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Inundation Depth and Duration Impacts on Wetland Soils and Vegetation

State of Knowledge

Glenn M. Suir and Jacob F. Berkowitz

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Abstract

The following synthesizes studies investigating plant and soil responses to increased inundation in order to support ecosystem restoration efforts related to the alteration of natural wetland hydrodynamics. Specific topics include hydrologic regimes, soil response to inundation, and implications for vegetation communities exposed to increased water depths. Results highlight the important interactions between water, soils, and vegetation that determine the trajectory and fate of wetland ecosystems, including the development of feedback loops related to marsh degradation and subsidence. This report then discusses the knowledge gaps related to implications of inundation depth, timing, and duration within an ecosystem restoration context, identifying opportunities for future research while providing source materials for practitioners developing restoration projects.

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Preface

The work reported herein was conducted as part of the Ecosystem Management and Restoration Research Program (EMRRP), Project No. 494199, under Funding Account Code U4368951; AMSCO Code 031342. The EMRRP is sponsored by Headquarters, US Army Corps of Engineers (HQUSACE), and is assigned to the US Army Engineer Research and Development Center (ERDC) under the purview of the Environmental Laboratory (EL), Vicksburg, MS. The EMRRP is managed under the Civil Works Environmental Engineering and Sciences Office, Dr. Jennifer M. Seiter-Moser, Technical Director. Dr. Brook D. Herman was the acting Program Manager of the EMRRP. Program Monitor during this study was Mindy Simmons, HQUSACE.

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1 Introduction

1.1 Background

The US Army Corps of Engineers' (USACE) environmental mission includes restoration of ecosystems exhibiting degradation, with a focus on re-establishing and maximizing beneficial ecological processes supporting sustainable, resilient, and healthy systems under current and future conditions. Long-term and large-scale restoration and management of wetland ecosystems are largely driven by hydrologic processes and pulses and therefore have focused primarily on hydrologic alterations and management as a way of creating or enhancing wetlands while promoting ecological function. As a result, proper management of water level and salinity regimes remain key components of many wetland ecosystem restoration and management projects. Furthermore, given current and future climate trends, the need for additional water management to support ecological and navigation missions is anticipated.

Recent studies have focused on relative sea-level rise (RSLR) and the gradual inundation of marshes, where plants have the ability to adapt and respond (Cahoon, McKee, and Morris 2020; Fagherazzi et al. 2020; Dang et al. 2021). However, hydrologic alterations and water management actions that induce intermittent and instantaneous changes in inundation impose entirely different vegetation and soil stresses than the gradual inundation due to sea-level rise. Currently, high levels of uncertainty exist regarding the effects of hydrologic alterations and water management actions on wetland resources.

1.2 Objective

No comprehensive, inundation-induced vegetation mortality data across fresh, intermediate, brackish, and saline (FIBS) marsh systems currently exist. The inundation-related knowledge gaps of specific importance are the impacts of inundation depth and duration on wetland soil condition and function and the effects of water level on productivity and tolerance across FIBS wetland plants. To fill the gap in the data, researchers need to evaluate the direct and indirect effects of inundation depth and duration across a gradient of fresh to saline wetlands. This special report seeks to (1) characterize the general nature of hydrologic alterations on wetland

function, (2) identify the state of knowledge pertaining to inundation depth and duration impacts on wetland soils and vegetation, and (3) identify other knowledge gaps to provide increased certainty in future exploration and modeling of inundation factors in wetland ecosystems.

1.3 Approach

The effects of frequency, duration, and seasonality of inundation within FIBS marshes are not well understood. This data and knowledge gap has limited resource managers in their ability to make informed decisions about freshwater diversions and lacustrine-stage operations for ecosystem management. This effort provides a comprehensive search and review of all literature related to inundation effects on wetland soil and vegetation and then provides a summary outlining the current state of the science, identifying knowledge gaps to be addressed. The information obtained in this work will provide useful data supporting decision-making and modeling efforts associated with ecosystem restoration, planning, and management, including wetland productivity, floristic quality, accretion, soil strength, and biogeochemical cycling.

2 Review of Inundation-Related Impacts on Soil and Vegetation Processes

2.1 Hydrologic processes

2.1.1 Hydrologic regime

Wetlands are among the most productive and beneficial ecosystems in the world whose services are largely controlled by fluctuations in inundation and soil saturation (Jin et al. 2017; Suir and Sasser 2017). Wetlands provide benefits that range from regulating services (floods, drought, and land degradation) to supporting services (soil formation and nutrient cycling) to provisioning services (food and freshwater) to cultural services (recreational and aesthetic) and to maintaining high biological productivity, serving as critical habitat for fish and wildlife (Millennium Ecosystem Assessment 2003; USACE 2013).

Hydrologic regime, a primary driver within wetland ecosystems, significantly affects the response patterns of vegetated communities as well as ecosystem form and function. Water level changes, whether temporary or continuous, often occur as a result of seasonally fluctuating water tables, riverine flooding, storms, over-irrigation, inadequate drainage, and impoundment of water by dams (Kozlowski 1997; Steyer et al. 2005; Jackson, Ishizawa, and Ito 2009; Suir et al. 2014). Wetland plants exhibit a wide range of responses to hydrologic regime, especially when considering associated flooding frequency, duration, water depth, flow rates, water chemistry, nutrient availability, and soil and sediment characteristics (Kushlan 1989; Fennessy, Cronk, and Mitsch 1994; Mortsch 1998; Roberts and Marston 2011). The hydrologic regime can be thought of as a master variable, since it controls hydroperiod and critical nutrient and sediment delivery and defines many characteristics and processes of wetland ecosystems (Fennessy, Cronk, and Mitsch 1994; Jones et al. 2013). Hydrologic regimes are highly dynamic and can vary substantially through time and space (Jin et al. 2017). While hydrologic regime can affect wetland processes, wetlands also have been shown to influence flood effects by reducing wave action, retaining stormwater runoff, and providing buffers to populated areas and critical infrastructure (Highfield, Brody, and Shepard 2018).

Water regime and hydropattern are defined by the depth, duration, frequency, timing and predictability of soil saturation, and surface inundation in a wetland (Casanova and Brock 2000). Tidal wetlands, defined as wetland areas between mean lower water and mean higher water, are typically structured according to water regimes, ambient salinity levels, and the slope and elevation of the marsh (Park et al. 1989; Gough and Grace 1998; Todd et al. 2012; Short et al. 2016). Whereas the frequency of flooding, as well as the timing, duration, depth, and nutrients, contributes to plant community variability observed in river-dominant wetland systems (DeLaune, Pezeshki, and Jugsujinda 2005; Barrett, Nielson, and Croome 2010; Finlayson 2016).

As environmental gradients, inundation depth and duration influence soil conditions and plant establishment, survival, and colonization (Coops, Van Den Brink, and Van Der Velde 1996; Forbes, Alexander, and Dunton 2008; Howard and Rafferty 2006). The distribution and diversity of plant species is linked to the distribution of hydrogeomorphological processes and landforms (Bendix and Hupp 2000). For plants, hydrological patterns determine the zonation of vegetation, where low marsh is the zone inundated at least once daily for most of the year, and high marsh consists of the zone submerged less frequently (Blom et al. 1990; Davidson-Arnott et al. 2002; Tiner and Milton 2016). This zonation is primarily determined by inundation resistance (especially in the low marsh), competition (in the high marsh), and habitat plasticity (Blom et al. 1990; Ferreira and Stohlgren 1999; Luo, Song, and Xie 2008).

2.1.2 Plant and soil processes

Inundation depth, duration, and timing, in addition to zonation, involve a number of distinct forces that can act as discriminatory variables (separately, or in concert) to control important plant and soil processes (Davy, Noble, and Oliver 1990; Todd et al. 2010). Inundation depth and duration influence plant productivity, distribution, competition, richness, and reproductive biology while also controlling many soil characteristics (Ferreira and Stohlgren 1999; Casanova and Brock 2000). In some ecosystems, hydropattern remains a key determinant of critical abiotic and biotic processes (Kneitel 2014). Inundation timing and extent affects seed dispersal and other conditions required for vegetation germination, establishment, and persistence (Galat et al. 1998; Bendix and Hupp 2000).

Elevation and accretionary processes are also linked to hydrologic regime and are primary influences of plant processes, vegetation community composition, and species distribution (Davis 2016). Wetland plants have the ability to tolerate or adapt to episodic inundation events or counteract long-term increases in inundation (that is, sea-level rise) through accretionary processes, like soil surface elevation gains via biological and physical feedbacks (Kirwan and Megonigal 2013). Depth and duration of inundation, along with other intrinsic (plant species and nutrient content) and environmental conditions (temperature and presence of microbes and shredders), contribute to the wetland accretion feedback loop (Foote and Reynolds 1997). The aboveground vegetation traps suspended sediment during flood events, vegetation productivity (above- and belowground) contributions to soil organic matter accumulation, and flood waters assist in the breaking down of plant matter and influencing the rate at which those plant materials are decomposed—all of which increase the marsh elevation by above- and subsurface expansion (Anderson and Smith 2002; Kirwan and Megonigal 2013; Short et al. 2016).

2.1.3 Inundation events—natural and anthropogenic effects

For all wetlands, there are optimal inundation rates and depths that result in maximum function and productivity (Morris et al. 2002). Additionally, wetlands can achieve equilibrium conditions, which are related to wetland stability, where the system exhibits resiliency against changes in hydropattern (Morris et al. 2002). For natural systems, the pulse, amplitude, duration, and timing of inundation events typically result in beneficial impacts on wetland landscapes (Wittmann and Junk 2016). Although natural fluctuations in these hydrological elements can be important ecological drivers in wetland ecosystems, the increasing anthropogenic manipulation of hydrology (that is, timing, depth, and duration) and the exacerbation of climatic change forcing have resulted in substantial and detrimental changes in hydropattern worldwide (Miao, Zou, and Breshears 2008). The influence of altered hydrological dynamics is of growing interest, as increasing threats from human action and climate change will increase marshes' susceptibility (Todd et al. 2010; Tobias and Nyman 2017).

High levels of ecosystem biodiversity are often associated with naturally occurring intermediate levels of disturbance (Jones et al. 2013). However, hydrological modifications—which include channelization, diversions, and climate change—can alter hydrologic regimes beyond the normal range,

subsequently impairing ecosystem structure and processes (Steyer et al. 2008; Jones et al. 2013; Lai, Jiang, and Huang 2013). Generally, these impairments include increased frequency and severity of flooding, increased salinity stress, reduced sediment inputs, lowered water tables, reduced groundwater recharge, spread of invasive species, reduction in aquatic biodiversity, and increased wetland loss (Bunn and Arthington 2002; Kercher and Zedler 2004; Nyman et al. 2009; Glick et al. 2013; Galatowitsch 2016). Previous studies show that anthropogenically altered hydrology has reduced flow and overbank flooding in some systems and increased inundation durations in others (Flynn, McKee, and Mendelssohn 1995; Field and Philipp 2000; Beilfuss 2018). Anthropogenically induced global climate change is expected to increase the frequency and severity of inundation. Coastal regions like the lower Mississippi River Delta are expected to be most affected, with RSLR rates as high as 10 mm/yr¹ contributing to rates of marsh loss as high as 10 ha per month (Arnell and Liu 2001; Sherman and Gares 2002; Snedden, Cretini, and Patton 2015). Continued RSLR and associated increases in coastal inundation will be among the most significant impacts of climate change to coastal wetlands (Heberger et al. 2011). Although established wetlands have been shown to exhibit an equilibrium level related to tidal parameters and thus respond to natural changes in sea level and frequency of tidal inundation, the impacts of anthropogenically induced changes in tidal parameters on coastal wetlands are expected to show nonequilibrium rates of accretion and surface change (Davidson-Arnott et al. 2002; Cahoon et al. 2006). Ultimately, future combinations of degrading wetlands with altered hydrology are expected to increase marsh deterioration, erosion, and collapse (Couvillion et al. 2013).

Not all anthropogenic alterations of hydrology are detrimental. Some previous hydrologic management activities have sought to control inundation depth, frequency, and duration (Richardson 2017). Controlling these hydropattern parameters allows for the management of water resident times, sediment settling, salinity, and biogeochemical transformations, which are then used to increase plant production and floristic quality, restore wetland structure, and enhance habitat (David

1. For a full list of the spelled-out forms of the units of measure used in this document, please refer to *US Government Publishing Office Style Manual*, 31st ed. (Washington, DC: US Government Publishing Office, 2016), 248–52, <https://www.govinfo.gov/content/pkg/GPO-STYLEMANUAL-2016/pdf/GPO-STYLEMANUAL-2016.pdf>.

1999; Randall and Foote 2005; Drinkard et al. 2011; Bond et al. 2014; Richardson 2017; Sonnier et al. 2018).

2.1.4 Negative impacts of increased inundation

High water tables and surface inundation are generally not stressful to wetland plants, which exhibit morphological, physiological, and reproductive adaptations to life in saturated soils (Otte 2001). However, hydrologic regime shifts beyond the normal range induce plant stress—where negative concurrent challenges to metabolic functions increase according to changes in inundation depth, frequency, duration, and timing (Kozlowski 1997; White and Ganf 2002; Jackson and Colmer 2005; Wang et al. 2014). Previous studies have noted a wide range of soil and macrophyte responses to inundation depth and duration. These responses are typically observed in aboveground (shoots), belowground (roots), reproduction, and soil conditions. Saturated-soil impacts on macrophytes include decreased availability of oxygen and carbon dioxide (via respiration and reduced diffusion), nutrient uptake, stomatal conductance and rate of photosynthesis, and changes in rhizome functions, mode of reproduction, and carbon allocation (Fennessy, Cronk, and Mitsch 1994; DeLaune, Pezeshki, and Patrick 1987; Mauchamp, Blanch, and Grillas 2001; Jackson and Colmer 2005; Pessarakli 2016).

Inundation depth and duration tolerance in plants varies with plant species, genotype, rootstock, and age as well as the timing and composition of the water inputs (Kozlowski 1997). Increasing inundation depth and duration can also reduce light penetration and water quality (that is, temperature and salinity), which can alter biomass allocation to seed, inhibit seed germination, and negatively affect seed banks (Zedler 1983; Rejmanek, Sasser, and Gosselink 1987; Peterson and Baldwin 2004). Furthermore, prolonged inundation can deprive plants of critical resources because of the indirect effects on plant physiology (that is, alteration of metabolic pathways, generation of toxic byproducts) (Lucas and Berry 1985; Jackson, Ishizawa, and Ito 2009). In addition to reduced photosynthesis, prolonged soil saturation can also curtail gas exchange from air, water, and soil to shoots and roots (Armstrong, Brändle, and Jackson 1994). These conditions alter cellular metabolism, as well as reduce root growth and respiration, often resulting in complete or near-complete loss of roots (Gleason and Zieman 1981; Ernst 1990; Armstrong, Brändle, and Jackson 1994; Sauter 2013).

Plants living in flooded soils often face two major problems: (1) obtaining an adequate oxygen supply for their roots and rhizomes and (2) protecting themselves against anoxic conditions (that is, generation of phytotoxins) (Končalová 1990, 128; Flynn, McKee, and Mendelssohn 1995). As soil saturation and inundation duration increases, available oxygen is removed from the water column and soil pore spaces via respiration, resulting in the development of anoxic or reduced soil conditions (Vepraskas, Polizzotto, and Faulkner 2016). Prolonged exposure to reduced soil conditions adversely affects plant physiological functions such as nutrient uptake, cellular water balance, activity of photosynthetic enzymes, gas exchange, photosynthetic electron transport, photosystem activity, hormonal imbalances, and accumulations of potentially toxic metabolic byproducts (Davy, Noble, and Oliver 1990; Howard and Mendelssohn 1995; Li, Yang, and Li 2007). The consequences of these changes in the plant-soil system include decreased survival, growth, and productivity, which can induce shifts in species composition, distribution, and successional patterns (DeLaune, Pezeshki, and Patrick Jr. 1987). Inundation impacts can further alter large-scale ecological processes, which determine the rate of wetland ecosystem goods and services delivery (for example, biodiversity, floristic quality) (Kercher and Zedler 2004; Maltby and Acreman 2011).

Prolonged inundation can also induce a negative feedback loop, in which excessive soil saturation impairs wetland vegetation primary productivity. Decreased productivity reduces sediment trapping efficiency and diminishes organic matter accumulation, ultimately increasing submergence and surface inundation (Snedden, Cretini, and Patton 2015). The failure or inability of plants to adjust or adapt to these changing hydrologic conditions can result in substantial marsh and habitat deterioration (Lessman et al. 1997).

2.1.5 Plant response and adaptations

While soil saturation or inundation is commonly regarded as inhibitory to plant growth, a sizeable minority of extant taxa have adapted metabolic and morphological adaptations allowing for robust growth in wetlands (McKee, Mendelssohn, and Burdick 1989; Howard and Mendelssohn 1995; Jackson 2004; Jackson, Ishizawa, and Ito 2009). Even within these specialized plants, the adaptations for exposure to regular or permanent hydropatterns differ significantly from those subjected to more irregular inundation (Blom et al. 1990). Plants within this highly diverse group

exhibit a wide range of growth habits, physiological attributes, and degrees of water tolerance as a result of adaptive plant and environment interactions (Jackson, Ishizawa, and Ito 2009). Two primary groups exist: the flood tolerant plants, which consist of plants that can withstand sudden surface inundation, and wetland tolerant plants, which consist of plants with the ability to survive and compete under permanently saturated soil conditions or where water regimes are outside of the normal range (that is, increased duration, intensity, and frequency) (Justin and Armstrong 1987; Ernst 1990).

In wetland vegetation, soil saturation induces specific above- and belowground stresses that can adversely affect plant establishment, survival, and growth (Blom et al. 1994; Blom 1999; Crossle and Brock 2002; Van Eck et al. 2006; Mony et al. 2010, 99). Adaptive responses in morphology and reproductive strategy are developed in wetland plants to overcome or minimize these effects (Wang et al. 2014). Under severe hydrologic conditions, a number of plant and ecosystem adaptations may take place to overcome or compensate for water-related stress (Laan and Blom 1990): life-cycle strategies (seedbanks, dormancy status, germination, vivipary) (Blom 1999); signaling and hormone changes (ethylene, auxin, and gibber) (Voesenek et al. 1996); morphological adaptations and acclimations (leaf anatomy, aerenchyma, adventitious roots, rapid early shoot growth, stem and lenticel hypertrophy) (Gomes and Kozlowski 1980; Jackson and Colmer 2005); physiological adaptations (anaerobic respiration, malate production) (Mendelssohn, McKee, and Patrick 1981); and ecosystem adaptations (accretion, transition, avoidance) (van Wesenbeeck et al. 2017).

The life-cycle strategies in wetland-tolerant plants include biennial and perennial life histories, the ability to germinate in the intervals between successive inundation events, and the ability to maintain large persistent seed banks (Blom et al. 1990). Water-induced hormone changes (that is, increasing ethylene) often signal and initiate other adaptations, like stem and leaf elongation (Kirkman and Sharitz 1993). Stem and leaf elongation, and petiole formation, are responses to saturated soil conditions (low oxygen availability) and are used as a means of exposing stems and leaf blades to the atmosphere, thereby increasing gas exchange (Grace 1989; Wang et al. 2014). Adventitious roots and root elongation are other morphological adaptations that increase the root porosity, promote oxygen exchange, and improve physical stability for life in saturated soils (Justin

and Armstrong 1987; Mony et al. 2010; Sauter 2013). Another important hydrophytic plant adaptation is the formation of aerenchyma (Colmer and Voesenek 2009). Aerenchyma provide “an unobstructed pathway for atmospheric oxygen transport from the leaves to the tips of the roots,” lowering the metabolic demands of the plant and reducing the volume of respiring tissue (Teal and Kanwisher 1966; Arenovski and Howes 1992; Maricle and Lee 2002, 110). These acclimations, along with metabolic adaptations (that is, switch from alcoholic fermentation) and accretionary processes (counteracting SLR), are all mechanisms that allow wetland plants to tolerate a wide range of flood stresses (McKee, Mendelssohn, and Burdick 1989; Sandi et al. 2018).

2.2 Soils studies

2.2.1 Implications of increased inundation on wetland soils

Wetland soils (also known as hydric soils) encompass a diverse array of sediment types, forms, and functions (Vepraskas and Craft 2016) that support some of the most productive ecosystems on the planet (Rocha and Goulden 2009). These soils occur in a variety of landscape positions, including freshwater and tidal fringes, riverine floodplains, backswamp deposits, depressional areas, extensive flats, slope-discharge locations, and in areas with a restrictive layer capable of supporting high water tables (Smith, Noble, and Berkowitz 2013). Wetland soils remain an area of ongoing research because of the high productivity of many wetland soils for agriculture (often requiring drainage or manipulation; Skaggs, Breve, and Gilliam 1994), the ability to reduce pollution and improve water quality (Gilliam 1994), the capacity to sequester large amount of carbon (Chmura et al. 2003; Suir et al. 2019), and the support of valuable habitat for aquatic and terrestrial species including many threatened and endangered species (Flather, Knowles, and Kendrall 1998).

While the presence of water serves as a master variable driving wetland soil function, potential long-term implications of increased inundation remain unknown. Recent work has focused on the implications of increasing sea-level rise and storm surge on wetland soil systems as storm severity and frequency is expected to increase (Hauser, Meixer, and Laba 2015). For example, Kirwan and Megonigal (2013) investigated tidal wetland stability under sea-level rise scenarios; Turner et al. (2006) reported changes in wetland soils following hurricanes Katrina and Rita; Cahoon (2006) evaluated storm-induced sediment dynamics in wetlands;

and Howes et al. (2010) examined hurricane-initiated wetland failure in marshes. These studies (and others) highlight the need for additional research regarding the impact of increasing sea levels, storm frequency and intensity, and surface inundation of wetlands soils across broad geographic areas. In response, the following sections discuss wetland soils and factors related to potential shifts in wetland soil function induced by increased inundation periods. Specifically, a review of wetland soil formation, morphology, and classification is provided, followed by sections addressing typical wetland soil hydroperiods and hydropatterns, ecological responses to soil inundation (morphology, chemistry, physical effects), and approaches to monitoring wetland soil conditions in field settings. Knowledge gaps and opportunities to improve the management of wetland soils under increasing inundation regimes are also identified.

2.2.2 Defining wetland soils and identifying them in the field

In order to examine potential impacts of increased inundation on wetland soils, a discussion of how wetland soils are defined, identified, and classified is warranted, as acknowledging differences between wetlands and uplands (and various wetland types) will dictate soil response to changes in environmental conditions. As a result of wetland soils diversity, they have been defined and classified according to their function, landscape position, hydrologic regime, and morphology (Brinson 1993; Tiner 2017). Operationally, wetland soils are defined as soils that “formed under conditions of saturation, flooding, or ponding long enough during the growing season to develop anaerobic conditions in the upper part”.¹ This definition has been used to identify wetland soils according to characteristic morphologies that form in response to prolonged anaerobic conditions via the development and application of field indicators (United States Department of Agriculture–Natural Resources Conservation Service 2016). In general, wetland soil field indicators use readily observable characteristics to conduct on-site evaluations on the basis of (1) the accumulation of soil organic matter, (2) the chemical reduction and translocation of iron or manganese oxides or both, or (3) the evolution of hydrogen sulfide gas (Figure 1; United States Army Corps of Engineers 2012). As a result, practitioners can rapidly conduct field evaluation of wetland soils using direct observations of the way a soil looks, feels, and smells (Vasilas and Berkowitz 2016). Field evaluations of hydric soils typically focus on the upper 50 cm of the soil surface, as this is the zone

1. Docket No. NRCS–2011–0026, 77 Fed. Reg. 40 (Feb. 29, 2012), 12234–12235.

having the greatest influence on the presence and vigor of rooted herbaceous and woody vegetation (National Research Council 1995). However, in some cases deeper investigation is required (for example, Mollisols; Thompson and Bell 2016).

Figure 1. Left: example of soil organic matter accumulation from a wetland soil on the island of Hawaii; right: soil profile displaying the effects of iron and manganese oxide reduction and translocation resulting in the presence of rust-colored features and the predominance of a gray soil matrix color. Photos reproduced from United States Department of Agriculture–Natural Resources Conservation Service (2016) and Vasilas and Berkowitz (2016).

2.2.3 Using soil maps to identify wetland soil areas

The Natural Resources Conservation Service (NRCS) publishes regional Soil Surveys that contain the most comprehensive information for locating hydric soils. The Natural Resources Conservation Service maintains the official Soil Surveys on Web Soil Survey, available at

<http://websoilsurvey.sc.egov.usda.gov/>. Web Soil Survey uses information on soil properties from the NRCS database to generate hydric soils lists and interpretive maps identifying areas that likely contain hydric soils. The Natural Resources Conservation Service also creates a National List of Hydric Soils (ftp://ftp-fc.sc.egov.usda.gov/NSSC/Hydric_Soils/Lists/hydric_soils.xlsx) representing a yearly compilation of all map units through to contain hydric soils throughout the US.

The National List of Hydric Soils uses four criteria for evaluating soil map unit components to determine whether they classify as hydric soils in the Natural Resources Conservation Service database. The criteria are as follows:

1. All Histels except Folistels and Histosols except Folists; or
2. Map unit components in Aquic suborders, great groups, or subgroups, Albolls suborder, Historthels great group, Histoturbels great group, or Andic, Cumulic, Pachic, or Vitrandic subgroups that:
 - a. Based on the range of characteristics for the soil series, will at least in part meet one or more Field Indicators of Hydric Soils in the United States, or
 - b. Show evidence that the soil meets the definition of a hydric soil;
3. Map unit components that are frequently ponded for long duration or very long duration during the growing season that:
 - a. Based on the range of characteristics for the soil series, will at least in part meet one or more Field Indicators of Hydric Soils in the United States, or
 - b. Show evidence that the soil meets the definition of a hydric soil; or
4. Map unit components that are frequently flooded for long duration or very long duration during the growing season that:
 - a. Based on the range of characteristics for the soil series, will at least in part meet one or more Field Indicators of Hydric Soils in the United States, or
 - b. Show evidence that the soils meet the definition of a hydric soil.
(National Technical Committee for Hydric Soils 2012, 1)

If a map unit component meets any of the criteria, then that component classifies as hydric in the Natural Resources Conservation Service soils database, and all map units that contain that component will be identified as containing hydric soils on the National List of Hydric Soils and other hydric soils lists (Figure 2).

Web Soil Survey allows researchers to produce interpretive tables and maps for an area according to soil characteristics. The survey provides valuable off-site information concerning the likelihood that a wetland soil exists in an area of interest. Within Web Soil Survey, the Hydric Rating by Map Unit represents a valuable tool for identifying areas that likely

contain wetland soil (Figure 3). This approach categorizes map units according to the percentage of that map unit considered a hydric soil. The interpretive map uses a five-category system of hydric (100%), predominately hydric (66% to 99%), partially hydric (33 % to 65%), predominately nonhydric (1% to 32%), and nonhydric (0%). The Web Soil Survey also produces several other soil reports on hydric soils, including a list of all map unit components in an area of interest that classify as hydric as well as their percentages.

Figure 2. An explanation of the various hydric soils lists provided by the Natural Resources Conservation Service (National Technical Committee for Hydric Soils 2012).

What's Included on the Hydric Soils Lists (NTCHS Website, Nov, 11, 2013)

National List

“Hydric Soils of the United States” is a compilation of all map units with either a major or minor component that is at least in part hydric. This could include components that are soil series, components that are classified at categories higher than the series level in Soil Taxonomy, and miscellaneous land types. Because the list includes both major and minor (small) percentages for map units in some cases most of the map unit may not be hydric. Also, some components may have a range of characteristics that cross the hydric/non-hydric soil boundary and, therefore, only a portion of that component may in fact be hydric. The list is useful in identifying map units that may contain hydric soils. The National List is generated once per calendar year (usually in January or February) to satisfy legislated mandates.

The Natural Resources Conservation Service (NRCS) Hydric Soils Lists, available here, are by soil map unit component. Detailed and up-to-date hydric soil lists (for example, by soil survey area map unit component) are maintained by the [NRCS State Offices](#) and local field offices and can be downloaded from the Web Soil Survey at <http://websoilsurvey.nrcs.usda.gov/>.

State Lists

The state lists are subsets of the national hydric soils list by state. For more detailed state lists by map unit, contact the appropriate [NRCS State Office](#).

Local Lists

NRCS has developed local lists of map units that contain hydric soils for each county, parish or soil survey area in the United States. These local lists are available at the [NRCS State Offices](#), local NRCS field office, and on the Web Soil Survey at <http://websoilsurvey.nrcs.usda.gov/> and are the preferred lists for use in making preliminary wetland determinations. Information from Web Soil Survey is the most up-to-date information as well as the official soil survey information. Since the National List is only compiled once a year and Web Soil Survey may be updated on a more frequent basis, Web Soil Survey should be deferred to when there is a discrepancy.

Figure 3. Maps generated by Web Soil Survey depicting Hydric Rating by Map Unit at (top) large and (bottom) fine scales. Note that wetland soils are more likely to occur in the red and orange shaded portions of each map (outside of Sacramento River corridor and in center, south, and east of map, respectively), and field investigations for wetland soils should focus in those areas.

2.2.4 Wetland soil morphology

Evaluating soil morphological features informs studies of wetland inundation and soil responses to increased inundation frequency, as soil physical characteristics reflect environmental conditions at both short (hours to days) and long (century to millennia) time scales. The establishment of anaerobic conditions occurs following periods of saturation, in which water fills available pore spaces in the soil matrix (Buol et al. 2011). Once soils saturation occurs, the rate of oxygen exchange from the atmosphere into the soil is greatly reduced, since the diffusion rate of oxygen through soil water is approximately 10,000× slower than through air (Pezeshki 2001). As aerobic microbial respiration continues, any remaining oxygen in soil solution is depleted, resulting in the onset of anaerobic conditions (Reddy and DeLaune 2008).

Once established, soil microbial communities use alternative electron acceptors during anaerobic respiration resulting in a decline of soil oxidation-reduction potentials (that is, redox) and the sequential chemical reduction of redox-active species (for example, nitrate, manganese oxides, ferrous iron, sulfate, and carbon dioxide; Faulkner 2004). The onset of anaerobic conditions and chemical reduction leads to a number of changes in the biogeochemistry of hydric soils (Reddy and DeLaune 2008). These changes alter the morphology of the soil and result in observable patterns and features used to identify and delineate hydric soils within the landscape (Tiner 1999; National Research Council 1995). The paragraphs below address common morphological features and terminologies used in hydric soil research (Vasilas and Berkowitz 2016). The anaerobic and reduced conditions associated with hydric soils lead to decreases in microbial respiration efficiency and organic matter decomposition (Lee 1992; Schink 1988). As a result, organic carbon accumulates in hydric soils (Gambrell and Patrick 1978; Mausbach and Richardson 1994). Several wetland soil characteristics are related to organic matter accumulation, including the development of Histosols, histic epipedons, organic bodies, and organic-matter coatings on mineral soil particles.

Histosols are defined as organic soils exhibiting 40 cm or more of organic soil material within the upper 80 cm (United States Department of Agriculture–Natural Resources Conservation Service 2016). Histosols also include soils containing organic materials of any thickness if the organic soil materials are underlain by rock or fragmental materials with interstices filled with organic soil materials. See Soil Taxonomy (Soil

Survey Staff 1999) for a complete definition. Histosols are found throughout the United States. However, Histosols are common across the north-central United States, including large expanses of organic flats in northern Minnesota, large portions of southeastern and northern Alaska, the Atlantic and Gulf coastal plains, and peninsular Florida (Moore and Bellamy 1974; Ping, Michaelson, and Kimble 1997; Ping et al. 2002). A histic epipedon is defined as a thick (20–60 cm) organic soil horizon that saturates with water during some period of the year (unless artificially drained). The organic soils materials must be underlain by a mineral soil with a chroma of 2 or less (United States Department of Agriculture–Natural Resources Conservation Service 2016). Histic epipedons predominantly occur in northern and high-elevation regions of the United States, although they also occur in other areas subject to prolonged soil saturation (Quideau 2002; FitzPatrick 1984). They are also found across many portions of the northeastern states, including marsh ecosystems as well as in coastal plain portions of the south Atlantic states (Bridgham and Richardson 1993).

The term *organic bodies* describes accumulations of highly decomposed (muck) organic soil material occurring at the tips of fine roots. To identify organic bodies, remove a large clump of roots from the soil and gently shake to remove loose soil material. If clumps of soil material remain on the tips of fine roots, rub them between the thumb and forefinger to determine whether they are composed of highly degraded organic soil material (that is, muck; Vasilas and Berkowitz 2016). If the organic material present is classified as muck, the soil would qualify as organic bodies. Organic bodies occur in the southern Gulf of Mexico and the Florida Panhandle as well as other areas along the Atlantic coast including coastal Georgia, Alabama, Georgia, and the Carolinas. These features are often associated with wet pine flat landscapes (Florida Department of Environmental Protection 2011). Sand grains may become coated or masked with organic material under anaerobic and reduced soil conditions associated with extended periods of saturation (Lindbo, Vepraskas, and Rhoton 2001). The features are called *masked sand grains* because the organic matter covers the sand particle in black organic material, hiding the original soil color (Vepraskas 2013). Other potential masking agents include silicate clays, iron, aluminum, or some combination of these; however, organic matter masking remains most common in hydric soils. Typically, if at least 70% of sand grains in a sandy soil appear masked with organic matter, the soil is considered to undergo extended periods of

saturation and reduced conditions. When examining coated or masked sand grains, using a 10× magnification hand lens will determine whether the 70% threshold is met or exceeded. When observed with the naked eye, the percentage of masked grains appears closer to 100% (Vasilas and Berkowitz 2016).

In addition to wetland soil features related to organic matter accumulation, many hydric soils exhibit characteristic morphologies associated with the reduction and translocation of iron, manganese, and sulfur. As noted above, the anaerobic and reduced conditions associated with wetland soils result in the chemical reduction and translocation of redox-active soil constituents (Reddy and DeLaune 2008). These elements may reprecipitate following the onset of aerobic conditions associated with decreasing water tables, producing rust-colored iron oxide coating along pore linings and within the soil matrix (Megonigal, Patrick, and Faulkner 1993; Megonigal, Faulkner, and Patrick 1996; Vepraskas 2004). Additionally, areas where iron and manganese have been removed create areas of metal oxide depletions that display the underlying gray colors of the naked mineral grain (Figure 4). As a result, iron and manganese often develop a characteristic pattern of redoximorphic features associated with hydric soils (Vepraskas and Sprecher 1997). Sulfur reduction results in an olfactory indication of chemical reduction (Castro and Dierberg 1987). Together, the reactions of these elements and compounds, and associated terminology, aid in the identification of wetland soils.

Some of the most common morphological features of wetland soils are redoximorphic (that is, redox) concentrations of iron, manganese, or iron-manganese complexes. As described above, redoximorphic concentrations occur where bodies of iron or manganese or both accumulate through the process of dissolution, translocation, and reprecipitation (Fanning and Fanning 1989). Concentrations of iron are typically red, orange, brown, or yellow depending on the form of iron, while concentrations of manganese are black (Birkeland 1999). If the iron and manganese concentrate together, the color will be blackish red, purple, or black. Manganese oxide concentrations can be positively identified in the field by applying a solution of 2% hydrogen peroxide. If the application of hydrogen peroxide causes the soil to effervesce, the presence of manganese redoximorphic concentrations is confirmed (Needelman, Ruppert, and Vaughn 2007). The compound α - α -dipyridyl is a colorless liquid dye that produces a pink or red color when placed on a saturated and chemically reduced soil

containing ferrous iron (National Technical Committee for Hydric Soils 2012). The dye provides a positive indicator of anaerobic conditions in a soil if a reaction is observed. However, because of several limitations associated with the dye (for example, soils must be saturated for several weeks prior to application, uncertain shelf life of the dye) a lack of a reaction does not provide a negative indicator of reducing conditions or the presence of a hydric soil (Vepraskas 2004). Further information on the application and use of α - α -dipyridyl dyes is provided in Berkowitz et al. (2017c), who reported that the liquid dye as well as paper strips implanted with the dye proved useful for the identification of anaerobic conditions in a field context.

Figure 4. Example of redoximorphic features in which anaerobic conditions have induced the formation of iron depletions and concentrations resulting from the dissolution, translocation, and reprecipitation of iron oxides. Photo from United States Department of Agriculture–Natural Resources Conservation Service (2016).

In soils subject to extended periods of saturation and anaerobiosis, a variety of sulfur compounds become reduced, and hydrogen sulfide gas is produced (Hedin, Hammack, and Hyman 1989). The odor of hydrogen sulfide gas provides another indicator of anaerobic and chemically reduced conditions in soils. Hydrogen sulfide exhibits a strong odor, most often

described as the smell of rotten eggs (Devai and DeLaune 1999). Hydrogen sulfide odors often happen in coastal and tidal areas, where regular inundation occurs throughout the tidal cycle and ample sulfur compounds exist (Koch, Mendelssohn, and McKee 1990). The presence of hydrogen sulfide in soil under anaerobic conditions results in an obvious, fetid odor that can often be detected prior to excavation or shortly after excavation of the soil pit. If an odor is observed only when the soil is held closely to the nose, it is not hydrogen sulfide odor. Also, hydrogen sulfide odor can only exist in the presence of saturated soils and anaerobic conditions. Therefore, the hydrogen sulfide odor will not occur in unsaturated soils.

2.2.5 Wetland soil taxonomy

Being able to classify wetland soils provides a basis for differentiating between soil types and targeting appropriate sampling approaches to investigate the influence of inundation patterns on soil properties and functions. In soil taxonomy, wetland soils are classified in several ways, including both soils dominated by organic soil materials and those exhibiting color patterns related to constituent translocation in mineral soil materials (Presley, Sprecher, and Kenney 2016).

As discussed above, the soil order Histosols encompasses many wetland soils composed of organic materials (excluding permafrost) that contain 12% to >18% soil organic carbon that typically remain saturated for >30 days during the growing season (Figure 5; Soil Taxonomy 1999). The organic soil horizon must occupy >40 cm of the upper 80 cm of the soil profile (United States Department of Agriculture–Natural Resources Conservation Service 2016). Histosols occur across a number of landscape features, including cold areas such as boreal forests in Alaska, poorly drained soils in the southeastern coastal plain, large freshwater wetland systems such as the central valley of California and the Florida Everglades, high-altitude freshwater bogs and fens, and in freshwater and tidal fringes near oceans and lakes (Buol et al. 2011). The organic matter associated with Histosols occurs in the form of peat (fibric), mucky-peat (hemic), and muck (sapric) materials representing poorly decomposed, partially decomposed, and well-decomposed soil organic materials, respectively (Figure 5; Table 1).

Figure 5. Relationship between organic carbon content and percent clay used to differentiate between organic and mineral soil materials. Adapted from Vasilas and Berkowitz (2016).

In principle, Histosols and other wetland soils identified based upon the presence of organic carbon accumulate carbon at rates that exceed microbial decomposition (Bohn 1976). Notably, the build-up of organic carbon is driven more by the slow rate of carbon decomposition than by the rate of biomass production (Mausbach and Richardson 1994). The process of organic matter accumulation is referred to as *paludification* or *paludization*, and once formed, organic soils display low rates of hydraulic conductivity, creating a positive feedback loop in which Histosol formation supports additional carbon storage (Malmer 1975; Hartshorn, Southard, and Bledsoe 2003). The carbon accumulation results in the development of thick organic soil horizons at or below the soil surface, as observed via the presence of identifiable plant materials or dark slippery organic matter deposits. Many Histosols display low bulk density (often well below 1 g/cm³), with poorly decomposed peat soils exhibiting densities as low as 0.02 g/cm³, increasing to 0.3 g/cm³ as organic materials decompose and soil void space is filled over time (Boelter 1965, 1969). As a result of these characteristics, natural and anthropogenic alterations of Histosols and other organic soils can have dramatic implications for ecosystem health, infrastructure, and land-use practices (Stephens 1956). For example, drainage of organic soils in Florida resulted in >1.8 m of ground subsidence with 50 years of soil disturbance (McDowell, Stephens, and Steward 1969). Histosols are categorized into suborders according to the degree of organic matter decomposition and hydroperiod. For example, Folists are organic soils that remain saturated for <30 days per year;

Wassists remain saturated >21 hours per day; and Saprist, Hemist, and Fibrists represent well decomposed, moderately decomposed, and poorly decomposed materials, respectively (Boul et al. 2011).

Table 1. Strategy for determining organic soil texture categories. After Vasilas and Berkowitz (2016).

Organic soil texture	Prior to 8–10 rubs	After 8–10 rubs
Muck	<33%	<17%
Mucky peat	33%–67%	17%–40%
Peat	>67%	>40%

Wetland soils dominated by mineral constituents are classified according to the presence of poorly drained and very poorly drained classes (Presley, Sprecher, and Kenney 2016) and those exhibiting aquic soil moisture regimes as described in *Soil taxonomy* (1999). Very poorly drained soils maintain water at or near the soil surface during much of the growing season. Internal free water is shallow and persistent or permanent. Commonly, very poorly drained soils occupy depressional or flat landscape positions, including prairie potholes, pinewood flats, and coastal marshes. Poorly drained soils periodically or for long periods remain wet at shallow depths during the growing season. Internal free water is shallow or very shallow and common or persistent. The soil in poorly drained systems, however, is not continuously wet and often experiences seasonal fluctuation in the water table. The high water tables commonly result from low or very low saturated hydraulic conductivity class or persistent rainfall, or a combination of both factors. Agricultural production of most crops remains severely limited in most poorly and very poorly drained soils without the implementation of artificial drainage (Holden, Chapman, and Labadz 2004).

Aquic moisture regimes define the period of the growing season when soil moisture persists in the upper portion of the soil profile for long enough to induce a reducing regime that is virtually free of dissolved oxygen because it is saturated by water (Soil Survey Staff 1999). Notably, the time required for a soil to remain saturated before it is said to have an aquic moisture regime is not precisely quantified, but the duration must be at least a few days, because it is implicit in the concept that dissolved oxygen is virtually absent (National Research Council 1995). Because dissolved oxygen is removed from groundwater by respiration of microorganisms, roots, and soil fauna as described above, it is also implicit that soil saturation occurs

during the growing season. Operationally, the United States Army Corps of Engineers (2005) has defined the minimum period required for wetlands soil saturation as 14 consecutive days during the growing season at a return interval (that is, frequency) of 5 years in 10, or a 50% probability.

The presence of aquatic conditions occurs across a number of soil orders (and suborders) including Alfisols (Aqualfs), Andisols (Aquands), Inceptisols (Aquepts), Mollisols (Aquolls), Oxisols (Aquox), Spodosols (Aquods), Ultisols (Aquults), and Vertisols (Aquerts) (Boul et al. 2011). Many soils containing permafrost (that is, Gelisols) also display wetland characteristics and may be dominated by either organic (Histels) or mineral (Orthels and Turbels) soil components (Bridgham et al. 2016). Additionally, the presence of other soil moisture regimes does not preclude the presence of wetland soils within the landscape. For example, many seasonally ponded soils exist under xeric (that is, Mediterranean) climatic conditions in central and southern California, where seasonal rainfall patterns supply water to wetlands during the winter and spring followed by gradual drying during the summer and autumn seasons (Bauder 2005; Rains et al. 2006). Other soil moisture regimes that support wetlands include udic and, to a lesser extent, ustic conditions (Miller, Boettinger, and Richardson 2016). In rare cases, wetland soils may persist in low landscape positions with perched tables under aridic soil moisture regimes (Moormann and Van de Wetering 1985).

2.2.6 Wetland soil hydroperiods and hydropatterns

Understanding patterns of wetting and drying cycles is essential to identifying changes in wetland soils resulting from increasing inundation frequency and duration. The hydroperiod and hydropattern of a wetland are considered controlling or master variables on soils, flora, fauna, and biogeochemical cycling within wetlands (Jackson, Thompson, and Kolka 2014). Hydroperiod refers to the duration of soil saturation or surface-water inundation within a wetland, while incorporating both the duration and frequency (or time series) of water-table or surface-water fluctuations. For example, much work evaluating natural and disturbed wetland hydrology has occurred in the Florida Everglades, in which the re-establishment of natural hydroperiods and patterns has been highlighted as an important factor for wetland characterization and restoration (Acosta and Perry 2001, 2002; King et al. 2004). Jackson, Thompson, and Kolka (2014) provide a list of terms related to wetland hydropattern and soil saturation intervals (Table 2), and Wissinger (1999) provides a

graphical representation of hydroperiods associated with various hydrologic regimes and wetland types.

Hydroperiods range from short-duration infrequent event (for example, ephemeral pools) to semipermanently saturated fringe wetlands such as tidal marshes (Jackson, Thompson, and Kolka 2014). Figure 6 presents a variety of hydropatterns and -periods from recent case studies in which a single, continuous high water table event (>90 days) occurred along a freshwater lacustrine fringe wetland in comparison to short duration (~20 day) high water table and inundation patterns with interstitial periods during which groundwater levels fell below 50 cm for extended periods (Berkowitz and Sallee 2011; Berkowitz, Pietroski, and Currie 2017a). Understanding wetland soil hydrology and how patterns of soil saturation and inundation are affected by changing environmental conditions remains essential for resource management and restoration (Jackson, Thompson, and Kolka 2014; Leeds, Garrett, and Newman 2009).

Table 2. List of terms related to wetland soil saturation duration and pattern.

Term or metric	Description	Example
Duration of inundation period	Annual period of soil surface saturation	Miller and Zedler (2003)
Pattern of wetland hydrology	Wetlands may have equivalent inundation periods (that is, number of days), but vary in the distribution of high water table events. For example, one wetland may exhibit a single 50-day period of inundation while another wetland undergoes five separate 10-day inundation events	Meeker, Wilcox, and Harris (2018)
Stage duration frequency	Frequency of time a water level exceeds a threshold value	Zhang, Thomas, and Mitsch (2017)
Stage excursion frequency	Histogram displaying the number of stage excursions exceeding a given threshold	Feng, Brubaker, and McCuen (2017)
Stage recurrence curve	Annual probability of exceedance or return interval of annual peak stage	Vogel and Fennesey (1995)
Mean water-level fluctuation	How often the water level rises and falls, informing the degree of flashiness within the system	van der Valk (2005)
Daily, monthly, or seasonal water table level	A simple plot of water table level over time	Berkowitz, Page, and Noble (2014).

Figure 6. Comparison of varying hydropatterns identified in a freshwater fringe wetland soil in Michigan (left panel; growing season data 2009; Berkowitz and Saltee 2011) and a formerly irrigated wetland in the Central Valley of California (wet season data 2015–2016; Berkowitz, Pietroski, and Currie 2017a). Note that the first panel displays a single continuous high water table event in response to seasonal increases in lake levels, while the second panel shows short-duration increases in water because of precipitation events with a dry-down period during which groundwater levels decreased to <50 cm. Scales are in inches (1-inch equals 2.54 cm).

2.2.7 Wetland soil responses to changes in hydropattern

In order to identify changes in wetland soil conditions resulting from alteration of natural or historic patterns of hydrology, practitioners have developed a number of tools and approaches to monitor soil processes. These tools and approaches include measurements of water table level, oxidation-reduction potentials, surface elevations and sediment deposition patterns, shear strength and basic soil physical properties, microbial activity, and biomass. Other more specific measures are also used, but the following section focuses on the most common and basic approach to wetland soil evaluation.

Water table and soil moisture measurements are essential to determine how a wetland soil is functioning and to detect changes in soil conditions over time (Shaffer, Kentula, and Gwin 1999). Most on-site water table measurements use pressure transducers, which determine the water table level within shallow surface-water wells (Figure 7). Wetland hydrology is generally considered present when the water table remains within 25 cm of the soil surface for a period of 14 consecutive days during the growing season. United States Army Corps of Engineers (2005) provides details on the construction, design, installation, monitoring, analysis, and

interpretation of groundwater data. Other approaches include repeated measures of water levels within an unlined borehole, installation of tensiometers, direct measures of soil moisture content, and hydrologic modeling (National Research Council 1995; Arndt, Emanuel, and Richardson 2016). Additionally, a wide variety of remote sensing techniques have developed to evaluate surface-water inundation and soil saturation. These approaches range from interpretation of aerial imagery (Suir, Saltus, and Barras 2011; Suir et al. 2013; Berkowitz, Pietroski, and Krenz 2016) to application of radar, satellite, and other approaches (Smith 1997; Lillesand, Kiefer, and Chipman 2014; Prigent et al. 2011; Broussard, Suir, and Visser 2018).

Figure 7. Diagram of water table monitoring well used to measure wetland hydrology (left panel; United States Army Corps of Engineers 2005) and deployed water table monitoring wells in adjacent wetland and upland soils. Courtesy of Berkowitz and Noble (2015).

Surface elevation and sediment deposition measurements have similarly evolved from basic approaches using survey techniques to fine-scale evaluations in the millimeter range (Lane, Day, and Day 2006). Cahoon et al. (2006) and Callaway, Cahoon, and Lynch (2014) describe the use of surface elevation tables to determine rates of wetland soil subsidence or accretion. Another common technique involves the deployment of artificial marker horizons, often created using feldspar that can be sampled over time, yielding direct measures of sediment accretion at the wetland soil

surface. Additionally, a number of isotopic techniques have also been applied to evaluate sediment accretion, erosion, and movement in wetland soils (Callaway et al. 1997). These techniques include measures of lead, cesium, mercury, beryllium, and other elements (DeLaune et al. 1989; Fry 2006).

Oxidation-reduction (that is, redox) potentials are required to determine whether wetland soils are anaerobic and inform the chemical speciation of elements and compounds within the soil matrix (Vepraskas, Polizzotto, and Faulkner 2016). As Berkowitz and Noble (2015) explain, “measurements of soil . . . [redox] potential require the installation and monitoring of five replicate platinum electrodes within near-surface soil layers” (3). Berkowitz and Noble (2015, 3) continue:

Redox potential measurements must undergo a reference electrode correction to reflect values [according to] the standard hydrogen electrode and must account for the soil pH (Faulkner, Patrick, and Gambrell 1989). For example, soil with a pH value of 7.0 requires a redox potential below 175 mV to be considered chemically reduced, while a soil with pH value of 5.0 is considered chemically reduced below a redox potential 295 mV (Reddy and DeLaune 2008) (Figure 8).

Figure 8. Example of data from two platinum electrodes measuring soil redox. The dashed line represents the level required for the soil to be considered anaerobic, which fluctuates because of changes in soil pH during the study period. Measurements occurring below the line are considered anaerobic. The Eh values represented by squares never display anaerobic conditions. The Eh values represented by triangles become anaerobic on 12 January 2009 and remain reduced through the monitoring period. Reproduced from National Technical Committee for Hydric Soils (2015).

The redox status of wetland soils can also be monitored using direct observations of soil reactivity through the application of chemical dyes and the installation of indicator of reduction in soils (IRIS) tubes (Berkowitz and Noble 2015). Application of α - α -dipyridyl dye in soil that is reduced with respect to iron yields a positive reaction and rapid development of a pink color on the soil surface (Figure 9; Berkowitz and Noble 2015; Berkowitz, Pietroski, and Currie 2017a). Installation of IRIS tubes for several weeks will result in removal of iron oxide paint on the tube surface if anaerobic conditions are present, further documenting wetland soil conditions and integrating results over space and time (Figure 9; Castenson and Rabenhorst 2006). Rabenhorst and Burch (2006), Rabenhorst (2008), and Berkowitz (2009) provide additional guidance on the application and analysis of IRIS tubes. Additional recent research also demonstrates the use of IRIS tubes to verify sulfur reduction in wetland soils, with rapid (<1 h) precipitation of iron sulfides in the IRIS tube

surface in strongly reduced wetland soils (Vaughan et al. 2016; Rabenhorst, Megonigal, and Keller 2010). Finally, the redox status of a wetland soil can be indirectly inferred by measuring the presence of dissolved constituents in soil solution. For example, observation of ferrous iron, sulfides, and reduced forms of manganese indicate that soils are anaerobic (Luther and Madison 2013). These tools and techniques discussed are essential to monitoring changes in wetland soils resulting from changes in hydroperiod, as they affect soil chemical, physical, and biological processes (DeLaune et al. 2013).

Figure 9. Examples of techniques to document anaerobic conditions in wetland soils including (a) application of α - α -dipyridyl dye to anaerobic wetland soils result in development of pink or red color (Berkowitz and Noble 2015); (b) typical iron removal patterns from IRIS-tube study. IRIS tubes display 5%, 10%, 30%, 45%, and 85% iron removal from left to right (Berkowitz 2009).

The soil physical properties of wetlands are also unique compared to nonwetland areas because of increased soil moisture content, higher organic carbon content, and other factors (Reddy et al. 2013). Common measurements to characterize soils and monitor response to environmental changes (that is, increased inundation) include bulk density, particle-size

distribution, water content, infiltration rate, and porosity. Additionally, Leach and Johnson (1994) identify a suite of physical properties related to wetland soil strength, compressibility, and permeability. Kuang et al. (2012) provide a review of methods to assess soil physical properties, including shear stress, penetration resistance, and other factors related to soil strength. These measures, in conjunction with the parameters discussed above, provide a comprehensive approach to documenting soil responses to changing patterns of saturation and inundation.

Finally, because soil properties, vegetation, and fauna (for example, microbial communities, burrowing animals) are closely linked, analysis of ecological factors also reflect changes in soil conditions. For example, changes in wetland nutrient loading have been shown to decrease soil strength in coastal marshes (Turner 2011), and DeLaune, Nyman, and Patrick Jr. (1994) and others have linked plant death with increased inundation, resulting in subsequent microbial decomposition of root materials and subsequent soil surface subsidence. As a result, soil microbial activity is often measured in addition to the chemical and physical properties discussed above and have been shown to exhibit rapid responses to changing environmental conditions (VanZomeren et al. 2018). Kang, Kim, and Freeman (2013) as well as Rinklebe and Langer (2013) provide a review of microbial measures in wetland soils systems.

2.2.8 Response of wetland soils to inundation

Many publications have investigated the implications of increasing inundation via hydrologic alteration and RSLR on wetland ecosystems, including several literature reviews and synthesis documents (Janssen-Stelder 2000; Friedrichs and Perry 2001; Willard and Bernhardt 2011). However, studies focusing on soil implications are less common than evaluation of flora and fauna effects. In response, the following section evaluates available literature investigating potential changes in soil conditions resulting from increasing inundation. Specific emphasis is placed on marsh geomorphology, nutrient cycling, and physical effects (for example, accretion, erosion, subsidence).

Coastal marshes and other wetlands display a variety of morphologic features because of gradients of elevation, inundation, salinity, and other factors (Wilson et al. 2009). Recent research demonstrates that soil properties differ between these wetland components (Berkowitz et al. 2018), and on the basis of the amount of recent and ongoing research,

many questions remain regarding the condition and fate of morphologic features under different flooding scenarios. Formation and subsequent infilling of open-water features result from natural cyclical process and anthropogenic alterations within the marsh environment (Berkowitz et al. 2018, 311, citing Wilson et al. 2014; Day et al. 2000), yet the associated mechanisms and timescales remain unclear, especially in areas displaying disturbance (for example, marsh ditching) and areas subject to increases in inundation (Berkowitz et al. 2018, 311). Notably, several studies document fragmentation and subsequent degradation of wetland soils following changes in wetland hydrology (that is, increased inundation and salinity), resulting in conversion of vegetated areas to unvegetated open water areas (Turner 1997; Steyer et al. 2008; Berkowitz et al. 2018, 311). As Berkowitz et al. 2018 explain, “DeLaune et al. (1994) described this process as pond initiation, in which newly formed open-water areas allow for further marsh degradation via erosion, [soil subsidence], and other mechanisms” (311). DeLaune, Nyman, and Patrick Jr.’s (1994) data suggested marsh inundation and plant mortality induced rapid loss of root turgor pressure, resulting in root structural collapse and soil subsidence. Berkowitz et al. 2018 also found that other research links disturbances and increasing inundation of marsh vegetation “with the formation of open-water features (Figure 10), which subsequently exacerbates [wetland] fragmentation via erosion and widening of tidal channels” (311, citing Kirwan, Murray, and Boyd 2008 and Temmerman et al. 2012).

Figure 10. Loss of root turgidity leads to soil collapse, subsidence, and initiation of open water areas within the wetland platform. Recently degraded marsh area showing remnant island of vegetation surrounded by subsided soil (Berkowitz, Vanzomeren, and Piercy 2017b).

Janssen-Stelder (2000) highlights the increases in soil submergence leading to poor drainage and waterlogging in the southeastern United States, and Linhoss et al. (2014) predict >5,000 ha of coastal areas in northeast Florida will be affected by increased inundation. Once additional inundation patterns are established, several changes in soil processes are induced. Multiple studies link increased coastal inundation with prolonged waterlogging, decreases in redox potentials (see previous sections), increases in ammonium concentration, and generation of toxic sulfide compounds that stress marsh vegetation (see Mendelssohn and McKee 1988 and DeLaune, Smith, and Patrick 1983, for example). Soil inundation and decreasing redox potentials also reduce rates of photosynthesis, further stressing plant growth (Pezeshki 2001). Spatially, increased inundation is most often observed at the wetland interior (that is, furthest from higher elevation tidal creek fringe) where elevation, redox potentials, and vegetative productivity remain low. Several studies evaluate the impact of increased inundation on nutrient cycling and soil microbial activity. For example, Kadiri and Spencer (2018) reported that microbial communities in wetland soils responded quickly to increasing inundation and frequency. However, increased salinity and inundation induced short-term carbon dioxide release and carbon mineralization, likely related to the release of previously protected organic matter in soil aggregates and the decomposition of labile biomass.

The duration of coastal saltwater intrusion into freshwater marshes can also affect soil and pore water chemistry. Higher flood durations can increase pore water salinity, which can remain high until flushing by, or diffusion into, subsequent floodwaters from precipitation or other freshwater sources (Steyer et al. 2005). Sjøgaard, Valdemarsen, and Treusch (2018) investigated increasing flooding and salinity on coastal soils in sediment microcosms, reporting an initial increase in heterotrophic activity and ammonium production followed by increased denitrification and eventual declines in nutrient availability. Labile organic matter availability and microbial activity decreased over the four-month study period, suggesting both short- and long-term shifts in microbial communities as soil systems respond to changes in saltwater inundation patterns.

Weston, Dixon, and Joye (2006) reported that introduction of salinity into freshwater wetland sediments increased nutrient release, initial iron reduction–driven organic matter mineralization, and subsequent dominance of sulfate reduction–driven mineralization. Overall, the

mineralization rate doubled during the month-long laboratory study, suggesting rapid organic matter processing following salinity manipulation. van Dijk et al. (2017) observed similar trends in sulfate reduction dynamics while evaluating soil properties during a three-year mesocosm study. An experimental increase in salinity from 0.9 to 9 practical salinity units increased pore water salinity and sulfate reduction rates. This increase resulted in higher sulfide concentrations and lower methane concentrations in the soil pore water. Additionally, hydraulic conductivity of the hyporheic zone increased 280%. Findings suggest that both physical and biogeochemical shifts in soil conditions should be considered when evaluating increasing saltwater inundation.

In one of the longer-term field manipulation studies, Neubauer, Franklin, and Berrier (2013) examined wetland soil processes in experimental mesocosms exposed to increased salinity over a 3.5 year period. During the experiment, the authors observed decreasing soil carbon content, organic matter availability, and associated reduction in microbial activity—and thus greenhouse gas emissions. Results suggest that long-term studies of increasing salinity in freshwater systems affected the entire carbon mineralization process, in some cases diverging from results observed in short-term disturbance-based experiments. Jones et al. (2018) also investigated the impact of elevated carbon dioxide and flooding on coastal marsh soil carbon processes. As seen in other studies, increased flooding resulted in decreases in the soil carbon pool coupled with increases in aboveground biomass but ultimately leading to an overall loss of carbon from the system. The authors also suggested that RSLR and associated increases in inundation will likely overwhelm the effects of elevated atmospheric carbon dioxide in C₄-dominated marshes. The loss of soil carbon to the atmosphere with additional flooding will likely threaten marsh carbon stocks.

Kirwan and Mudd (2012) used a modeling approach to investigate the effects of increasing inundation, RSLR, and temperature on accretion rates and organic matter accumulation over a 100-year period. Results suggest that plant productivity and organic-matter accumulation rates will increase with progressive inundation of the marsh surface, followed by a dramatic organic-matter and soil accretion due to plant mortality as inundation thresholds are exceeded. Following plant death, carbon ceases to be deposited while existing carbon near the soil surface continues to decay, resulting in subsidence. Stagg et al. (2017) also applied a modeling

approach to investigate causal pathways for changes in carbon cycling and soil organic-matter decomposition resulting from increased inundation and salinity. The pathways identified included (1) the effects of flooding on soil moisture, (2) the effects of salinity on decomposer microbial communities and soil biogeochemistry, and (3) an indirect pathway representing the effects of salinity on litter quality through changes in plant community composition over time (Stagg et al. 2017, 2003). Results suggest that organic-matter decomposition will decrease in freshwater forested wetlands exposed to drought, hurricane salinity pulses, and long-term RSLR. However, oligohaline marsh model results predicted increased organic-matter decomposition following a hurricane salinity pulse, with no effect detected for the other factors examined. Findings suggest that both disturbance magnitude and duration must be considered when examining inundation effects on carbon dynamics. Valiela et al. (1984) investigated litter decomposition in marsh ecosystems during field studies, reporting that decomposition rates were reduced under anaerobic conditions. The rate of soil organic-matter decomposition was closely related to nitrogen content and litter chemical composition. Additionally, three phases of decomposition were identified, consisting of a rapid leaching phase, decomposer-driven degradation, and finally a slow processing of refractory compounds.

Kirwan and Guntenspergen (2012) evaluated changes to productivity, accretion, and wetland stability in a rapidly subsiding marsh. Results suggest that marsh morphologic features presently above the optimum elevation may display increased productivity and accretion with increased inundation; however, features below optimum elevation will experience reduced root growth, decreasing elevation gain, and rapid degradation. Additionally, the authors highlighted the finding that belowground responses to RSLR provide more valuable insight than aboveground feedbacks because they are consistent across species and do not depend on the input of mineral sediment at the soil surface. Morris et al. (2002) reported similar results, predicting optimum rates of RSLR to maintain marsh elevation via increases in productivity and accretion, while indicating that marshes below optimal elevation will become increasingly unstable.

2.2.9 Knowledge gaps in soil inundation research

While much research has evaluated the implications of increased inundation on soil properties and functions, significant knowledge gaps

remain. Notably, many studies rely on laboratory manipulations or modeling approaches, which may not reflect field conditions. Further, most studies are limited to several months, with only a few studies evaluating conditions over multiyear periods. As a result, large-scale manipulation or retrospective studies are recommended to investigate the link between soil inundation, stability, and biogeochemistry. Additional research is also required to highlight the fate of various wetland morphologic features, which have been shown to respond differently to disturbance. Finally, linking remote sensing approaches and ground-based measurements of soil processes, including applying novel techniques, will help inform the resource management community as wetland inundation increases.

2.3 Vegetation studies

2.3.1 Implications of increased inundation on wetland plants

Research into plant tolerance and adaptation to inundation has increased considerably in recent decades (Armstrong, Brändle, and Jackson 1994; Ellison and Bedford 1995; Luo, Song, and Xie 2008; Nyman et al. 2009). Early inundation studies occurred primarily in riparian environments or on row crop plants (Bendix and Hupp 2000; Berning, Viljoen, and Du Plessis 2000). However, the physiological responses of crop species and plants grown in aerobic soil environments have been shown to differ significantly (DeLaune, Pezeshki, and Pardue 1990). More recently, researchers focused primarily on fine-scale differences and relationships between long-term incremental changes (that is, RSLR; Sharpe and Baldwin 2012) in hydrologic regime components (inundation depth, duration, frequency, and timing) and wetland plant responses (establishment, growth, and reproduction of individual plants; composition and diversity of assemblages) (Jackson, Ishizawa, and Ito 2009; Araya, Gowing, and Dise 2010; Webb, Wallis, and Stewardson 2012).

Though water regime is the predominant physical force acting to shape plant composition and behavior in many wetlands, the response to changing hydrodynamic conditions varies significantly between wetland plants, and those responses are often nonlinear (Blom et al. 1990; Kirkman and Sharitz 1993; Janousek and Mayo 2013). Previous studies evaluated a wide range of cause-effect hypotheses (cause typically based on 6 common regime components; waterlogging, inundation, depth, duration, frequency, and timing) related to plant inundation responses.

Webb, Wallis, and Stewardson (2012) identified 38 hypotheses (most related to biomass, shoot density, and length; reproductive output; plant assemblages; and plant establishment, growth, and diversity) across 102 qualifying studies; a large number of hypotheses (17) showed inconsistent effects of water-regime components, and 10 contained insufficient evidence to test the hypotheses. These findings reflect the diversity of responses of different plant taxa, wetland types, and the methods used in these studies and represent true knowledge gaps that can only be filled by future targeted research (Webb, Wallis, and Stewardson 2012). Webb, Wallis, and Stewardson (2012) provides tables summarizing causal criteria analyses of hypotheses linking changes in wetland plant establishment, wetland plant growth, reproductive output, and plant assemblage to increases in components of the water regime.

2.3.2 Flooding impacts on foundational plants

Given the large number of plant species that exist in wetland landscapes globally, and the wide range and species-specific nature of inundation impacts, evaluating those impacts can prove difficult. However, only a small number of species are able to survive and thrive in wetland environments with inundation depth and duration beyond the normal range. The functional role and abundance of these foundational plants (defined because of their important functional role in the ecosystem) are largely influenced by inundation depth and duration (Osland, Enwright, and Stagg 2014). Using these foundational indicator species to study plant survival mechanisms against different hydrologic regimes provides some clear advantages (Blom et al. 1990). However, little information presently exists on the range of indicator (foundational) plant tolerance to flooding stressors that exist beyond the normal range (Howard and Rafferty 2006). Therefore, the review of previous research and existing literature will be confined to a selection of foundational wetland plants. Table 3 provides a list of scientific and common names and descriptions (origin, duration, form, family, and dominant vegetation zone) about the foundational plants selected for this review. A summary of inundation impacts for each foundational plant species is provided below.

Table 3. Foundational plant species that occur in riverine and coastal wetlands, all are native perennial plants in the United States.

Scientific name	Common name	Form	Family	Water composition
<i>Panicum hemitomon</i>	maidencane	Graminoid	Poaceae	Fresh

Scientific name	Common name	Form	Family	Water composition
<i>Sagittaria lancifolia</i>	bulltongue arrowhead	Herb	Alismataceae	Fresh
<i>Sagittaria latifolia</i>	broadleaf arrowhead	Herb	Alismataceae	Fresh
<i>Typha domingensis</i>	southern cattail	Graminoid	Typhaceae	Fresh/Interm.
<i>Typha latifolia</i>	broadleaf cattail	Graminoid	Typhaceae	Fresh/Interm.
<i>Phragmites australis</i>	common reed	Graminoid	Poaceae	Intermediate
<i>Spartina patens</i>	saltmeadow cordgrass	Graminoid	Poaceae	Brackish
<i>Spartina alterniflora</i>	smooth cordgrass	Graminoid	Poaceae	Saline

2.3.2.1 *Panicum hemitomon*

Panicum hemitomon Schult. (maiden cane) is a warm-season, aquatic or subaquatic, extensively rhizomatous perennial that forms almost pure stands that range in heights of 1–1.5 m (Newman and Gates 2006). *P. hemitomon*, which is commonly located in coastal and freshwater marshes, edges of ponds, lakes, ditches, canals, and cypress-gum ponds, is native to Louisiana and has historically dominated freshwater wetlands in coastal states reaching from New Jersey to Texas (Figure 11) (United States Department of Agriculture–Natural Resources Conservation Service 2018). However, over the past century, *P. hemitomon* has experienced high rates of loss in some coastal areas, potentially due to inundation and salinity shifts (Willis and Hester 2004). *P. hemitomon* has vigorous vegetative clonal growth and high biomass production that allows it to tolerate broad ranges of hydrology—from nearly continuous inundation to relatively dry conditions (Holm and Sasser 2008). Few studies exist, however, that evaluate impacts on *P. hemitomon* from inundation depths and duration that extend beyond the normal range.

Figure 11. Species distribution map of *Panicum hemitomon* in North America. Courtesy of United States Department of Agriculture–Natural Resources Conservation Service (2018). (Shaded: native to state.)

Kirkman and Sharitz (1993), Lessmann et al. (1997), and Willis and Hester (2004) all examined the effects of controlled water regimes (under greenhouse conditions) on the height, biomass allocation patterns, reproduction, and water relations of *P. hemitomon*. These studies found that the stem lengths in aboveground segments of *P. hemitomon* were strongly related to water depth. Kirkman and Sharitz (1993) found the lengths were positively related ($r^2 = 0.78$) to water depth above the soil, with approximately a 1:1 cm relationship; the range of water depth exceeded 1.0 m. Stem emergence, however, decreased significantly when inundation occurred after culm removal (Kirkman and Sharitz 1993); apparently, the erect, dead culms of *P. hemitomon* may be critical for maintaining ventilation to roots and emerging tillers (Jordan and Whigham 1988). Maximum height, aboveground mass, and stem number of *P. hemitomon* was also attained under the inundated treatments, while plants in inundated-moist and saturated treatments showed intermediate growth responses (Kirkman and Sharitz 1993).

Under field conditions, the maximum biomass of *P. hemitomon* has been reported where inundation depths were between 30 cm and 60 cm

(optimum range), and where inundation frequency was between 85% and 90% (Schalles and Shure 1989, Holm and Sasser 2008). *P. hemitomon* has been observed in inundation depths between 80 cm and 100 cm for up to nine months, but significant reductions occur with water depths around 120 cm (Loveless 1959, Kirkman and Sharitz 1993, Holm and Sasser 2008). Researchers theorize that the increase in biomass production with inundation may be due to adventitious roots, which may increase nutrient uptake and improve tolerance (Lessmann et al. 1997).

Previous studies have also evaluated combined impacts of disturbance (herbivory and salinity) and water stress for management and control of *P. hemitomon*. Though increased salinity alone resulted in reductions in *P. hemitomon* biomass (aboveground, belowground, and total), moderate increases in salinity (1.5 ppt) with moderate inundation depths (10 cm) resulted in increased production and biomass (Willis and Hester 2004). Willis and Hester (2004) also found that increasing inundation depths from 10 cm to 20 cm resulted in significantly greater water root biomass. The combined effects of culm removal followed by inundation resulted in the reduced production of tillers and culms (Kirkman and Sharitz 1993). *P. hemitomon* has been shown to be a competitive and vigorous plant that persists in freshwater wetlands, even during stressful extremes such as drought, flooding, and fires (Holm and Sasser 2008).

2.3.2.2 *Sagittaria lancifolia* and *latifolia*

The *Sagittaria* assemblage consists of aquatic plants that grow in swampy ground or standing water in ponds, lakes, stream edges, and ditches (Jepson and Hickman 1993). *Sagittaria lancifolia* (bulltongue arrowhead) is an herbaceous, aquatic, native, monocotyledonous perennial plant with distribution ranges along the Gulf and East Coasts of the United States and parts of Oklahoma (Figure 12). *S. lancifolia* has a high propagation potential and readily colonizes new wetland habitats (Martin and Shaffer 2005). *S. latifolia* (broadleaf arrowhead) is a highly competitive perennial obligate wetland plant that is found in marshes and wetlands throughout temperate North America (Figure 13) (Baldwin and Mendelssohn 1998a; Martin and Shafer 2005; Dorken and Barrett 2003; Stevens 2016). Stresses associated with excessive plant submergence, due to marsh subsidence and RSLR, has caused decreased plant growth in *S. lancifolia* throughout portions of its range (McKee and Mendelssohn 1989). *S. latifolia* exhibits both dioecious and monoecious reproductive character in its northern and southern ranges, respectfully (Stevens 2016).

Figure 12. Species distribution map of *Sagittaria lancifolia* in North America. Courtesy of United States Department of Agriculture–Natural Resources Conservation Service (2018). (Shaded: native to state.)

Figure 13. Species distribution map of *S. latifolia* in North America. Courtesy of United States Department of Agriculture–Natural Resources Conservation Service (2018). (Shaded: native to state.)

Previous inundation studies of *Sagittaria* spp. consisted of controlled water regimes as part of laboratory (greenhouse) experiments and more natural fluctuations as part of field experiments (though many experimental designs altered the elevations of the plants). In 1995, Howard and Mendelssohn evaluated inundation impacts on *S. lancifolia* by increasing water depths by 7.5 cm and 15 cm above the marsh surface. Plants subjected to the 15 cm increase in water depth (inundation was not continuous for field study) had significantly greater mean and maximum leaf heights compared to disturbed controls, and 7.5 cm treatment plants had significantly greater mean height (Howard and Mendelssohn 1995). These findings are similar to those by Martin and Shafer (2005), who found no difference in aboveground biomass of the two species of *Sagittaria* exposed to 5 cm and 30 cm water depths over a three-year study (Wilks' lambda $F(5) = 2.963$, $p(5) = 0.061$). In most studies, *Sagittaria* spp. responded favorably to increased hydroperiods. This favorable response was primarily by virtue of the underground reserves and by the changing of leaf morphology with fluctuations in water depth (David 1996; Martin and Shafer 2005).

Grace (1989) evaluated *S. latifolia* response to inundation along a gradient to a maximum depth of 1.25 m. The height of *S. latifolia* increased with increasing depth, adjusting to deep water through changes in biomass allocation but eventually died with depths greater than 95 cm (Grace 1989). Studies by Howard and Mendelssohn (1995), Baldwin, McKee, and Mendelssohn (1996), and Grace and Ford (1996) are in general agreement that water levels in the range of +5 cm to +20 cm are also not a significant impairment to the productivity of *S. lancifolia* (Spalding and Hester 2007). A few studies, however, have shown that *S. lancifolia* has exhibited some decrease in plant growth with moderate increasing water depths and percentage of time inundated (Webb and Mendelssohn 1996; Visser and Sandy 2009). Pezeshki, DeLaune, and Patrick (1987) also observed a significant decrease (–26%) in net photosynthesis in *S. lancifolia* with increased inundation, compared to control levels. Similar relationships (linear) between plant growth (cm) and belowground biomass with increasing water level were observed in *S. lancifolia* (Visser and Sandy 2009).

Although high water tables can reduce the germination of many freshwater marsh plant species, many *Sagittaria* spp. are able to germinate even when inundated (Webb and Mendelssohn 1996). Baldwin, McKee, and

Mendelssohn (1996) reported that *S. lancifolia* seedlings and *S. latifolia* achenes were able to germinate in moderately inundated conditions (Martin and Shaffer 2005; Spalding and Hester 2007). These findings, showing no strong relationship between inundation and *Sagittaria* seedlings, are consistent with tolerance responses reported by others (van der Valk and Davis 1978; Keddy and Ellis 1985; Marburger 1993; Martin and Shaffer 2005; Kenow and Lyon 2009; and Kenow et al. 2018).

Inundation alone may not significantly affect *Sagittaria* spp. productivity, but the rapid decline in abundance of plants like *Sagittaria* spp. from some ecosystems provides an example of how multiple stresses can adversely affect an otherwise common, wetland plant (Holm and Sasser 2001). Previous studies have shown that the combination of inundation with other disturbances, like salinity stress (Baldwin and Mendelssohn 1998b; Holm and Sasser 2001) and herbivory (Shaffer et al. 1992) can significantly reduce biomass, seedling recruitment, and ultimately plant survival.

2.3.2.3 *Typha*

The *Typha* are herbaceous, rhizomatous perennial plants with long, unbranched flowering shoots (1.5–3 m) terminating with a cylindrical pistillate spike that is persistent and drying to brown and a staminate spike of deciduous flowers. Leaves are clustered, originating from the base and typically concave-convex proximally, becoming flat distally with parallel venation (Stevens and Hoag 2006a). *Typha latifolia* L. (broadleaf cattail) are common in coastal and valley marshes throughout the United States and other temperate and tropical places worldwide, especially where elevations are lower than 2,000 m (Figure 14) (Hickman 1993). *Typha domingensis* Pers. (southern cattail) are also common in the warm temperate and tropical coastal and valley marshes with elevations lower than 1,500 m, and range through most of the lower continental United States (Figure 15) (Stevens and Hoag 2006b).

Figure 14. Species distribution map of *Typha latifolia* in North America. Courtesy of United States Department of Agriculture–Natural Resources Conservation Service (2018). (Shaded: native to state.)

Figure 15. Species distribution map of *T. domingensis* in North America. Courtesy of United States Department of Agriculture–Natural Resources Conservation Service (2018). (Shaded: native to state.)

Previous research has shown that *Typha* spp. can persist and survive in relatively deep water and spread vegetatively for many years (Fennessy, Cronk, and Mitsch 1994). Li, Pezeshki, and Goodwin (2004) found that the biomass and height of *T. latifolia* were highest when grown under continuous inundation (compared to periodic flooding and drought). Dark fluorescence yield, net photosynthesis, stomatal conductance, and chlorophyll content are potential explanations as to why *Typha* spp. have higher biomass increment and height growth when continuously inundated (Li, Pezeshki, and Goodwin 2004). Early research reported that the maximum depth of *T. latifolia* was <70 cm (Grace and Wetzel 1998, Sloey, Willis, and Hester 2015). Until recently, the response of *Typha* spp. to water depths of greater than 100 cm was relatively unknown (Chen, Zamorano, and Ivanoff 2010). Recent studies have evaluated the ability of *Typha* spp. to grow in deeper waters, and found that *T. latifolia* was able to grow in water depths up to 95 cm and *T. domingensis* in depths up to 115 cm (Grace 1989; Chen, Zamorano, and Ivanoff 2010).

Typha spp. have morphological, anatomical, and physiological characteristics commonly found in wetland plants, including the capacity to pressurize their leaves for oxygen transport and the ability produce adventitious roots (Tornbjerg, Bendix, and Brix 1994; Pezeshki et al. 1996). Such capacity is critical for continued functioning under continued saturated and reduced conditions (Pezeshki et al. 1996). Chen, Zamorano, and Ivanoff (2010) found that inundation depths greater than 115 cm can significantly affect new shoot, leaf, root, rhizome, shoot base, belowground biomass, total biomass, and chlorophyll production as well as root-leaf ratio. Increasing inundation depths from 91 cm to 137 cm also significantly decreased new shoot, leaf, root, rhizome, belowground, and total biomass production.

Grace (1989) and Chen, Zamorano, and Ivanoff (2010) reported that even though *T. domingensis* increases its shoot height and produces fewer but larger ramets as water levels increase, flowering incidence is restricted to a relatively narrow range of water depths. Increasing inundation depth also reduces the anchorage capacity, photosynthesis and chlorophyll fluorescence, seed production and germination, and shoot density (because of competition) of *Typha* spp. (Smith 1967; Grace 1989; Grace and Wetzel 1998; Ekstam and Foresby 1999; Lorenzen et al. 2001; Li, Pezeshki, and Goodwin 2004). Though *Typha* spp. are capable of persisting at relatively high water depths and moderate durations, Van den

Brink et al. (1995) and Chen, Zamorano, and Ivanoff (2010) results indicate that *Typha* spp. have high negative growth rates under prolonged submerged conditions. *T. latifolia* exhibited significant reductions after total submergence for four to eight weeks (Van den Brink et al. 1995). For *T. domingensis*, total submergence for one week resulted in chlorosis and plant mortality by week six (Chen, Zamorano, and Ivanoff 2010). However, the adaptive abilities of *Typha* spp. under increased flood and nutrient conditions have led to *Typha* spp. dominance in many wetland ecosystems (Newman, Grace, and Koebel 1996; Todd et al. 2010).

2.3.2.4 *Phragmites australis*

Phragmites australis (Cav.) Trin. ex Steud. (common reed) is a large rhizomatous and stoloniferous cool season grass obtaining heights of up to 4 m with stems averaging 0.5 to 1.5 cm in diameter. *P. australis* has an extensive system of scaly rhizomes and stolons (up to 18 m long), which allows the plants to spread into dense monotypic stands (Welsh et al. 2003). *P. australis* has been described as the most broadly distributed flowering plant in the world (Good 1974), with the North American subspecies covering most of the United States and portions of southern Canada (Figure 16) (Tilley and St. John 2012; United States Department of Agriculture–Natural Resources Conservation Service 2018). Some recent reductions in *P. australis* distribution have been caused by increased flooding, erosion, and storm-related damage (Suir, Saltus, and Reif 2018).

Figure 16. Species distribution map of *Phragmites australis* in North America. Courtesy of United States Department of Agriculture–Natural Resources Conservation Service (2018). (Shaded: both native and introduced to state or province.).

Hydrology is among the most profound abiotic factor impacting *Phragmites* growth, especially at early growth stages (Chambers et al. 2003). *P. australis* has well-developed mechanisms of inundation tolerance, particularly through biomass allocation, pressurized ventilation, physiological tolerance of its rhizome to anoxia, and low oxygen consumption by the roots (Gries, Kappen, and Löscher 1990; Mauchamp, Blanch, and Grillas 2001; Roberts and Marston 2011). With these adaptations, *Phragmites* can withstand water stress or drought for a relatively long period of time (Pagter, Bragato, and Brix 2005). Previous studies have reported that *Phragmites* have thrived in both stable and fluctuating water levels up to 1.5–2 m deep and can tolerate periodic full submergence (Gries, Kappen, and Löscher 1990; Roberts and Marston 2011). The initial response of *Phragmites* to water stress is to reduce and modulate leaf area, converting the remaining leaves from functioning to physiological until the stress level becomes critical (Pagter, Bragato, and Brix 2005). Previous studies show that *Phragmites* shoot height increased or remained steady with increasing water depth (up to 65 cm) (Yamasaki and Tange 1981; White and Ganf 2002), while shoot density, length, and total biomass decrease significantly with water depths greater than 90 cm (less biomass is allocated to underground parts) (Mauchamp, Blanch, and

Grillas 2001; White and Ganf 2002). In a study by Squires and van der Valk (1992), *Phragmites* shoot density, total shoot height, and total biomass were highest when water depths were between 20 cm and 45 cm. These results are similar to those by Coops, Van Den Brink, and Van Der Velde (1996), which showed stem densities were low and irregularly patterned over the water-depth gradient (–20 cm to 80cm). The stem lengths were greatest near the upper portion of the inundation range, but max densities were between 30 cm and 55 cm. There was also a significant increase in the aboveground to belowground biomass ratio as water depths increased. The increasing stem height with increasing inundation depth is the result of increasing concentrations of ethylene, a gaseous hormone that stimulates stem elongation (Howard and Rafferty 2006). However, total submergence did inhibit growth of *P. australis*, which is potentially due to the limited ability of *Phragmites* to photosynthesize under water (Mauchamp, Blanch, and Grillas 2001).

In addition to depth, fluctuations in water levels also have some control over *Phragmites*' growth response. *Phragmites* are not able to modify internal stem structure to increase aeration, so large fluctuations over long periods (45 cm over six weeks) have been shown to reduce plant growth in deeper waters but not in shallow (Roberts and Marston 2011). However, some aerenchyma development in *Phragmites* has been observed 10–15 cm behind both the growing tips of the culms and long, little branched roots (Gries, Kappen, and Lösch 1990). Inundation can also affect seed germination, since *Phragmites* seedlings can only survive if they can extend through the water and into the air (Roberts and Martson 2011). The aboveground production of *Phragmites* and the subsequent reserve storage were also reduced when subjected to the combination of increased inundation with increasing salinity (Hellings and Gallagher 1992). Therefore, water depth in conjunction with other stressors can be important for *Phragmites* productivity, distribution, and colonization, all of which rely on vegetative growth by above- and belowground rhizomes (Mauchamp, Blanch, and Grillas 2001).

2.3.2.5 *Spartina alterniflora* and *patens*

Spartina patens (Aiton) Muhl. (salt-meadow cordgrass) is a native warm season perennial grass that grows from 30 cm to 120 cm tall and spreads extensively by seed, long slender rhizomes, or vegetative fragments (Xiao et al. 2010; Leif 2013). *S. patens* has been observed from the Atlantic shorelines in Newfoundland, Canada, to the coastlines of Florida and

Texas, along Lake Huron in Michigan, and in Oregon and Washington, where it is considered invasive (Figure 17) (Leif 2013). *S. alterniflora* Loisel. (smooth cordgrass) is a robust, rapidly spreading (by vegetative propagation) native warm-season grass that forms tall (60 cm to 183 cm), dense, vegetative colonies along shorelines and intertidal flats in coastal wetlands (Materne 2009). *S. alterniflora* tends to grow parallel to and continuous along shorelines in the United States, where stands are typically controlled by conditions such as elevation, shoreline-slope, and hydropattern (Figure 18) (Materne 2009).

Figure 17. Species distribution map of *Spartina patens* in North America. Courtesy of United States Department of Agriculture–Natural Resources Conservation Service (2018). (Shaded: native to state.)

Figure 18. Species distribution map of *S. alterniflora* in North America. Courtesy of United States Department of Agriculture–Natural Resources Conservation Service (2018). (Shaded: native to state.)

S. alterniflora are typically dense monocultures that dominate frequently inundated and physically stressful tidal saline marsh (Bertness 1991; Xiao et al. 2010), while *S. patens* are monocultures that dominant higher, brackish marshes, with wider ranges of salinity (Gleason and Zieman 1981; Pezeshki and DeLaune 1993). *S. patens* marshes are dense, competitive, and persist year-round, which often inhibit colonization by other species in the high marsh, including *S. alterniflora* (Bertness 1991; Baldwin and Mendelssohn 1998b).

Previous research has evaluated the influence of inundation depth, particularly moderate fluctuations in water levels, on *Spartina* spp. growth. For *S. alterniflora*, although small to moderate increases in water depth did cause reductions in plant height, density, and biomass, those changes were not significant (Reed and Cahoon 1992; Ding, Zhang, and Cai 2010). These changes were potentially due, in part, to significant changes to soil oxidation-reduction potential, sulfide accumulations, and methane concentration, which were observed with a 10 cm increase in water level (Ding, Zhang, and Cai 2010; Reed and Cahoon 1992). Inundation impacts in *S. alterniflora* also differed according to the age and connectivity of ramets. In young connected ramets, the total biomass remained unchanged with increasing water depth (Xiao et al. 2010). At all

depths, the total biomass of disconnected ramets was significantly lower than the connected ramets, and those biomasses decreased with increasing water depths (Xiao et al. 2010).

Multiple studies have shown that small increases in water depth had insignificant impacts on *S. patens* biomass, but when depths were increased to +30 cm, there were significant reductions in fine roots, total belowground biomass, aboveground biomass, and total biomass (–40%); reductions in stem density and length; and in some cases, the plants barely survived (Broome, Mendelssohn, and McKee 1995; Spalding and Hester 2007; Langley et al. 2013; Snedden, Cretini, and Patton 2015). These results are consistent with other research that concluded *S. patens* has limited tolerance for excessive inundation (Naidoo, McKee, and Mendelssohn 1992).

Prolonged inundation can lead to increased plant stress (that is, oxygen deficiencies), biomass reductions, and decreases in stem densities, even in tolerant plants such as *S. alterniflora* and *S. patens* (Naidoo, McKee, and Mendelssohn 1992). Xiao et al. (2010) reported *S. alterniflora* biomass and stem density reductions of 46.7% and 40.6%, respectively, with prolonged exposure to 9 cm of standing water. Research by Visser and Sandy (2009) and Snedden, Cretini, and Patton (2015) revealed significant linear and exponential relationships between above- and belowground biomass of *S. alterniflora* and *S. patens* and inundation duration. Snedden, Cretini, and Patton (2015) also reported that prolonged inundation of *S. patens* could result in approximately 90% reduction in belowground biomass, leading to substantial reductions in soil shear strength and erosional resistance. Conversely, Smith and Medeiros (2013) report that discrete stands of *S. alterniflora* were better able to tolerate constant immersion for 2.5 months without any visible signs of stress. It is assumed that this resilience to inundation is capable over the course of one growing season, but even tolerant plants could not withstand multiple years of persistent inundation (Smith and Medeiros 2013).

Because *S. alterniflora* is better at transporting oxygen to the roots, it is more flood and salt tolerant than *S. patens* (Snedden, Cretini, and Patton 2015), and it is more capable of colonizing regularly inundated marsh (DeLaune, Pezeshki, and Patrick 1987; Pezeshki and DeLaune 1993; Ewing et al. 1995). Tolerance to inundation in *S. alterniflora* and *S. patens* involves morphological as well as physiological strategies, with the

dominant adaptation being the development and significant increase in root aerenchyma (Naidoo, McKee, and Mendelssohn 1992). However, *S. alterniflora* possesses more aerenchymatous tissue than *S. patens* and therefore can typically supply more oxygen to its belowground organs (Gleason and Zieman 1981). The maximum amount of aerenchyma in *S. patens* is often not sufficient to provide enough oxygen to completely support aerobic respiration (Kludze and DeLaune 1994). Aerenchyma development does allow *S. patens* to tolerate some inundation and contributes to its ability to successfully exploit wetland habitats (Lonard, Judd, and Stalter 2010). *S. alterniflora* has also demonstrated the ability to adapt to inundation depth and duration through shoot and leaf elongation. Although roots and shoots elongate rapidly when the water stress is initiated, the rate of leaf elongation in both *S. alterniflora* and *S. patens* generally declines (rates actually increased for *S. alterniflora* at 30 flood days) with increased duration (Lessman et al. 1997).

2.3.3 Knowledge gaps in plant inundation research

Degradation and loss of wetlands and associated goods and services remains a national and worldwide concern. Many uncertainties persist regarding the effect of inundation depth and duration on wetland ecosystems in altered and natural settings. Significant gaps in data and guidance exist pertaining to proper ecosystem management and restoration in relation to how inundation affects wetland structure, function, and sustainability. One area of need relates to the impacts of increased water levels (and related flow rates, nutrient dispersal, freshwater pulses, salt intrusion) on the establishment and growth patterns of foundational plant species present in FIBS wetlands. Additionally, few studies have examined the impacts of inundation timing and frequency on plant function as well as inundation depth and duration necessary for plant mortality. Likewise, previous modeling studies exhibit extensive uncertainty because of poorly defined inundation-vegetation response relationships. For example, understanding the critical depth of inundation at which vegetation becomes stressed (and mortality rates) is essential to effectively parameterize vegetation biomass models. Finally, linking geospatial data and field measurements of vegetation processes will improve resource management, especially as wetland inundation increases.

3 Summary and Conclusion

The USACE environmental mission is to restore ecosystem structure and processes, manage land and water resources, and support cleanup and protection activities efficiently and effectively. This stewardship often requires restoration and management activities, including manipulations of hydrology and hydraulics, or protective measures to reduce harmful impacts—whether natural or anthropogenic. Generally, the focus of water-level impacts on ecosystem resources has been on RSLR and the gradual inundation of marshes, where plants and soils have the ability to adapt and respond. However, hydrologic alterations and water management actions that induce intermittent and instantaneous changes in inundation impose entirely different vegetation and soil stresses than the gradual inundation due to RSLR. Currently, high levels of uncertainty exist regarding the effects of hydrologic alterations and water management actions on wetland resources. These uncertainties represent increasingly critical gaps, since there is also an increasing thrust to implement large-scale hydrologic alteration applications (that is, freshwater diversions, lake drawdowns, and reservoir management) to support ecological and navigation missions.

Hydrologic alterations and water management actions, specifically those that induce intermittent and instantaneous changes in water level, may alter wetland plant, soil, and elevation trajectories by modifying the physical, biological, and hydrological processes that control wetland structure and function (Teal et al. 2012). Understanding the interactions and feedbacks that drive landscape change and dynamics resulting from water level management and hydrologic fluxes in wetlands is critical. However, high levels of uncertainty exist regarding the effects of hydrologic alterations and water management actions on wetland resources. With significant ongoing or planned ecosystem management activities affecting or being affected by water level, a comprehensive understanding of short- and long-term inundation effects on wetlands is necessary. This special report evaluated and summarized key inundation-related impacts on wetland plants and soils. These impacts included basic assessments of hydrologic regime, plant and soil processes, flooding events, negative impacts, and plant response and adaptations to floods and other stressors. More detailed reviews of soil and plant morphology and response to inundation were also performed to better assess the current state of knowledge, identify gaps for future research, and to inform ecosystem management decisions.

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14. ABSTRACT The following synthesizes studies investigating plant and soil responses to increased inundation in order to support ecosystem restoration efforts related to the alteration of natural wetland hydrodynamics. Specific topics include hydrologic regimes, soil response to inundation, and implications for vegetation communities exposed to increased water depths. Results highlight the important interactions between water, soils, and vegetation that determine the trajectory and fate of wetland ecosystems, including the development of feedback loops related to marsh degradation and subsidence. This report then discusses the knowledge gaps related to implications of inundation depth, timing, and duration within an ecosystem restoration context, identifying opportunities for future research while providing source materials for practitioners developing restoration projects.					
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