals have become common in the United States as many have begun to lose their function and efficiency, often with the goal of improving safety and/or restoring riverine ecosystems. In 2017, American Rivers Association, Massachusetts Fish & Wildlife and, the Sturbridge Conservation Commission removed three small dams along Hamant Brook in Sturbridge, Massachusetts. The removals provided a compelling opportunity to examine how areas nearly devoid of life—the three newly exposed former impoundments—revegetate. The aim of this research was to understand how plant communities develop in these newly exposed areas and have been influenced, in composition and richness, by proximity to the abutting stream or forest edge. A total of 29 taxa were recorded at the sites; 19 of the recorded species were classified as native and six nonnative. There was no relationship between native and nonnative species richness and relation to forest or stream edge but, the results of this study provide American Rivers Association and other stakeholders with new information regarding plant regeneration and community composition following the small dam removals along Hamant Brook. The implications of this may prompt reevaluation of existing conservation plans to meet broader conservation goals.

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INTRODUCTION & BACKGROUND

Small dams, which bolstered the economy for a large part of the 19th century, populate streams across the Midwest and northeastern United States (Magilligan et al., 2017 as cited by Orr et al., 2004; American Rivers, 2014). The deteriorating function of these aging structures not only pose safety hazards that require expensive upgrades, but degrade habitats for riverine species (Born et al., 1998). Removal of dams is often less expensive than retrofitting to meet new dam safety standards, (which have been developed by many states in response to fatal dam failures) (Silva et al., 2017) but can also help to restore ecosystems closer to conditions before the installation of the dams (Bellmore et al., 2017; Ding, Chen, Ding & Tao, 2019 as cited by Mullens and Wanstreet, 2010; O'Connor et al., 2015; Bohrerova et al., 2017).

The dams along Hamant Brook in south-central Massachusetts are among the 1,400 that have been removed in the United States since the 1970s (Doyle et al., 2005). Hamant Brook is a small stream (three quarters of a mile in length) in the Quinnebaug watershed which contained three small (defined as those under 15 feet in height), 100 year old dams, until 2017, (*Hamant Brook restoration project: Dam removal design plans*, 2015; Silva et al., 2017).

Hamant Brook is contained within a natural area called the Sturbridge Trek, a space used mostly for recreation by the surrounding community. The brook area is owned by Old Sturbridge Village, an interactive colonial museum in the town of Sturbridge, Massachusetts (Semon, 2017).

After a contentious debate in the town of Sturbridge about the potential harm to habitat of beavers, geese, and other species that had established within and around the reservoirs, the Conservation Commission voted 3-2 in favor of removing the Hamant Brook dams in 2009 (Semon, 2017). In 2013, the American Rivers Association consulted with Massachusetts Fish and Wildlife and the Sturbridge Conservation Commission to create a plan to remove the three dams with the goal of restoring stream and riparian habitats while making several miles of hiking trails more accessible (*Hamant Brook restoration project: Dam removal design plans*, 2015).

The dam removals also sought to improve the water quality of the Quinebaug River (Semon, 2017). Poor water quality was due to a nearby electrical utility warming the river too much for wildlife; the removals have allowed the utility to continue to operate (Davis, 2013). In addition, the dam removals aimed to restore native trout habitat, their migration routes, and restore habitat for the endangered wood turtle (*Hamant Brook restoration project: Dam removal design plans*, 2015; Semon, 2017).

The initiation of the removal process required several steps. Restoration crews first opened the low level dam outlets in order to drain the highest volume of water possible (Semon, 2017). Once a significant amount of the water was drained, the crew was able to remove the outer pieces of the dam, then removed the middle structure before allowing the remaining water to be released (Semon, 2017). Tree saplings were then planted to stabilize the banks of the new stream channel (*Hamant Brook restoration project: Dam removal design plans*, 2015).

The stated conservation goals of the project did not include documentation and analysis of the plant communities that have established since the dam removals. American Rivers, however, sought information on the composition and richness of plant species in the stages following the impoundment draining so they may be able to improve their restoration plans in ways that are most fitting for the establishing plant communities and thereby improve the ecosystem to meet their broader conservation goals.

LITERATURE REVIEW

Reasons for Removal of Small Dams in the United States

The density of small dams is highest in New England, where water provided power to mills for the greater part of the 19th century (Magilligan, Sneddon, & Fox, 2017 as cited by Hunter 1979; Steinberg, 1991). Though some large dams have been removed due to their decreasing function and negative ecological impacts, e.g. the Elwah River in Washington state, most damn removals have been small ones (Magilligan et al., 2017 as cited by Orr et al., 2004; American River, 2014). There are approximately 18,000 dams less than or equal to 15 feet in height in the U.S. (Silva et al., 2017).

The removal of dams continues to be a source of concern for citizens and scientists as a number of reasons warrant their removal. A major driver of dam removal among the conservation community is restoring riverine ecosystems. Water quality over time can suffer greatly from impoundment (Born et al., 1998). Rapid sedimentation often causes shallowing, increased productivity, and accumulation of contaminants; this creates a eutrophic system, leading to harmful algal blooms and excessive aquatic vegetation (Born et al., 1998). These conditions can be deadly for fish and other aquatic species while hindering recreation for the community. Rivers can be restored close to their original state by removing dams and therefore improving the water quality, as indicated by a number of studies (Ding, Chen, Ding & Tao, 2019 as cited by Mullens and Wanstreet, 2010; O'Connor et al., 2015; Bohrerova et al., 2017).

In addition to issues concerning water quality, dams can also disrupt the migration of fish, hindering reproduction (Born et al., 1998). Stream channel fragmentation can make it difficult for fish to cross (Born et al., 1998). Restoring these habitats is often to replenish species of economic and cultural importance (Bellmore et al., 2019 as cited by Bednarek, 2001). Dam removals aim to restore ecosystems to their original state, though this varies on spatial and temporal scales (vegetation takes longer recover to after a removal than benthic macroinvertebrates, for example) (Bellmore et al., 2017). Still, an ecosystem disrupted by a dam and its removal may never reach its original state pre-dam installation (Bellmore et al., 2017).

Many of the dams in the United States have experienced structural deterioration as they approach or exceed their 50-year life expectancy (Orr and Stanley, 2006 as cited by USDA, 2000; Bednarek 2001). Because of this this, dams become a safety and liability issue for private owners and communities (Orr and Stanley, 2006). This is usually caused by the volume of sediment which has accumulated throughout the life of the dam (Born et al., 1998). According to the U.S. Army Corps of Engineers (USACE) (1998), 85% of dams in the U.S. will reach their operational life expectancy in the year 2020. There are 4,400 unsafe dams in the U.S., according to a 2016 assessment done by the Association of State Dam Safety (2017). Dam owners are held liable for dam failures, injuries to visitors or trespassers, and environmental or property damage (Schiermeier, 2018). In response to several devastating dam failures killing hundreds of people, several states have implemented legislation enforcing dam safety (Silva et al., 2017).

The high cost of repairing dams often leads to neglect, and dangerous conditions (Born et al., 1998; Silva et al., 2017). To combat this, some states have developed funding programs to repair unsafe dams (Silva et al., 2017). Even still, most states cannot afford to fully fund public dam safety projects; relatively speaking, most dams are small and privately owned, and do not qualify for those programs, so individuals who must meet safety regulations because of legislation, are left to source their own funding (Silva et al., 2017). To rehabilitate non-federal dams, deemed "high hazard" by The Association of State Dam Safety, $18.71 billion in funding is required (Silva et al., 2017). Because of this, many turn to dam removal rather than repair.

In many cases, dam stakeholders have no choice but to remove dams whether they care about ecological impacts of the dam or not. Dams can be aesthetically pleasing, which is why some argue they are worth repair (Schiermeier, 2018). Dams also provide communities with opportunities for recreation including boating, swimming and fishing. The prospect of losing those activities to restore a stream may lead to push back from the community against draining reservoirs. Others fear that removal will decrease properties values; homeowners often believe that lake front homes have higher property values than river front homes, though there is little information to support this (Schiermeier, 2018).

Though reliance on large scale hydroelectric dams is increasing as much of the world aims to reduce its reliance on fossil fuels, small, aging dam structures in the north-eastern and mid-western U.S. are, in general, no longer being used for their original purpose. They can be harmful to ecosystems, and it is often more of a threat to human safety by leaving these deteriorating structures in place, and costly to make necessary upgrades in order to uphold regulations established for dam safety.

The Nature of Sediment Deposition from Small Dams and their Removal

Sediment deposition is thought to be one of the most disruptive agents of change in dewatering a reservoir during the dam removal process (Foley et al., 2017). The installment of a dam almost certainly increases sedimentation rates due to the disruption of natural stream flow (Foley et al., 2017). In addition to increased sediment loading behind the dam, contaminants, nutrients, and organic material typically increase in mass (Foley et al., 2017). The possible impacts of these materials are concerning, so evaluation of sediment load before the removal process is often deemed essential (Foley et al., 2017).

These factors often are indicative of the soil composition which might be left behind after a removal and therefor greatly affect the surrounding ecosystem at its base level (Lafrenz, Bean, & Uthman, 2013). Bellmore et al., (2019) and Cui et al., (2016) research aims to understand how this will play out before the dewatering process by constructing models which can be used more widely by technicians. Even still, discrepancies in the predictability of dam removal and sediment movement make it difficult to fully understand how removal will impact stream hydrology and ecosystems (Lafrenz, et al., 2013)

Two distinct layers of sediment typically form as a result of build up behind a dam (Cui et al., 2016). The top deposit is composed of larger granules including sand to gravel sized sediment, while the bottom deposit is composed of smaller particles of silt, but can include some amounts of clay and sand (Cui et al., 2016). It is expected that a significant amount of this sediment will be displaced and redistributed downstream as a result of hydrologic changes of the stream or river (Cui et al., 2016). However, the composition of the sediment is essential in understanding how the particles will move. By nature, bottom sediment erodes much faster than the heavier top layer sands and gravels (Cui et al., 2016). This is measured by "cohesiveness" (Sawaske & Freyberg, 2012). Cohesion and grain size of particles affect the ease of erodibility (Sawaske & Freyberg, 2012). Cohesion in this context means consolidated segments of sediment which is a measure of their resistance to erosion; particles defined as "non-cohesive" have a lower degree of resistance to erosion (Sawaske & Freyberg, 2012). On average, 39% of sand and gravel volume are eroded while 11% of fine sediment deposits are eroded from the areas where the dams were (Sawaske & Freyberg, 2012). Bottom set deposits contain other organic materials which may affect erosion and can make their erodibility more difficult to predict for stakeholders in a dam removal (Cui et al., 2016). Testing in the field or lab is usually necessary to understand how bottom sediment composition may erode in a case by case basis (Cui et al., 2016). If there is relatively small volume of the bottom layer sediment, then it may not be necessary to factor into models (Cui et al., 2016). Top set sediment is more studied. Numerical simulation of coarser top layer sediment is usually feasible in transport modeling because it is a longer term process (Cui et al., 2016). A greater understanding of these factors allows for predictions to be made about how and where the sediment behind a dam will redistribute after a dam removal project.

The flow of the newly reestablished river or stream affects the way in which sediment moves (Foley et al., 2017). High flow is not required to move sediment; it may have effect on timing and trajectory of sediment deposits, but typically does not affect the eventual volume of sediment which will be deposited following removal (Foley et al., 2017). Stream geology (affecting hydrology) indirectly affects the way sediment is deposited downstream (Cui et al., 2016). Deposition also may promote "lateral channel migration" which occurs as sediment deposits continuously build up on stream beds until the channel gradient becomes similar to pre-dam conditions (Cui et al., 2016). In other words, lateral channel migration means the eventual reestablishment of the stream flow path (Cui et al., 2016).

Ultimately, the redistribution of sediment downstream will impact short and long-term water quality, distribution of nutrients in and outside of the water and ecosystems in and out of the water (Foley et al., 2017). Potential toxins in sediment loads should be handled with extreme care as to not disrupt an ecosystem more so than initial shock of a dam removal (Tomsic, Granata, Murphy, & Livchak, 2007). More research is needed to understand movement of sediment and how toxic buildup behind dams affects aquatic and terrestrial ecosystems.

Small Dams and Their Removals Impact Future Soils

The removal of a dam provides an interesting area for studying succession; with a newly exposed impoundment, left behind is an expansive mudflat free of vegetation. Sediment and organic matter left behind from a former stream impoundment provide a basis for soil formation (Perry et al., as cited by Steiger & Gurnell, 2003; Adair et al., 2004; Bechtold & Naiman, 2003; Cabezas & Comin 2018; Bats et al., 2015). The exposed area from the previously inundated reservoir, along with the gradual formation of a stream channel forms a new riparian zone. The dewatering process influences the availability of water and nutrients in soil, and therefore impacts water and nutrient availability for riparian vegetation (Lafrenz et al., 2013). It is important to note that some toxins may be contained within this soil and pose potential hazards to plants and animals coming in contact with the newly exposed soil; these toxins may be freed from sediment when the water is released from the dam, so they are likely to be deposited along stream banks (Tomsic et al., 2007). New soils may impact the success of plants establishing in these areas (Tomsic et al., 2007).

Pedogenesis is defined the formation of soil (Lafrenz et al., 2017). The impacts dams have on riparian soils is not well understood (Perry, Shafroth, & Perakis, 2017). Lafrenz et al. (2017) identifies three ways in which soil "ripens" subsequently after exposure. "Ripening" refers to "the initial soil formation processes that render a soft alluvial deposit or peat suitable for agricultural use" (J. Pons & Van Der Molen, 1973). The three types of development are physical, chemical and biological (Lafrenz et al., 2013).

Physical ripening is calculated from soil moisture, organic matter content and texture (Lafrenz et al., 2013). When the impoundment is first drained, the dewatered material begins to dehydrate; it shrinks and cracks, which increases the permeability of the soil (Lafrenz et al., 2013). Soil goes from soft and wet to a crumbly texture—the first stage of soil formation (Lafrenz et al., 2013 as cited by Pons & van der Molen, 1973). Physical ripening is identified as the onset of pedogenesis (D. J. Kim, Feyen, Vereeken, Boels, & Bronswijk, 1993). Clay and organic matter retain water much better than sands, and a former impoundment is likely to have high amounts of organic matter, so it will therefore retain high amounts of water for an extended period of time (Lafrenz et al., 2013 as cited by Pons and Zonneveld, 1965). Though the physical ripening index was first established for agricultural use, it has been used to understand pedogenesis in other newly drained areas (Rhody, 2013). The large amount of organic matter which is left behind after being trapped behind dams before their removal takes a long time to decompose in anaerobic conditions because of its high moisture retention; eventual drying and physical ripening increases exposure to oxygen, therefore increasing the rate at which it can decompose and enrich the soil (Lafrenz et al., as cited by Josette, Leporcq, Sanchez & Phillipon, 1999). The physical ripening process must first be complete before subsequent biological and chemical ripening (Lafrenz et al., 2013).

The changes initiated by physical ripening lead to the onset of chemical ripening (Lafrenz et al., 2013). Aeration of the soil is largely from chemical ripening which is defined as increased oxidation of the soil from aeration (Lafrenz et al., 2013). The organic matter left from dams decomposes slowly in anaerobic conditions (Lafrenz et al., 2013 as cited by Josette, Groendijk & Kroes 1999). The chemical ripening process increases organic matter decomposition because of the availability of oxygen (Lafrenz, 2013). In sediments that are low in calcium, oxidation causes the production of acids, lowering soil pH (Lafrenz, 2013). In soil formation, the oxygen reaction initiates the availability of plant nutrients in soil (Rhodes, 2013 as cited by Vepraskas & Faulkner, 2001). Plants will be able to start to establish in the newly exposed soil once aeration accelerates.

The third and final type of soil ripening is biological. This defined as the “result of activity of all kinds of flora and fauna” (Pons & Zonneveld, 1965). Pons & Zonneveld (1965) identify homogenization as one part of the biological ripening process by soil organisms digging, eating, and excreting causing mixing of soil materials and altering soil structure; plant root growth also contributes to this process. Mixing can occur from large animals, plants and microscopic organisms before the onset of pedogenesis and physical ripening (Pons & Zonneveld, 1965). Conditions best for biological activity are high humidity, high levels of organic matter, medium soil moisture and medium textured soils (Pons & Zonneveld, 1965).

Ultimately, the fate of riparian communities is largely dependent on soil formation. Ripening occurs on varying temporal scales and impacts the ability of vegetation to successfully establish along newly forming stream channels. These plant communities provide bank stability and habitat for riparian animal species. Soil formation has the ability to impact an ecosystem at its very basic level because of the implications it can have on vegetation and therefore the greater trophic cascade.

Impacts of Small Dam Removals on Riparian Vegetation

The impact of dam removal on riparian vegetation provides a compelling area for study of colonization and succession due to the large, available expanse of open mud flat following a removal (Doyle et al., 2005). Dewatering of an impoundment offers a large area of bare soil containing sediment that contributes to soil quality and composition in months to years following the removal (Orr & Stanley, 2006). Depending on the nutrient richness of bare soils, the presence or absence of “inhibitory factors” (such as soil toxicity), re-vegetation can occur quite rapidly (Orr & Stanley, 2006 as cited by Rebele, 2001).

Doyle et al., (2005), Kim et al., (2015), and Orr & Stanley (2006) found that within a month of dam removal, nearly no bare sediment remains. In fact, it is highly unlikely, according to Doyle et al. (2005), that after a month, any bare sediment will remain. Plants which initially colonize a newly exposed soil area are “weedy” vegetation which generally grow fast, have high seed production and efficient dispersal mechanisms (Doyle et al., 2005). Doyle et al. (2005) found that at removal sites under 10 years old, areas were dominated by grasses and forbs. Orr & Stanley (2006) examined subsequent vegetative succession following 30 small dam removals in Wisconsin and only five out of the 650 quadrats studied after one month were bare—all of which were in areas where walking trails bisected the transects. In addition, a comparative study of 13 dam removals varying in age from one to 30 years old found that younger sites are typically dominated by grasses and forbs (Kim et al., 2015 as cited by Orr & Stanley, 2006). There was no difference detected in tree or shrub taxa cover in studies sampling vegetation one year after removal, indicating their establishment is typically later in succession (Stephens, 2017). Though there is variation in plant species depending on location and surrounding vegetation, general patterns arise in successional processes following dam removal (Orr & Stanley, 2006).

There are typically changes in downstream riparian zones in addition to the exposed mudflat of the former reservoir (Doyle et al., 2015 as cited by Shafroth et al., 2002). Previously inundated reservoir sediment is redistributed downstream creating new surfaces for vegetative colonization (Doyle et al., 2005 as cited by Shafroth et al., 2002). Again, these areas are likely to be first dominated by early successional vegetation because they are efficient in seed production and dispersal (Doyle et al., 2005).

Species richness was also found to be variable at a number of small dam removal sites (Orr & Stanley, 2006). Orr and Stanley (2006) found this to be true among all of their sites, particularly, at newly removed dam sites. The lowest species richness in these “young” cites was seven and the highest, 29 species, which consisted mostly of common herbaceous plants (Orr & Stanley, 2006). Over time, later successional species are expected to colonize the newly exposed sediment (Doyle et al., 2005). In these mid successional stages, it has been found that within the first ten years post removal, herbaceous plant composition was highly variable (Kim, Toda, & Tsujimoto, 2015). After a decade, riparian vegetation may gradually transition to more tree and shrub species along the channel margins; they’re expected to become much more abundant (Kim et al., 2015).

Variation exists between newly removed sites and those which have been without a dam for several decades (Kim et al., 2015). At sites thirty years or older, grasses and forbs, in one study, were still common, but the frequency of trees and shrubs was much higher (Doyle et al., 2005; Orr & Stanley, 2006). Orr and Stanley (2006) found, consistent with their hypothesis, that species richness was highest at older sites and that species diversity and frequency of trees were positively correlated with time since dam removal. Species diversity was found consistently high at the oldest removal sites in multiple studies (Doyle et al., 2005; Orr & Stanley, 2006).

Introduced species have the potential to inhibit native species colonization (Orr & Stanley, 2006). Orr & Stanley (2006) found a mean frequency of introduced species to be 75% among their 13 Wisconsin small dam removal sites. Low species diversity occurred in areas dominated by non-native grasses and forbs and was negatively correlated with native vegetation (Orr & Stanley, 2006).

Because the presence of riparian trees, shrubs and grasses is vital to stabilizing stream/river banks, their establishment over time post dam removal is essential. There are significant differences between the effects of grasses and trees and their implications on channel stability (Doyle et al., 2005 as cited by Simon & Collison, 2002). Their strong root systems aid in stabilizing stream channels, reducing long term erosion, which helps to prevent excessive sediment deposition downstream (Doyle et al., 2005). Simon & Collison (2002), aiming to quantify the mechanical effects of riparian vegetation on streambank stability, found that tree roots increased soil strength by 2-8kPa while grass roots increased stability by 6-18 kPa. With this rapid revegetation, stream channels become larger and it is assumed that over time, the simple stream channel produced by the removal may become compound, providing space for riparian vegetation and long term reforestation (Kim et al., 2015).

The establishment of riparian vegetation can have serious implications for ecosystem restoration, soil strength and channel stability (Doyle et al., 2005; Simon & Collison, 2002). Though patterns of recolonization are often site specific, a general trend in herbaceous plants, especially grasses and forbs, as the pioneer species, and then later establishment of shrubs and trees, was seen across several study sites (Doyle et al., 2005; Kim et al., 2015; Orr & Stanley, 2006; Simon & Collison, 2002).

METHODS

To document a possible relationship between plant species richness and proximity to forest edge and stream edge, the locations of the three former impoundments were identified using a map provided by American Rivers Association (Appendix 1). The former impoundment areas, which were drained in 2017, served as the three study sites for the project. The three sites, referred throughout this paper, as site one, site two and site three, are all contained within the three-quarter mile Hamant Brook, which flows north into the Quinebaug River on the western side of Sturbridge, Massachusetts.

The area within and around Hamant Brook contains Merrimac-Hinckley-Windsor soils: sandy loam with generally good drainage (Sturbridge Master Plan, 2011). The former impoundments themselves contain soils which are much wetter compared to the surrounding forest. The brook is surrounded by varying topography with elevations ranging from 520ft to 1,100ft above sea level.

Hamant Brook is part of the Quinebaug River watershed which is in the southern part of central Massachusetts and extends to Connecticut and western Rhode Island spanning 850 square miles (Sturbridge.gov). Within a half of a mile of the brook is an access road for the nearby Old Sturbridge Village Museum and an on ramp for Interstate Highway 84.

Site one, also referred to as the “lower pond dam,” is approximately 120m in diameter and is the site of reconnection between Hamant Brook and the Quinebaug River. The surrounding forest overstory is largely eastern white pine (*Pinus strobus)* and other early to mid-successional species including paper birch *(Betula papyrifera)*; the understory is sparse, including partridge berry (*Mitchella repens),* jewel weed (*Impatiens capensis)* and several species of nettle (Urticaceae). Within three meters of the forest edge (also the edge of the former impoundment) is a well established walking trail. Site two, also referred to as the "middle pond dam," is approximately 130m in width and 180m in length. The surrounding overstory also contains largely white pine and eastern red cedar (*Juniperus virginiana)* among rocky outcrops which are visible to the east of the brook as elevation increases. Understory species are sparse, containing mostly jewel weed and jump seed (*Persicaria virginiana)* within the canopy. Site three, also called the "upper pond" is about 180m in length and 110m in width. The overstory is consistent with the other two sites, but the understory is denser and contains several fern species, dominated mostly by evergreen woodfern (*Dyropteris intermedia)* and ostrich fern (*Matteuccia struthiopteris)*.

At each site, three points for transect origins were randomly selected. The selection was made by creating three grids in excel (one for each site). Using an excel randomization function, three cells on each grid were randomly selected. The grids were then placed over their respective former impoundments on the map (Appendix 1). The randomly selected cells served as a marker for where the transects would be placed in relation to distance down the stream at each site. If randomly selected cells would place two transects in the same spot, the randomization process was repeated. Transect locations are marked with a star on the map (Appendix 1).

The sampling occurred on August 12-14, 2019. Late summer was chosen in order to record the most species possible. A measuring tape was run from the transect's origin at the stream edge perpendicularly toward the forest edge. Along the entire transect (30m), the dominant species, and the length at which they ran along the transect, were recorded in addition to their distance from the stream edge. This method was chosen to avoid possible bias that could occur when estimating percent plant cover in square plots. Where transects intersected the forest edge, the distance from the stream edge was noted. All taxa recorded were identified to species where possible. In areas where there wasn't a clear dominant species (especially those areas containing grasses, sedges and rushes), several taxa were recorded as a mix. Taxa were classified as woody or herbaceous; they were defined as native or non-native, and as invasive or non-invasive (GoBotany; Conte, 2005). The transects were broken up into three meter increments and the taxa in those increments on any of the transects were recorded with their corresponding site number(s).

All data were recorded and organized by site, transect, and distance from stream edge (meters) for analysis. A one-way analysis of variance (ANOVA) was used to test if proximity to forest edge had a positive relationship with average native and nonnative species richness. The same test was used to assess a potential relationship between proximity to stream edge and native and nonnative species richness.

RESULTS

The sampling resulted in 29 different plant taxa recorded, with 24 of the recorded plants identified to species, and four identified to genus. Seven woody taxa were recorded and 22 herbaceous (Table 1). Thirteen species were found at all three sites; three species were found at two of the sites, and 13 species were found at only one site (Table 1).

Table 1. Plant species identified at three study sites along Hamant Brook, Sturbridge, MA. Identified on August 13-14, 2019. Numbers in site presence column indicate sites where species were found. Site numbers correspond to each dam removal location, see Appendix 1 for site locations. Plants not identified to species are represented with a question mark. Asterisks indicate planted species.

Species	Site presence	native (N), nonnative (NN)	woody (W) or herbaceous (H)
Agrostis sp.	1, 2, 3	?	H
*Alnus incana**	1, 2, 3	N	W
Ambrosoia sp.	1, 2, 3	?	H
Carex scoparia	2, 3	N	H
Carex sp. 1	1, 2, 3	?	H
Carex sp. 2	1, 2, 3	?	H
Carex versicaria	1, 2, 3	N	H
Comptonia peregrina	2	N	W
Daucus carota	2	NN	H
Eupatorium perfoliatum	1, 2, 3	N	H
Gleditsia triacanthos	2	NN	W
Hemerocallis fulva	2	NN	H
Impatiens capensis	1	N	H
Juncus bufonius	1, 2, 3	N	H
Juncus effusus	1, 2, 3	N	H

Kalmia sp.	2	N	W
Lathyrus pratensis	1, 2, 3	NN	H
Lycopus virginicus	2	N	H
Mitchella repens	2	N	H
Oxalis stricta	2, 3	N	H
Persicaria maculosa	1, 3	NN	H
Phragmites americanus	1, 2, 3	N	H
*Pinus strobus**	2	N	W
Sagittaria latifolia	2	N	H
*Salix discolor**	3	N	W
Scutellaria galericulata	1, 2, 3	N	H
Securigera varia	2	NN	H
Tsuga canadensis	1	N	W
Typha latifolia	1, 2, 3	N	H

Site two had the highest species richness with 25 recorded species; 17 species were recorded at site three and 16 were recorded at site one (Figure 1; see appendix 1 for site locations). Site two also had highest woody species richness, with five recorded. At both sites two and three, two different woody species were recorded.

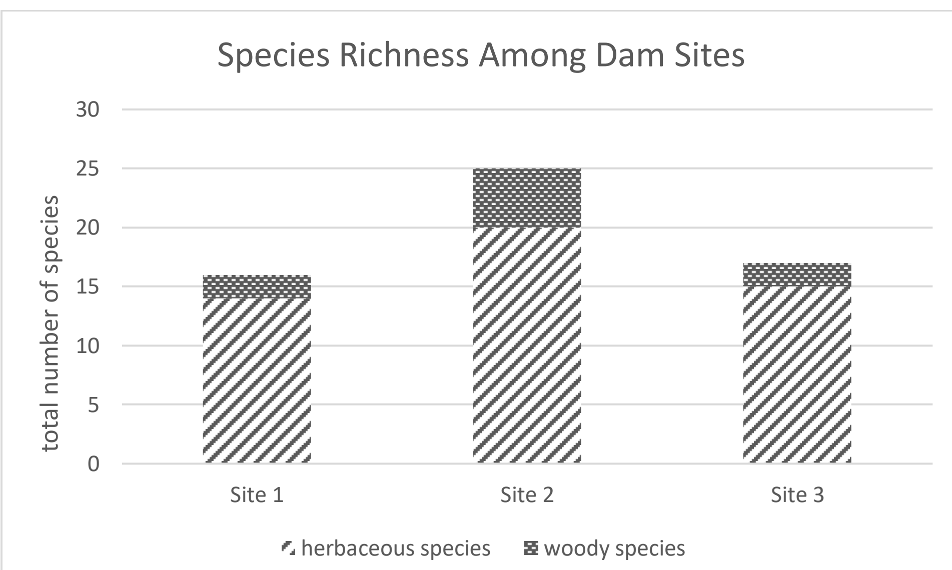

Fig 1. Comparison of species richness of three dam removal sites along Hamant Brook in Sturbridge, Massachusetts on August 12-14, 2019. Each site number corresponds to a dam removal location (see Appendix 1).

Nineteen of the recorded plant species were classified as native, six were classified as nonnative and four species classifications were unknown (Table 1). One nonnative species (*Lathyrus pratensis),* was recorded at all three sites. No nonnative taxa were classified as invasive.

Table 2. All species recorded along Hamant Brook, Sturbridge, MA on August 12-14, 2019. Corresponding numbers indicate the sites at which taxa were recorded within the distance from stream edge.

	Distance from stream edge (m)									
Species	**0-3**	**3-6**	**6-9**	**9-12**	**12-15**	**15-18**	**18-21**	**21-24**	**24-27**	**27-30**
Agrostis sp.		3	3	1,2,3	2,3	1	1,2,3	2	1,2	1,3
Alnus incana		3	2	2		1	2,3	2,3	2,3	3
Ambrosoia sp.				1	2	2	2		2,3	2
Carex scoparia		1,3	1,2,3	2	2	2	2		1	
Carex sp. 1	2	2,3	2	2,3	1,2,3	1,2,3	1,3	1	1,3	1,3
Carex sp. 2		3	2	2	2	2			1,3	1
Carex versicaria	2	1,2	1,2,3	1,2,3	1,2,3	3	1,2	1,2	1,2	1,2
Comptonia peregrina						2				
Daucus carota			2							
Eupatorium perfoliatum		2,3	3	1	1	2	1,2	1	1,2	1
Gleditsia triacanthos						2				
Hemerocallis fulva										3
Impatiens capensis										1,2
Juncus bufonius		1	1,2	2,3	1,2,3	1,2,3	1,3	1	1,3	1,3
Juncus effusus		1	1,2	1,2,3	1,2,3	1,2,3	1,3	1,3	1,3	1,3
Kalmia latifolia								2	2	2
Lathyrus pratensis	1,2,3	1,3	1,2,3	1,2,3	1,3	3	3	1,3	3	1
Lycopus virginicus							2		2	
Mitchella repens							2			
Oxalis stricta				3		2				3
Persicaria maculosa	1	1	1		1					1,3
Phragmites americanus	1,2	2,3	2	1,2						
Pinus strobus					2					
Sagittaria latifolia	2									
Salix sp.	3							3		
Scutellaria galericulata	2,3	3		1	3					1
Securigera varia							2	2		

Tsuga canadensis					2					1
Typha latifolia	1,2,3									

Site one was dominated by grasses, sedges, rushes, and other herbaceous species including a large population of meadow vetchling (*Lathyrus pratensis).* Day lilies (*Hemerocallis fulva)* and crown vetch (*Securigera varia)* populated the area where the former impoundment abutted the forest at site two. Sites two and three also contained similar herbaceous species as site one. Three of the recorded species, *Carex* sp. 1*, Carex versicaria,* and *Lathyrus pratensis,* were found at every three meter increment at at least one of the sites (Table 2). Three species, *Eupatorium perforliatum, Juncus bufonius,* and *Juncus effusus,* were found at every increment except for the first 0-3m. *Impatiens capensis* and *Hemerocallis fulva* were found furthest (27-30m) from stream edge (Table 2).

Average native species richness among all sites was highest, 2.5 taxa, at the forest edge and decreased by one. (Figure 2). Average invasive species richness was stagnant, at less than 0.5 taxa, but increased slightly about 12m from the forest edge and then decreased again (Figure 2). There was no significant difference in native species richness (p=0.57) or nonnative species richness (p=0.95) closer to forest edge versus farther away (Figure 2).

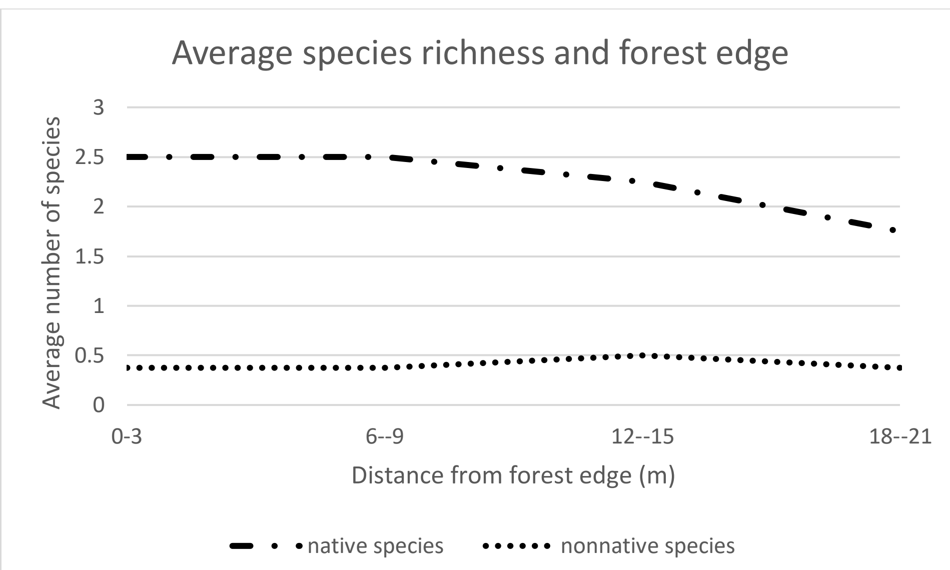

Fig. 2 Average species richness of native and nonnative species in relation to distance (m) from forest edge along Hamant Brook in Sturbridge, Massachusetts sampled on August 12-14, 2019. Y values are represented by the average number of species recorded among all transects found at a given location from the forest edge. Distance (m) from the forest edge is represented along the x-axis. There is no significant difference in native (p=0.57) or nonnative (0.95) species richness closer to forest edge versus farther from forest edge.

Average native species richness was lower at the stream edge than the forest edge (Figure 2; figure 3). Richness increased as distance from the stream increased, but decreased to an average of two at 18-21m. Average nonnative species richness at the stream edge was similar to forest edge (Figure 2; figure 3). It increased slightly at 3m from the stream edge, but never above 0.5 taxa (Figure 3). It decreased below the initial stream edge average as distance increased. There was no significant difference in average native (p=0.68) or nonnative (p=0.47) species richness recorded closer to the stream edge versus farther away (Figure 3).

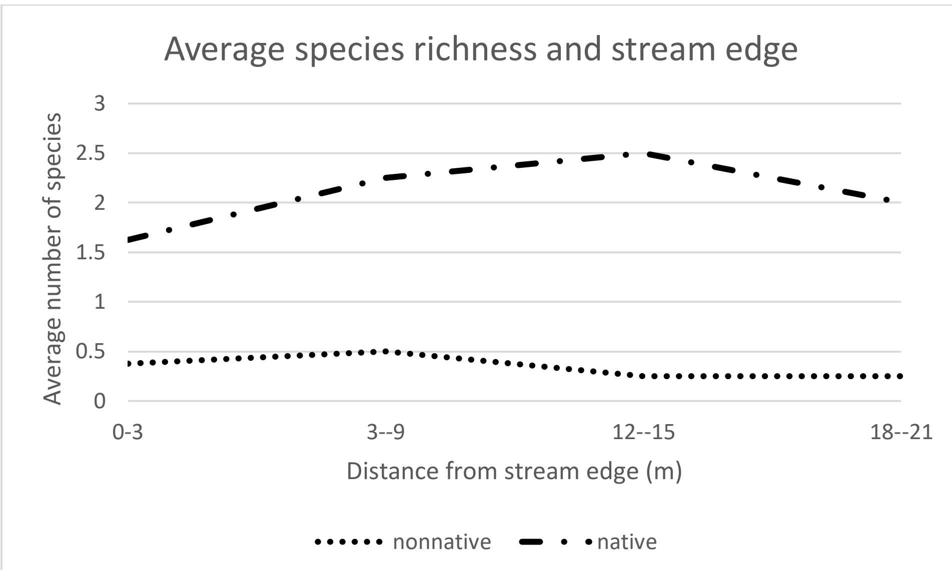

Figure 3. Average species richness of native and nonnative species in relation to distance (m) from stream edge along Hamant Brook in Sturbridge, Massachusetts sampled on August 13-14, 2019. Y values represent average number of taxa recorded among all transects found at a given location from the stream edge which is represented on the x-axis. A p-value of 0.68 found using ANOVA indicates no significant difference in average species native richness closer verses farther from stream edge. A p-value 0.47 also indicates the same is true for nonnative species.

DISCUSSION

With the data collected, the broader goal of this research was to document the species composition and richness of three former impoundments along Hamant Brook in Sturbidge Massachusetts so that American Rivers Association may potentially adapt their future restoration plans to better suit the early successional communities that have developed since the removals in 2017.

According to several studies concerning plant succession post dewatering a reservoir, almost no bare soil remains within a month of dam removal due to the nature of early colonizing species (Doyle et al., 2005; Kim et al., 2015; Orr & Stanley, 2006). This was the case in all three of the sites. Unsurprisingly, given the length of time since the dam removals, the majority of recorded plants were herbaceous along Hamant Brook. Doyle et al., (2005) described colonizing plants in early states of succession as “weedy vegetation.” The high seed production, fast growing ability, and efficient dispersal of these plants are likely why the study sites for this project were largely dominated by grass, sedge, and rush species. This is supported by research from Kim et al. (2015), a comparative study of sites varying in age from one to 30 years where younger sites were dominated by grasses and forbs as well (Kim et al., 2015 as cited by Orr & Stanley, 2006). These pioneer species play a vital role in stabilizing stream banks to prevent further erosion caused by the initial shock of dam removal and channel formation (Simon & Collison, 2002). Of course, there is variation in plant species depending on surroundings, location, and other environmental factors but, the findings of this study support the general expected successional patterns following dam removal (Orr & Stanley, 2006).

As is common with dam removal restoration plans, several woody species were planted by American Rivers at the sites to improve bank stabilization as new water flow establishes. These species were included in data collection and contributed to the species richness values. However, there were only seven total woody species recorded during the data collection process and not all of those recorded were human planted (see table 1). Lower woody species richness was expected because of the age of the dam removal and is consistent with the findings of other studies which found that sites thirty years or older tend to have a much higher number of tree and shrub species (Doyle et al., 2005; Orr & Stanley, 2006). Site two had the highest woody species richness, but this does not necessarily indicate a higher quality site. Potential reasons for this difference in richness could be that the randomly chosen transects at site two happened to intersect more woody species planted during the restoration project, or the transects at site two, on average, intersected forest edge sooner than other transects (which would likely include more woody species given their location in a more established, later successional, area).

Newly exposed areas are especially vulnerable to colonization by nonnative species (Orr & Stanley, 2006). Orr & Stanley (2006) found a mean frequency of introduced species to be 75% among 13 small dam removal sites in Wisconsin. In comparison to Orr & Stanley’s (2006) findings, only 21% of the recorded species of this study were nonnative, which is relatively low. None of the nonnative species were classified as invasive (“non-native species that have spread into native or minimally managed plant systems in Massachusetts") (Conte, 2005). These plants "cause economic or environmental harm by developing self-sustaining populations and becoming dominant and/or disruptive to those systems," (Conte, 2005). The findings suggest that it is likely taxa close enough to the sites to disperse and establish there are largely native or nonnative, but not invasive. While it is important to note the number of nonnative species recorded, because none of them are considered invasive, they may have naturalized to the community and could play a role in its function. As the sites age and reach mid successional stages, herbaceous plant composition will likely change as sites transition from pioneer species to more woody and later successional taxa.

The total species richness for the sites along Hamant Brook supported Orr & Stanley's (2006) findings that recent removals' species richness was highly variable among their study sites, with the lowest richness being seven and the highest 29. When compared to Orr & Stanley's (2015) study, the sites had an average species richness that is fairly high (Figure 1). Notably, site two had a species richness that was higher, while the other two had under 20 taxa recorded, though this could be by chance. Variation in richnesses among the three sites is also supported by Orr & Stanley (2015). Potentially, different and/or more species between sites could impact future successional patterns differentiating the three sites from one another.

The dam removals provided a clear edge in which the former impoundment abutted the forest. This provided an open area for seed dispersal and pollination. This research aimed to document if proximity to forest edge correlated positively with average species richness (though species richness values appear low in figures two and three [all under three taxa], it is important to note that these values represent average species richness in only three-meter increments). Though this was rejected, if the species within the abutting forest require different soil, water, or light levels than the newly exposed mudflat, then they may have been less successful—potentially a reason no correlation was found. Another reason could be that the newly exposed sediment is likely to have higher amounts of organic matter, which retains more water for a longer period of time, so taxa which are successful in the abutting forest may not be able to thrive in soils saturated with water (Lafrenz et al., 2013 as cited by Pons and Zonneveld, 1965). These discrepancies in adjacent land areas could cause different a plant community to form within a larger surrounding habitat.

The study also documented if proximity to the stream edge impacted native and nonnative species richness. The stream edge not only served as a starting point for all of the transects but, was also the furthest point from forest edge. Without the influence of nearby understory, the stream edge could be considered more vulnerable to nonnative and/or invasive species. The results indicated this is not the case, however. Comparative studies indicate that nonnative species richness was relatively low along Hamant Brook (Orr & Stanley, 2006).

Plant communities influence habitat, biological soil formation, stream channel development and water quality. A better understanding of the plant communities that have established in the two years since the dam removals will hopefully help American Rivers Association, Massachusetts Fish and Wildlife, the Sturbridge Conservation Commission evaluate the extent to which dam removal has affected these parameters.

CONCLUSION

This research indicated a relatively high species richness along Hamant Brook when cross referenced with other comparative studies (Orr & Stanley 2006). The hypothesis, suggesting that native and nonnative plant species richness was affected by proximity to forest or stream edge was rejected. These findings will, however, provide American Rivers with an understanding of how their restoration project has unfolded since their work in 2017 and 2018 in regard to plant communities. Though their ultimate goal was improved water quality for the Quinebaug River (and therefor the greater Quinebaug watershed), the outcomes of this study may be useful in this broader goal. Future research may include resampling the sites in several years to see how plant community richness and composition has changed since this sampling in 2019. In having a grasp on the plant composition of the sites along Hamant Brook, American Rivers may be able to better predict future successional patterns.

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APPENDIX 1 Apendix 1. Map created by American Rivers Association. Sites referred to throughout this paper correspond to the following labels on the map: site 1, lower pond dam impoundment; site 2, middle pond dam impoundment; site 3, upper pond impoundment. Transect locations are marked with a star.