JOURNAL OF THE AMERICAN WATER RESOURCES ASSOCIATION

Vol. 55, No. 3 AMERICAN WATER RESOURCES ASSOCIATION June 2019

Modeling Connectivity of Non-floodplain Wetlands: Insights, Approaches, and Recommendations

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Research Impact Statement: We utilize four case studies to explore recent advances in process-based modeling of non-floodplain wetlands in low-gradient, wetland-rich landscapes.

ABSTRACT: Representing hydrologic connectivity of non-floodplain wetlands (NFWs) to downstream waters in process-based models is an emerging challenge relevant to many research, regulatory, and management activities. We review four case studies that utilize process-based models developed to simulate NFW hydrology. Models range from a simple, lumped parameter model to a highly complex, fully distributed model. Across case studies, we highlight appropriate application of each model, emphasizing spatial scale, computational demands, process representation, and model limitations. We end with a synthesis of recommended "best modeling practices" to guide model application. These recommendations include: (1) clearly articulate modeling objectives, and revisit and adjust those objectives regularly; (2) develop a conceptualization of NFW connectivity using qualitative observations, empirical data, and process-based modeling; (3) select a model to represent NFW connectivity by balancing both modeling objectives and available resources; (4) use innovative techniques and data sources to validate and calibrate NFW connectivity simulations; and (5) clearly articulate the limits of the resulting NFW connectivity representation. Our review and synthesis of these case studies highlights modeling approaches that incorporate NFW connectivity, demonstrates tradeoffs in model selection, and ultimately provides actionable guidance for future model application and development.

(KEYWORDS: wetlands; hydrologic connectivity; process-based models.)

CHARACTERIZING HYDROLOGIC CONNECTIVITY OF NON-FLOODPLAIN WETLANDS

Hydrologic connectivity between non-floodplain wetlands (NFWs) and downstream water bodies has been a recent focus of research, management, and policy. NFWs lack bidirectional hydrologic flows with an adjacent stream or river (USEPA 2015; Lane et al. 2019), and notably, include systems often referred to as geographically isolated wetlands (Tiner 2003; Leibowitz 2015) or upland embedded wetlands (Mushet et al. 2015; Calhoun et al. 2017). Examples of NFWs include Prairie Pothole wetlands in the north-central United States (U.S.) and southern Canada, Delmarva and Carolina bays in the Eastern U.S., and vernal pools common on the U.S. West Coast (Tiner 2003). These landscape features provide a critical portfolio of physical (e.g., McLaughlin et al. 2014; Epting et al.

Paper No. JAWRA-18-0030-P of the *Journal of the American Water Resources Association* (JAWRA). Received February 22, 2018; accepted January 17, 2019. **Discussions are open until six months from issue publication**.

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Citation: Jones, C.N., A. Ameli, B.P. Neff, G.R. Evenson, D.L. McLaughlin, H.E. Golden, and C.R. Lane. 2019. "Modeling Connectivity of Non-floodplain Wetlands: Insights, Approaches, and Recommendations." *Journal of the American Water Resources Association* 55 (3): 559–577. https://doi.org/10.1111/1752-1688.12735.

TABLE 1. Key modeling terms in reference to hydrologic connectivity of non-floodplain wetlands (NFWs).

Term	Definition
Hydrologic connectivity	The hydrologically mediated exchange of materials or energy between watershed units (e.g., wetland and downstream waters)
Hydrologic fluxes	Fluxes of water between landscape elements typically described by the mode, magnitude, duration, and timing of the specific flux
Empirical model	Application of a statistical tool (e.g., correlation, time-series analysis, spatial analysis) to empirical observations
Process-based model	Mathematical representation of hydrological processes
Model complexity	Characterized by both number of factors (parameters, variables) and hydrologic processes represented
Spatial representation	The method used to discretize the landscape into control volumes (e.g., spatially lumped, semi-distributed, and distributed models)
Model domain	The portions of the landscape simulated. In wetland systems, these are normally wetland surface water, shallow subsurface, and deep groundwater systems.
Model fidelity	A model's ability to faithfully represent hydrologic processes
Conceptually based model	Process-based models that route water between user-defined control volumes often using equations that presuppose physical processes (e.g., Manning's and Darcy's equations) and predefined thresholds.
Physically based model	Process-based models that employ first principles (e.g., conservation of mass and momentum) and are often spatially distributed
Lumped model	Process-based models that spatially aggregate landscape properties of a single landscape unit to simulate hydrological processes
Semi-distributed model	Process-based model that utilizes a series of spatially lumped models used to simulate hydrologic processes
Distributed model	Process-based model that discretizes the landscape into small units, typically in the form of a grid or link-node network, to simulate hydrological processes

2018), chemical (e.g., Marton et al. 2015; Cheng and Basu 2017), and biological (e.g., Fairbairn and Dinsmore 2001; Zamberletti et al. 2018) services; yet, these systems are being lost at increasing rates (Van-Meter and Basu 2015; Serran and Creed 2016; Sofaer et al. 2016; Jones et al. 2018a). In recognition of this loss, NFWs have been the focus of policy debates at both state and federal levels, and central to this debate is uncertainty associated with their hydrologic connectivity to downstream waters and associated influences to the physical, chemical, and biological integrity of such waters (e.g., "significant nexus"; see Alexander 2015; Creed et al. 2017 for more details).

Generally, hydrologic connectivity is described as the water mediated transfer of materials, energy, and organisms between watershed components (Pringle 2003; Oldham et al. 2013; Ali et al. 2018), and it is often characterized by the frequency, magnitude, duration, and intensity of hydrologic exchange flows (Table 1) (Harvey and Gooseff 2015; Covino 2017). Here, we define *wetland connectivity* as hydrologic connectivity between wetlands and their surrounding watershed components (Forman 1995; Golden et al. 2017), and we specifically focus on the hydrologic connectivity of NFWs. Relevant hydrologic connections include both surface- and groundwater exchange flows between individual NFWs and their surrounding upland aquifer, other neighboring NFWs, and adjacent flowing waters (Cohen et al. 2016; Rains et al. 2016). NFW connectivity varies over time and space (Evenson et al. 2018), and when considered at watershed scale, NFW connectivity can influence watershed hydrology and downstream flows, with associated influences on biogeochemical (e.g., Cheng and Basu 2017) and biological (e.g., Zamberletti et al. 2018) function and condition.

Recent improvements in high-resolution sensors (Epting et al. 2018; Haque et al. 2018), environmental tracer analyses (Ali et al. 2017; Thorslund et al. 2018), and remote sensing technology (DeVries et al. 2017; Vanderhoof et al. 2017a) have all significantly increased our ability to characterize hydrologic fluxes between NFWs and other watershed components. These empirical characterizations are important for developing a fundamental understanding of the system in question (see Burt and McDonnell 2015) but are often limited in spatial domain and measurement period. As such, these techniques can inform modeling efforts to project empirical inferences over time and space, but alone do not provide adequate spatial or temporal resolution to address many research, regulatory, or management questions.

Recent advances suggest that the combination of conceptual understanding, empirical measurements, and process-based hydrologic models will improve estimates of hydrologic connectivity between watershed components (Golden et al. 2017). Models can be used to extend empirical findings to larger scales and under different scenarios of system change (e.g., climate and land-use change; Weiler and McDonnell 2004). For example, knowledge gained from several recent studies of Prairie Pothole wetlands, a type of NFW, was combined to improve understanding of their connectivity to a specific river system in North Dakota. Empirical investigations included temporal measurements of surface-water hydrologic fluxes in a wetland complex (Leibowitz et al. 2016), remotely sensed imagery to determine spatial and temporal patterns of wetland inundation (Vanderhoof and Alexander 2016; Vanderhoof et al. 2017a), and water isotope analysis to examine potential contribution of wetlands to streamflow (Brooks et al. 2018). These empirical studies then informed modeling studies that examined variability and drivers of hydrologic fluxes at the watershed scale and, importantly, how different scenarios of wetland loss (and restoration) may impact the watershed and downstream hydrology (e.g., Evenson et al. 2018; Neff and Rosenberry 2018). Specifically, scenarios of wetland loss (and restoration) may impact watershed and downstream hydrologic flows, aquatic habitat provisioning, and water residence times (e.g., Evenson et al. 2018; Neff and Rosenberry 2018). Similarly in the Canadian portion of the Prairie Pothole Region, Ameli and Creed (2018) used a combination of empirical measurements and hydrologic modeling to investigate the loss of NFW connectivity and its impact on water age and downstream baseflow.

Until very recently, model representation of wetland hydrologic connectivity and its variation (e.g., mode, magnitude, and timing) has been limited (Golden et al. 2014, 2017). However, such representation is needed to examine wetland influences on hydrologic, biogeochemical, and biological regimes at scales relevant for management and policy (Wellen et al. 2015). In this review, we discuss considerations for representing hydrologic connectivity of NFWs and their hydrologic exchange flows, present four examples of models that are designed to represent NFW hydrology, and end with several "best practices" for modeling wetland connectivity.

REPRESENTING WETLAND CONNECTIVITY IN PROCESS-BASED MODELING

Representing hydrologic exchange flows of NFWs in process-based models is an emerging challenge relevant to many research, regulatory, and management questions (Golden et al. 2017). Here, it is important to distinguish process-based from solely empirically based models (see Table 1), noting the focus of this review is on the former. Solely empirical-based models utilize statistical tools to link observations of drivers and responses (e.g., correlations between wetland attributes and measured surface-water flows; Epting et al. 2018). Because empirical models are based on observed data, they often provide predictions for the location and spatiotemporal scale from which the empirical measurements were taken. However, extrapolating beyond these boundaries can lead to diminished predictive capabilities. In contrast, process-based models are founded on mathematical representation of hydrological processes (e.g., Darcy's Equation, mass balance and momentum equations; Clark et al. 2011; Golden et al. 2014) and, as such, have the potential to quantify water storages and fluxes at defined spatial scales and temporal resolutions (Kuppel et al. 2018). Process-based models are particularly useful for projecting and hindcasting when detailed data are not available to parameterize empirical-based models. However, process-based models can suffer from over-parameterization and equifinality (i.e., when divergent parameter sets result in similar simulation results; see Beven 2006).

It is worth noting that the difference between empirical- and process-based models is somewhat nuanced, and there is a gradient from purely empirical-based models to purely process-based models. In particular, many modeling efforts along this gradient rely on comparing model predictions to empirical measurements; and to further muddy the waters, process-based models often employ empirical methods. For example, many process-based hydrologic models utilize the Natural Resources Conservation Service Curve Number approach to estimate runoff volumes, which at its core is an empirical model, where a parameter (i.e., the Curve Number) is used to fit a curve between observed rain and runoff data (see Walter and Shaw 2005).

Process-based models vary in their physical representation of specific landscape elements and their hydrologic flows and storages. Until recently, most process-based watershed models were calibrated using observations of flow at the watershed outlet (Wellen et al. 2015), and the simulations of hydrologic processes like the magnitude, duration, and timing of NFW connectivity within the watershed were not validated and often unrepresentative (Golden et al. 2014). For example, many hydrologic models often represent multiple wetlands as a single control volume, omitting potentially important factors such as cumulative wetland shore line length (but see Cheng and Basu 2017), horizontal fluxes between wetlands and adjacent upland (but see McLaughlin and Cohen 2013; Evenson et al. 2019), and spill–fill hydrology (but see Evenson et al. 2015; Hayashi et al. 2016).

Recent progress has been made to better represent NFWs and their hydrologic connectivity in process-based models. Such models range from simple, spatially lumped models (e.g., McLaughlin et al. 2014) that estimate mass balances to fully distributed models (e.g., Amado et al. 2016) that solve conservation

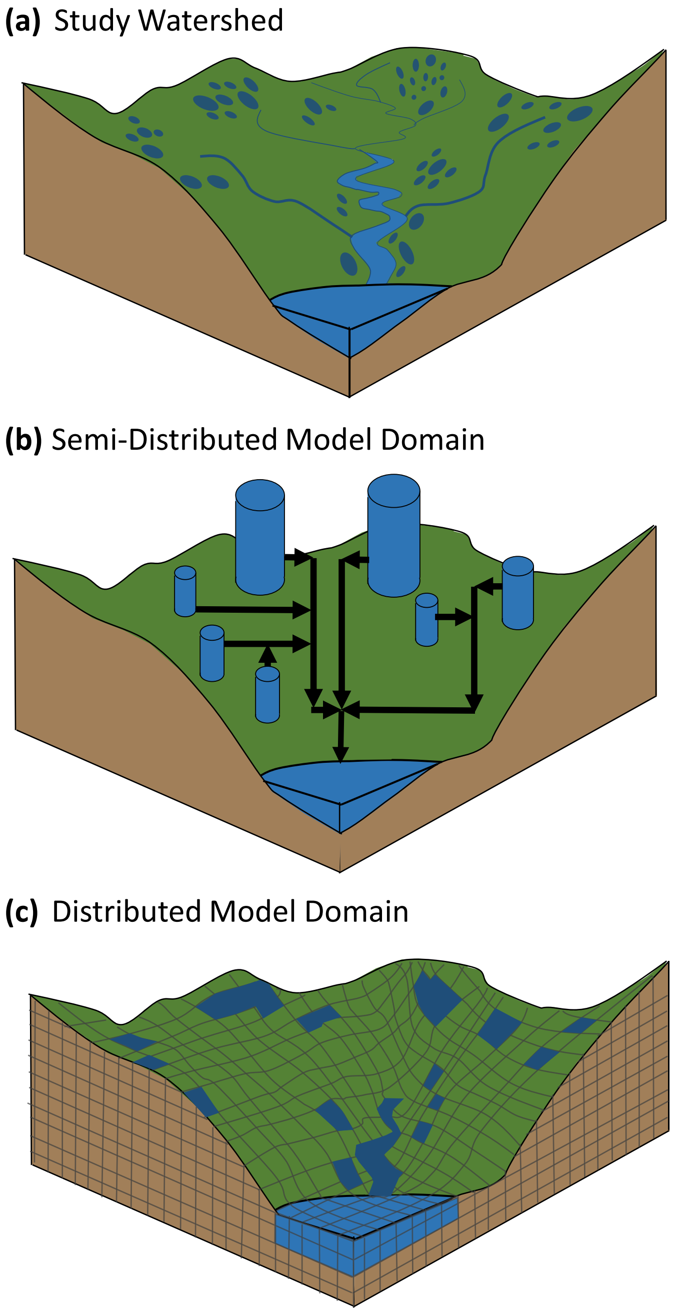

FIGURE 1. Illustration of how (a) study watersheds are discretized into smaller landscape units and gridded networks in (b) semi-distributed models and (c) distributed model domains, respectively. Note: Semi-distributed models are often referred to as *conceptually based models* because hydrologic fluxes are based on empirical equations and initiated by predefined thresholds like wetland storage capacity, whereas fully distributed models are sometimes referred to as *physically based* models because hydrologic fluxes are based on emergent properties, or mass and momentum exchange between individual grid cells.

of mass and momentum at increasingly high spatial and temporal resolutions (Figure 1; Table 1). These models occur along gradients of complexity, fidelity, practicality, and data availability (Clark et al. 2015; Golden et al. 2017). Here, model complexity refers to the number of parameters and hydrologic processes required to sufficiently satisfy modeling objectives; fidelity refers to the ability of the model to faithfully represent those processes (Table 1).

Given the gradient in modeling demands and capabilities, we argue that defining the modeling objective is an important first step to selecting the appropriate modeling approach. Specifically, defining (and iteratively reassessing) the modeling objective helps the modeler assess several aspects of the associated study. Namely, what are the spatial and temporal scales, processes, and empirical data required to facilitate the modeling activity, and what level of model fidelity is needed? Golden et al. (2017) recently provided several considerations for defining modeling objectives and selecting appropriate models, describing both different modeling approaches and the associated tradeoffs. Here, we briefly review these considerations in reference to (1) conceptualizing wetland connectivity and associated model domain, (2) the spatial representation of hydrologic processes, and (3) tradeoffs between model fidelity and resource requirements.

Wetland Connectivity Conceptualization and Model Domain

Modelers should strive to understand key hydrological processes in the system they are modeling and ensure the selected model can represent those processes. All process-based models are simplifications of reality, and the resulting simulations are inherently biased by those simplifications (Freeze and Harlan 1969). As such, there are important steps when approaching most modeling problems: (1) ensure relevant field observations are available to assist in parameterizing and representing specific processes simulated by the model, (2) develop a conceptual understanding of the system based on those field observations, and (3) either select or construct a model that comports with the conceptual understanding of the system (Sivapalan et al. 2003; Fenicia et al. 2008). (Note, these guidelines do not necessarily apply to modeling activities supporting decision-making in data-poor environments; see Hrachowitz et al. 2016; Jaramillo et al. 2018 for more details.)

Conceptualizing and simulating wetland connectivity has been challenging due to limited consensus across scientific and management communities on the definition of hydrologic connectivity (Ali and Roy 2009; Larsen et al. 2012; Bracken et al. 2013), and how and at which resolution to incorporate wetland flux exchanges with the other watershed components

(Golden et al. 2014). In part, this ambiguity is likely due to the common practice of *combining* Lagrangian and Eulerian reference frameworks in the fields of hydrology and water resources (Salamon et al. 2006; Doyle and Ensign 2009). Simply stated: hydrologic connectivity has been described in terms of both the movement of water particles (i.e., the Lagrangian framework) and by describing the mass balance of individual watershed components (i.e., the Eulerian framework).

In practice, most process-based hydrologic models utilize a combination of Lagrangian and Eulerian frameworks. This is accomplished by representing watersheds as systems of connected control volumes, allowing for the discretization of the landscape into distinct units where hydrologic processes can be simulated (see Bernhardt et al. 2017). This compartmentalization of the watershed allows for both conceptualization of specific, dynamic flow paths (i.e., Lagragian reference frames) and representation of individual storage zones or control volumes (i.e., Eulerian reference frames; McDonnell and Beven 2014; McDonnell 2017).

In wetland-rich systems specifically, the landscape is often partitioned into surface water, shallow subsurface, and deep groundwater model domains (Golden et al. 2014), where storage capacity of each domain and the hydrologic exchanges between domains are conceptualized using Eulerian and Lagrangian frameworks, respectively. Depending on both the goals of the modeling exercise and the landscape being represented, modelers often need to configure their models for a subset of these compartments. Here, it is important to note that hydrologic storage and flux representations and the spatial and temporal scales of the model are intrinsically linked.

Spatial Representation of Hydrological Processes

Simulated control volumes within a model diminish in size as spatial representation transition from a spatially lumped representation to fully distributed representation. Most process-based models are, at least in part, based on the first principles of conservation of mass and momentum; and these first principles move water between the specified control volumes. This means the simulated landscape is often discretized into units (e.g., control volumes) of sufficient size to distribute water across time and space. The spatial representation of control volumes may have consequences for the representation and estimates of hydrologic fluxes in the model. For example, models that solve both mass and momentum balances typically must satisfy the Courant Conditions (i.e., Courant et al. 1967), a condition that relates control volume size, time step, and model stability.

Often called conceptually based models, lumped and semi-distributed models spatially represent and aggregate specific watershed elements (e.g., uplands, a single NFW, or a group of colocated NFWs) into user-defined storage units and then route water between those units often using equations that presuppose physical processes, such as Manning's and Darcy's equations (Table 1; Figure 1b). In most semi-distributed models, hydrologic fluxes are predefined by model architecture and are typically initiated based on assigned thresholds (e.g., water-level thresholds triggering wetland surface-water outflows; Golden et al. 2017). Historically, semi-distributed models like the Soil and Water Assessment Tool (SWAT) (Arnold et al. 2012) have been developed to represent streamflow at the outlet of a watershed, and connectivity between watershed components is implicitly enforced by predefined model architecture. This simplified process representation is often too coarse for specific management or science needs (Wellen et al. 2015). For example, SWAT simulates multiple colocated wetlands as single control volume. While this may be an effective approach to developing weekly or monthly water balances at the watershed scale, it severely limits the ability of the model to simulate management decisions associated with individual wetlands (see Case Study on Evenson et al. 2016, 2018 presented below).

In contrast, fully distributed models often use physically based approaches to simulate hydrologic processes (e.g., the MODFLOW model; Harbaugh 2005; Table 1). Fully distributed models discretize the watershed into relatively small spatial units, typically arranged in a grid or mesh structure within which physical processes (and governing equations) of water and material movement are simulated (Figure 1c). As such, fully distributed models that employ physically based equations can be readily coupled with transport equations (e.g., Fiori and Russo 2008) or particle tracking approaches (e.g., Kollet and Maxwell 2008), allowing for detailed representation of the dynamics of water and material transport among wetlands and between wetlands and downstream waters (see Cui et al. 2014). In this way, hydrologic fluxes are emergent properties of model simulations, potentially providing more detailed characterizations of relevant hydrologic processes.

While fully distributed models often provide insights that lumped or semi-distributed conceptual models may not, fully distributed models often come at potentially prohibitive high resource requirements, including intensive data, modeling capacity/experience, and computational needs. For example, NFW management questions are often associated with land-use change, and more specifically, the conversion of wetlands, or their surrounding uplands, to impervious surfaces (Bierwagen et al. 2010). While a description of modeling impervious surface is beyond the scope of this review, detailed simulation at the watershed scale is often challenging because of the complex nature of engineered structures and lack of data to characterize the distribution of those structures (see Salvadore et al. 2015 for more information).

Managing Tradeoffs between Model Fidelity and Resources

Using hydrologic models involves tradeoffs between the quality and type of information produced by the model and the allocation of limited resources such as funding, time, personnel, modeling experience, and computing power. In addition, the availability of data for model parameterization, calibration, and validation often limits the quality of information produced from modeling.

In terms of the quality of information produced, the ability of a model to faithfully represent hydrologic fluxes is known as *fidelity* (Table 1) (Evans et al. 2013; Getz et al. 2018). There is a direct, but not necessarily linear, relationship between model fidelity and required resources (Golden et al. 2017). In this case, the allocation of additional resources essentially increases confidence that the information produced by the model is "correct." The type of information sought from a model can be broad and generalizable (e.g., McLaughlin et al. 2014; Neff and Rosenberry 2018), or it can be detailed information about the mode, magnitude, and timing of hydrologic fluxes (e.g., Ameli et al. 2017; Evenson et al. 2018). Generally, there is both an inverse relationship between model fidelity and generalizability (Harvey 2016).

Different types of empirical data may be needed for the model depending on both the dominant hydrologic fluxes in the system and target research or management objectives. For example, for quantifying wetland contributions to recharge and maintenance of baseflow in landscapes dominated with groundwater flow, baseflow and groundwater table measurements would be necessary for calibrating and validating the groundwater-based models as done in Ameli and Creed (2018). Alternatively, in landscapes with dominant surface flow processes and for management goals including flood risk mitigation, measurements of stormflow responses are required. Similarly, a management objective may focus on precise locations in a watershed where nutrient reduction is necessary. In this case, spatial and temporally distributed environmental tracer measurements may complement a fully distributed model. Synergistically, focusing on both surface and subsurface flow paths can be combined to better characterize both relevant hydrologic fluxes between landscape elements (e.g., Wellen et al. 2015) and legacy nutrients that experience relatively long transit times (McDonnell and Beven 2014; VanMeter and Basu 2015; Ameli et al. 2018).

CASE STUDIES: INTEGRATING WETLAND CONNECTIVITY INTO PROCESS-BASED MODELING

In the context of modeling considerations detailed in the previous section, we present four case studies that showcase advances in simulating hydrologic exchanges between NFWs and from NFWs to downstream waters. The case studies occurred in many ecoregions of North America, extending from Delmarva bays and similar NFWs of the Atlantic Coastal Plain and Mid-Atlantic ecoregions to Prairie Pothole wetlands of the Northern Glaciated Plains and into Canada. The process-based models we review cover a range of complexities, from relatively simple model structures to highly complex ones (Figure 2). Further, the four studies range in scale from individual wetland scale (e.g., <1–10 ha) to larger watershed scales (e.g., >1,000 ha). Given that representing wetland connectivity in process-based models is an emerging challenge (Golden et al. 2014, 2017), these examples provide context for future research, policy, and management questions and applications, and may find further utility in process-based modeling of other systems (e.g., small lakes, beaver pond management).

Lumped Wetland Model (U.S. Atlantic Coastal Plain)

The Wetland Hydrologic Capacitance (WHC) model is a simple, lumped hydrologic model designed to represent surface water–groundwater connectivity in low-gradient, wetland-rich landscapes. The WHC has been utilized to examine the effects of upland forest management on NFW function (Jones et al. 2018b), explore potential feedbacks between smoldering peat fires and wetland hydroperiod (Watts et al. 2015), and elucidate the role of NFWs in downstream waters (McLaughlin et al. 2014). The WHC simulates climatic-driven water level and soil moisture variation in two spatially lumped and exchanging control volumes: a lumped upland module and a lumped wetland module (top part of Figure 3). Major hydrologic fluxes include upland–wetland groundwater exchange (via Darcy flow) and cumulative groundwater flux out

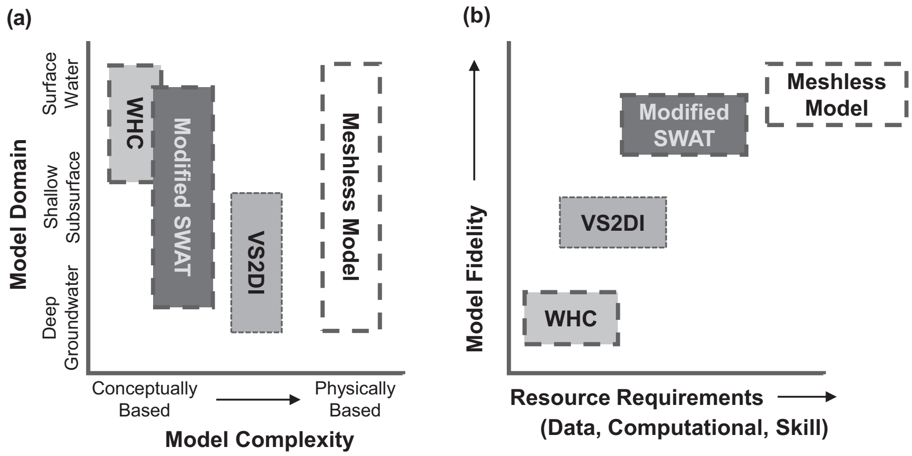

FIGURE 2. Presented models across (a) spatial representation and model domain and (b) model fidelity and resource requirements. WHC, Wetland Hydrologic Capacitance; SWAT, Soil and Water Assessment Tool.

of the entire model domain. The model can either be applied at the individual wetland scale (one lumped wetland and upland) or larger landscape scales (multiple pairs of lumped wetland and upland components), and for various scenarios of wetland size and density, climate forcing, and soil types.

McLaughlin et al. (2014) used the WHC to explore the role of groundwater–wetland hydrologic fluxes in landscape hydrology. More specifically, the modeling objective was to examine the influence of area and density of NFWs on surficial aquifer variation and resultant downgradient baseflow dynamics. This modeling effort was motivated and informed by field observations of NFWs transitioning from sinks of water (groundwater inflow) during wet periods to sources of water (groundwater outflow) during dry periods (see McLaughlin and Cohen 2013). The modeler scaled these small-scale observations and modeling results to larger scales to test the hypothesis that distributed wetlands provide landscape hydrologic capacitance, defined as the ability of wetlands to modulate both high and low water tables and subsequent baseflows to downstream systems. McLaughlin et al. (2014) simulations comport with the landscape hydrologic capacitance hypothesis and generally suggest that variation in baseflow decreases with increases in both wetland area and number of wetlands (bottom part of Figure 3). Further, simulations also suggest hydrologic capacitance increases with both precipitation and soil hydraulic conductivity (e.g., sandy soils). McLaughlin et al.'s (2014) modeling efforts clearly support the hydrologic capacitance hypothesis and provide general information about the influences of wetland size and density, climatic variables, and soil characteristics on aquifer and baseflow dynamics in low-gradient, wetland-rich landscapes.

The WHC requires low data and computational costs, is relatively robust, and provides generalizable results. Model inputs include readily available soil information (e.g., saturated hydraulic conductivity, porosity, wilting point), wetland morphology (e.g., wetland storage capacity), and climatic information (e.g., mean annual precipitation, daily temperature). Further, utilizing a normal desktop computer, a 1,000-year simulation can be completed in minutes. Finally, because simulations can be based on a simplified, synthetic landscape (i.e., a landscape based on modeler conceptualization), inference can be generally applied to many wetland-rich landscapes such as cypress domes in Florida, Carolina and Delmarva bays in the eastern U.S. coastal plain, and vernal pool systems in California and Texas. However, the WHC is relatively simple and does not take into account local variation in hydrologic storage (e.g., variable upland soil depth and texture; Deemy and Rasmussen 2017) and fluxes (e.g., preferential flow, Hester et al. 2016). Thus, site-specific application of this model would likely be unrepresentative without significant model improvements to allow for such parameterization. As such, lumped models such as the WHC may not be the best option for site-specific decision-making and management (e.g., prioritization and selection of specific wetlands for restoration or conservation; see Babbar-Sebens et al. 2013), which would likely benefit more from higher resolution semi- or fully distributed models but with potentially higher computational costs and data needs (Figure 2).

Semi-Distributed Watershed Model (U.S. Prairie Pothole Region)

Evenson et al. (2016) modified the SWAT model (a semi-distributed, watershed-scale model) to specifically

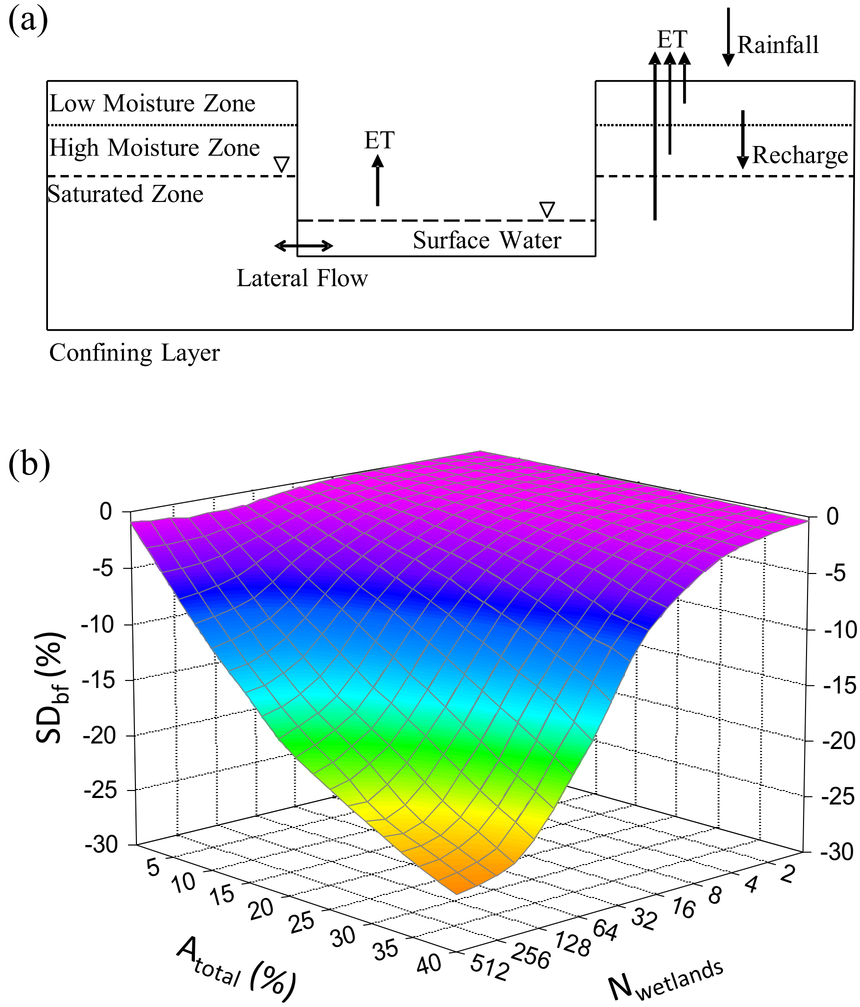

FIGURE 3. (a) Conceptualization of the WHC model, highlighting lumped upland and wetland model components. (b) The effect of increasing total wetland area (A_{total}) and total number of wetlands (N_{wetlands}) on simulated baseflow variation (i.e., relative change in baseflow standard deviation; ΔSD_{bf}). Figures were adapted from McLaughlin et al. (2014) and were reproduced with permision. ET, evapotranspiration.

represent spill–fill hydrology common in wetland-rich environments dominated by surface-water connections (i.e., a wetland first "fills" with inputs then "spills" via overland flow paths across the landscape or to another system; Spence and Woo 2003). SWAT is a process-based, watershed-scale hydrologic model commonly applied in rural and agriculturally dominated watersheds (Neitsch et al. 2011). As noted in previous section, the original SWAT model represents colocated wetlands as a single control volume, thus limiting the representation of the hydrologic fluxes of individual wetlands and their hydrologic connections to other wetlands and to the stream (Golden et al. 2014; Evenson et al. 2015, 2016; Wellen et al. 2015). In response, Evenson et al. (2016) modified SWAT to depict individual NFWs as distinct hydrologic response units (HRUs), where HRUs are SWAT's most elemental spatial unit of simulation. Evenson et al. (2016) also modified the model to estimate the probability of inter-wetland "fill–spill" hydrologic connections whereby upgradient wetlands fill to capacity and then spill to downgradient wetlands (Figure 4). A previous iteration of the model was used to examine hydrologic connectivity between NFWs and downstream waters in the southeastern and mid-Atlantic U.S. coastal plains (Evenson et al. 2015; Lee et al. 2018), and an updated version of the model has been used in two watersheds in the Prairie Pothole Region (Evenson et al. 2018; Muhammad et al. 2019).

Evenson et al. (2018) applied the modified SWAT model to a study with an objective of quantifying the cumulative impacts of NFWs on downstream waters in a large North Dakota watershed. The modeling objective was to inform wetland conservation and restoration efforts by assessing wetland loss impacts to watershed functions. To inform potential

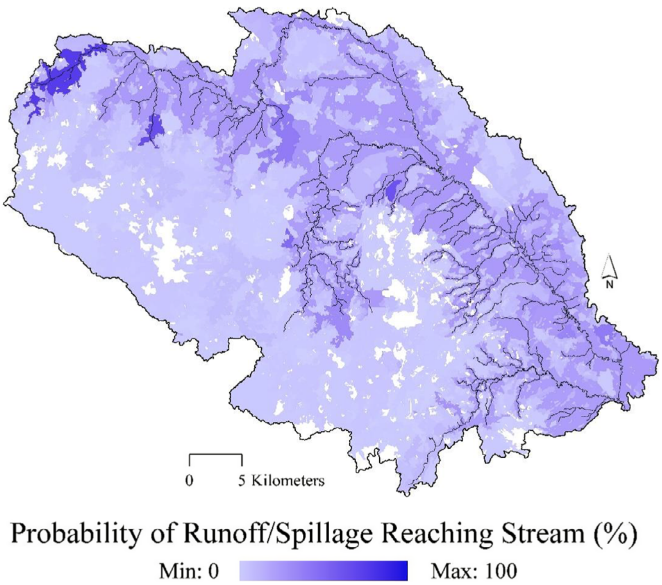

FIGURE 4. Simulation results from Evenson et al. (2018) highlighting the probability of wetland-generated runoff (or spillage) to downstream waters.

management decisions, authors executed a series of model scenarios in which all or particular subsets of Prairie Pothole wetlands were removed from a baseline model. Scenario results suggested that wetland management strategies should balance protection of both small and large wetlands for conservation of multiple wetland functions. Further, results indicated that large wetlands played an important role in reducing storm peaks while smaller wetlands were hotspots for biogeochemical processing. In doing so, Evenson et al. (2018) provided both site-specific recommendations for land management (e.g., priority areas for wetland conservation derived from the scenario analysis) and generalizable management approaches for wetland-rich landscapes (e.g., small wetlands provide increased nutrient removal capacity whereas large wetlands provide attenuation of stormflows).

The modified SWAT model used by Evenson et al. (2018) is more computationally intensive when compared to the traditional SWAT model, especially when applied to large quantities of wetlands. For example, Evenson et al. (2018) simulated ~13,000 NFWs within the ~1,700-km^2 study watershed, requiring approximately one hour of runtime for a five-year simulation with a daily timestep on a conventional desktop computer. In contrast, a traditional SWAT model would require only minutes to complete the same simulation, though without the spatial specificity of the modified model. In addition to the traditional data requirements of SWAT (i.e., elevation, soil, and land-use data as well as weather observations), the modified version of SWAT requires wetland spatial data that include maximum storage capacity estimates (e.g., as provided by Lane and D'Amico 2010; Jones et al. 2018a; Wu et al. 2019). Further, as wetland spatial boundaries are "hard-wired" into the model structure (i.e., the model utilizes static HRU boundaries), the modified model does not explicitly simulate "fill–merge" hydrologic connections (i.e., when two HRUs are hydrologically connected by surface inundation; see Leibowitz et al. 2016).

Fully Distributed Gridded Groundwater Model (U.S. Prairie Pothole Region)

VS2DI is a two-dimensional, groundwater model developed to examine flow and transport in variably saturated porous media along a specified vertical transect (Rossi and Nimmo 1994; Hsieh et al. 2000). Developed and distributed by the U.S. Geological Survey (USGS), VS2DI has been widely used for a variety of purposes, ranging from aquifer recharge simulations (e.g., Heilweil et al. 2015), to evaluating performance of bioretention cells (e.g., Zhang and Chui 2017) and investigating connectivity of NFWs to downstream waters (e.g., Neff and Rosenberry 2018). VS2DI utilizes a finite-difference approach in conjunction with Richard's equation to simulate unsaturated groundwater flow. In the modeling domain, a two-dimensional vertical cross section of a groundwater system is simulated, with boundary conditions assigned and reassigned for multiple time periods, a feature helpful to simulating climate variability such as seasonality and drought. Spatial and temporal scales can vary from a single wetland shoreline cross section of a few meters or smaller (Rosenberry 2000), to cross sections 75 km long or greater, with time scales of seconds to hundreds of thousands of years (Neff and Rosenberry 2018).

In the same setting described in Evenson et al. (2018), Neff and Rosenberry (2018) used VS2DI to develop a narrative of how landscape factors such as geologic setting and topography affect groundwater connectivity between NFWs and downstream waters (e.g., Figure 1). Specifically, their modeling objective was to simulate hypothetical landscapes that typify the Prairie Pothole Region and determine the range of geologic conditions that permit groundwater connectivity between wetlands and downstream waters. In essence, they manipulated the model to determine what conditions, realistic or not, would be necessary to allow groundwater to flow between wetlands and downstream water bodies. This required use of a fully distributed and ideally physically based model, but the exercise of modeling hundreds of combinations of

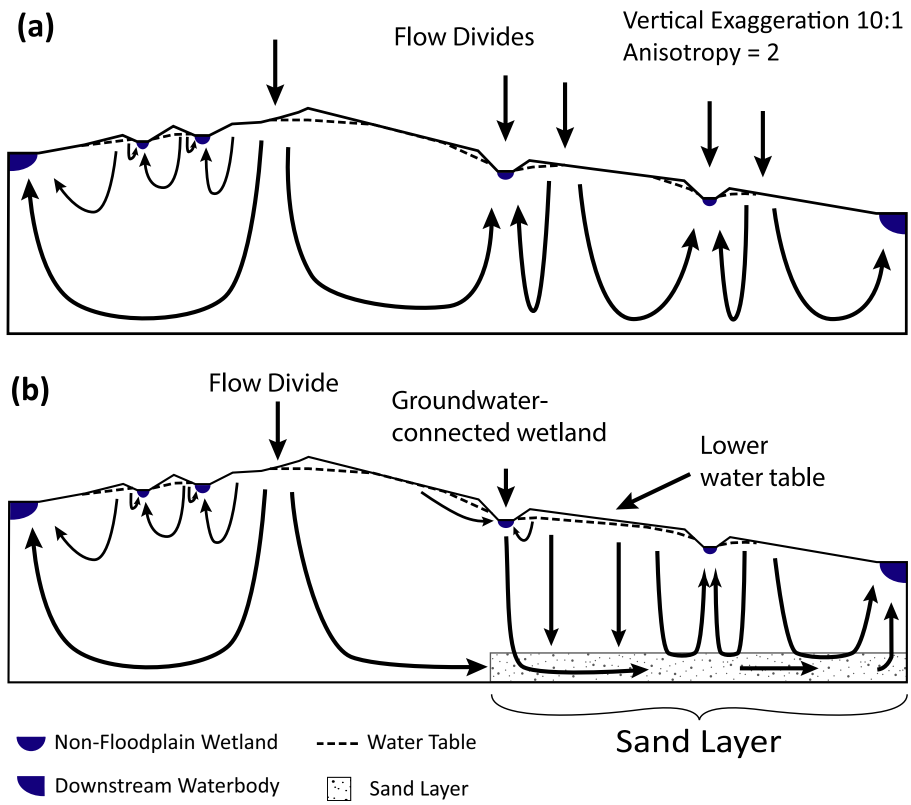

FIGURE 5. Groundwater flow simulation showing water table mounds, represented as flow divides, as a barrier to groundwater flow. The addition of a sand layer (b) lowers the water table and also allows groundwater to flow from a NFW, through the sand layer, under water table mounds, and to a downstream water body. Adapted from Neff and Rosenberry (2018).

parameters and domains favored the use of a relatively simple and fast two-dimensional modeling program. Simulations showed water table mounding, a condition where the differential water table gradients create a barrier to subsurface flow, existed around most wetlands and served as a barrier to groundwater flow between the wetland and downgradient waters (Figure 5a). However, in some situations sand deposits acted as a conduit to groundwater flow under a water table mound or lower the water table locally, sometimes sufficiently to dissipate a mound to allow downgradient connectivity (Figure 5b). Neff and Rosenberry (2018) also found anisotropy, or differences in vertical and horizontal hydraulic conductivity, could account for regional groundwater flow observed in some studies of the Prairie Pothole Region, but only if it is unrealistically high. Sand deposits, however, are common in the Prairie Pothole Region and likely account for observed regional groundwater flow.

Here, it is also important to highlight that Neff and Rosenberry's (2018) modeling results comport with results from Brooks et al. (2018), an empirical study that used water isotopes (e.g., δ^2H and δ^{18}O) as a hydrologic tracer to examine connectivity between Prairie Pothole wetlands and an adjacent water body. Both studies were based on the same watershed (e.g., The Pipestem Watershed in North Dakota — USGS Gage 06469400), and together, provide strong evidence that groundwater mounding can limit subsurface connectivity between NFWs and downstream water bodies in the Prairie Pothole Region.

The conceptual nature of the modeling results from Neff and Rosenberry (2018) has interesting implications for the generalizability, limitations, and utility of findings. In the context of wetland management, Neff and Rosenberry (2018) highlight a specific hydrogeomorphic setting (see Figure 5) where groundwater connectivity between wetlands and downstream water bodies is limited. In settings where groundwater mounding may be prevalent, conceptual knowledge from Neff and Rosenberry (2018) could be utilized in wetland management decisions, especially when more detailed empirical and modeling data are not available. Further, future work could use this same approach to examine NFW connectivity in other physiographic settings, highlighting differences across landscapes relevant to regional decision-making and conservation efforts.

From a management perspective, the two-dimensional nature of VS2DI presents strengths and limitations. VS2DI is well suited for building conceptual knowledge of connectivity within a small group of wetlands or the long-distance, regional connectivity of individual wetlands. However, a three-dimensional model, such as MODFLOW (Harbaugh 2005), allows users to import digital elevation models of actual landscapes and simulate connectivity across a three-dimensional landscape. In this case, VS2DI is still useful to efficiently build conceptual understanding of a complex setting and illuminate how more complex models should be used.

Fully Distributed Coupled Surface–Subsurface Model (Canadian Prairie Pothole Region)

Ameli and Creed (2017) utilize a new *grid-free*, physically based subsurface flow model to simulate hydrologic connectivity of individual wetlands to downstream water bodies at the watershed scale (Figure 6). Notably, this new model attempts to address a key point of uncertainty relevant to management, decision-making, and research questions: how does the hydrologic functioning of *individual* wetlands affect the patterning of both streamflow hydrograph and regional groundwater variation? For most physically based models, the incorporation of wetlands of different sizes requires fine-scale grid discretization that in turn can be computationally inefficient in large wetland-dominated landscapes (Golden et al. 2014). Ameli and Creed (2017) address this challenge by employing a semi-analytical solution of groundwater mass and momentum transfer, which importantly, does not require discretizing the landscape into individual control volumes. Thus, this new grid-free model allows for both the accurate representation of wetland geometry and use of Lagrangian reference frame (e.g., allowing the user to track movement of individual particles of water).

Ameli and Creed (2017) applied their model to a large watershed in the Canadian Prairie Pothole Region. The modeling objective was to examine the hydrologic connectivity between individual wetlands and downstream water bodies, with a specific focus on the effects of the historical loss of individual wetlands on watershed hydrology. The simulated watershed is roughly 4,000 km^2 in size with over 100,000 Prairie Pothole wetlands. Through the explicit mapping of surface and subsurface hydrologic connections of each individual wetland to a major stream network, the authors showed that the distance between wetlands and stream network is not always a proxy for hydrologic connectivity. Further, Ameli and Creed (2018) suggested that historical loss of wetlands in the studied watershed significantly increased the age of groundwater baseflow as well as the frequency of hydrologic connections between NFW and downstream water bodies. Notably, Ameli and Creed (2018) suggested persistent loss of individual wetlands enhance the potential of drought at the larger watershed scale, highlighting the need for continued wetland conservation and restoration in the greater Prairie Pothole Region.

Similar to most physically based models, the coupled surface–subsurface model of Ameli and Creed (2017) requires data for soil properties, climate, and land-surface characteristics (e.g., Manning's roughness parameters). The model then can be calibrated and validated against streamflow and baseflow observations and/or tracer measurements. Despite the accuracy, resolution, and time efficiency of the model, it suffers from two issues that weaken its applicability in some landscapes. First, the subsurface portion of the model by Ameli and Creed (2017) is steady-state and cannot detect fast, dynamic subsurface connections within the landscape. This might be problematic in environments with a high proportion of macropores and fractures (e.g., dissolution wetland features in karst landscapes), wherein fast, shallow lateral movement of water and material is the norm (e.g., Cao et al. 2018). Second, the linkage between the subsurface model and surface processes is not fully integrated and leads to difficulty in characterizing wetland connectivity in landscapes with pronounced feedbacks between surface and subsurface pathways.

LESSONS LEARNED: WETLAND CONNECTIVITY AND PROCESS-BASED MODELING

The four modeling studies we present herein focus on different modeling domains (e.g., groundwater, surface water), spatiotemporal scales, and levels of fidelity, providing considerations and potential insight for model selection in wetland-dominated watersheds. For instance, the Evenson et al. (2018) and Neff and Rosenberry (2018) models were designed to simulate hydrologic fluxes of surface water and groundwater, respectively. Even though these studies were conducted on the same watershed, the transit time of simulated hydrologic fluxes occurs at vastly different time scales (e.g., days to years in Evenson et al. 2018; 1,000s of years in Neff and Rosenberry 2018). These differences in temporal scales are directly related to differences in each study's research objectives, as well as notable differences in the transit time distributions of surface and subsurface flow paths, highlighting a critical consideration when defining modeling objective and domain.

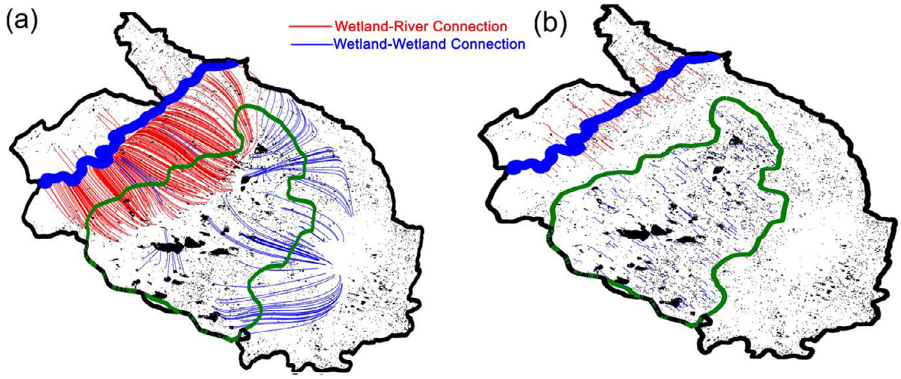

FIGURE 6. Hydrologic connectivity among wetlands (blue lines) and between wetlands and North Saskatchewan River (red lines). (a) Map of subsurface connections: only particles released from recharge wetlands located in the moraine (shown by green line) that reached the discharge wetlands (blue lines), and particles that discharged into North Saskatchewan River from recharge wetlands (red lines) are shown. (b) Map of surface connections for the period when the largest net surface-water fluxes since 2000 occurred.

Similarly, spatial scale is an important consideration when simulating wetland hydrology. Across the four presented studies, the latter three studies occurred across relatively large watersheds (approximately 1,700–4,000 km^2). However, this is not always the appropriate scale for inference, which varies based on the objectives of the study. For wetland-rich systems, relevant scales of interest include the individual wetland scale (e.g., roughly <1–10 ha in size; Park et al. 2014; Bertassello et al. 2018); the wetland complex scale (e.g., on the order of tens to hundreds of hectares in size; Leibowitz and Vining 2003; Hayashi et al. 2016); and large-watershed scale (e.g., on the order of hundreds of hectares to thousands of square kilometers; Fossey and Rousseau 2016; Golden et al. 2016). In wetland-rich landscapes like the Prairie Pothole Region or portions of the southeastern U.S. coastal plain, determining the appropriate scale is often difficult because of complexities associated with fill–spill–merge patterns (Shaw et al. 2012; Vanderhoof and Alexander 2016), nested catchment structures (McCauley and Anteau 2014; Wu and Lane 2017), and surface water–groundwater exchanges across wetland complexes (McLaughlin et al. 2014; Brannen et al. 2015). Focusing on the individual wetland (e.g., Jones et al. 2018b) could result in poor predictions because the interacting, emergent properties of connected wetlands are not accounted for. However, at the other extreme of model complexity (e.g., Figure 2a), representatively modeling large spatial extents with sufficiently high levels of detail can require prohibitive amounts of resources (e.g., computational time, modeling capacity/experience) and require high-resolution data. Thus, the choice of spatial and temporal scales requires careful consideration of study objectives, balancing the requirements for hydrologic flux data, ancillary data availability, available personnel, time, and temporal resources.

The four modeling studies we reviewed also highlight tradeoffs between model complexity, spatial representation, and model fidelity. While McLaughlin et al. (2014) and Neff and Rosenberry (2018) utilize relatively simple models, their results are broadly generalizable because of the range of simulations possible with their models. For example, McLaughlin et al. (2014) highlight potential linkages between the spatial abundance, or wetland density, and hydrologic capacitance across three different soil types and across 1,000-year stochastic climate simulation. If this modeling exercise had been confined to a specific site and/or time period, the broad inference (i.e., the degree of WHC varies with wetland configuration, soil texture, and climatic regimes) would be lost. However, the WHC is also limited because it does not simulate potentially dominant hydrologic processes like preferential flow paths (e.g., Hester et al. 2016) and spatially variable water storage. Therefore, while McLaughlin et al. (2014) provide a compelling narrative of wetland functioning and are suggestive of several potentially fruitful areas for further inquiry, it has limited practical application for site-specific management.

In contrast, managers could use the spatially explicit models employed by Evenson et al. (2018) and Ameli and Creed (2017) for targeted wetland conservation and management. Simulations from Ameli and Creed (2017, 2018) directly link specific wetlands to downstream waters. Thus, the impact of land-use change and wetland loss could be tested and used to optimize management activities. However, in these studies, results are still sensitive to uncertainties in both process representation and input data. Simulated fluxes at the individual wetland scale may prove inaccurate, but simulated spatial and temporal trends at the watershed scale may be representative. For example, SWAT and other highly parameterized models exhibit a large degree of uncertainty due to potential equifinality. To address uncertainty associated with equifinality, Evenson et al. (2016) utilized an iterative calibration and uncertainty analysis to identify over 250 possible model parameter sets. The range of simulation output across these parameter sets constituted the uncertainty range of the model results (see also: Gallagher and Doherty 2007; Matott et al. 2009).

Moving forward, we suggest the water resources community needs to continue developing improved models that focus on and represent NFWs. In particular, while we are seeing "off-the-shelf" modeling products like SWAT and Hydrogeosphere (Therrien et al. 2008) used to model wetland connectivity, more work needs to be done to explicitly represent hydrologic fluxes within these models. In the case of SWAT specifically, the evolution of an object-oriented model (i.e., Bieger et al. 2017) will allow users to simulate individual wetlands, reservoirs, and lakes, and thus, greatly improve the ability of users to represent wetland hydrology like the modified model in Evenson et al. (2018). For physically based models like Hydrogeosphere, MODFLOW, and VS2DI, we recommend that improved techniques are developed for an efficient incorporation of wetland bathymetry as well as efficient incorporation of connectivity across subsurface and surface flow paths. Nonetheless, the recent development of physically based, grid-free models is promising (e.g., Ameli and Creed 2017), facilitating an accurate incorporation of wetland geometry and characterization of Lagrangian movement.

In addition to continuing to develop model structure to better represent wetlands, our community needs to consider new validation and calibration data and techniques. While there are general guidelines for calibration/validation procedures (e.g., Grimm et al. 2010; Rose et al. 2015) and assessing acceptable model fit in watershed modeling (e.g., Moriasi et al. 2007), traditional calibration/validation procedures are based on characterizing flow at the catchment outlet. Moving beyond streamflow as the primary variable will allow for better representation of watershed patterns of hydrologic storage and flows. Additional data for calibration/validation include remotely sensed inundation data (e.g., Lang et al. 2012; DeVries et al. 2017; Vanderhoof et al. 2017b), distributed water-level sensor networks (e.g., McLaughlin and Cohen 2013; Epting et al. 2018), a suite of environmental tracer data (e.g., Fossey and Rousseau 2016; Thorslund et al. 2018), and nutrient isotope data (Kendall et al. 2015), as well as qualitative data to conceptualize both expert knowledge and local residents' experiences (e.g., Seibert and McDonnell 2002). As an example, Ameli et al. (2017) used spatially distributed water-level measurements from 1,400 groundwater wells and tracer measurements from 200 wetlands to validate their grid-free, physically based wetland connectivity model.

Finally, managing wetlands using a watershed or systems approach that considers a portfolio of wetland functions requires understanding how wetlands hydrologically connect to each other and to downstream surface waters (Creed et al. 2017). As a tool, models are well suited for this task. The four studies reviewed here primarily aim to characterize hydrologic connectivity between NFWs and their effects on downstream waters, and, as such, are directly useful in management applications. For example, Evenson et al. (2018) simulate the magnitude, duration, and timing of surface-water fluxes between NFWs and downstream waters — and in doing so highlight the importance of both small and large wetlands for hydrologic and ecosystem functions, respectively. Evenson et al. (2018) also present a tool that practitioners could utilize for specific questions (e.g., the modified SWAT model could be used to examine how a large conservation easement could potentially affect downstream flows). Furthermore, the models cover a gradient of approaches that can be used for different targeted management goals, for example, addressing different levels of wetland management challenges in the Lake Winnipeg Watershed (see Golden et al. 2017).

BEST PRACTICES FOR MODELING WETLAND CONNECTIVITY

Continued use of models to quantify the movement of water, material, and organisms will be required to effectively manage and conserve NFWs and their river networks (*sensu* Hynes 1975; see also Creed et al. 2017; Harvey et al. 2018; Leibowitz et al. 2018; Lane et al. 2019). In this analysis and synthesis, our goal was to assess and compare four different modeling approaches to provide examples that researchers, regulators, and land managers can use to effectively simulate and assess the influences of wetland connectivity. In addition, we hope this review can further stimulate subsequent innovations in aquatic connectivity science.

We recognize that, ultimately, the "best" approach for selecting a model to simulate wetland connectivity is contextually dependent on the questions that are being asked and the data available to answer them. However, we conclude the paper with five major recommendations (listed below), based on the example studies herein, to improve NFW connectivity modeling techniques and applications. Notably, our list is not meant to be a comprehensive guide to modeling wetland hydrology; that responsibility falls to others (e.g., Golden et al. 2014, 2017).

Clearly Articulate the Goals of the Modeling Exercise to Best *Represent the System's Wetland Connectivity*

This remains the most important step in any modeling exercise, and it should be revisited regularly and the goals updated as new skills are learned, new information is obtained, or new methods are developed. Doing this will affect the type of model that is chosen, along with the necessary spatial and temporal scales required to simulate wetland connectivity and other watershed hydrological processes. Specifying the goals of the study should also clarify the required model domain (e.g., surface water, shallow subsurface, and deep groundwater flow paths) and model fidelity for connectivity questions. In the case studies we reviewed, the importance of articulating clear modeling goals is evident when comparing Evenson et al. (2018) and Neff and Rosenberry (2018). While both studies focused on NFW hydrologic connectivity in the same watershed, the goals of these two modeling studies were different. Evenson et al. (2018) focused on cumulative impacts of surface and shallow subsurface fluxes at the watershed scale; whereas Neff and Rosenberry (2018) focused on deep groundwater fluxes. As articulated, their respective objectives led to a selection of different model domains, and time scales, and, ultimately, divergent conclusions.

Develop a Conceptualization of Wetland Connectivity Using a Combination of Qualitative Observations, Empirical Data, and Process-Based Modeling

Begin by using landscape cues to develop a conceptual understanding of wetland connectivity with the surrounding area. Then use this conceptual understanding to guide field investigations. Finally, use both the conceptual understanding and field observations to guide hydrologic modeling and further improve knowledge of wetland connectivity relevant to the management question at hand. Using these approaches together improves understanding through triangulation and, from a management perspective, reduces the resources required to generate information needed to support decision-making. We further submit that modeling can then improve relevant conceptual knowledge and better guide future measurements of connectivity. This general advice is based on early hydrologic modeling literature (e.g., Freeze and Harlan 1969) and a widely accepted tradition within hydrologic sciences and water resources communities.

Development of the WHC model provides an example of this general process. McLaughlin and Cohen (2013) provide an empirical study of NFWs' role in stabilizing downstream baseflows. This led the authors to the Landscape Hydrologic Capacitance hypothesis, and to develop what would be later named the WHC model to explain observed field conditions, extend inferences to other landscape settings, and design future experiments and sampling campaigns. While many practitioners and researchers will not have time or resources for this type of cyclic activity, this example highlights the value of synergistically combining conceptual understanding, empirical measurements, and process-based modeling to better understand NFW connectivity (see Golden et al. 2017 for more in-depth discussion).

Select a Model to Represent NFW Connectivity by Balancing Both Modeling Objectives and Available Resources

Selecting the "right tool for the right job" will usually lead to better, more efficiently generated, results. One of our primary goals of this review was to provide a menu of options that practitioners and researchers can use to guide their own model selection and development decisions when considering questions related to wetland connectivity. To this end, we articulated the model domain, spatial representation, fidelity, and computational requirements for four available wetland connectivity modeling approaches (see Tables 2–3). In addition to these factors, our case studies suggested that model selection decisions should be based on informal cost–benefit analyses, where the largest costs may be the time required to learn new models such as VS2DI, MODFLOW, and SWAT; time investments to develop or modify modeling code to address the research or management questions; and computational expenses (i.e., simulation times). Therefore, while there are formal resources to guide the model selection process (e.g.,

TABLE 2. Model domain, spatial representation, fidelity, and resource requirements of models used in case studies.

Model	Study	Model domain	Spatial representation	Fidelity	Resource requirements
WHC	McLaughlin et al. (2014)	Wetland and shallow subsurface	Lumped	Low	Low
Modified SWAT	Evenson et al. (2018)	Wetland and shallow subsurface	Semi-distributed	Medium-high	High
VS2DI	Neff and Rosenberry (2018)	Deep groundwater	Distributed (2D)	Medium	Medium
Grid-free model	Ameli and Creed (2018)	Coupled surface and subsurface	Distributed (3D)	High	Very high

TABLE 3. Primary results from the presented case studies.

Study	Major finding
McLaughlin et al. (2014)	Groups of NFWs can attenuate both high and low shallow subsurface flows to downstream waters
Evenson et al. (2018)	Large NFWs are important for attenuating flooding, while small NFWs are important for nutrient retention and downstream water quality
Neff and Rosenberry (2018)	Groundwater mounding can limit connectivity between NFWs and downstream waters
Ameli and Creed (2018)	Wetland loss decreases downstream baseflow, increases catchment transit time, and increases the vulnerability of water resources to drought

Golden et al. 2014), model selection decisions — for wetland connectivity questions and many others — are context specific and will often need to balance modeling objectives with available resources and/or modeling experience.

Use Innovative Calibration/Validation Techniques for Simulating Wetland Connectivity

Current calibration and validation techniques based on comparing simulated and measured flow at watershed outlets are typically insufficient when modeling hydrologic fluxes and associated transport processes, particularly for NFW connectivity simulations. Model performance may be sufficient for traditional calibration and validation outputs (e.g., comparing simulated and measured streamflow at the outlet of a watershed; Moriasi et al. 2007), but can fail to accurately represent wetland water storages and fluxes and thus the role of NFWs in watershed hydrology (e.g., inundation patterns) and downstream flows (Evenson et al. 2019). We suggest keeping abreast of the continually improving validation and calibration literature, and becoming part of it: be creative, and share your results! Examples of nontraditional, but potentially useful sources of calibration/validation data for wetland connectivity simulations include distributed hydrometric data (e.g., Ameli and Creed 2017) and remotely sensed inundation data (e.g., Evenson et al. 2019). Others that could be used in wetland connectivity simulations include daily evapotranspiration estimates (e.g., Herman et al. 2018; Rajib et al. 2018) and qualitative data (e.g., Seibert and McDonnell 2002). Further, similar to the uncertainty analysis utilized by Evenson et al. (2016), there are increasingly accessible techniques researchers and practitioners can employ to quantify and bound uncertainty associated with calibration and validation results (e.g., Abbaspour 2013; Beven and Binley 2013).

Articulate the Limitations of Your Model and How This Affects the Type of Wetland Connectivity Simulated

Quantifying the effects of, or adequately characterizing, NFW connectivity remains a nascent science, and there remain many uncertainties associated with modeling NFWs and their effects on downstream waters. For example, all four of the reviewed models have their shortcomings: simplified flow equations in McLaughlin et al. (2014); equifinality in Evenson et al. (2016); and lack of subsurface data to parameterize both Ameli and Creed (2018) and Neff and Rosenberry (2018). Clearly articulating these limitations allows readers and users to understand the extent to which results can be applied to management questions. Moreover, including concise model limitations affords a clearer understanding of the insights that can be gained from each approach.

However, adequately characterizing NFW connectivity continues to be a challenge for management, regulatory, and research communities. There remain many uncertainties associated with modeling NFWs and their effects on downstream waters. Therefore, in addition to providing analyses and discussion on effective and varied contemporary modeling approaches that end-users and/or readers may choose for their needs, we hope this review can further stimulate subsequent innovations in aquatic connectivity science.

ACKNOWLEDGMENTS

CNJ was supported by the National Socio-Environmental Synthesis Center under funding received from the National Science Foundation DBI-1052875. This work was conducted as a part of the North American Analysis and Synthesis on the Connectivity of Geographically Isolated Wetlands to Downstream Waters Working Group supported by the John Wesley Powell Center for Analysis and Synthesis, funded by the U.S. Geological Survey and U.S. Environmental Protection Agency. The views expressed in this paper are those of the authors and do not necessarily reflect the views or policies of the U.S. Environmental Protection Agency. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

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