Evaluation of Successful Forested Wetland Mitigation in Iowa

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16. Abstract

Wetlands contribute to important landscape-level functions such as nutrient retention and cycling, sediment capture, flood attenuation, groundwater recharge, wildlife habitat, and recreational opportunities. Historic losses of wetlands due to land use conversion led to federal regulations requiring mitigation for unavoidable adverse impacts to them. Road development projects are among activities that cause unavoidable impacts to wetlands, and agencies such as the Iowa Department of Transportation (Iowa DOT) have expended considerable effort and funding to meet regulatory requirements.

This study evaluated tree seedling performance and site characteristics on 25 forested wetland restoration sites in Iowa, where Root Production Method (RPM) stock, bare root (BR) seedlings, and/or balled-and-burlapped (B&B) stock were planted. Initially, researchers assessed 2,533 seedlings representing 22 species, including 1,994 BR and 539 RPM trees. Overall, BR seedlings had higher mean survival rates (91%) than RPM stock (74%). Trees protected by shelters had lower survival rates. Seedling survival was also lower on sites associated with higher order streams and relatively fine surface soil textures. A more detailed assessment of 1,050 of the seedlings indicated considerable variation in performance across species, but on average BR seedlings had greater survival, height, crown depth, and root collar diameter than RPM seedlings. The study also compared stem densities for these plantings with naturally regenerating stands and reference forest areas near Iowa DOT project sites. Average stem densities for RPM stock were considerably lower than the averages for BR seedlings and naturally regenerating stands. An experimental site assessment indicated BR and RPM seedlings were similar to each other in size and morphology, and both were significantly smaller than B&B stock. Stem densities on this site were also low for RPM and B&B stock compared to naturally regenerating stands.

Overall, the researchers recommend selection of sites near lower order streams with surface soil clay contents less than 25%, and they suggest a strategy of planting BR seedlings at high densities without tree shelters to achieve more consistent success, specifications that clearly identify contractor responsibility for removal of any foreign material from seedlings at planting, and greater attention to site maintenance to promote seedling success. Finally, more realistic benchmarks for compliance should be set on a site-specific basis and tied to features of the surrounding landscape.

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EVALUATION OF SUCCESSFUL FORESTED WETLAND MITIGATION IN IOWA

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EXECUTIVE SUMMARY

Wetland area throughout the US has declined dramatically due to conversion to other land uses such as intensive agriculture or expansion of urban land areas. Historic wetland losses led to regulations contained in Section 404 in the US Clean Water Act that require a permit for any projects with impacts to wetland areas. Mitigation for damage or loss of wetlands is required of permittees, which may include restoring, creating, enhancing, and/or preserving additional wetlands. Wetland restoration is a relatively common approach to mitigation. Permittees are advised to provide in-kind compensation, which often involves identifying similar sites in nearby areas that may already be supporting growth of aquatic plants and contain hydric soils. Mitigation permits typically require that restored sites meet specific requirements in terms of vegetation, soil, and hydrological characteristics to be released from interventions or continued monitoring.

Success in mitigation wetland establishment has been mixed and has been of particular concern with respect to forested wetland mitigation, for which regulatory compliance nationwide has been low. Substantial mortality of planted trees is often a cause of non-compliance. Because of poor performance for forested wetland mitigation, new rules for permittees were released (the new Final Rule on Compensatory Mitigation for Losses of Aquatic Resources of 2008), which include longer monitoring periods, higher mitigation-area-to-impact-area ratios, and new minimum species diversity and stem density requirements. As an agency whose road construction projects sometimes have unavoidable impacts on wetlands, the Iowa Department of Transportation (Iowa DOT) is among agencies that seek such permits, and Iowa DOT personnel have sought to mitigate wetland impacts through establishment of in-kind restoration of wetland areas throughout the state. Although many wetland mitigation projects in the state have been successful, concerns about compliance for forested wetland mitigation in particular have arisen due to variable survival rates for trees planted on mitigation sites.

The goal of this research was to investigate site selection, project design, and performance of tree species and stock types to develop recommendations leading to timely release from permit requirements on forested wetland mitigation sites in Iowa. Specific project objectives were to (1) conduct a literature review examining theory and practice related to forested wetland mitigation, with particular attention to methods relevant for Iowa; (2) assess the degree of success achieved by existing forested wetland mitigation projects in Iowa by examining a set of project sites as well as areas of natural regeneration and reference forest areas; (3) evaluate the performance of different species and stock types on a relatively new experimental planting site; and (4) develop recommendations for forested wetland mitigation design and implementation leading to successful compliance and timely release from mitigation permits.

Both the literature and this study’s field research point toward greater seedling survival in the riparian areas of lower order streams (likely to have less prolonged flooding) and lower soil clay content (≤ 25% clay). Seedling survival rates across all sites (ranging from 65% to 100% depending on species and stock type, averaging 87%) were relatively high compared to previous reports in the literature (for which they range from 54% to 76%). Several species (e.g., Kentucky coffeetree, pin oak, river birch, and swamp white oak) had consistently high survival rates, although others (e.g., American sycamore and black walnut) were somewhat less successful. Overall, survival rates of bare root (BR) seedlings (91%) were greater than those of Root Production Method (RPM) seedlings (74%). Also, the research found that trees protected by tree shelters had lower survival rates than trees without shelters. Analyses of stem density indicate that plantings of RPM and balled-and-burlapped (B&B) stock have lower densities than BR stock, much lower densities than are typical of naturally regenerating stands, and even a short time after planting are not sufficient to meet regulatory requirements.

Based on this research, the authors recommend selecting sites associated with lower order streams and with relatively low soil clay content (< 25%) to limit exposure of seedlings to the effects of long-duration flooding. If it is necessary to choose sites associated with higher order streams, it may be beneficial to modify site topography to create microsites for seedling placement at slightly higher elevations that will limit duration of seedling exposure to inundation and/or prolonged high water tables. The authors also suggest that weed control activities be conducted both before and after tree seedlings are planted and be continued until tree seedlings have "captured" the site. Species and stock type should be chosen based on location in the state, site characteristics, species' adaptability to flooding, and previous performance in mitigation projects. On the basis of their empirical work, the authors recommend a general strategy of planting bare root seedlings at relatively high densities (e.g., 600 to 1,000 stems/acre) and use of tree shelters on a limited number of trees (30 to 50 trees/acre) or more on sites with known strong herbivore pressure. To accelerate seedling capture of the site, RPM and/or B&B stock could be used to supplement BR seedlings in order to speed canopy closure. In addition, the research team's assessments of plots in existing bottomland hardwood forests (reference forests) and nearby areas of volunteer natural regeneration indicate that natural regeneration should be considered a viable option for establishing forested wetlands and that creating conditions to support natural stand development should be considered a viable approach to forested wetland mitigation in Iowa.

1. INTRODUCTION

Background

Wetlands are defined as “lands transitional between terrestrial and aquatic systems, where the water table is usually at or near the surface, or the land is covered by shallow water” (Cowardin et al. 1979). Wetland ecosystems contribute to a number of important landscape-level functions, such as carbon storage, nutrient retention and cycling, sediment capture, flood attenuation, groundwater recharge, wildlife habitat, and recreational opportunities (Bruland and Richardson 2005, Broussard and Turner 2009, Mitsch et al. 2009, Johnson et al. 2012). However, widespread loss of wetlands across the US has occurred due to their conversion to intensive agricultural and urban land uses, including construction of roads (Ouchley et al. 2000; Dahl 2000, 2006). Some of the highest documented wetland losses are in the states of California, Indiana, Illinois, Iowa, Kentucky, Missouri, and Ohio, where wetland area coverage has declined by at least 80% (Dahl 2006). Historically, wetland losses were the result of decision-making at the site scale rather than the landscape scale, although over time cumulative impacts have led to the loss of landscape-scale structure and functional integrity (Johnson and McCormick 1978, Richardson 1981). This is particularly true of riparian wetlands in the Midwest, many of which were naturally forested riparian areas that played a crucial role in nutrient and sediment capture and protected water quality in the region’s streams and rivers (Bruland and Richardson 2005, Theriot et al. 2013, Jacob et al. 2013, Passeport et al. 2013, Maillard and Imfeld 2014).

Concerns over dramatic losses of wetlands and reductions in functional capacity have led to recent revisions of federal regulations to limit additional disturbance and require mitigation to repair or replace impacted areas (NRC 2001, Hough and Robertson 2009). Section 404 of the Clean Water Act (CWA) (as revised in 2008) articulates a national policy goal of “no net loss” of wetland area or function (Federal Register 2008). Under this legislation, conversion of wetlands to other land uses has been allowed on a case-by-case basis via permits, typically administered by the US Army Corps of Engineers (US ACE), which stipulate specific mitigation activities that the permittee must conduct related to the hydrology, soil, and vegetation on mitigation sites (US ACE 1987). Wetland restoration and/or reconstruction is a relatively common approach for mitigating unavoidable disturbances to wetlands (Sweeney and Czapka 2004, Matthews and Endress 2008). Permittees generally propose to provide in-kind compensation, that is, to seek sites similar to the impacted wetland to conduct restoration of the same wetland type. This usually involves identifying sites with hydrologic characteristics that are similar to and in areas near impacted wetlands as locations for restoration activities, and efforts to establish vegetation that is similar to what was originally present at the disturbed site.

Agencies such as the Iowa Department of Transportation (Iowa DOT) have been engaged in wetland mitigation activities for many years. A previous assessment of the ecological performance of 12 Iowa DOT mitigation wetlands indicated that they performed similarly to reference (i.e., undisturbed) wetlands and that the area of all sites taken together exceeded the total required by permits (VanDeWalle et al. 2007). However, this assessment did not directly address forested wetland mitigation, and attempts to restore/reconstruct forested riparian wetlands have met with less success, both nationwide (Brown and Veneman 2001, Cole and

Shafer 2002, Morgan and Roberts 2003, Matthews and Pociask 2015) and in Iowa (Marler 2014). Difficulties with establishment of bottomland hardwoods have been attributed variously to wildlife damage (e.g., McLeod et al. 2000, Riley et al. 2015), soil characteristics (e.g., Richardson and Bruland 2005, Pennington and Walters 2006) site hydrology (e.g., depth and duration of flood events [Matthews and Pociask 2015]), and extreme weather (both floods and droughts) during the establishment period.

Problem Statement

Results of recent research demonstrate mixed success for mitigation wetland establishment (Kihslinger 2008, Matthews and Endress 2008). Some success has been reported for meeting soil, hydrology, and vegetative cover criteria, but goals for vegetative composition, structure, and/or wetland area criteria are less often achieved (e.g., Wilson and Mitsch 1996, Brown and Veneman 2001, Morgan and Roberts 2003, Matthews and Endress 2008). A recent assessment of the ecological performance of 12 Iowa DOT mitigation wetlands, however, indicated a high degree of success and that these wetlands performed similarly to undisturbed wetlands (VanDeWalle et al. 2007). However, this earlier assessment did not include forested wetland mitigation projects, for which regulatory compliance has been low across the US (often because of substantial mortality of planted trees, as per Brown and Veneman [2001], Robb [2001], Cole and Shafer [2002], Pennington and Walters [2006], Matthews and Pociask [2015]) and about which there are concerns in Iowa (Marler 2014). Because of their unique functional role in the Midwest landscape, there is strong interest in additional assessment of riparian/bottomland forest wetland restoration projects to identify factors that could lead to greater mitigation success.

Over time, many agencies and entities have been involved in experimental efforts to establish and/or restore bottomland hardwoods, including investigations of performance by different species and stock types, and silvicultural aspects of establishment (e.g., Barton et al. 2000, McLeod et al. 2000, Stanturf et al. 2001, Lockhart et al. 2003, Patterson and Adams 2003, Sweeney and Czapka 2004). For bottomland/riparian plantings, such evaluations of seedlings of different species have indicated differential performance according to elevation (at the microtopographic scale) and flood event frequency and duration (Barry et al. 1996, Bruland and Richardson 2005, Randall and Herring 2012). Several investigators have reported greater success of species (e.g., *Acer saccharinum*, *Populus deltoides*, and *Salix nigra*) that are better adapted to lower micro-elevations with relatively frequent and sometimes prolonged inundation, whereas other (often later-successional) species (e.g., *Quercus bicolor* and *Quercus palustris*) are more easily established in locally elevated microsites with less frequent and shorter-duration flooding (Pennington and Walters 2006, Simmons et al. 2011).

Previous evaluations of stock types that are relevant to this study include assessments of Root Production Method (RPM) seedlings and bare root (BR) stock. RPM seedlings are produced by placing carefully selected seed in open-bottomed trays to germinate and subsequently transplanting them to successively larger containers that use air pruning to promote production of a more a fibrous root system over a typical culture period of two years (Lovelace 2002, Dey et al. 2004). Both the root systems and stems of RPM stock are larger than those of typical one-year-old bare root plants. A number of studies have documented greater survival and more rapid growth of RPM seedlings compared to BR stock in bottomland planting projects (Dey et al. 2004, Krekeler et al. 2006, Walter et al. 2013). In spite of their greater cost, researchers have recommended their use, particularly to enhance establishment of later-successional species such as oaks, to extend the planting season, and to hasten the process of canopy closure on planting sites. In contrast, BR seedlings are grown from seed broadcast in nursery beds for one or two season(s) before being lifted from the nursery bed and packaged/chilled to maintain seedling moisture levels until the time of planting. These plants are much less expensive and relatively easy to handle and transport to planting sites, and large numbers of BR seedlings can be planted relatively quickly using planting machines or even by hand.

A number of investigators have also evaluated use of tree shelters to enhance establishment success. These can be solid, corrugated, or mesh tubes that are placed around individual seedlings at the time of planting. Originally produced in the United Kingdom (UK) in the early 1980s to protect seedlings from animal damage (Tuley 1983), they were also found to provide favorable micro-environmental conditions that enhance seedling growth (Lantagne et al. 1990; Costello et al. 1991; Lantagne 1995, 1997; Ponder 2000). Although performance has varied among species and stock types, in most cases height growth of sheltered trees has increased to a greater extent than diameter growth. Some researchers have recommended use of more light-transmitting and ventilated shelters that may promote a better balance between height and diameter growth (Sharew and Hairston-Strang 2005). However, other researchers have reported little advantage in terms of survival or growth of sheltered seedlings compared to unsheltered seedlings (e.g., Stuhlinger 2013), that their effect diminishes over time (Drayer et al. 2017), or that they actually reduce growth and survival of some species under particular circumstances (Bardon et al. 1999).

Personnel in the Iowa DOT are among those who have expended considerable effort to establish forested wetland mitigation/restoration projects in Iowa that meet the requirements set forth by federal policy as overseen by the US ACE, but results have been mixed (Marler 2014). This project was undertaken to address specific concerns related to tree seedling survival by investigating site selection, project design, and performance of tree species and stock types in order to make recommendations that support timely release from permit requirements on forested wetland mitigation sites in Iowa.

Project Objectives

This project had four specific objectives:

- Conduct a literature review to examine theory and practice related to forested wetland mitigation and restoration, with particular attention to methods relevant for Iowa.

- Assess the degree of success achieved by existing forested wetland mitigation projects in Iowa by examining a set of project sites as well as areas of natural regeneration and reference forest areas.

- Evaluate the performance of different species and stock types on a relatively new experimental planting site.

- Develop recommendations for forested wetland mitigation design and implementation leading to successful compliance and timely release from mitigation permits.

Results for each project objective are summarized in the four sections of the text that follow.

Objective 1: Literature Review

The research team conducted an extensive literature search and review beginning in fall 2014 using available search tools and library resources at Iowa State University as well as Interlibrary Loan to acquire relevant books and peer-reviewed literature. The initial literature synopsis was based on 96 sources, reviewed by the Technical Advisory Committee in summer 2015, and delivered to Iowa DOT personnel in fall 2015. The team continued to acquire additional literature sources as the project progressed; these are cited in more recent project-related documents.

The literature review indicated that the establishment of forested wetlands is difficult, often due to substantial mortality of planted trees. This has generally been attributed to the ecological complexity (a high degree of variability across both space and time) of the natural systems they are meant to replace (Anderson and Mitsch 2008a) and to factors, such as extreme flooding or drought and/or herbivore pressure, that negatively affect seedling survival. Temporal changes in composition of natural forested wetlands occur on time scales of decades, outside the scope of regulatory timelines for mitigation projects (Hodges 1997, Ouchley et al. 2000).

Further, the influence of these changes on other system characteristics is just beginning to be understood, and the effectiveness of restored forested wetlands in providing important biogeochemical functions appears to lag significantly behind establishment of wetland hydrology and vegetation (D'Angelo et al. 2005, Theriot et al. 2013). In addition, although many mitigation/restoration sites are located near existing bottomland hardwood stands that could contribute to natural regeneration, designs that rely on this form of forested wetland establishment are uncommon and rarely described in the literature. Finally, locating restored forested wetland project sites so as to purposefully connect existing remnant riparian forests could potentially increase their landscape-scale function, an approach that is also rarely addressed in the literature (but see the suggestion of Shaffer et al. 1992).

Objective 2: Assessment of Forested Wetland Mitigation Projects

The team conducted an assessment of 24 sites located in central and southeast Iowa to evaluate performance of different tree species and seedling stock types in relation to each other and to natural regeneration and reference forests. Specifically, researchers compared survival rates for BR and RPM seedlings planted on 14 of those sites and evaluated species diversity and stem density on those sites as well as on 10 additional sites (natural regeneration and reference forests).

Objective 3: Assessment of an Experimental Forested Wetland Restoration Site (Goose Pond)

In 2015, the research team conducted an assessment of an additional experimental planting established by the Iowa DOT in 2014 at the Goose Pond site located in Linn County. This site was included to evaluate the performance of different tree species and three seedling stock types in relation to each other and to characteristics of natural regeneration and reference forests. Specifically, for this site, researchers examined survival rates for BR, RPM, and balled-and-burlapped (B&B) seedlings and evaluated species diversity and stem density for them compared to the same set of five natural regeneration and reference forests already described.

Objective 4: Recommendations for Tree Establishment in Forested Wetland Mitigation

The researchers used information from their literature review and the empirical site studies to develop a set of recommendations for establishment of trees on forested wetland mitigation project sites. The purpose of the recommendations is to provide guidance to Iowa DOT personnel responsible for site selection and the planning, design, and development of specifications for wetland mitigation projects throughout the state. The recommendations emphasize the importance of setting realistic performance standards in each permit, considering the landscape-watershed context of proposed mitigation sites, potentially creating microtopographic variation on restoration sites to increase seedling survival rates, attending to pre- and post-planting weed control, selecting species and stock types based on previous performance, and strengthening specifications and increasing monitoring of work performed by contractors on mitigation sites.

Overall, the team recommends selecting sites associated with lower order streams and with relatively low soil clay content (< 25%) to limit exposure of seedlings to the effects of long-duration flooding. If it is necessary to choose sites associated with higher order streams, it will likely be beneficial to modify site topography to create microsites for seedling placement at slightly higher elevations that will limit duration of seedling exposure to inundation and/or prolonged high water tables. The team also suggests that weed control activities be conducted both before and after tree seedlings are planted and be continued until tree seedlings have "captured" the site. Species and stock type should be chosen based on location in the state, site characteristics, species' adaptability to flooding, and previous performance in mitigation projects. On the basis of the team's empirical work, they recommend a general strategy of planting bare root seedlings at relatively high densities (e.g., 600 to 1,000 stems/acre) and use of tree shelters on only a limited number of trees (up to 30 to 50 trees/acre) or more primarily on sites with known strong herbivore (e.g., deer and beaver) pressure. To accelerate seedling capture of the site, RPM and/or B&B stock could be used to supplement BR seedlings in order to speed canopy closure.

This report is organized by chapters as follows. Chapter 2 presents the team's review of the current literature on the topic of forested wetland mitigation. Chapter 3 explains the team's research methodology. Chapter 4 includes a summary of the team's findings and its conclusions. Chapter 5 includes the recommendations for Iowa DOT personnel for successful forested wetland restoration strategies.

2. LITERATURE REVIEW

Introduction

General Characteristics of Wetlands and Wetland Types in Iowa

Wetlands are ecosystems such as swamps, marshes, bogs, seeps, river oxbows, and overflow areas where surface water or groundwater flow patterns cause water to stay at or near the land surface for significant periods of time (Mitsch et al. 2009). Cowardin et al. (1979) defined wetlands as “lands transitional between terrestrial and aquatic systems, where the water table is usually at or near the surface or the land is covered by shallow water… Wetlands must have one or more of the following three attributes: (1) at least periodically, the land supports predominantly hydrophytes; (2) the substrate is predominantly un-drained hydric soil; and/or (3) the substrate is non-soil and is saturated with water or covered by shallow water at some time during the growing season of each year.”

Historically, Iowa’s landscape contained significant wetland areas, with many poorly drained, closed depressions in glaciated upland areas, and extensive riparian wetlands in floodplains of major rivers and streams throughout the state (Thompson 1992). Following settlement, it is estimated that approximately 89% of these wetlands were drained or lost due to channelization (USDA 1995). Remaining natural wetlands that occur in Iowa include several forms of palustrine wetlands: emergent wetlands, fens, and wet meadows dominated by herbaceous vegetation, as well as forested wetlands containing significant woody vegetation (Cowardin et al. 1979, Bishop and van der Valk 1982, Iowa NRCS 2005, USFWS-NWI 2012). In Iowa, forested wetlands may occur in small upland closed depressions or, more commonly, in low-lying areas associated with rivers and streams (Cowardin et al. 1979, Iowa NRCS 2005). Generally, upland depression wetlands may be recharge areas where precipitation inputs and overland flow contributions in excess of evapotranspiration move through the soil and reach the water table, whereas riverine wetlands where groundwater levels tend to be shallow or at the surface may be either recharge or discharge areas (e.g., Mitsch et al. 2009, Schaetzl and Thompson 2015).

Characteristics of Forested Wetlands

Both forested wetlands that occur as upland systems and bottomland forests have plant communities that are dominated by tree and shrub species. Common tree taxa that occur in these wetlands include several species of willow (*Salix* spp.), ash (*Fraxinus* spp.), and elm (*Ulmus* spp.), as well as eastern cottonwood (*Populus deltoides*), hackberry (*Celtis occidentalis*), boxelder (*Acer negundo*), silver maple (*Acer saccharinum*), red maple (*Acer rubrum*), river birch (*Betula nigra*), swamp white oak (*Quercus bicolor*), American sycamore (*Platanus occidentalis*), Kentucky coffeetree (*Gymnocladus dioicus*), bur oak (*Quercus macrocarpa*), and black walnut (*Juglans nigra*). Shrubs include species of dogwood (*Cornus* spp.), as well as chokecherry (*Prunus virginiana*), elderberry (*Sambucus canadensis*), and wahoo (*Euonymous atropurpureus*) (USDA Forest Service 1995, Herring 2012, NRCS 2005). Forest composition varies spatially and temporally (more specifically described below) in these systems as influenced by topographic and soil moisture gradients and by change over time as natural or human-influenced successional processes lead to site alterations (Hodges 1997, Lockhart et al. 2010, Gee et al. 2014).

Landscape Ecological Roles of Forested Wetlands in Iowa

Protection of water quality and attenuation of water quantity. Wetlands serve many important ecological roles at a landscape scale (Iowa NRCS 2005). They provide protection for adjacent terrestrial habitats and protect soils from erosion by capturing sediments, thus acting as buffers or filters between intensive urban and agricultural land uses and groundwater or surface water systems (Mitsch et al. 2005, Mitsch et al. 2009). These functions are especially critical in Iowa and throughout the Upper Midwest, where the soil surface is often exposed for significant periods of time seasonally in the spring and fall on a large portion of the landscape.

Wetlands also provide sequestration and processing of environmental contaminants, including nutrients, especially nitrogen and phosphorus, excess quantities of which (at least in part due to widespread loss of wetlands) have contributed to serious impairment of surface water quality throughout the Upper Midwest and areas downstream, especially in the Mississippi River basin (Broussard and Turner 2009, Theriot et al. 2013). These nutrients can be captured and stored as organic material in wetlands and, in the case of nitrogen, can be removed from the terrestrial system under periodically anaerobic soil conditions that support microbial denitrification (Bruland and Richardson 2005, Theriot et al. 2013). Additionally, through a variety of non-degradative (volatilization and sorption) and degradative processes (photolysis, hydrolysis, biodegradation), wetlands can remove pathogens and synthetic chemicals, including pesticides and hydrocarbons from groundwater and surface water (Passeport et al. 2013, Douglass et al. 2014, Maillard and Imfeld 2014).

Because of high rates of biomass production and relatively low rates of decomposition, wetlands provide significant carbon storage and globally contain close to 30% of the organic carbon on the Earth (Mitsch et al. 2009). However, under extensive anaerobic conditions wetlands can release carbon as methane gas, although the quantities and fluxes of gaseous carbon released at a global scale are difficult to estimate. Importantly, forested wetlands in particular provide for long-term carbon storage in the form of living tissue, given their relatively large quantities of perennial biomass both above and below ground.

Hydrologically, wetlands store significant quantities of water, which is especially important in response to intense and long-duration precipitation events, and thus provide for groundwater recharge in upland settings and attenuation of peak flows of rivers in bottomland settings (Thompson 1992, USDA 1995, Bruland and Richardson 2005, Iowa NRCS 2005, Mitsch et al. 2009). Certain features of wetlands, such as microtopographic variation, are thought to contribute to their ability to retain greater quantities of water (Bruland and Richardson 2005). Again, the particular role of forested wetlands with respect to flood attenuation is significant, since trees remove a relatively large quantity of water from the soil and release it to the atmosphere via evapotranspiration (Johnson et al. 2012). As wetlands absorb and retain surface water and function as depositional areas for soil and sediment, nutrients, metals, and synthetic chemicals entrained in groundwater and surface water, these ecosystems improve water quality downstream from these sites (Jacob et al. 2013, Passeport et al. 2013, Maillard and Imfeld 2014).

Habitat for wildlife. Wetland systems also provide critical habitat for both vertebrate and invertebrate wildlife species, and, again, forested wetlands in particular play several key roles in this regard. Many vertebrate species that must inhabit wetlands for some or all of their lives (wetland-dependent species) also require forest habitat to maintain viable populations (Peterson and Westmark 2013, Alix et al. 2014, Calhoun et al. 2014, Masse et al. 2014, Peterman et al. 2014, Quesnelle et al. 2015). Likewise, many forest-dwelling species without an aquatic life stage (such as birds) depend on resources generated by nearby wetlands, including constituent plant taxa and amphibians and insects with aquatic larval stages (Alsfeld et al. 2010, MacDade et al. 2011, Hagy et al. 2014, Masse et al. 2014, Quesnelle et al. 2015).

Trees in these systems function as ecosystem engineers, providing terrestrial life stages of wetland-dependent vertebrates with substrate, shelter, and food resources that are essential for growth, breeding, migrating, hibernating, and avoiding competitors and predators (Calhoun et al. 2014, Quesnelle et al. 2015). Shading from tree canopies reduces temperature and increases moisture content of terrestrial soils, mitigating heat and water stress among vertebrates and their prey (Alix et al. 2014, Masse et al. 2014). Forest habitat also indirectly benefits wildlife by improving wetland condition. Sediment and nutrient inputs to wetlands themselves can be reduced by trees absorbing water and nutrients and the structure created by woody vegetation that slows overland water flow and promotes infiltration (Jackson 2006, Quesnelle et al. 2015). Inputs of woody debris and leaf litter to wetlands increase habitat and food resources in the basin itself, and overhead canopy cover creates more hospitable conditions by reducing water temperature, increasing dissolved oxygen, and preventing noxious algal blooms (Jackson 2006, Calhoun et al. 2014).

Conservation management of wetland-dependent species often focuses on preserving and increasing habitat within the wetland of interest. The abundance and diversity of aquatic invertebrates found in wetlands provide a critical link in the detrital-based food webs of these communities in addition to increasing the biodiversity of these systems (Fritzell 1988). In addition, it is increasingly apparent that for many wetland vertebrates, quality of the surrounding landscape is as important, or even more important, than the wetland itself. In many recent investigations, wetland-dependent vertebrate species diversity and population densities were found to be significantly and positively correlated with tree abundance near the wetland basin, with amphibians, reptiles, and birds responding especially favorably to the quantity of adjacent upland forest habitat (Peterson and Westmark 2013, Peterman et al. 2014, Quesnelle et al. 2015). In fact, for many amphibian and reptile species, forest area proximate to a wetland appears to be more important than the size of the wetland itself in determining occupancy of a site (Alix et al. 2014, Quesnelle et al. 2015). Therefore, an integrated approach that addresses restoration and maintenance of both the wetland basin and surrounding habitat is usually necessary to achieve wildlife conservation objectives (Peterson and Westmark 2013, Calhoun et al. 2014).

Whereas restoring and maintaining trees within and near a wetland generally has positive effects on wildlife, attributes of the forest system itself influence the composition, densities, and diversity of wildlife species at a site. Forest composition and age have been found to be important determinants of bird population/community structure (Masse et al. 2014). Specific management actions found to increase wildlife species diversity include maintenance of young and old forest patches within or near the same wetland site, as well as actions that enhance tree species diversity and provide mast crops (Masse et al. 2014). Finally, interactions between trees and wildlife are reciprocal, so effects of wildlife on the integrity of the forested wetland itself must also be considered in management plans. Herbivory by deer, rabbits, rodents, and beaver can kill planted trees, especially in early growth stages when trees are most vulnerable to damage (McLeod et al. 2000, Sweeney and Czapka 2004, Riley et al. 2015).

Regulatory Requirements for Wetland Mitigation

General Requirements Stipulated in Original Guidance

Widespread loss of wetlands across the nation (due to agricultural expansion, growth of urban areas, and road construction) led to federal legislation to limit wetland disturbance and require mitigation to repair damage/replace loss of wetlands (Section 404 of the Clean Water Act). As part of this legislation, compensatory mitigation is allowed in cases of unavoidable wetland damage or loss as per requirements outlined in permits granted on a case-by-case basis. Allowable wetland mitigation approaches have included enhancing, restoring, or creating wetlands or protecting existing wetlands (CWA, as amended in 1977). Among these approaches, many mitigation projects have restored or created wetlands that were established with single permits (NRC 2001, Spieles 2005).

Although specific requirements/goals are stipulated in individual permits, general requirements for wetland mitigation as per the original guidance provided (US ACE 1987) include characteristics of the vegetation, soil, and hydrology on wetland restoration sites (Table 1). Criteria for vegetation indicate that dominant plant species must be native and hydrophytic, such that the species in a stratum first exceeding 50% of dominance, plus any additional species that individually comprise 20% for each stratum, have a Wetland Indicator Status (WIS) of facultative, facultative wetland, or obligate wetland (US ACE 1987). The same guidance stipulates that soils be hydric or be affected by conditions that would support hydric soil formation, as evidenced by low-chroma colors, mottles, and other visible indicators of anaerobic conditions (US ACE 1987). The third general criterion is that the site be inundated or saturated continuously for 5% or more of the growing season in a majority of the years the site is monitored for compliance (US ACE 1987). For forested wetlands in particular, additional requirements for diversity (including species that contribute hard mast) and density of live stems have also been specified in individual permits (e.g., Matthews and Endress 2008, Marler 2014).

Table 1. Original and updated requirements for forested wetland mitigation projects

Element	Original guidance and requirements[a]	Updated guidance and requirements[b, c]
Monitoring period	5 years[d, f]	10 years[d] (alternating year basis)
Mitigation ratio	As per individual permit	3:1 ratio for mature forested wetlands[d] 2:1 for sub-mature or other stand types[d]
Wetland criteria parameters	Minimum acres of wetland by cover type as per three wetland parameters	
Vegetation, general	More than 50% of dominant plant species are native and hydrophytic ("50/20 Rule") with specified Wetland Indicator Status (WIS)[a]	"50/20 Rule" with dominance test and WIS as per Revised Iowa Wetland Plant List (2012)[c, d, e]; Chapter 2[c]
Soils	Hydric soils or conditions that support hydric soil formation[a]	Hydric soils (as per local Soil Survey hydric soils list [c]) or conditions that support hydric soil formation (Chapter 3[c])
Hydrology	Inundated or saturated continuously for 5% or more of the growing season in majority of monitoring years[a]	As per Wetland Hydrology Indicators list and growing season criteria (Chapter 4[c])
Additional criteria specific to forested wetland mitigation		
Species diversity	As per individual permit[f]	5 native species minimum, each no more than 20% of total (shrubs minimal)[d]
Hard mast requirement	As per individual permit[f]	3-5 species, unless out of natural range[d]
Density of live stems	As per individual permit[f]	100 trees per acre (with live growth at or above 5 feet) in 10th year[d]

After Pociask and Matthews (2013)
(a) US ACE (1987), (b) Federal Register (2008), (c) US ACE (2010), (d) S. Marler (2014) and M. Carlson (2014, personal communication), (e) Lichvar (2012), (f) Matthews and Endress (2008)

Reports of success in mitigation wetland establishment based on the original guidance have been mixed (reviewed by Kihslinger 2008). Overall, some degree of success has been reported for meeting soil, hydrology/hydroperiod, and vegetative cover criteria (Wilson and Mitsch 1996). However, criteria for vegetative composition and structure, or area of the replacement wetlands, are less often met (e.g., Wilson and Mitsch 1996, Matthews and Endress 2008). For example, based on assessments conducted in Illinois, Matthews and Endress (2008) concluded that of 76 mitigation sites examined, 23 achieved all goals specified in their respective permits, 45 sites met some but not all goals, and 8 sites did not achieve any goals. In their study, although many individual sites did not create the minimum area of wetland required, at a project scale a realized mitigation ratio of 1.1:1 was noted, representing a small net gain of wetland area overall (Matthews and Endress 2008). In Iowa, a previous assessment of 24 mitigation sites by VanDeWalle et al. (2007) similarly indicated that while just 58% of individual wetland mitigation sites examined were successful in terms of wetland area established, taken together the wetland area for all projects exceeded the total required. They also examined a subset of 12 wetlands that were deemed successful by standards that included vegetative composition measures (species richness and abundance) and several measures of water quality.

Revised Regulatory Guidance

Based on evidence that accrued from a growing number of evaluations of mitigation wetland area and function, concern about the failure of mitigation programs to meet the overarching goal of "no net loss" of wetland area and function led to examination of the original regulatory guidance and the conclusions that (1) permits issued did not define performance goals clearly enough and (2) the permit program lacked adequate mechanisms to ensure compliance (NRC 2001). Subsequent studies to evaluate mitigation efforts continued to indicate insufficient area of restored/created wetlands and insufficient creation of in-kind (same wetland type) replacements (e.g., Brown and Veneman 2001, Morgan and Roberts 2003). This has been especially true of forested wetland mitigation projects, for which regulatory compliance has historically been low in many areas of the country (0% to 30%), often because of substantial mortality of planted trees (e.g., as reported by Brown and Veneman [2001] in Massachusetts, Robb [2001] in Indiana, Balzano et al. [2002] in New Jersey, Cole and Shafer [2002] in Pennsylvania, Morgan and Roberts [2003] in Tennessee, Matthews and Endress [2008] in Illinois, and, more generally, Kihslinger [2008]).

Therefore, current regulations provide more explicit descriptions for wetland vegetation, soils, and hydrology, as well as performance goals and requirements for monitoring and reporting, as stipulated in the new 2008 Final Rule on Compensatory Mitigation for Losses of Aquatic Resources (33 CFR Parts 325 and 332, 40 CFR Part 230, 2008). The requirements included in the 2008 Final Rule document are still interpreted on a case-by-case basis; for forested wetlands in particular, these now include a longer monitoring period, higher mitigation-to-impact-area ratio, new minimum species diversity and stem density requirements, and number of mast-producing species, in addition to earlier stipulated requirements for other wetland parameters as generally described above and as per the Midwest Regional Supplement (US ACE 2010) and Iowa Wetland Plants list (Lichvar 2012). (Table 1 includes a comparison of the previous and current guidance and requirements.)

Ecohydrological Characteristics and Functions within Natural Forested Wetlands

General Characteristics

In addition to the general ecohydrological roles of wetlands described earlier, natural forested wetlands have features that support landscape functions based on their vegetation, soils, and hydrology. Because preserved natural systems can serve as reference points or benchmarks to gauge restoration success, those characteristics for extant forested wetlands are briefly described in the paragraphs that follow.

Vegetation. Depending on the setting, natural forested wetlands may contain a variety of understory and overstory plants with specific anatomical, morphological, and/or physiological features that allow them to survive and grow on sites that are at least periodically saturated or submerged by water. Morphological adaptations include features such as adventitious roots, lenticels, multiple-stemmed habits, buttressed root collars, and shallow root systems; physiological features that allow roots to persist in anoxic conditions include the ability to produce specific enzymes and organic acids and/or oxidize the rhizosphere, as well as alterations of metabolic products (e.g., malate rather than ethanol) and metabolic rates. Many wetland species also have specific reproductive characteristics that allow establishment and growth in saturated soil conditions, such as long seed viability, germination under low oxygen conditions, and/or seedlings that tolerate flooding (US ACE 1987).

Inundation has differential effects depending on whether it occurs during the vegetation establishment period and/or later in plant community development. In terms of plant community composition, Pociask and Matthews (2013) noted that flood exposure affected species richness and the proportion of perennial species. They documented strong correlations for loss of species richness and perennial species with increases in magnitude of flooding in the same year and for gain of non-hydrophytic annual and non-native species in the year following flooding.

For woody plants in particular, inundation has both positive (provides abundant moisture) and negative (causes anaerobic conditions) effects on establishment and growth, which vary by species and have differential effects across their potentially long lifespans (McCurry et al. 2010, Gee et al. 2014). This has been referred to as the “subsidy-stress” model (Odum et al. 1979), wherein brief inundations provide an influx of moisture and nutrients that can enhance growth, whereas prolonged flooding creates stress that can diminish growth or lead to mortality. For example, Anderson and Mitsch (2008a) and Rodriguez-Gonzalez et al. (2010) reported changes in tree growth related to frequency and duration of flooding that were consistent with the subsidy-stress model and indicated that best growth was typical of systems with brief flooding events that did not cause extended anaerobic soil conditions. Seasonal timing of inundation is also important; flooding during the previous dormant season can actually increase tree growth in the following growing season (Mitsch and Rust 1984).

In bottomland riparian systems, tree/stand establishment and subsequent growth are subject to the interacting influences of flood frequency, intensity, depth, and duration coupled with velocity of water movement. Species respond differently to these characteristics of flood events, although flooding duration has been found to explain more variation in damage to and mortality of trees, followed by the depth and lastly the velocity of floodwaters (Kramer et al. 2008). These systems are also characterized by dynamic interactions among the species themselves. For example, fewer species and a smaller number of individual plants persist in the most active flood zones with frequent, rapid water movement, so there is less vegetation in the ground layer, fewer tree stems (both young and old), and increased resource availability for plants that do persist (Gee et al. 2014). These areas typically have a relatively open understory, and establishment and recruitment of new overstory trees may occur only sporadically during consecutive dry years, since seedlings are very vulnerable to flood damage. Within these forested communities, natural flooding regimes may also favor establishment of more flood-tolerant but less shade-tolerant species, or more heavy-seeded species, such as oaks, compared to light-seeded species, such as elms, maples, and hackberry (Battaglia et al. 2000, Bledsoe and Shear 2000, Ouchley et al. 2000, Gee et al. 2014).

Soils. The availability of oxygen in the root zone of wetland soils is an important determinant of the survival of trees (Pennington and Walters 2006). As previously mentioned, wetland (e.g., hydric) soils have specific features related to flooding and saturation that at least periodically cause anaerobic soil conditions. Oxygen availability can be measured in terms of redox potential, or Eh. Values of soil Eh above +300 mv generally indicate aerobic conditions, while values between +100 and +300 mv indicate moderately anaerobic conditions, and values less than +100 mv indicate strongly anaerobic conditions (Pennington and Walters 2006). Certain soil morphological traits related to their redox potential are recognizable in extant wetlands and even in soils that have been artificially drained for a number of years, such as olive and/or gray soil matrix colors (chromas ≤ 2), mottled soil color patterns of brown/orange "splotches" in a gray or gray-brown soil matrix, and accumulations of organic matter in surface soil horizons (possibly even to the degree that organic soils develop) (US ACE 1987, Parker 1988, Richardson and Vepraskas 2001, Schaetzl and Thompson 2015). There can be a wide degree of variation in other properties of wetland soils, such as soil texture, pH, and nutrient availability, that are the result of soil formation processes and can strongly influence the duration of saturation and the types of vegetation/species of trees that may naturally grow on a given site (e.g., Bledsoe and Shear 2000). These soil characteristics are also closely tied to topographical features and related site hydrology.

Microtopographic Effects within Forested Wetlands

Based on studies of natural bottomland hardwood forests, practitioners and researchers have documented the presence and the effects of very small changes in topography/surface elevation on seedling success and species distribution. Causes of natural small-scale topographic variation in these systems include soil erosion and sediment deposition, tree fall (creating pits and mounds), and animal activity (e.g., Bruland and Richardson 2005). Among the many tree species that occur in wetland areas, most have relatively narrow ecological amplitude in terms of soil texture, soil moisture content, and soil drainage characteristics (Merritt 1980, Hodges 1997, Wray 2004a, Herring 2012). Differences in soil properties and the length of inundation associated with different flood regimes lead to these species distributions. In Iowa and the Upper Midwest generally, willows, silver maple, boxelder, cottonwood, and sparse understory plants are typical in frequently inundated areas with poorly drained (clay and clay loam) soils (Mitsch and Rust 1984, Thompson 1992, Wray 2004b, Randall and Herring 2012). In slightly elevated areas that are less frequently flooded and characterized by moderately drained to well-drained loam and sandy loam soils, swamp white oak, sycamore, bur oak, Kentucky coffeetree, hackberry, and black walnut occur, often with more species of associated woody trees and shrubs, as well as herbaceous plants in the understory (Thompson 1992, Randall and Herring 2012).

Natural Temporal Changes in Forest Communities

The characteristic distribution patterns of species within bottomland forest communities are also influenced by the outcomes of intra- and inter-species competition and successional processes, resulting in predictable changes in composition over time. For example, Hodges (1997) described species occurrence and typical successional patterns for bottomland hardwoods in the lower Mississippi River valley. Different patterns emerged for large-river floodplains with fine-textured (poorly drained) soils (succeeding from black willow to boxelder/sugarberry and eventually oak-hickory stands) compared to those with coarser-textured soils (succeeding from cottonwood to sycamore/sweetgum and eventually to oak/hickory) and smaller floodplain areas (succeeding from river birch to sycamore/sweetgum). Hodges (1997) specifically noted that small differences in elevation and deposition regimes (considered allogenic influences) are more important in bottomland systems than in uplands, affecting both starting and ending points of successional sequences. In Iowa bottomland forests, successional change is often characterized by transitions from shade-intolerant early colonizers such as willow, boxelder, birch, and cottonwood to intermediate stages with sycamore and species of oak to later stages with dominance by shade-tolerant species such as hard maples and basswood (Wray 2004a, Herring 2012). In addition to these naturally driven changes over time, the present-day composition of forests may also be strongly influenced by past human management activities that may have altered stand characteristics (e.g., Ouchley et al. 2000).

Temporal Changes in Hydrology

Forest plant community changes that are driven by hydrological alterations over time can also occur. These changes are in response to either small-scale, short-term events, such as periodic seasonal floods and droughts, or longer-term landscape-scale hydrological changes caused by more extensive flooding events or increased/decreased depth to water tables (Bledsoe and Shear 2000, Kramer et al. 2008). Localized, short-term changes affect seedling establishment for woody plants in bottomland systems and can lead to differences in forest stands over time (Bledsoe and Shear 2000, Spencer et al. 2001, McCurry et al. 2010). Gee et al. (2014) indicated that light-seeded species are particularly vulnerable to flooding and can be eliminated by small, short-duration inundation events. Temporal changes in hydrology may occur laterally across a short distance (between a channel and floodplain), vertically (surface to subsurface), or longitudinally (along a floodplain upstream to downstream), creating a variety of gradients along which different species/species groups typically occur (Bledsoe and Shear 2000).

Natural flood regimes in riparian forests have also been altered by human activities such as levee building (and breaching), often causing long-term and large-scale hydrological modifications that affect stand development processes and the trajectory of successional change (Gee et al. 2014). Gee et al. (2014) described the long-term effect of decreased flood disturbance in the Mississippi River valley on species growth and recruitment over a 90-year period following levee construction. They found a shift in species dominance from overcup oak to sugarberry related primarily to exclusion of flooding disturbance. This change was thought to be related to greater opportunity for sugarberry seedling establishment in the absence of flooding, and changes in tree growth rates for both species (sugarberry increasing and oak decreasing) over time related to the change in site hydrology.

Overall, although extant natural bottomland forest systems provide an important reference point, it is also important to note that both historic forest management activities and large-scale and long-term changes in water table levels (due to regional-scale agricultural drainage systems and/or global climatic shifts) mean that these systems may either be “relict” or highly modified stands that should be used with caution for setting restoration goals (Richter et al. 1996, Ouchley et al. 2000, Stanturf et al. 2001).

Traditional Approaches to Forested Wetland Restoration

A variety of agencies and entities have been involved in relatively large-scale experimental efforts to restore/establish bottomland hardwoods, particularly in the riparian wetlands of the south; prominent among these were forestry researchers working in the Lower Mississippi River Alluvial Valley (LMAV) and ecology researchers associated with the Savannah River Ecology Laboratory (SREL). Restoration projects associated with both efforts were initiated in the early 1990s and involved investigations of species selection and the silvicultural aspects of establishment (Barton et al. 2000, McLeod 2000, McLeod et al. 2000, Lockhart et al. 2003, Patterson and Adams 2003). Given the scale of these projects and careful documentation of site preparation, artificial regeneration (species, stock type, and planting methods), and post-planting treatments, the results obtained in selected experiments provide some valuable lessons learned that are described, together with information from less extensive work in the Upper Midwest, in the paragraphs that follow.

Site Preparation

In their experimental bottomland hardwood planting established in the LMAV in 1993, Lockhart et al. (2003) tested several site preparation tillage methods. Based on stocking levels six years after planting seeds and bare root seedlings, these researchers concluded that more intensive methods (e.g., double disking prior to sowing acorns or planting trees) resulted in greater densities of established oak seedlings. In an experimental planting conducted by SREL researchers (also established in 1993), site preparation methods included either mowing or spraying entire plots compared with treatment of strips within plots where seedlings were planted (McLeod et al. 2000). In this experiment, tree survival and growth after two years were less affected by weed control than by planting site elevation and flood tolerance of the different species tested. Other workers in the LMAV also tested the effects of bedding (mounding soil in parallel ridges) in fine-textured soils to increase soil aeration and reduce waterlogging before planting seedlings (Patterson and Adams 2003). They observed better survival rates (up to 95% for direct-seeded oak versus 47% on non-bedded areas) using bedding on very poorly drained soils and recommended the use of bedding under that set of conditions.

In the Upper Midwest, competition from grass and weeds is a well-documented problem in establishing tree seedlings, and the amount of effort required for site preparation depends on soil and vegetation characteristics of the site. For areas with coarse-textured soils that do not have a dense weed or sod layer, little site preparation may be necessary, although for other sites with finer-textured soils and more established weed populations, initiating weed control before planting seed or seedlings has been recommended (Wray 2004a). Generally, this has been accomplished through using mowing or tillage (mechanical control) or herbicide (chemical control) applications, or a combination of the two (Thompson 1992, Wray 2004a).

Tree Establishment

Species and microsite. Several studies conducted by workers in the LMAV and at SREL included examination of relative performance of different species, stock types, and planting methods. Many of these studies included a number of species of oaks (nuttall, willow, cherrybark, and overcup, together representing close to 80% of planted trees in the LMAV [King and Keeland 1999]), water tupelo, green ash, bald cypress, hackberry, and sweetgum. Although several of these species do not grow in Iowa, these studies' findings are informative with respect to differential performance of species related to the topographic placement of seedlings and their relative tolerance of flooding.

For example, Lockhart et al. (2003) reported greatest success with nuttall oak compared to water oak and willow oak on flood-prone sites with fine-textured and poorly drained soils; they surmised that the better performance of nuttall oak was due to larger acorn size and delayed germination on flooded sites. Patterson and Adams (2003) similarly found better growth of nuttall oak compared to green ash on poorly drained, frequently inundated fine-textured soils. In the SREL studies, McLeod (2000) reported on a number of species (24), of which only 8 exhibited greater than 50% survival in any experiment (out of 12 combinations of stock type, species, and weed control treatments). Based on the full set of experiments, McLeod (2000) concluded that bald cypress and water tupelo performed best across a range of soil moisture/flooding conditions and were suitable for lower elevation and more frequently flooded microsites and that green ash and water hickory, as well as overcup, swamp chestnut, willow, and nuttall oaks, were characterized by adequate survival at slightly higher microtopographic locations.

Stock type and planting methods. Different kinds of planting stock have been evaluated for their suitability for use in forested wetland restoration, with the recognition that there is a trade-off in terms of cost of establishment and time to canopy closure. Costs are relatively low for direct seeding methods and bare root seedling stock and increase as planting stock age and size increases and time to canopy closure decreases (as summarized by Stanturf et al. 2001). A survey of agencies involved in early efforts toward bottomland reforestation with large-scale planting projects in the LMAV by King and Keeland (1999) indicated that use of bare root seedlings was most common, followed by direct seeding methods. At that time, out of 27 survey respondents, only 1 agency reported using containerized seedlings.

Based on determinations of stem density six years after planting Lockhart et al. (2003) working in the LMAV reported acceptable survival rates and no significant differences between direct-seeded (planted with either a seed drill or via broadcast seeding) and bare root (planted either with a hand auger or tree planting machine) seedling stock. Researchers at SREL also reported satisfaction with survival of hand-planted (dibble bar) bare root seedlings, especially so if initial seedling height exceeded water depth during occasional flooding (Barton et al. 2000, McLeod et al. 2000). Based on experimental plantings of bare root seedlings (northern red oak, white oak, and black walnut) in six states across the Upper Midwest, Schultz and Thompson (1996) concluded that planting method had significant effects only on marginal sites, where hand-planting in holes excavated with an auger resulted in greater height and diameter growth after four years when compared to machine planting. Although these experimental plantings were conducted on upland sites rather than in wetlands, these researchers also concluded that seedling characteristics (in particular larger seedling root systems) were more important for success than planting method, a finding that is likely generalizable to a range of site types.

Tree shelters. Tree shelters (corrugated plastic tubes, available in different heights, which are placed around individual planted seedlings) were introduced in the UK in the early 1980s primarily to protect trees from animal damage (Tuley 1983). Researchers in the US began testing them in field plantings in the late 1980s and generally reported increased survival (due to protection from animal browsing) and more rapid height growth (due to the altered growing environment within the shelter) compared to non-sheltered trees (Lantagne et al. 1990, Costello et al. 1991, Schultz and Thompson 1996, Ponder 2000). Although performance varied somewhat among species, in most cases height growth of sheltered trees increased but diameter growth did not, leading some researchers to express concern about the effects of the potential imbalance that resulted (for northern red oak and eastern black walnut, Schultz and Thompson [1996]). More recent research resulted in recommendations for use of higher light-transmitting and ventilated shelters that both promoted growth and a better balance between height and diameter increments (Sharew and Hairston-Strang 2005).

In some of the previously described experimental restoration projects in the southeastern US, researchers also examined the effectiveness of tree shelters and determined that their use led to increased survival and higher growth rates of planted trees. For example, in one of the previously described SREL experiments, half of the seedlings were protected with 1.5 m tall tree shelters (Conner et al. 2000). Five years after planting, they reported survival rates of 67% to 100% for different species of seedlings in shelters compared to 2% to 90% for seedlings without shelters. These investigators also documented more rapid early height growth of seedlings in tree shelters, although differences in total tree height at the end of five years were no longer significant. In work conducted in Maryland, Sweeney and Czapka (2004) also reported better survival and growth of seedlings in shelters compared to those without, finding on average (across five species) 39% greater survival and 300% better growth five years after planting. Most researchers who evaluated tree shelters noted the significant advantage provided by protection against herbivory, especially on sites with documented populations of deer, rabbits, and beaver. Tree shelters can also provide some protection to newly planted trees from both mower and herbicide damage during post-planting maintenance activities.

Post-planting site maintenance. In some of the previously mentioned experimental studies, post-planting weed control was not as important to seedling success as the use of tree shelters (e.g., McLeod 2000, Sweeney and Czapka 2004). However, in areas where soils and climatic conditions provide an environment more conducive to weed growth, maintenance of tree planting projects usually requires some form of ongoing weed control during the establishment period. In particular, intensive competition from weed growth has been identified as an important post-planting limitation for establishment and growth of woody plants in the Upper Midwest. Wray (2004b) indicated that in Iowa lack of effective weed control was a primary reason for failure of tree plantings. Both mowing and herbicide application have been used for varying durations following tree planting. Generally, mowing has been found to be less effective (and can potentially cause more damage to seedlings) than careful herbicide application (Wray 2004b). In their experimental planting, Sweeney and Czapka (2004) found that weed control was more effective in combination with use of tree shelters and that tree shelters were the more important of the two. For trees in shelters, they rated weed control with herbicides as most effective, followed by use of ground mat covers, with mowing rated least effective.

Practices Recommended in New Approaches to Forested Wetland Restoration

Although many of the experimental plantings described above helped to identify common problems as well as effective practices, overall widespread difficulties in forested wetland establishment have persisted, primarily related to lack of success in establishing trees. Problems often cited include slow growth or high mortality of newly planted trees due to competition with weeds, excess water and/or drought, and herbivory (King and Keeland 1999, Stanturf et al. 2001, Sweeney and Czapka 2004, Pociask and Matthews 2013, Marler 2014). These documented problems have led to experimentation with new approaches that may contribute to greater success for forested wetland mitigation. On the basis of findings from earlier approaches, as well as results from more recent restoration work, the general consensus is that there is not a simple one-size-fits-all approach and that site-specific conditions require site-specific and nuanced restoration designs (e.g., Sweeney and Czapka 2004, Lockhart et al. 2008, Johnson et al. 2012).

Site Selection and Site Preparation

Recent reports indicate many previous wetland mitigation sites have been chosen with a strong emphasis on ensuring that regulatory requirements for hydrological standards are met (e.g., Matthews and Endress 2008, Johnson et al. 2012, Pociask and Matthews 2013). These workers contend that a common result is mitigation sites with greater depth of water or greater duration of flooding than natural wetlands in similar geomorphic settings (Matthews and Endress 2008). This can cause increased mortality of planted vegetation (especially trees), delays in moving along expected trajectories of vegetative succession, and thus slow progress toward achieving mitigation performance standards (Pociask and Matthews 2013). This is particularly true of riparian systems at lower-elevation locations within a watershed, and the implication is that selection of sites at slightly higher elevations in a watershed (along tributary streams) with less intense flood exposure might contribute to greater success in attaining mitigation goals related to species richness and the proportion of perennial species that persist over time. A possible strategy for site selection, particularly for riparian wetland restoration, would be to examine updated soil survey information to find suitable areas along smaller streams in higher watershed positions that contain hydric soils (in Iowa, as per the updated Iowa Soil Properties and Interpretations Database [ISPAID 2015]).

A second strategy that has recently been studied in constructed wetland systems is preservation or creation of artificial microtopography as a part of site preparation for restoration. Although it is an important feature of natural forested wetlands, the creation of microsite-scale variation in topography during restoration has not been common (Simmons et al. 2011). Similar in concept to construction of bedding on finely textured soils in the LMAV to increase woody plant survival (the previously described work of Patterson and Adams [2003]), creation of "mound and pool" topography (considered more natural than bedding in rows) has recently been investigated in both closed depressions and riparian system wetlands as a means to create spatial heterogeneity of aerobic/anaerobic soil zones and enhance establishment success of a broad variety of plants (Barry et al. 1996, Bruland and Richardson 2005, Simmons et al. 2011). Researchers have described addition of soil materials to construct mounds that were between 60 cm and 1 m above the elevation of pool bottoms, with very gradual slopes between the mounds and pools. In their experimental study, Simmons et al. (2011) documented uniform survival of early-successional species across the topographic gradient that was created and differential survival of late-successional species (different species of oaks), which was greater on constructed mounds. They also noted, however, significant declines in survival for all species in the second year (with significant flooding). Notwithstanding, these researchers suggested that such microtopography should accommodate a more diverse species assemblage. Depending on the characteristics of the site, if excavation is done to create pools, it may also be possible to remove recent anthropogenic sediments (from human-caused acceleration of erosion) above the previous soil surface, with the added benefit of exposing the former seed bank (Smith et al. 2016). Based on the relatively small set of experimental studies conducted thus far, this approach has particular promise for areas with relatively poorly drained fine-textured soils that limit woody plant establishment, but there are indications that it is not as effective on sites with more well-drained coarse-textured soils (e.g., Dey et al. 2004).

Species and Placement on Site

With regard to woody species included in forested wetland projects, several recent studies have also proposed use of mixed-species stands and carefully designed approaches to placement of particular species and stock types within them (Bruland and Richardson 2005, Pennington and Walters 2006, Lockhart et al. 2008, McCurry et al. 2010, Simmons et al. 2011). All of these researchers have recommended that mixed-species tree plantings be used and carefully stratified according to site microtopography and the relative flood tolerance of the species to be introduced. In addition, based on their long-term experience and variable success with afforestation efforts in the LMAV, Lockhart et al. (2008) proposed that woody plant regeneration on such sites include single-cohort mixed-species stands that include both early- and late-successional species. They contend that such species mixes will contribute to more rapid initiation of natural succession processes.

Based on their examination of natural vegetation zonation in created wetlands with microptopographic variation, Pennington and Walters (2006) suggested that greater success could be achieved by proportionally focusing greater tree planting effort on woody species with intermediate flooding tolerance and placing them in transitional zones that are typically only periodically inundated and have intermediate soil redox potentials. They noted dramatically improved survival rates for four species after five years (92% for green ash and swamp white oak, 81% for silver maple, 90% for pin oak) when planted in transitional zones compared to trees in more continuously inundated zones (only 12% survival for green ash and silver maple, 14 % for swamp white oak, and exact rate unknown but also low for pin oak). In their analysis, greater photosynthetic production was characteristic of trees planted in the transitional zones, leading to a more positive carbon balance and thus greater survival and growth. Based on close correlations between zonation of herbaceous plant assemblages that rapidly established on restoration sites and soil redox potentials, they proposed that practitioners could grade sites to create appropriate microtopography, allow them to revegetate naturally for one growing season, and subsequently use naturally occurring vegetative zonation to identify transitional areas as foci for tree planting.

Stock Type, Planting Methods, and Post-Planting Care

Several of the previously described traditional afforestation projects involved the use of direct seeding or bare root seedlings with reasonable success (Schultz and Thompson 1996, Lockhart et al. 2003, Sweeney and Czapka 2004). In their efforts to evaluate bare root seedling quality, Schultz and Thompson (1996, 1997) found that seedling performance (survival and growth) for upland hardwoods was greater for trees with larger root systems (based on numbers of first-order lateral roots). Typically, these seedlings were also characterized by larger diameters and taller stems. Based on these and other early studies of differential seedling performance based on seedling morphology, relatively new stock production methods have focused on root system development. One such method is the Root Production Method used to develop what are now known as RPM seedlings (Lovelace 2002). The RPM system uses air root pruning to generate a large, fibrous root system and relies on use of carefully selected seed, movement of plants through a succession of containers as they grow, and careful culling processes to eliminate inferior stock as the seedlings develop (Lovelace 2002, Dey et al. 2004). Both root systems and stems of RPM seedling stock are larger than those of typical bare root planting stock.

On the assumption that transplant success in bottomland areas could be improved by planting relatively large seedlings with large root systems, Dey et al. (2004) compared survival and growth of bare root versus RPM seedlings for pin oak and swamp white oak in Missouri. They reported significantly greater survival for RPM seedlings (94%) compared to bare root stock (which varied from 54% to 76% depending on species) and larger early diameter growth increments for RPM seedlings. In another study, Krekeler et al. (2006) compared direct-seeded, bare root, and RPM seedlings of pin oak and found that while all stock types had greater than 80% survival during the first year, the RPM planting stock was the largest and grew most rapidly. In a longer-term study, other researchers compared bare root and RPM stock of black oak, white oak, and swamp white oak seedlings and documented 5-year survival rates and 14-year growth patterns (Walter et al. 2013). These researchers documented greater survival of RPM seedlings (100%) compared to bare root seedlings (63% for black oak and 75% for white oak) but found no difference for swamp white oak seedlings (which had 100% survival for both stock types). They also noted greater growth for RPM black oak and white oak at 14 years after planting and generally recommended the use of RPM planting stock for establishment of late-successional species. Overall, although RPM seedlings are costly, consistent reports of better survival and growth in research settings indicate that careful placement in wetland designs and careful planting can result in a more rapid trajectory toward crown closure (evidence of success as per Stanturf et al. [2001]) and that this may justify the added cost.

Additional Considerations for Restoration of Forested Wetland Function

As suggested by a number of researchers, establishment of created/restored forested wetlands is made difficult by a number of challenges related to the ecological complexity (high degree of variability across both space and time) of the natural systems they are meant to replace. Temporal changes in the composition of naturally occurring forested wetlands occur on time scales of at least decades (if not much longer) and are outside the scope of regulatory timelines. Further, the influence of these changes on other system characteristics, for example productivity (and carbon capture), is just beginning to be understood in relation to cycles of inundation (e.g., Anderson and Mitsch 2008b). Another unknown is related to the effectiveness of forested wetland restoration in terms of reestablishing wetland biogeochemical functions, which appear to lag significantly behind those of hydrology and vegetation (D'Angelo et al. 2005, Theriot et al. 2013). Further, although many restoration sites are near existing bottomland hardwood stands that could contribute to natural regeneration in the surrounding landscape, designs that rely heavily on this form of regeneration as a significant component of woody revegetation are not common. Finally, strategic location of restored forested wetlands to connect existing large and small remnant patches at whole watershed/landscape scales has been proposed as a means to increase their functions at that level, but this suggestion has not been rigorously tested (e.g., Shaffer et al. 1992).

3. RESEARCH METHODOLOGY

The authors collaborated with Iowa DOT and Iowa Department of Natural Resources (DNR) personnel to select study sites. The team assessed 25 sites overall in central and southeastern Iowa. These included nine Iowa DOT (RPM and BR planting stock) wetland mitigation sites, five Iowa DNR (BR planting stock) forested riparian restoration sites, five reference forest sites, five natural regeneration sites, and one Iowa DOT experimental planting site (Goose Pond Natural Area).

Objective 2: Assessment of Forested Wetland Mitigation Projects

The team initially conducted an assessment of 24 sites located in central and southeast Iowa to evaluate performance of different tree species and seedling stock types in relation to each other and to natural regeneration and reference forests. Specifically, researchers compared survival rates for BR and RPM seedlings planted on 14 of those sites and evaluated species diversity and stem density on those sites as well as on 10 additional sites (natural regeneration and reference forests).

Study Site Selection and Characteristics

The research team collaborated with Iowa DOT and Iowa DNR personnel to select study sites. For evaluation of forested wetland mitigation projects, the research team chose 9 riparian-area plantings (sites that Iowa DOT personnel identified as representative of the range of ages, locations, and types of planting stock used) from a total of 44 such sites that were established at the initiation of this study. These sites included seven on which RPM seedlings had been planted and two on which BR stock were used. To balance the number of sites with RPM and BR stock, the researchers evaluated five bottomland forest restoration sites that were planted using BR stock under the guidance of Iowa DNR Forestry Section staff members. They also evaluated five sites with areas of natural regeneration and five sites with mature reference forests. All 24 sites are located in central and eastern Iowa (Figure 1). The team conducted tree assessments on all sites from May through October 2015 and collected soil samples from May through August 2016.

At each mitigation/restoration site, trees were planted by forestry contractors between 1995 and 2012 in areas that were formerly in row-crop production or grassland. Plantings were implemented according to practices and specifications established by the two agencies. Both agencies required standard site preparation activities that included use of initial weed control prior to planting and cover crops if desired. For Iowa DOT sites, RPM seedlings were hand-planted into auger-drilled holes at densities of 50 to 200 trees per acre. Iowa DOT plantings with BR seedlings were planted at densities of 80 to 300 trees per acre and were machine-planted in rows. For Iowa DNR sites, BR seedlings were machine-planted in rows to obtain densities of approximately 600 to 725 trees per acre. Both agencies required post-planting maintenance that included use of herbicide treatments and/or mowing for weed control for at least two to three years.

The research team used aerial photographs and site letting plan maps to verify locations, planting design, species, and stock type used, as well as to review site preparation and weed control specifications for each site. The team used soil survey maps (SSURGO 2017) to identify soil mapping units present, soil series, and their characteristics (USDA NRCS 2017). They used Iowa Geological Survey maps to characterize stream order (Strahler 1952) at each site (Iowa Geological Survey Bureau 2017). At each mitigation/restoration site, the researchers collected one 60 cm deep soil sample within each soil mapping unit using a hand-held 1.27 cm diameter soil probe to verify soil classification and hydric soil status. They also collected three surface soil samples for determination of bulk density within each mapping unit at each site using a 4 cm^3 cylindrical soil corer. Bulk density samples were weighed, dried at 65 °C for 24 hours, and reweighed to determine initial soil moisture content and dry weight. Soil bulk density ($g\ cm^{-3}$) was calculated as the dry weight of soil per unit volume of the core. The researchers composited and air-dried the three surface soil core samples for subsequent particle size analyses (conducted by Minnesota Valley Testing Laboratories, Inc., of New Ulm, MN) using the hydrometer method (Gee and Bauder 1986).

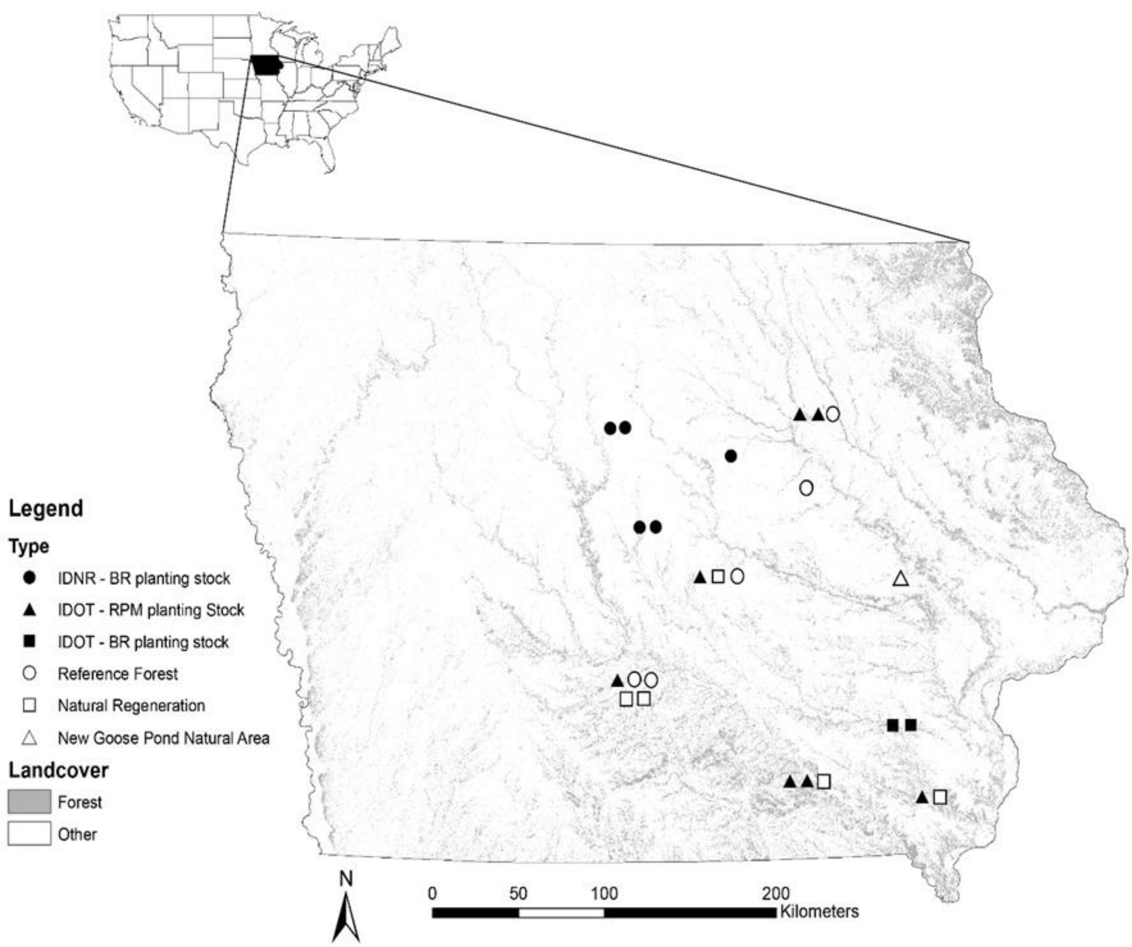

Figure 1. Approximate locations of nine Iowa DOT (RPM and BR planting stock) wetland mitigation sites, five Iowa DNR (BR planting stock) forested riparian restoration sites, five reference forest sites, five natural regeneration sites, and one Iowa DOT experimental planting site (Goose Pond Natural Area) in central and southeastern Iowa

Tree Seedling Sample Design and Measurements

The research team used aerial photographs to locate a transect line across the long axis of each site. Additional shorter transects oriented perpendicular to the main one were used for irregularly shaped sites. Fixed-radius plots (from 1/1000th to 1/5th acre, as per site type) were placed along each transect with a random start and systematic between-plot spacing. Plots were then located in the field based on their photo locations.

Each site contained from 3 to 5 (reference forests) or from 5 to 32 (all other site types) plots determined in proportion to the size of the planting project (an average of 1 plot per acre of site area [Husch et al. 2003]). Data collected for trees on each plot included number of trees and species present, and for mitigation and restoration sites included stock type, number of living and dead seedlings, and whether tree shelters were used. The team also identified a subset of seven focal species based on their representation in both growing stock types and presence across a range of sites for further analysis: silver maple (*Acer saccharinum L.*) river birch (*Betula nigra L.*), shellbark hickory (*Carya laciniosa Michx. F. Loud*), Kentucky coffeetree (Gymnocladus dioicus L. K. Koch), American sycamore (Platanus occidentalis L.), swamp white oak (*Quercus bicolor Willd.*), and pin oak (*Quercus palustris Muenchh.*). For these species, the researchers analyzed survival rates as well as seedling morphology (height, crown depth, root collar diameter, and height-to-diameter ratio).

Percent survival for each plot was calculated by dividing the numbers of living trees by all trees, both living and dead. For dead trees, species was determined using letting plans and/or by tags still present on seedlings. Survival rates for each site and for the seven focal species were calculated based on the sum of living trees divided by all trees for all plots at a site. Researchers calculated stem densities based on the total number of living stems per area of all sample plots.

For the seven focal species, the team measured the height of each living tree on each plot using a telescoping height pole. Crown depth was measured using the height pole to determine stem length from the top of the canopy to the lowest branch with leaves. Seedling root collar diameter was measured using digital calipers. Height-to-diameter ratio was calculated by dividing the total height of the seedling by the root collar diameter (after converting to the same unit of measure).

Data and Statistical Analyses

The research team used linear mixed-effects logistic regression models fit by restricted maximum likelihood to examine the effects of stock type (BR, RPM), presence or absence of tree shelters, soil characteristics (percent sand and clay, bulk density), and age of planting on seedling survival for all trees, and for the seven focal species in the initial study and the four focal species on the experimental site to examine the effects of stock type on seedling morphology (height, crown depth, root-collar diameter, and height-to-diameter ratio).

Analyses were conducted using the LMER function (Cook 2014) in the R statistical package (R Core Team 2013) in the initial study. For the seven focal species, data for height, crown depth, root collar diameter, and height-to-diameter ratio were log-transformed prior to statistical analysis to create linear data distributions and reduce heteroscedasticity prior to analyses. The team used the Laplace approximation to degrees of freedom for analyses of survival and the Satterthwaite approximation for all other variables examined for the seven focal species.

Objective 3: Assessment of an Experimental Forested Wetland Restoration Site (Goose Pond)

In 2015, the research team conducted an assessment of an additional experimental planting established by the Iowa DOT in 2014 at the Goose Pond site located in Linn County. This site was included to evaluate the performance of different tree species and three seedling stock types in relation to each other and to characteristics of natural regeneration and reference forests. Specifically, for this site, researchers examined survival rates for BR, RPM, and B&B seedlings and evaluated species diversity and stem density for them compared to the same set of five natural regeneration and reference forests already described.

Tree Seedling Sample Design and Measurements

The research team used aerial photographs to locate a transect line that crossed areas on the site with each stock type. Fixed-radius plots (1/10th acre) were placed along the transect with a random start and systematic between-plot spacing. Plots were then located in the field based on their photo locations and were balanced to ensure representation of the different stock types.

The research team assessed trees on 18 plots (6 for RPM stock, 3 for BR, and 9 for B&B stock). Data collected for trees on each plot included number of trees and species present, stock type, number of living and dead seedlings, and whether tree shelters were used. For this assessment, the researchers identified a subset of four focal species (silky dogwood [*Cornus obliqua*], black walnut, swamp white oak, and bur oak) based on their representation in at least two of the three stock types and measured their height, crown depth, and root collar diameter, as previously described. Percent survival for each plot and for each stock type was calculated by dividing the number of living trees by all trees, both living and dead. At this site, species were not identified for dead/missing trees. For the four focal species, the team measured the height, crown depth, and root collar diameter, as per the previous study.

Data and Statistical Analyses

The research team used linear mixed-effects logistic regression models fit by restricted maximum likelihood to examine effects of stock type on seedling morphology (height, crown depth, root-collar diameter, and height-to-diameter ratio) for the four focal species. These analyses were based on the Satterthwaite approximation to degrees of freedom, and statistical differences for the three stock types were determined using the Tukey method for comparing a family of three estimates. Because this analysis was performed for seedlings on a single site, it was not possible to include site factors that may have contributed to seedling performance in this analysis, as was possible for the previous study.

4. RESULTS AND CONCLUSIONS

Summary of Results

The research team conducted an assessment of 25 sites located in central and southeast Iowa to evaluate performance of different tree species and seedling stock types in relation to each other and to natural regeneration and reference forests (Heber 2017). Specifically, they compared survival rates for BR and RPM seedlings planted on 14 sites (Heber 2017) and evaluated species diversity and stem density on those sites and on 11 additional sites (experimental mitigation, natural regeneration, and reference forests). The authors determined overall survival rates of 91% for BR and 74% for RPM stock, with 5 to 8 species and average stem densities ranging from 67 to 317 trees per acre (Table 2). To benchmark these numbers relative to natural stands, they determined that regenerating stands had on average 4 species and a stem density of 410 trees per acre, and reference forests (mature stands) had an average of 6 species and 60 stems per acre (Table 2). Thus, relative to both regulatory targets and characteristics of natural stands, stem densities and survival rates for BR seedlings appear to be sufficient for stand establishment, whereas they are lower for RPM seedlings (and for B&B seedlings; however, this stock type was assessed on only one experimental site). Species diversity on mitigation and restoration sites met or exceeded regulatory requirements and was comparable to that of natural stands.

Table 2. Summary of site assessments conducted for empirical study of tree performance (2015-2016)

Site type	Number of sites	Number of plots	Total number of trees in plots	Average number of species/site	Average stem density (stems/acre)	Average survival (%)
Wetland mitigation - RPM	7	79	539	5	67	74%
Wetland mitigation - BR	2	21	333	6	158	88%
Bottomland restoration - BR	5	68	1,661	8	317	92%
Experimental mitigation	1					
RPM		6	53	7	88	100%
BR		3	43	7	143	100%
B&B		9	44	4	49	95%
Natural regeneration	5	57	234	4	410	NA
Reference forest	5	19	229	6	60	NA
Total	25	262	3,136	NA	NA	NA

Heber 2017

The results and conclusions of Objectives 2 and 3 are described in more detail below.

Objective 2: Assessment of Forested Wetland Mitigation Projects

Study Site Characteristics

Mitigation and restoration study sites (Table 3) were located in the floodplain areas of streams ranging from third- to seventh-order for the Iowa DOT sites (RPM and BR stock) and from third- to fifth-order for the Iowa DNR sites (BR stock) (Strahler 1952, Iowa Geological Survey Bureau 2017). Soils present on the sites ranged from relatively coarse sandy loam and loam soil textural classes (Dickinson, Hayfield, Saude, and Zenor soil series) to finer clay loam, silty clay loam, and silty clay textures (Colo, Kennebec, Ladoga, Wabash, and Zook series) (USDA NRCS 2017). Actual clay percent in surface soil samples collected at all study sites ranged from 5% to 45% (Table 3). Silty clay loam soils were more common on Iowa DOT sites, and sandy clay loam soils were more common on Iowa DNR sites. Average soil bulk density on Iowa DOT sites ranged from 1.00 to 1.86 g cc^{-1} and on Iowa DNR sites ranged from 1.16 to 1.70 g cc^{-1} (Table 3). Study areas were planted between 1995 and 2012; sample plots contained a range of 38 to 701 trees, and project sites varied in size from 1 to 63 acres (Table 4).

Table 3. Responsible agency and general site and soil characteristics for 14 riparian planting sites in central and southeastern Iowa

Agency/site	Stream order	Soil series	Surface soil bulk density (g cc^{-1})	Soil textural class	Surface soil clay (%)
Iowa Department of Transportation					
137 Bridge	7	Colo (overwash)	1.42	Silty clay loam	35
		Landes-Perk	1.38	Silty clay loam	25
		Nodaway-Landes	1.54	Silty clay loam	33
137 Mitigation	7	Colo	1.86	Sandy clay loam	13
		Colo-Ely	1.74	Sandy loam	20
Ainsworth	3	Ladoga (benches)	1.45	Silty clay loam	33
		Zook	1.28	Silty clay loam	38
		Lawson	1.25	Loam	30
Cox	3	Marsh (depressional)	1.47	Sandy loam	8
Jarvis	6	Klums-Perk-Nodaway	1.43	Silty clay loam	30
Painted Ridge	3	Coppock	1.16	Silty clay loam	30
		Aquents (ponded)	1.10	Silty clay loam	35
		Lawson	1.05	Silty clay loam	35
North River I	5	Kennebec	1.22	Silty clay loam	28
		Nodaway (channel)	1.00	Clay	30
		Zook	1.29	Silty clay loam	-
		Colo	1.39	Sandy clay loam	38
		Wabash	1.36	Silty clay	45
Wapsipinicon	3	Marsh (depressional)	1.68	Sandy loam	18
		Hayfield (moderate)	1.48	Sandy loam	15
		Hayfield (deep)	1.44	Loam	23
Woods Farm	5	Hanska	1.45	Sandy clay loam	15
		Saude	1.72	Sandy loam	13
		Lawson	1.45	Loam	40
		Waukee	1.48	Clay loam	25
Iowa Department of Natural Resources					
Cooper	4	Spillville	1.16	Sandy clay loam	33
Evans	3	Zook	1.36	Silty clay loam	38
		Nevin	1.35	Silty clay	38
		Bremer	1.31	Silty clay	35
		Colo	1.70	Sandy clay loam	33
		Colo (channel)	1.44	Silty clay	30
Gannon	5	Coland	1.27	Clay loam	45
		Spillville	1.30	Sandy clay loam	40
		Hanlon	1.52	Sandy clay loam	40
Jones	3	Spillville	1.50	Sandy clay loam	28
		Zenor-Storden	1.39	Sandy loam	20
		Terril	1.60	Clay loam	33
		Dickinson	1.59	Sandy loam	5
Sadler	3	Spillville-Coland	1.51	Sandy loam	10
		Spillville	1.37	Sandy clay loam	13

Table 4. Agency, site name and location, year planted, size of planting area, growing stock type, number of trees, and percent survival of 2,533 trees on 14 sites, and number of trees and stem densities for five natural regeneration and five reference forest sites

Agency/site	Location (County)	Year planted	Size of area (ac)	Stock type	Number of trees in plots	Survival (%)	Average stem density (stems/ac)
Iowa Department of Transportation							
137 Bridge Site	Wapello	2011	1	RPM	44	50	
137 Mitigation Site	Wapello	2005	4	RPM	38	42	
Ainsworth	Washington	1995	11	BR	173	80	
Cox	Bremer	2007	2	RPM	42	71	
Jarvis	Henry	2003	10	RPM	112	100	
Painted Ridge	Washington	1997	16	BR	160	96	
North River I	Warren	2011	63	RPM	168	73	
Wapsipinicon River	Bremer	2006	2	RPM	74	68	
Woods Farm	Marshall	2009	6	RPM	61	79	
Total RPM					539	74	67
Total BR					333	88	158
Iowa Department of Natural Resources							
Cooper	Story	2009	16	BR	150	95	
Evans	Grundy	2010	45	BR	560	83	
Gannon	Story	2010	7	BR	111	98	
Jones	Wright	2010	42	BR	701	95	
Sadler	Wright	2012	6	BR	139	100	
Total BR					1661	92	317
Natural regeneration				Total	234		410
Reference forests				Total	229		60

Generally, Iowa DOT plantings of RPM seedlings were located in higher order stream floodplains that likely have experienced greater depth and duration of flooding, whereas both agencies' plantings with BR stock were located near lower order streams. Although frequency of flooding in different sized watersheds may be similar, depth and duration of flooding is likely to be greater in higher order watersheds and could have negative effects on seedlings (e.g., Matthews and Pociask 2015). Prolonged flooding can lead to slower and diminished growth and increased seedling mortality (McCurry et al. 2010, Kabrick et al. 2012, Gee et al. 2014, Matthews and Pociask 2015). Finer-textured and poorly drained soils were also more common on Iowa DOT sites with RPM stock, possibly prolonging the effect of flooding by consistently retaining higher soil water content over longer time periods, which is also known to negatively affect seedling survival and growth (Pennington and Walters 2006, Matthews and Pociask 2015).

Overall Seedling Survival

The researchers assessed 2,533 trees representing 22 species on the 14 mitigation and restoration planting sites with an overall survival rate of 87% (see also Appendix A). This included 539 RPM seedlings with shelters on Iowa DOT sites and 1,994 BR seedlings without shelters on 2 Iowa DOT and 5 Iowa DNR sites (Table 4). Mean survival for RPM stock was 74%, with survival rates ranging from 42% to 100% on different sites. Mean survival for BR stock was 91%, with survival rates ranging from 80% to 100% on different sites. Percent survival of RPM stock was significantly lower than that of BR stock ($p = 0.021$, Table 5). Based on a binary logistic regression model, estimated log-odds of survival were 40 times lower for RPM growing stock. The presence of shelters had a marginal negative effect ($p = 0.071$). Based on the site factors the research team investigated in this overall analysis (soil percent sand, percent clay, and bulk density), they detected a negative relationship between tree survival and percent clay ($p = 0.019$, Table 5). Stem densities were highest on natural regeneration plots, followed by BR on Iowa DNR sites and BR and RPM seedlings on Iowa DOT sites.

The analysis indicated a much higher overall survival rate (87%) over a longer period of time than has been reported for similar wetland restoration/mitigation plantings (e.g., 54% to 76% survival depending on species as reported by Dey et al. [2004], 59% survival for experimental plantings reported by Roquemore et al. [2014], and 57% survival at the end of monitoring periods on forested wetland mitigation plantings in Illinois as reported by Matthews and Pociask [2015]).

Table 5. Linear mixed-effects logistical regression model for survival using Laplace approximations to degrees of freedom for 2,533 trees on 14 sites in central and southeastern Iowa

Fixed effects	Estimate	Standard error	df	z value	p value
Intercept	7.107	1.971	2524	3.606	< 0.001
Stock type (RPM)	-3.739	1.624	2524	-2.302	0.021
Shelter (true)	-1.494	0.827	2524	-1.805	0.071
Percent sand	-0.436	0.352	2524	-1.237	0.216
Percent clay	-1.027	0.437	2524	-2.351	0.019
Age	-0.063	0.175	2524	-0.361	0.718
Bulk density	0.269	0.219	2524	1.230	0.219

The analysis also indicated lower survival of RPM stock (74%) compared to BR plants (91%). This finding is contrary to a number of earlier reports of comparisons between RPM and BR planting stock. For example, Dey et al. (2004) reported significantly greater survival for RPM seedlings (94%) compared to bare root stock (which varied between 54% and 76% depending on species). In a longer-term study, Walter et al. (2013) noted 100% survival of RPM seedlings after 14 years compared to bare root stock (which varied from 63% to 75% depending on species). Although the researchers did not study the direct mechanism(s) that could lead to the difference they observed, it is possible that the RPM stock experienced greater transplant shock because of a number of factors that they could not control for in this observational study (e.g., exact provenance of the stock, timing of planting, and/or the contrast between nursery cultural conditions and post-planting site conditions). In newer plantings, the research team also noted improper planting and that a number of RPM trees had been girdled by tags that encircled the base of the seedlings' stems.

The marginal negative relationship between tree shelters and survival is also contrary to the results of a number of earlier studies in which shelters enhanced survival largely by protecting seedlings from herbivory (Lantagne et al. 1990, Costello et al. 1991, Schultz and Thompson 1996, Ponder 2000, Conner et al. 2000, Sweeney and Czapka 2004). However, there have also been reports indicating no effect of shelters on seedling survival (e.g., Andrews et al. 2010, Stuhlinger 2013, Drayer et al. 2017) and other evidence, as in this study, of negative effects from their use (e.g., Bardon et al. 1999). Because of the expense associated with tree shelters and their installation, the negative relationship of shelters with seedling survival in this study suggests that they should not be used to protect all seedlings in riparian/bottomland restoration plantings unless certain site-specific characteristics (e.g., very high herbivore pressure) warrant their use.

Stem densities on planted sites were lower than in naturally regenerating areas (which had an average density of 410 stems/acre and a range from 0 to 23,000) but greater than the average of 60 stems/acre (range from 15 to 145) for mature forests (Table 4). In particular, average stem densities for RPM stock (67 trees/acre) were below the regulatory requirement of 100 stems/acre after a 10-year establishment period, whereas BR planting stock exceeded that density on both the Iowa DOT (158 trees/acre) and Iowa DNR (317 trees/acre) sites the researchers assessed. Survival rates, especially those the research team documented for RPM stock, indicate that these seedlings will need to be "overplanted" at densities considerably above 100 stems per acre in order to maintain desired densities at 10 years and beyond the monitoring period. Average number of species present (five to six species) on mitigation sites was comparable to that observed in natural stands (four to six species).

Survival and Seedling Morphology for Seven Focal Species

The seven focal species represented in the plantings of both stock types include 1,050 of the seedlings described above, of which 234 were RPM seedlings and 816 were BR seedlings (Table 6). For this subset of seedlings, survival of RPM trees ranged from 65% to 100%, with an average of 86%, and survival of BR stock ranged from 94% to 100%, with an average survival rate of 99% (Table 6). For Kentucky coffeetree and pin oak, both stock types had 100% survival. Although on average stock type was not significantly associated with survival for this subset of the focal species, the presence of tree shelters was negatively associated with survival ($p = 0.036$, Table 7). Soil percent clay was also negatively related to survival ($p = 0.041$, Table 7). Laplace approximations were used for degrees of freedom for survival estimates (Table 7), and Satterthwaite approximations to degrees of freedom were used for t tests for all other parameters.

The researchers observed considerable variation in survival among different species for the RPM stock. Previous analyses of seedling survival in forested wetland plantings also indicated substantial variation among species (e.g., Costello et al. 1991, Lockhart et al. 2003, Andrews et al. 2010) and among combinations of species and stock types (McLeod et al. 2000, Roquemore et al. 2014). Such differences have been attributed to microsite variation with respect to duration of flooding and soil saturation, and differential ability among species to tolerate these conditions (Barton et al. 2000, McLeod et al. 2000, Pennington and Walters 2006, Simmons et al. 2011), which appears to be exacerbated for the RPM seedlings.

Table 6. Percent survival for a subset of 1,050 trees of the most common species represented on both Iowa DOT and Iowa DNR planting projects in central and southeastern Iowa

Species	Number of Iowa DOT sites with species present	Number of seedlings (RPM)	Percent survival on Iowa DOT sites	Number of Iowa DNR sites with species present	Number of seedlings (BR)	Percent survival on Iowa DNR sites
Acer saccharinum	5	43	78	5	311	100
Betula nigra	5	35	88	1	33	100
Carya lacinosa	3	33	79	2	20	100
Gymnocladus dioicus	1	11	100	1	43	100
Platanus occidentalis	3	27	65	4	59	94
Quercus bicolor	8	78	98	4	249	100
Quercus palustris	3	7	100	2	101	100
Total		234	85		816	99

Table 7. Linear mixed-effects logistical regression models for survival, height, crown depth, diameter, and height-to-diameter ratio for 1,050 trees of 7 focal species on 14 sites in central and southeastern Iowa

Variable	Estimate	Standard error	df	z value	p value
Survival					
Intercept	5.254	1.500	1041	3.527	< 0.001
Stock type (RPM)	-1.933	1.951	1041	-0.990	0.322
Shelter (true)	-2.562	1.221	1041	-2.098	0.036
Percent sand	-0.466	0.435	1041	-1.070	0.285
Percent clay	-0.948	0.464	1041	-2.043	0.041
Bulk density	0.065	0.187	1041	0.347	0.729
Age	0.122	0.282	1041	0.433	0.665
Variable	**Estimate**	**Standard error**	**df**	**t value**	**p value**
Height					
Intercept	-0.122	0.284	15.8	-0.428	0.674
Stock type (RPM)	-0.656	0.219	12.8	-2.995	0.010
Shelter (true)	0.168	0.093	1051.3	1.800	0.072
Percent sand	-0.199	0.054	427.7	-3.675	< 0.001
Percent clay	0.220	0.117	974.7	1.876	0.061
Bulk density	-0.061	0.033	819.6	-1.820	0.069
Age	0.122	0.021	11.7	5.927	< 0.001
Crown depth					
Intercept	-0.370	0.320	15.8	-1.158	0.264
Stock type (RPM)	-0.743	0.245	12.3	-3.037	0.010
Shelter (true)	0.034	0.095	1112.9	0.353	0.724
Percent sand	-0.196	0.056	539	-3.524	< 0.001
Percent clay	0.211	0.119	1058.6	1.768	0.077
Bulk density	0.086	0.034	923.2	-2.512	0.012
Age	0.119	0.023	11.5	5.151	< 0.001
Diameter					
Intercept	0.196	0.414	13.20	0.473	0.644
Stock RPM	-0.904	0.338	11.3	-2.677	0.021
Shelter true	0.080	0.101	1206.9	0.789	0.430
Percent sand	-0.311	0.060	866	-5.146	< 0.001
Percent clay	0.465	0.128	1191.3	3.646	< 0.001
Bulk density	-0.051	0.037	1125.7	-1.387	0.166
Age	0.147	0.032	10.9	4.550	0.001
Height: diameter ratio					
Intercept	4.292e+00	2.188e-01	1.020e+01	19.6	< 0.001
Stock RPM	2.429e-01	1.909e-01	1.110e+01	1.272	0.229
Shelter true	9.425e-02	6.333-e02	1.180e+03	1.488	0.137
Percent sand	9.820e-02	3.755e-02	7.310e-02	2.615	0.009
Percent clay	-2.261e-01	7.955e-02	1.152e+03	-2.842	0.005
Bulk density	-6.174e-03	2.288e-02	1.060e+03	-0.270	0.787
Age	-2.441e-02	1.817e-02	1.050e+01	-1.343	0.207

Height

For this subset of trees, mean height varied by stock type and species, for RPM stock ranging from a low of 0.8 m for pin oak to a high of 3.0 m for American sycamore, and for BR stock ranging from 0.6 m for pin oak to 4.5 m for American sycamore (Figure 2). Boxes represent the middle 50%, lines within the boxes represent means, and whiskers are upper and lower limits of measured tree heights (Figure 2). Height of the RPM trees was less than that of the BR trees ($p = 0.010$, Table 7). Tree height was negatively related to soil percent sand ($p < 0.001$) and positively related to tree age ($p < 0.001$). Percent clay in soil ($p = 0.061$) and presence of tree shelters ($p = 0.072$) both had a marginal positive relationship to tree height, while soil bulk density had a marginal negative relationship ($p = 0.069$, Table 7).

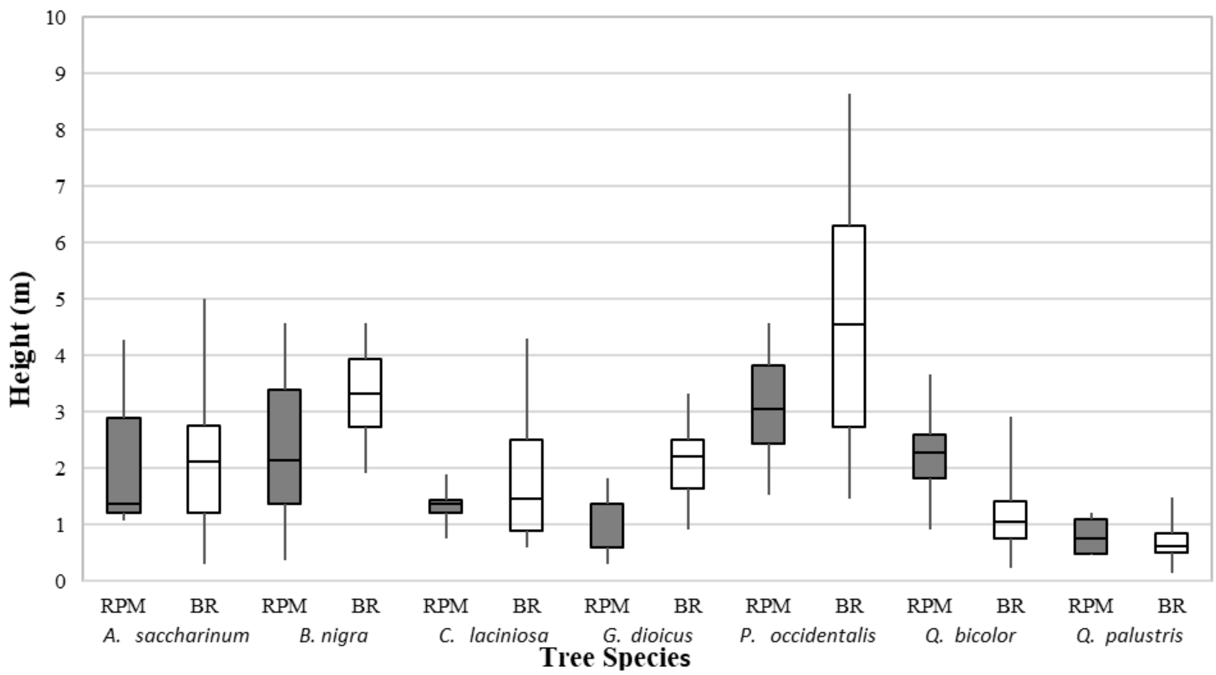

Figure 2. Tree height (m) by species for root production method (RPM, shaded boxes) and bare root (BR, white boxes) trees of 7 focal species on 14 sites (1,050 trees) in central and southeastern in Iowa

Mean Crown Depth

Crown depth also varied by stock type and species, for the RPM stock ranging from 0.3 m for Kentucky coffeetree to 1.5 m for American sycamore, and for BR ranging from 0.4 m for pin oak to 3.6 m for American sycamore (see Figure 3). Boxes represent the middle 50%, lines within the boxes represent means, and whiskers are upper and lower limits of measured crown depth (Figure 3). Crown depth for the RPM stock was less than that for the BR seedlings ($p = 0.010$, Table 7). Crown depth was negatively associated with soil percent sand ($p < 0.001$) and soil bulk density ($p = 0.012$) and positively related to tree age ($p < 0.001$, Table 7).

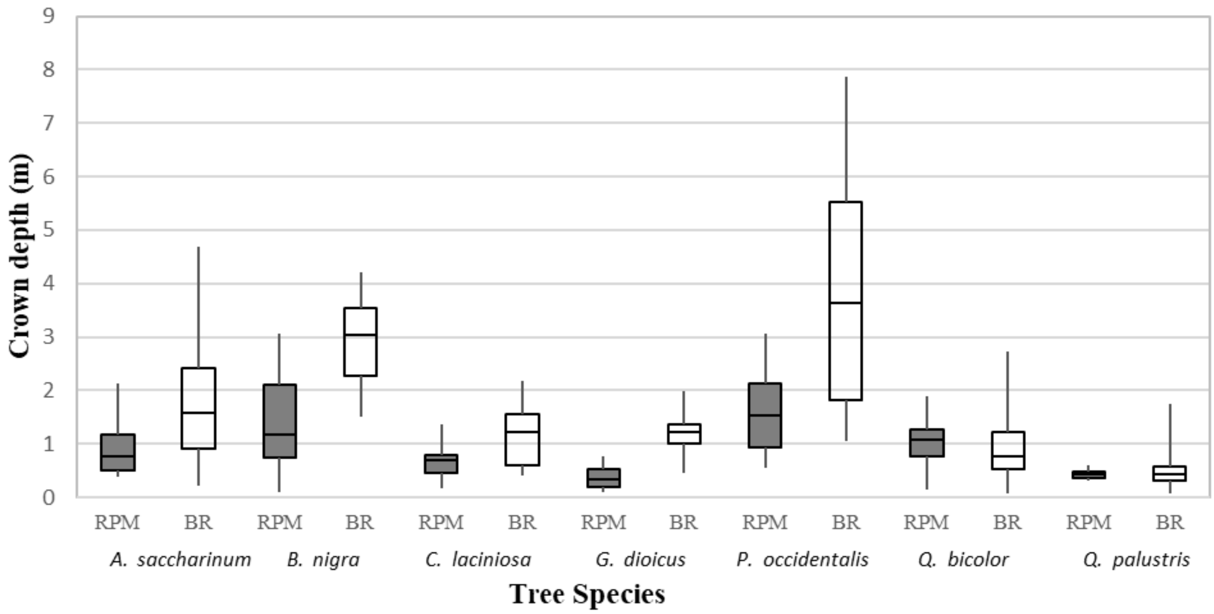

Figure 3. Crown depth (m) by species for root production method (RPM, shaded boxes) and bare root (BR, white boxes) trees of 7 species on 14 sites (1,050 trees) in central and southeastern Iowa

Diameter

Mean root collar diameter for RPM growing stock ranged from a low of 0.5 cm for Kentucky coffeetree to a high of 3.4 cm for river birch, and for BR stock it ranged from 1.7 cm for pin oak to 7.2 cm for American sycamore (Figure 4). Boxes represent the middle 50%, lines within the boxes represent means, and whiskers are upper and lower limits of measured diameter (Figure 4). Root collar diameter was smaller for RPM stock ($p = 0.021$, Table 7). This parameter was also negatively related to soil percent sand ($p < 0.001$) and positively related to percent clay and tree age (both p-values < 0.001, Table 7).

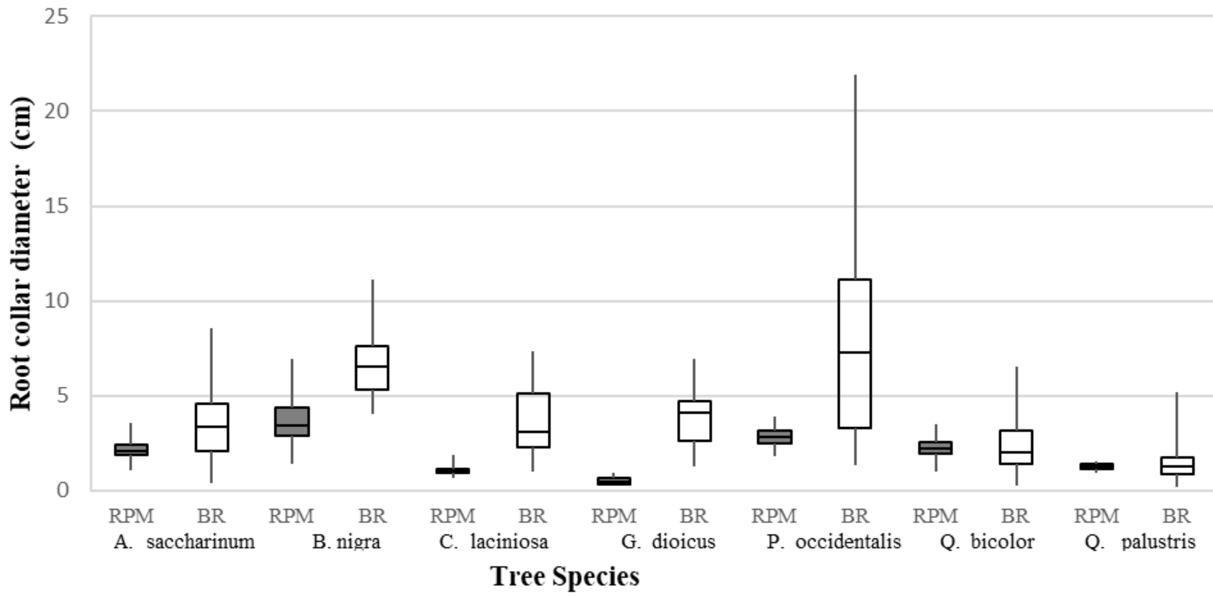

Figure 4. Root-collar diameter (cm) by species for root production method (RPM, shaded boxes) and bare root (BR, white boxes) growing stock for 7 species on 14 sites (1,050 trees) in central and southeastern Iowa

Height-to-Diameter Ratio

Mean height-to-diameter ratio for RPM stock ranged from a low of 57.2 cm cm^{-1} for pin oak to 144.1 cm cm^{-1} for Kentucky coffeetree, and for BR stock it ranged from 47.0 cm cm^{-1} for shellbark hickory to 67.0 cm cm^{-1} for American sycamore (Figure 5). Boxes represent the middle 50%, lines within the boxes represent means, and whiskers are upper and lower limits of calculated ratio (Figure 5). Although the RPM stock had height-to-diameter ratios that were nearly twice that of BR stock, variation among species precluded detecting a significant relationship for stock type in this analysis ($p = 0.229$, Table 7). The researchers detected a significant positive association with percent sand in soil ($p = 0.009$) and a negative relationship with percent clay ($p = 0.005$, Table 7).

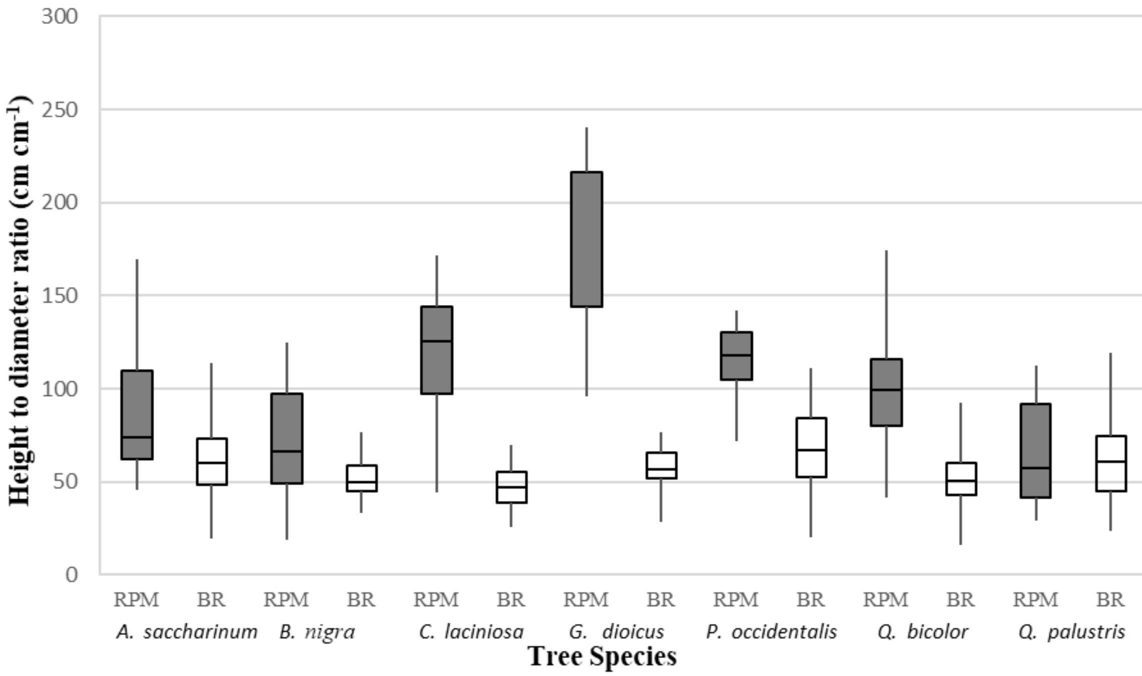

Figure 5. Height-to-diameter ratio (cm cm^{-1}) by species for root production method (RPM, shaded boxes) and bare root (BR, white boxes) trees for 7 species on 14 sites in central and southeastern Iowa

Overall, the RPM seedlings were shorter, had smaller crown depths, and had smaller root collar diameters than did BR seedlings. RPM seedlings were also characterized by height-to-diameter ratios that were relatively high, although the research team did not detect a significant difference between the stock types for this parameter across all seven species. Differences in height and diameter between stock types were consistent and most notable for river birch, Kentucky coffeetree, and American sycamore. However, differences were less pronounced, and RPM stock actually performed slightly better in terms of height and root collar diameter for the two oak species. Crown depth was closely associated with height for the BR seedlings, whereas it was limited for the RPM seedlings, possibly related to more prevalent use of tree shelters on RPM stock.

Similar to the researchers' conjecture related to survival rates for the two stock types in the overall analysis, it is likely that the BR stock experienced less "transplant shock" due to their more natural shoot-to-root ratio and seedling form at the time of planting, and thus they grew more rapidly in diameter on average than did the RPM stock. Differences in seedling diameter may also be attributable to the use of shelters primarily on the RPM seedlings. Previous work has documented, for example, that stems of trees grown in tree shelters tend to be elongated with little change in diameter over the length of the stem (Schultz and Thompson 1996, Sharew and Hairston-Strang 2005). Earlier researchers have suggested that modified "greenhouse" conditions within tree shelters can lead to altered patterns of resource allocation, generally away from support of root growth, root collar diameter expansion, and development of branches, and toward height growth of the main stem (Ponder 1995, Schultz and Thompson 1996, Bardon et al. 1999).

Conclusions for Objective 2

The research team determined that site characteristics, stock type, and use of tree shelters influenced seedling success on a range of mitigation/restoration projects. Overall, the findings suggest that forested wetland planting projects have higher tree survival rates on sites where seedlings are planted near lower-order stream systems, in soils with lower clay content, and that are established without use of tree shelters. The research team also determined that BR seedlings had higher survival rates and, for a set of focal species, better seedling morphology than did RPM planting stock. In addition, BR stock have been successfully established at densities that are likely to meet regulatory requirements and are likely to lead to stand structures typical of naturally regenerating stands.

It is important to note that survival and growth of planted trees are dependent on a number of additional factors, such as careful matching of species to sites and their placement within sites, planting method, seedling care up to and at the time of planting, immediate post-planting care, and ongoing site maintenance. Given the additional costs associated with RPM stock and tree shelters (for materials and installation), based on seedling performance in this study the research team recommends the use of BR seedlings at relatively high densities, without tree shelters, and with careful attention to species/seedling placement and post-planting maintenance. Based on measures of stem density in areas of natural regeneration, creating conditions to enhance natural regeneration should also be considered a viable approach to forested wetland mitigation in Iowa.

Objective 3: Assessment of an Experimental Forested Wetland Restoration Site

Seedling Survival

The team assessed 140 trees representing 12 species, with an overall survival rate of 99% (Table 2). This included 53 RPM seedlings, 43 BR seedlings, and 44 B&B plants. Average survival for all three stock types was relatively high, from 95% (B&B stock) to 100% (RPM and BR seedlings). There was no statistical difference in survival rates among stock types. Average stem density for these stock types ranged from 49 trees per acre for B&B stock to 143 trees/acre for BR seedlings.

These trees were measured just one year following planting, so relatively high survival rates were expected. Based on more recent follow-up visits to the site, the researchers are aware that survival rates have declined in the two field seasons after their earlier measurements. In several areas of the site, the research team observed mortality due to maintenance (primarily mower damage), which had a uniformly negative effect on seedlings of all stock types. Stem densities for all three stock types are considerably lower than those in areas of natural regeneration, and for the B&B stock stem densities are also lower than those of mature forest stands. Stem densities for RPM and B&B stock would not meet current regulatory requirements and are too low to allow seedlings to “capture” the site.

Survival and Seedling Morphology for Four Focal Species

The research team's analysis of seedling morphology included 76 of these trees, which included 4 species that were represented by at least 2 stock types: 7 silky dogwood (3 BR, 4 RPM), 28 black walnut (20 BR, 8 RPM), 18 swamp white oak (9 BR, 9 B&B), and 23 bur oak (1 BR, 8 RPM, and 14 B&B). Their overall survival rate was 98%, ranging from 94% for swamp white oak (mortality of 1 B&B plant) to 100% for the other 3 species.

For this subset of trees, mean height varied by stock type and species, for RPM stock ranging from a low of 1.1 m for black walnut to a high of 1.4 m for bur oak. For BR stock, mean height ranged from 1.1 m for bur oak to 1.3 m for black walnut, and for B&B stock it ranged from 2.9 m for bur oak to 3.3 m for swamp white oak. Overall, height of RPM and BR seedlings was similar, and for both stock types it was less than that of B&B stock ($p < 0.0001$, Table 8). Satterthwaite approximations to degrees of freedom and the Tukey method for comparing a family of three estimates were used to determine statistical differences (Table 8). Mean crown depth also varied by stock type and species, for RPM seedlings ranging from a low of 0.8 m for black walnut to a high of 1.2 m for dogwood. For BR stock, mean crown depth ranged from 0.6 m for bur oak to 1.0 m for dogwood, and for B&B stock it ranged from 1.6 m for bur oak to 2.1 m for swamp white oak. Overall, crown depth of RPM and BR seedlings was similar and for both stock types it was less than that of B&B plants ($p \leq 0.0038$, Table 8).

Mean seedling root collar diameter also varied, for RPM seedlings ranging from a low of 0.6 cm for black walnut to a high of 1.2 cm for dogwood. For BR stock, diameter ranged from 0.6 cm for dogwood to 1.3 cm for bur oak, and for B&B seedlings it ranged from a low of 5.4 cm for bur oak to a high of 6.6 cm for swamp white oak. Overall, diameter for RPM and BR seedlings did not differ, and both were less than that of B&B stock ($p < 0.0001$, Table 8). Average height-to-diameter ratio for RPM stock ranged from a low of 113.8 cm cm^{-1} for bur oak to a high of 182.5 cm cm^{-1} for black walnut. For BR seedlings, this parameter varied from 84.1 cm cm^{-1} for bur oak to 189.4 cm cm^{-1} for dogwood. For B&B stock, this ratio varied from 49.7 cm cm^{-1} for swamp white oak to 57.4 cm cm for bur oak. Overall, this ratio did not differ between RPM and BR stock but was greater for them compared to B&B seedlings ($p < 0.0001$, Table 8).

Table 8. Linear mixed-effects logistical regression models for height, crown depth, diameter, and height-to-diameter ratio for 140 trees of 4 focal species on the Goose Pond site in Linn County, Iowa

Variable/Contrast	Estimate	Standard error	df	t value	p value
Height					
BR – RPM	-0.146	0.123	54	-1.196	0.461
B&B – RPM	0.903	0.116	22	7.758	< 0.001
B&B – BR	1.050	0.112	25	9.333	< 0.001
Crown depth					
BR – RPM	-0.157	0.176	63	-0.890	0.649
B&B – RPM	0.556	0.165	63	3.358	0.004
B&B – BR	0.712	0.160	63	4.455	< 0.001
Root collar diameter					
BR – RPM	-0.271	0.137	63	-1.977	0.126
B&B – RPM	1.581	0.141	63	11.237	< 0.001
B&B – BR	1.852	0.130	62	14.252	< 0.001
Height-to-diameter ratio					
BR – RPM	0.102	0.139	62	0.723	0.745
B&B – RPM	-0.791	0.139	49	-5.691	< 0.001
B&B – BR	-0.892	0.131	60	-6.803	< 0.001

Conclusions for Objective 3

Species diversity at this site exceeds that of natural stands, which is desirable for increasing chances for successful establishment and for providing ecological benefits (e.g., habitat, hydrological function). However, stem densities are low relative to naturally regenerating stands, and for RPM and B&B stock they do not meet current regulatory requirements even at this very early stage in stand development**,** and it is already known that these densities have declined somewhat. In addition, such low stem densities are unlikely to lead to canopy closure in a desired time-frame.

In general, seedling characteristics reflect the nature of each stock type at the time of planting relatively closely, since they were only on the site for a year before the research team conducted measurements. Although the B&B plants are larger (height, crown depth, and diameter) and have a lower height-to-diameter ratio, they are also more expensive and require an intensive field operation for planting. It is unclear at this point if the added expense and effort are justifiable for long-term performance, and the research team recommends continued monitoring of the site.

General Conclusions

Concerns over the success of mitigation projects, particularly for forested wetland restoration, have led to regulatory updates that identify more stringent requirements, including relatively high mitigation area ratios and densities of live trees on sites at the end of 10-year monitoring periods. Road development projects are among activities that cause unavoidable impacts to wetlands, and agencies responsible for such projects have expended considerable resources to meet these requirements. However, efforts to establish forested wetlands have achieved variable and often limited success in terms of survival of tree seedlings on wetland sites. Although there have been very few studies that documented exact causes for failure of specific restoration projects, a small number of experimental studies emphasize the importance of relatively subtle site-scale variation in topography and soil characteristics, and larger surveys have identified landscape-scale factors that influence flooding regimes and determine the hydrological dynamics of project sites that strongly affect survival of perennial vegetation in general and tree seedling success in particular.

In this study, the researchers determined that in addition to site characteristics, stock type and use of tree shelters influence seedling success on a range of forested wetland mitigation/restoration projects. Compared to similar studies conducted elsewhere, the authors found that seedling survival rates on Iowa DOT forested wetland mitigation sites in Iowa were relatively high, ranging from 74% for RPM stock to 88% for BR stock. They also determined that tree shelters had a negative effect on tree survival and seedling morphology. In addition, analyses of stem density suggest that plantings of RPM and B&B stock have lower densities than BR stock, are much lower than those for naturally regenerating stands, and are not adequate to meet regulatory requirements. Overall, the findings indicated that forested wetland planting projects have higher tree survival on sites where seedlings are planted near lower order stream systems, in soils with lower clay content, and that are established without use of tree shelters. The researchers also determined that BR seedlings had better survival rates and seedling morphology than did RPM planting stock. It is important to note that survival and growth of planted trees are also dependent on a number of additional factors, such as careful matching of species to sites and their placement within sites, planting method, seedling care up to and at the time of planting, immediate post-planting care, and ongoing site maintenance.

Given the additional costs associated with RPM stock and tree shelters (both for materials and installation), based on seedling performance in this study the authors recommend a general strategy of planting BR seedlings at high densities, using a limited number of tree shelters, and paying careful attention to species and individual seedling placement on each site. The research team also recommends more detailed specifications in contracts with qualified forestry professionals for site preparation, planting, and post-planting maintenance to enhance project success. More detailed recommendations are provided in the following chapter.

It is not possible to control some factors influencing seedling establishment, such as extreme flooding or drought during the establishment period. One approach to minimize risk, even at the individual site scale (for large projects), would be to distribute tree planting activities over two to three years on different parts of a site to increase the overall probability of success. Other factors affecting seedling establishment could potentially be addressed in site selection, in project design, at the time of planting, and in the three to five years following planting. For example, based on the researchers' observations, complete removal of materials attached to seedlings (any tags or tape encircling the stem) at the time of planting would decrease seedling mortality due to stem girdling. Project designs that specifically facilitate post-planting maintenance, especially mowing, would likely also limit damage to and/or mortality of seedlings.

5. RECOMMENDATIONS FOR TREE ESTABLISHMENT IN FORESTED WETLAND MITIGATION PROJECTS IN IOWA

The purpose of this set of recommendations is to provide guidance for Iowa DOT personnel responsible for site selection, planning, design, and development of specifications for forested wetland restoration. The recommendations are based on information assembled during a literature review and an empirical study of 25 sites: 9 forested wetland mitigation projects, 5 natural regeneration areas, 5 reference forest sites, and 1 new experimental planting identified by Iowa DOT personnel, and 5 bottomland forest restoration sites (to balance representation of different stock types) identified by personnel in the Iowa DNR's Forestry Section (as described above and in Heber 2017).

General Recommendations

To support timely release from permit requirements on mitigation sites, the research points to the importance of (1) planning for forested wetland restoration; (2) setting a small number of realistic and site-specific goals/performance standards in each permit (Matthews and Endress 2008); (3) considering the characteristics of the proposed mitigation site within the landscape-watershed context (Matthews and Pociask 2015) and adjusting planting plans accordingly; (4) where possible, creating microtopographic variation on restoration sites to enhance seedling survival (e.g., USDA NRCS 2003, Simmons et al. 2011); (5) ensuring both pre- and post-planting weed control (Randall and Herring 2012); (6) selecting species and stock types from those known to have been successful in previous projects; and (7) enhancing specifications and closely monitoring to verify proper planting and maintenance when projects are installed through contractual agreements. More focused recommendations for each of these categories are provided in the sections that follow.

Restoration Goals and Performance Standards

Although a thorough analysis of permit documents was not conducted as a part of this project, a review of the literature indicated greater success in compliance and timely release of sites for which permits specified fewer and very site-specific goals (Kihslinger 2008, Matthews and Endress 2008, Pociask and Matthews 2013). As detailed by Pociask and Matthews (2013), specific goals should be set for different landscape-watershed contexts. For example, achieving high wetland plant diversity and representation of perennial species is more likely on sites with low levels of flood intensity (see Site Selection section below), whereas on sites with greater flood intensity the authors suggest setting functional goals that may apply only to specific areas within a site (Pociask and Matthews 2013).

Site Selection

Confirmation or restoration of hydric soil conditions and adequate site hydrology (hydric soils or conditions that lead to hydric soil development, and site inundation or saturation for specific periods of time) are crucial criteria that influence site selection for mitigation projects. Although identification of specific characteristics of hydric soils and site hydrology themselves are beyond the scope of this set of recommendations, it has been suggested that efforts to select sites that are certain to meet these criteria in a short time-frame may lead to a tendency to choose areas that are also impacted by deep and/or long flood events that can limit successful establishment of perennial vegetation, particularly tree seedlings (e.g., Kramer et al. 2008, Johnson et al. 2012, Pociask and Matthews 2013, Matthews and Pociask 2015).

For example, in two studies of wetland mitigation projects in Illinois, the effects of site flooding regimes on establishment and persistence of wetland vegetation were investigated (Pociask and Matthews 2013, Matthews and Pociask 2015). The researchers for these studies found that sites characterized by greater depth and duration of flood events ("flood exposure") had lower species richness and lower proportions of perennial species and that colonization of these sites by native, perennial, and hydrophytic species declined following flood events (Pociask and Matthews 2013). Based on a follow-up study, they also concluded that depth and duration of flooding (rather than flood frequency) created a "ceiling effect" on planted tree survival, suggesting that survival can be either high or low on sites with less flood exposure but that on sites with greater flood exposure tree survival was almost always lower (Matthews and Pociask 2015).

Although flood exposure per se was not measured in this study of forested wetland mitigation sites in Iowa, the research team did note relationships between stream order, soil texture, and seedling performance: seedling survival was lower on sites associated with higher order streams (in larger watersheds with several contributing tributaries) and finer soil textures (e.g., higher clay content), both characteristics that are typical of sites that likely experience flood events of greater depth and duration (Heber 2017). Therefore, where possible, analysis of physiographic position within the larger watershed context should be used to focus attention on selecting sites associated with lower order streams where shorter and shallower flood events may occur but where flood duration is brief. In addition, because fine-textured soils also retain water for longer periods of time, the authors recommend a review of soil survey documents to assess surface and subsurface soil texture and, when possible, selecting sites with lower proportions of clay (e.g., ≤ 25% clay) because lower clay content on the sites the team investigated was associated with higher tree survival (Heber 2017).

Another criterion that may be considered in site selection, depending on method of forest establishment, is proximity to natural bottomland forest stands that could provide a seed source for areas of natural regeneration. This is a viable option to consider for the establishment of native, light-seeded wind- and/or water-dispersed riparian species (e.g., black willow, eastern cottonwood, silver maple) in combination with planting other species not likely to "seed themselves in," based on assessment of successful natural regeneration at existing Iowa DOT mitigation sites. This approach is described more fully in the section below on methods for forest establishment. It has the further advantage of creating "connectivity" among stands of different ages and across the landscape.

Site Preparation

Site preparation can include modifications to topography and in Iowa must include attention to control of undesirable vegetation, particularly control of well-established perennial grasses such as brome and reed canarygrass, which can quickly dominate entire restoration sites. Both elements of site preparation are described more fully in the paragraphs that follow.

Modifications to Topography

Particularly if mitigation sites must be located in larger and higher order watersheds with fine-textured soils, a strategy that has been successful at an experimental scale involves preservation or creation of microtopography to mimic the “pool and mound” characteristics of natural bottomland forests (USDA NRCS 2003, Bruland and Richardson 2005, Simmons et al. 2011). This provides a range of soil moisture conditions in small areas across a site, and seedlings can be placed based on each species’ natural ecological amplitude (Table 9). If site plans include manipulation of topography to create other features (e.g., areas of shallow open water), it may be possible to incorporate additional earthwork on restoration sites that would create small-scale topographic variation. Differential species performance has been documented at elevational changes ranging from 3 inches to 3 feet, with better performance of tree species adapted to flooding in low positions (e.g., willows, silver maple, cottonwood, boxelder) or transitional zones (river birch, sycamore, hackberry, swamp white oak, pin oak), and better performance of species with intermediate or lower adaptability to flooding on mounds (black walnut, bur oak, Kentucky coffeetree, shellbark hickory) (Pennington and Walters 2006, Simmons et al. 2011, Kabrick et al. 2012, Randall and Herring 2012). There is also evidence to suggest that this approach would not be necessary on sites with loamy, well-drained soils, because such sites are likely to be suitable for a broader array of species without topographic manipulation (Dey et al. 2004).

Table 9. Characteristics of species planted on 14 bottomland forest restoration sites assessed in 2015

Species	Natural distribution in Iowa[a]	Adaptability to flooding[b]	Seedling survival[c] (%, BR[d])	Seedling survival[c] (%, RPM [d])
Silver maple	Throughout	Good	100	84
River birch	Eastern and south-central	Good	100	88
Shellbark hickory	Southeastern	Moderate	100	72
Shagbark hickory	Except extreme northwest (not bottomlands)	Low	100[e]	100
Kentucky coffeetree	Throughout, especially south-central	Moderate	100	100
Honey locust	Except extreme northwest	Moderate	100	100[e]
Black walnut	Throughout	Moderate	93	70
Mulberry	Southern half and northeast	Moderate	100	-
American sycamore	Southern two-thirds	Good to moderate	94	65
Eastern cottonwood	Throughout	Good	98	90
Chokecherry	Throughout	Moderate	100	-
Black willow	All except extreme northwest	Good	100	-
Swamp white oak	Eastern half	Good	100	92
Bur oak	Throughout	Good	100	96
Pin oak	Southeastern	Good	100	88[e]
Red oak	Throughout (not bottomlands)	Low	100[e]	100[e]

Heber 2017
(a) van der Linden and Farrar 2011; (b) Randall and Herring 2012; (c) Percent survival based on living and dead stems at each site; (d) BR = bare root seedlings, RPM = Root Production Method seedlings; (e) species for which either BR or RPM (as indicated) sample included ≤ 10 trees

Weed Control

Initiation of weed control is generally recommended up to two years before tree planting (in the case of sites dominated by grasses) and should be continued for at least five years or until canopy closure after planting (Herring 2010, 2011; Randall and Herring 2012). For sites dominated by grasses, mowing in late summer (August) followed by herbicide application to regrowth in late September in the year(s) before seed dispersal (natural regeneration) or planting (artificial regeneration) can effectively limit competition. Cover crops such as perennial rye can be used to limit weed invasion of the site. On land previously used for row-crop production, less intensive site preparation may be possible if consistent weed control over time has prevented establishment of grasses and other aggressive weedy species. Scarification and/or prescribed burning after herbicide treatment(s) can also be used to expose mineral soil to enhance natural regeneration if desired (Randall and Herring 2012).

Immediately after tree planting (best completed in the spring following one or two years of pretreatment), pre-emergent herbicide (e.g., Princep, Prowl) should be used to treat a three- to five-foot perimeter around each seedling (Randall and Herring 2012, Iowa DNR 2017), which is recommended in early spring for at least the first five years. Competing vegetation near seedlings should be controlled for 75% of the growing season (Herring 2010, 2011). If site monitoring indicates that pre-emergent herbicides lose effectiveness too early in the growing season, additional herbicide treatments may be necessary (and should be customized for control of the most prevalent weed competition on the site). Selective herbicides can be used to control broadleaf weeds (e.g., Transline, Stinger) and grasses (Fusilade) (Herring 2011). Roundup may be used for spot treatments, but it is nonselective and will damage tree seedlings on contact. Because wetland restoration sites are usually located in areas with hydrological connections to surface water, it is especially critical that any herbicide be used in accordance with label directions and authorization for use.

In addition to herbicide treatments, areas between trees should be mowed two to three times per summer for at least five years to prevent weeds from overtopping the seedlings (Herring 2010, 2011). Because mower damage to seedlings on these sites has frequently been observed, maintenance contracts should draw attention to site planting layouts and include stringent specifications (e.g., size and type of mower decks, consequences of damage) to limit damage to seedlings during site maintenance.

Species and Stock Type Selection

Species Selection

Planting a diverse mix of tree species (as much as possible within species' natural range limitations, representing different degrees of adaptability to flooding, early and late successional stages) and carefully placing seedlings of each species with respect to microsite characteristics within restoration sites are likely to increase chances of establishment success (as per discussion of microtopography in the previous section and in Table 9). Performance of seedlings in recent assessments of Iowa DOT and Iowa DNR riparian bottomland plantings indicate that a number of species can be successfully established on restoration sites (see survival rates for assessment sites in Table 9).

Based on these assessments, most species rated "good" or "moderate" for flood adaptation had high survival rates regardless of stock type, particularly Kentucky coffeetree, honey locust (*Gleditsia triacanthos*), and bur oak. Species for which survival was at least somewhat tied to stock type include silver maple, river birch, shellbark hickory (*Carya laciniosa*), and pin oak; for each of these, survival for bare root seedlings was more than 10% higher than for RPM stock. Species for which survival was lower (although in relation to the research literature it was acceptable) include black walnut and American sycamore, which have been reported as having less adaptability to flooding. Species found in reference bottomland forests that researchers did not observe on plots (although some are present on newer Iowa DOT restoration sites that the team did not assess on a species-by-species basis) could be considered for continued use on an experimental basis, including American hornbeam (*Carpinus caroliniana*), hawthorn (e.g., downy hawthorn, *Crataegus mollis,* and fleshy hawthorn, *C. succulenta*), bitternut hickory (*Carya cordiformis*), wild plum (*Prunus americana*), peachleaf willow (*Salix amygdaloides*), sandbar willow (*S. interior*), Missouri River willow (*S. eriocephala*), and slippery elm (*U. rubra*), as well as a variety of native shrubs (e.g., dogwoods [*Cornus*, spp.] and elderberry). Two of the species included in the assessment of restoration sites—shagbark hickory (*C. ovata*) and red oak (*Q. rubra*)—are not bottomland species, although they do appear to have become successfully established on the sites the research team evaluated.

In addition to careful placement of species according to topography, some research suggests that a useful approach to long-term stand development is to purposefully include early- ("pioneer") and late-successional species at all planting sites (Lockhart et al. 2008). The early-successional species (such as silver maple, river birch, honey locust, black willow [*S. nigra*], and cottonwood) grow rapidly and are useful to "capture" the site to maintain dominance by woody vegetation, and the late-successional species (hickories, oaks) generally have greater longevity and value for wildlife (e.g., most mast-bearing species).

Stock Type Selection

Choice of stock type can be informed by historic performance as well as consideration of individual site characteristics and goals for each restoration project. The team's assessment of seedlings on sites throughout the state included BR seedlings, RPM stock, B&B stock (limited to one new site), and seedlings/saplings in areas of natural regeneration. Of these, planting bare root seedlings and/or encouraging natural regeneration are the most cost-efficient and likely to be successful under a wide range of site conditions.

BR seedlings are grown from native seed in nursery beds for one or two season(s) before being lifted and packaged/chilled to maintain seedling moisture levels until the time of planting. These plants are smaller, less expensive, and relatively easy to handle and transport to planting sites. Large numbers of BR seedlings can be planted fairly quickly using planting machines or by hand. Disadvantages of BR seedlings include a longer establishment period and greater need for weed control.

The research team's assessment of bottomland restoration sites indicated very high survival rates for all species planted as BR seedlings on seven sites (two Iowa DOT sites, five Iowa DNR sites) between 1995 and 2012, although both black walnut and American sycamore had somewhat lower survival than other species. The authors conjecture that three factors may have contributed to better performance of BR stock on these sites: most plantings with BR stock had been planted more recently (Iowa DNR sites) than those with RPM stock (less time for attrition), seeds for these trees are more likely to have been locally collected and grown (therefore better adapted to local site conditions), and seedling morphology (root-to-shoot and height-to-diameter ratio) is more "natural" for the smaller seedlings than that of larger planting stock (Heber 2017).

RPM seedlings are produced by placing carefully selected seed in open-bottomed trays to germinate and subsequently transplanting them to successively larger containers that use air pruning to promote production of a fibrous root system over a typical culture period of two years

(Lovelace 2002, Dey et al. 2004). Both the root systems and stems of RPM stock are larger than those of typical one-year-old BR plants. A number of previous studies documented greater survival and more rapid growth of RPM seedlings compared to BR stock in bottomland planting projects where they have been directly compared (Dey et al. 2004, Krekeler et al. 2006, Walter et al. 2013). In spite of their greater cost (for both plant material and installation), researchers have recommended their use, particularly to enhance establishment of later-successional and/or coarse-rooted species such as oaks, to extend the planting season (because they have more intact root systems), and to hasten the process of canopy closure on planting sites.

Contrary to most reports in the literature, however, the team's assessment of RPM stock planted on seven sites between 2003 and 2011 indicated significantly lower overall survival (74% overall) for this stock type, and in particular for shellbark hickory (72%), black walnut (70%), and American sycamore (65%) (Heber 2017). Although the researchers were not able to conduct a rigorous analysis of causes for mortality, they did note improper planting, nursery tags that girdled seedlings at the root collar, and evidence of "transplant shock" among RPM trees on Iowa DOT sites. Use of RPM trees may be merited for species such as shagbark hickory, Kentucky coffeetree, honey locust, and bur oak (species for which this stock type has had greater success) to speed forest establishment. This stock type could also be used in combination with BR seedlings to enhance the species mix and more rapidly close the canopy on a site.

B&B trees are grown in the field for several years and then excavated with soil surrounding the roots, wrapped in burlap, and often reinforced with wire cages or twine. The stems of these trees are typically greater than two inches in diameter. The authors' search of the literature found only one experimental study of B&B stock used for forested wetland restoration (McLeod 2000), which reported survival rates slightly greater than or similar to those of BR seedlings (depending on species and site conditions). Because B&B trees are heavy, they are more cumbersome to transport and distribute on large planting sites. They are also relatively expensive and so typically only used in amenity landscapes. It is generally recommended that burlap and any other packaging (e.g., wire basket or twine webbing) be removed at the time of planting because they limit root growth, which also adds to the cost of planting (and/or the potential for complications/mortality later if this is not done properly).

The research team assessed B&B stock on one new restoration site, Goose Pond in Linn County, where the three stock types (B&B, BR, and RPM) were planted experimentally to compare their performance. The team conducted their assessment only one year after planting at this site and at that time observed 100% survival of BR and RPM seedlings and 95% survival of B&B plants. More long-term monitoring of all three stock types at this site may indicate whether the use of B&B stock is justified, most likely to hasten canopy closure on sites particularly prone to aggressive weed competition (especially grasses).

Natural regeneration and direct seeding are also possibilities for forested wetland establishment, especially on sites located near natural bottomland forests. These forests are often dominated by early-successional species that produce abundant crops of small seeds that are dispersed by wind and water and can quickly colonize sites where bare soil is exposed either by the scouring action of natural flood events or through intensive site preparation techniques (see previous section on site preparation). Many of these species (silver maple, cottonwood, willow) disperse seed in spring (March through June), so site preparation must be completed the previous fall to enhance germination (Randall and Herring 2012). For heavier-seeded and late-successional species, direct seeding is also an option (although less well studied). For direct seeding, recommended tactics are to use a high seeding rate (e.g., one to two bushels per acre of oak acorns or hickory nuts) and to monitor closely to determine the need for weed control by mowing with a raised deck or spot-spraying with an appropriate herbicide (Randall and Herring 2012). Either of these approaches may be most suitable to enhance other establishment methods.

The team's assessments of plots in existing bottomland hardwood forests (reference forests) and nearby areas of volunteer natural regeneration indicate that natural regeneration should be considered a viable option for establishing forested wetlands (Table 2, Thompson et al. 2018). The mature reference forests that the researchers evaluated contained between 15 and 145 stems per acre (an average of 60) and were dominated by silver maple, green ash, hawthorn, cottonwood, and hackberry, with a mix of other species that most frequently included elms, black willow, and river birch. Plots in areas of natural regeneration were dominated by black willow, cottonwood, and silver maple, at densities that ranged from 0 to 23,000 stems per acre (average of 410). These areas of typically very dense regeneration are representative of the inverted J-curve structure for early stages of forest stand development, and most of the sites that the research team assessed have achieved the degree of canopy closure necessary to prevent invasion by aggressive weeds and grasses.

Tree Planting and Stand Establishment Strategies

Recommended planting methods differ according to stock type. In general, attrition of planted seedlings of any type can be expected, so it is recommended that seedlings be planted at densities higher than that required to certify the site (e.g., more than the required density of 100 trees per acre at 10 years following restoration planting). Generally, attrition is expected to be greater for direct seeding or small (BR) seedlings given the longer period required for their establishment. Because seedling success on Iowa DOT forested wetland mitigation sites is dependent on proper planting of high-quality plant material, contractual agreements should be made only with qualified professional forestry contractors and the specifications therein should be stringent (Herring 2010, 2011).

For projects that include bare root stock, use of conservation-grade seedlings, planting at high densities (600 to 1,000 stems/acre), and maintaining density of 300 trees per acre until the stand is established has been recommended (Randall and Herring 2012). This approach mimics the natural structure of a dense, early-successional forest (as described above) and allows seedlings to capture the site and suppress weeds (reducing the need for site maintenance). It is also recommended that the species mix include fast-growing species that cast dense shade, such as silver maple, American sycamore, and cottonwood (and assuming some of these species will also seed in naturally), with other species such as bur oak, swamp white oak, pin oak, and shellbark hickory making up the balance (Randall and Herring 2012).

Bare root seedlings can be planted by hand or with a machine in spring from early April through mid-May or in fall from mid-October until there is frost in the soil. Seedlings should be planted as soon as possible after they are removed from storage/delivered and should be kept out of direct sunlight, and seedling roots should be soaked for an hour before planting (Iowa DNR 2017). Seedlings should be planted with the root collar at grade, and soil should be placed to accommodate spreading the root system and leaving no air pockets (as illustrated in Appendix B). Species should be placed according to topography and adaptability to flooding (as per Table 9). Planting seedlings in rows (eight-foot spacing both within and between rows will yield close to 700 trees per acre) provides space for mowing between rows and may help to limit damage to seedlings from mowers during early stages of stand development. Although rows will initially be apparent, over time seedlings will be more naturally distributed due to random mortality.

The literature the research team reviewed indicated mixed results for the use of tree shelters (Lantagne et al. 1990, Sharew and Hairston-Strang 2005, Stuhlinger 2013, Drayer et al. 2017). On Iowa DOT sites, the team found that tree shelters were negatively associated with tree survival (Heber 2017), although the exact cause for this was beyond the scope of this study. Thus, use of vented shelters is recommended only for 30 to 50 trees per acre to provide protection on sites known to have strong herbivore pressure (e.g., from deer and beaver). Correct placement of stakes and vented shelters (proper orientation of shelters, location of stakes external to shelters, and attachment with zip ties) should be specified for contractors. Further, site monitoring should include evaluation of tree shelters to determine when they should be (manually) removed.

For projects that include RPM stock, it is also necessary to plant at densities greater than 100 trees per acre to account for expected mortality. Based on the overall survival rate (74%) that the researchers observed for RPM trees, this would require planting approximately 135 to 150 trees per acre (more for species with poorer performance) to achieve 100 trees per acre after 10 years. A species mix that emphasizes fast-growing species to cast dense shade and species that have been successful on Iowa DOT sites would include silver maple, river birch, and cottonwood. Other species that have been successful as RPM stock include Kentucky coffeetree and honey locust, depending on site characteristics. Late-successional, mast-bearing species such as swamp white, bur, and pin oaks have performed well as RPM stock and could be included.

RPM seedlings can also be planted in spring or fall during planting windows similar to those described for BR stock. Planting holes should be prepared with power augers that are slightly larger than the seedling root bag size. Seedlings should be planted as soon as possible after delivery, be well hydrated at time of planting, and be placed in planting holes such that the seedling root collar is at or slightly above grade (also illustrated in Appendix B). All materials from the nursery should be removed, including root bags and stakes. Tape or tags that encircle tree stems must be removed at the time of planting to prevent girdling (which researchers observed causing mortality on several sites). Seedlings should be placed with consideration of topographic features of each site (Table 9) and post-planting maintenance. This may require planting trees in rows or other regular patterns that provide space for use of mowers after planting without damaging the seedlings themselves. Although the team’s overall assessment of prior plantings indicated tree shelters had a negative effect on survival, there were no RPM seedlings that had been planted without tree shelters to provide a direct comparison to those that did have them. Using shelters on some of these trees (50 to 70 trees per acre) may be warranted to protect the additional investment in this more expensive planting stock.

Monitoring and Post-Planting Care

Forested wetland mitigation sites should be monitored regularly and as frequently as possible to ascertain proper planting immediately following installation and during the first year for and maintenance for the subsequent three to five years. Recently planted stock should be closely inspected to verify the quality and species of seedlings and to ensure that they have been planted correctly (at proper depth with upright stems and complete closure of soil around them). Monitors should verify that planting occurred during the specified planting window and that all nursery-related materials were removed from the trees, especially any tape or tags on tree stems that could girdle seedlings. Because weed control is so essential for seedling success, early monitoring should also include assessment of the effectiveness of pre-emergent herbicide treatment and/or should determine the need for post-emergent herbicide treatments. Routine monitoring is also recommended to verify ongoing control of weeds using appropriate herbicides in a three- to five-foot zone around each seedling for at least 75% of the growing season during the first five years after planting (Herring 2010, 2011). Sites should be maintained such that seedlings are not over-topped by weeds. The authors also recommend monitoring the trees in shelters and removing the shelters after some period of time.

Other routine site maintenance should include mowing the areas between seedlings to eliminate cover/habitat for rabbits and rodents and to minimize seedling damage from those as well as other herbivores (Iowa DNR 2017). However, because the researchers observed substantial seedling mortality on wetland mitigation sites due to mower damage, they recommend the use of planting designs that easily accommodate the use of mowing for weed control, as well as strict specifications in maintenance contracts that describe the planting design, allowable size/configuration of mowing equipment ("belly mowers" are preferable to large rear-mounted mowing decks), and suggested patterns for mowing. Consequences to contractors for any damage to planted trees should be identified and strictly enforced. Continued regular monitoring of sites to evaluate routine maintenance activities is recommended. Over time, longer-term maintenance for successful plantings may include thinning of established tree stands at 15- to 20-year intervals, which may be negotiated with partner land-holders (especially if they are conservation agencies).

Conclusions

Challenges in restoration of forested wetlands on mitigation sites have been of concern nationwide, and substantial mortality of tree seedlings has often been a cause of non-compliance (Kihslinger 2008, Matthews and Endress 2008). Although there have been few studies that documented exact causes for failure of specific restoration projects, experimental studies emphasize the importance of relatively subtle site-scale variation in topography and soil characteristics (e.g., Pennington and Walters 2006), and larger surveys have identified landscape-scale factors that influence flooding regimes and determine the hydrological dynamics of project sites (Matthews and Pociask 2015), thereby affecting seedling success.

Some factors influencing seedling establishment can’t be controlled, such as extreme floods or drought events during the establishment period. One approach that might be considered, even at the individual site scale (particularly for very large projects), would be to distribute tree planting activities on different areas of a site over two to three years to increase the overall probability of success. Other factors affecting seedling establishment could potentially be addressed in site selection, in project design, at the time of planting, and in the three to five years following planting. For example, based on the research team’s observations, removal of materials (any seedling tags or tape encircling the stem) at the time of planting would decrease seedling mortality due to stem girdling. Project designs that specifically facilitate post-planting maintenance, especially mowing, would likely also limit damage to and mortality of seedlings.

The research team observed greater survival for bare root seedlings and for trees planted without tree shelters and noted success of natural regeneration at the limited number of sites that were evaluated. Planting bare root seedlings at relatively high densities (with or without shelters) and/or creating conditions that support natural stand development should be considered viable options for forested wetland mitigation in Iowa.

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APPENDIX A. SUMMARY TABLE

Table 10. Species and numbers of RPM and BR seedlings included in assessments of 14 riparian area planting projects in Iowa

Scientific name	Common name	Number of RPM seedlings	Number of sites	Estimated survival (%)	Number of BR seedlings	Number of sites	Estimated survival (%)
Acer saccharinum	Silver maple	152	6	84	321	6	100
Betula nigra	River birch	35	5	88	33	1	100
Carya laciniosa	Shellbark hickory	36	3	72	20	2	100
Carya ovata	Shagbark hickory	14	1	100	5	2	100
Cornus obliqua	Silky dogwood	8	2	50	18	2	100
Fraxinus pennsylvanica	Green ash	42	1	100	256	5	82
Gymnocladus dioicus	Kentucky coffeetree	11	1	100	43	3	100
Gleditsia triacanthos	Honey locust	5	1	100	27	3	100
Juglans nigra	Black walnut	36	3	70	247	4	93
Juniperus virginiana	Eastern red cedar	-	-	-	38	1	95
Morus alba	White mulberry	-	-	-	48	1	100
Platanus occidentalis	American sycamore	27	3	65	59	4	94
Populus deltoides	Eastern cottonwood	20	2	90	126	7	98
Populus tremuloides	Quaking aspen	10	1	100	-	-	-
Prunus virginiana	Chokecherry	-	-	-	33	3	100
Salix nigra	Black willow	-	-	-	37	1	100
Taxodium distichum	Bald cypress	5	1	100	-	-	-
Quercus bicolor	Swamp white oak	96	5	92	323	7	100
Quercus macrocarpa	Bur oak	25	3	96	198	3	100
Quercus palustris	Pin oak	7	3	88	131	3	100
Quercus rubra	Red oak	10	2	100	4	1	100
Ulmus americana	American elm	-	-	-	27	1	100
Total		**539**			**1994**		

APPENDIX B. TREE PLANTING SPECIFICATION DIAGRAMS

Guidelines for Proper Planting of Bare Root Seedlings (from Iowa DNR 2017)

1. Store unplanted seedlings in a cool place out of direct sunlight.

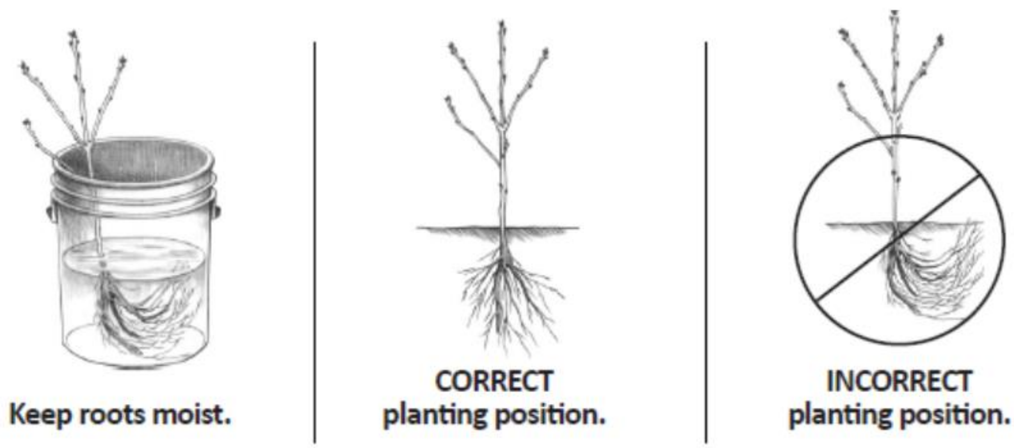
2. Soak seedling roots for an hour before planting.

3. Remove any tags or other foreign materials from seedlings.

4. Plant seedling with root collar at or slightly above grade.

5. Position seedling root systems to spread and grow out- and downward.

6. Plant seedlings upright and firmly pack soil around the roots, leaving no air pockets.



Iowa DNR 2017, *State Forest Nursery Seedling Catalog*

Figure 6. Proper and improper planting methods for planting bare root seedlings

Guidelines for Proper Planting of RPM or B&B Seedlings

1. Ensure the tree is well hydrated.

2. Make a planting hole that is *wider* than the root bag/container (RPM) or ball (B&B) and *shallow* enough allow placement of tree root flare at or slightly above grade.

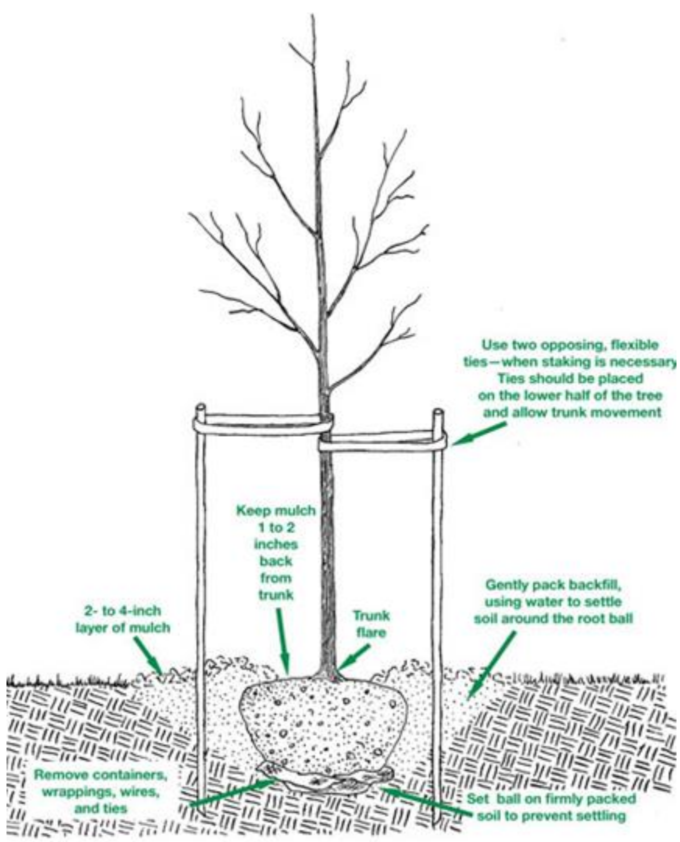
3. Remove any stakes/transit guards from tree.

4. Remove materials from around root ball completely – for a root bag (RPM) remove before planting, for burlap and wire cages (B&B) it may be necessary to place seedling in the hole first.

5. Ensure that the seedling is upright.

6. Fill the hole gently but firmly with soil leaving no air pockets.

7. Trees should be staked *only if necessary*.



http://greenspade.com/how-to-plant-a-tree

Figure 7. Guidance for proper tree planting

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