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Staff Report 23-SR 123
July 2023

Center for Agricultural and Rural Development
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A Review of Ecohydrological and Hydraulic/Flood models for Evaluating the Water Quality and Flood Mitigation Potential of Natural Infrastructure (NI) Practices in the Mississippi-Atchafalaya River Basin

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Abstract

Extraordinary interventions are needed to overcome the extensive flooding and water quality problems that continue to manifest within the Mississippi-Atchafalaya River Basin (MARB). The Mississippi River & Tributary levee system has resulted in successful flood control in recent decades in the Lower Mississippi River (LMRB). However, hundreds of levees have failed in other MARB subregions as a result of recent floods, and the extensive LMRB levee system and other factors are linked to the considerable loss of coastal wetlands and habitat areas in southern Louisiana. Water quality degradation continues essentially unabated across much of the MARB, especially in the Corn Belt region. Massive investment in a wide range of cropland conservation practices and other interventions has unfortunately yielded only minor improvement at best in MARB stream system water quality. Widespread adoption of natural infrastructure (NI) practices provides the potential to establish greater progress in mitigating both flood impacts and improving in-stream water quality in MARB waterways, especially if those practices are embraced in a more aggressive manner within federal and state conservation practice strategies.

The Agricultural Policy/Environmental EXtender (APEX), Soil and Water Assessment Tool (SWAT) and SWAT+ models developed at the co-located USDA and Texas A&M AgriLife laboratories in Temple, TX show the strongest overall abilities to accurately replicate these NI practice impacts. The SWAT+ model is particularly advantageous, due to the incorporation of previous APEX and SWAT modeling strengths and the current development of the National Agroecosystem Model (NAM) system. The utility of SWAT+ is further underscored by the relative ease of interfacing it with other models, such as previous studies that report model cascades between SWAT and Hydrologic Engineering Center-River Analysis System (HEC-RAS). This attribute provides the potential to expand an adapted NAM modeling system to not only capture MARB NI practice flood and water quality impacts, but also account for more detailed flood analyses if needed. It is recommended that the SWAT+/NAM modeling system be adopted for future MARB NI practice simulations. Further development will be required to implement the full suite of NI practices discussed here. This can be achieved via close collaboration with the SWAT+/NAM
developers and the required investment needed to incorporate physical representation of the full suite of NI practices in future MARB applications.
1. Introduction

The Mississippi-Atchafalaya River Basin (MARB) drains an area of approximately 3.3 million km² across parts or all of 32 U.S. states and 2 Canadian Provinces, which equates to 41% of the contiguous U.S. and ranks among the five largest basins worldwide (Miranda et al., 2020; Robertson and Saad, 2021). The MARB consists of six major water resource regions: Arkansas-White-Red River System (AWRRS), Lower Mississippi River (LMRB), Missouri River Basin (MRB), Ohio River Basin (ORB), Tennessee River Basin (TRB) and Upper Mississippi River (UMRB) Basins (USGS, 2022b; Wikipedia, 2021). The region provides critical habitat for an extensive array of North American wildlife (NPS, 2021), including serving as the major migratory flyway for both waterfowl (40% of total) and 326 species of other birds (60% of total), and hosting approximately 260 fish species (25% of the U.S. total) throughout the stream system (NPS, 2021). The Mississippi River and major tributaries have also served as important transport and commerce routes since the early 1800s, which has been expanded over time by: (1) channelization; (2) lock and dam systems built on the main stems of the Mississippi River (in the UMRB, ORB, and AWRRS); and, (3) other structural interventions on the main Mississippi artery and various tributaries (Alexander et al., 2012). There are also 1,700 reservoirs located within the MARB that cover a total surface area of 35,000 km² and hold a combined volume of 317 km³ (Miranda et al., 2020).

The MARB clearly offers substantial ecosystem and transportation services but is also severely stressed by acute flooding, ecological and water quality problems (Gonzalez and Kuzma, 2020; Newman et al., 2019; Mcdonald, 2017; American Rivers, 2022). Changnon (1998), Rogers (2008) and Criss and Luo (2017) collectively chronicle numerous severe floods that occurred in parts of the MARB stream system between the early 1800s into the 21st century. Several of these floods can be described as “massive” including floods that resulted in widespread destruction in large subregions during 1858, 1874, 1927, 1936, 1973, 1993, 1997, 2011 and 2019 (Noble, 1976; Barry, 1997; Changnon, 1998; Pitlick, 1997; Alexander et al., 2012; Criss and Luo, 2017; Almukhtar et al., 2019; Renfro, 2019).

Mississippi River flood control efforts were initiated in the New Orleans, Louisiana area, with the construction of levees shortly after the establishment of the city in 1718 (Rogers, 2008; 2009; Kelley
2008; Kates et al., 2006; Morris, 2017; Table 1). Levees have continued to serve as a primary method of flood protection within the MARB to the present as shown by the example development phases listed in Table 1. Ellet (1852) advanced the conviction nearly 170 years ago that a multi-pronged flood control approach should be used in the LMRB region, which would include other technology such as reservoirs and outlets in combination with levees. However, his views were immediately countered by Bayley (1852), and later by Humphreys and Abbot (1861), who advocated a “levees only” approach in their “Delta Survey.” This was the primary flood control method embraced by the seven-member Mississippi River Commission (MRC, 2021) created by Congress in 1879 (Changnon, 1998; Camillo and Pearcy, 2004). The reliance on levees dominated MARB flood control efforts until the enormous flood of 1927, which devasted the huge levee system built throughout the LMRB region between 1879 to 1926 (Changnon, 1998). The U.S. Army Corps of Engineers (USACE, 2014) report that no levees have failed within the Mississippi River & Tributary (MR&T) LMRB levee system that was implemented following the 1927 flood, including during ten major floods that occurred between 1937 and 2011. However, the MR&T system includes several floodway, spillway and backwater area structures that have been used to reduce pressure on the levees during the most severe floods (MRC, 2007). Extensive levee failures have also manifested elsewhere in the MARB during that time period (e.g., roughly 1,000 levee failures in the UMRB and MRB during the 1993 flood) (Hey and Philippi, 1995; Changnon, 1998) and dozens more levee failures in the MRB stream system alone during the record 2019 flood (Smith and Schwartz, 2019).

In general, levees can be viewed as a “two-edged sword” regarding flood protection, having provided considerable protection against many MARB floods but at the same time undermining protection due to increased river stage and velocity in channelized reaches of the MARB stream system (Hey and Philippi, 1995; Criss and Luo, 2017; Gonzalez and Kuzma, 2020). Munoz et al. (2018) further found that the magnitude of “100-year floods” has increased by 20% over the past 500 years, and that channelization of the LMRB was responsible for 75% of this increase. Pinter et al. (2008) attributed the largest component of these increases in flood magnitudes to wing dikes and other navigational structures, with progressive levee development contributing secondary impacts. “Base
Table 1. Examples of historical levee development within the MARB

<table>
<thead>
<tr>
<th>Source</th>
<th>Year</th>
<th>Total length (km)</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Rogers, 2008; 2009</td>
<td>1718-1727</td>
<td>1.65</td>
<td>Constructed to protect New Orleans, LA; 5.5 m wide at the crown, included elevated roadway (1.2 m); slope = 1:2</td>
</tr>
<tr>
<td>Morris, 2017</td>
<td>1732</td>
<td>80.5</td>
<td>Extension of levees designed to protect the New Orleans area</td>
</tr>
<tr>
<td>Changnon, 1998</td>
<td>1812</td>
<td>161</td>
<td>Built to protect Port of New Orleans</td>
</tr>
<tr>
<td>Changnon, 1998</td>
<td>1855</td>
<td>1,609</td>
<td>Levees were built and paid for by state or local sources (private and government); levees extended south from Cairo, IL in flood-prone areas of LMRB alluvial valley</td>
</tr>
<tr>
<td>Alexander et al., 2012</td>
<td>1920</td>
<td>2,700</td>
<td>Based on U.S. War Dept. report; a total volume of 276,694,000 m³ of material was used to construct the levees between Rock Island, IL and Head of Passes, LA</td>
</tr>
<tr>
<td>Alexander et al., 2012</td>
<td>2011</td>
<td>5,630</td>
<td>Estimated length of levees in just the LMRB; based on Mississippi River Commission data</td>
</tr>
<tr>
<td>USACE, 2014</td>
<td>2014</td>
<td>6,094</td>
<td>Mississippi River &amp; Tributary (MR&amp;T) LMRB levee system; 3,566 km constructed along Mississippi main steam</td>
</tr>
</tbody>
</table>

*Mississippi River Commission described at MRC (2021).

Floods” are now typically underestimated by 0.4 to 2 m at numerous locations in the central U.S. (Criss and Shock, 2001; Criss and Luo, 2017). Pinter et al. (2008) further confirm this, noting that approximately 2 m of the flood crest in the region affected by the 2008 UMRB flood was due to levees, channelization and other stream engineered manipulations. El Niño-Southern Oscillation (ENSO) and other climatic shifts and phenomena have also contributed to increased flooding levels in the MARB stream system (Munoz and Dee, 2017; Munoz et al., 2018; Zhang and Villarini, 2017; Criss and Luo, 2017; Bird et al., 2019; Neri et al., 2019; Merz et al., 2021).

The manipulation of the MARB stream system with levees, reservoirs, channelization and other interventions has also resulted in considerable ecological ramifications. Some of the most serious consequences pertain to sediment transport—as stated by Morris (2017): “Much of the river’s energy is spent moving dirt.” The Mississippi River meandered considerably over the past 7,000 years in its southern reaches, resulting in the accumulation of millions of tons of sediment annually across multiple
deltaic lobes that supported the development of the overall coastal delta region and lower quarter of Louisiana (Frazier, 1967; Morris, 2017; Restore, 2021a). However, the implementation of levees in the LRMB, coupled with upstream reservoirs and other interventions, off-shore industry (shipping channels, oil and gas exploration, etc.) and other factors, has resulted in a redistribution of sediment across the MARB and along coastal Louisiana (Morris, 2017; Restore, 2021c). This has led to an extraordinary high rate of annual land loss in southern Louisiana (Wells and Coleman, 1987; Bernier et al., 2006) that resulted in a cumulative land loss of roughly 4,000 km² at the start of the 21st century (Bernier et al., 2006) and is now estimated to exceed 5,000 km² (Restore, 2021b). In turn, this has resulted in an accompanying massive loss of coastal wetlands and wildlife habitat (Restore, 2021b). Extensive wetland and habitat loss have also occurred across much of the other MARB subregions, in both riparian areas and upland landscapes, due to engineered stream infrastructure and/or the installation of subsurface drainage tiles and drainage ditches (Hey and Philippi, 1995; Mitsch et al., 2001; Mitsch and Day, 2006; Miller et al., 2009; Morris, 2017).

MARB stream water quality degradation began shortly after European settlement in various subregions (e.g., mercury and lead contamination of UMRB stream segments that started in the mid-1800s) (Wiener and Sandheinrich, 2010). Contamination of UMRB and/or LMRB stream networks by heavy metals, bacteria, sulphates, Polychlorinated biphenyls (PCBs), DDT and other point-source pollutants increased greatly during the 20th century (Sparks, 2010; Wiener and Sandheinrich, 2010; Turner, 2021). However, in-stream levels of these pollutants have declined markedly across the MARB during the past five decades primarily due to improved control of sewage treatment plant discharge and other point sources in response to the passage of the Clean Water Act in 1972 (NRC, 2008; Wiener and Sandheinrich, 2010; Turner, 2021).

However, limited progress has occurred in reducing nonpoint source pollutant export from cropland landscapes and other nonpoint sources, especially nitrogen (N), to the MARB stream system during the same time period (NRC, 2008; Sprague et al., 2011). Nitrate levels in nine Iowa agricultural watersheds, located in five landform regions, increased 18% on average during the second 15 years of the 30-year
period of 1987 to 2016 (Jones et al., 2018b). Sprague et al. (2011) further reported that little change occurred in UMRB nitrate loads between 1980 and 2008 based on evaluations of long-term monitoring data at eight sites on the main stem of the Mississippi River. Kreiling and Houser (2016) found that monitoring data collected in six UMRB tributaries (located in Minnesota, Wisconsin, Iowa and Missouri) during 1991 to 2014 revealed that concentrations and/or fluxes of total suspended solids (TSS) and phosphorus (P) generally declined but nitrate did not. In contrast, monitoring data collected across 46 Iowa watersheds during 2000 to 2017 revealed a weighted average total P (TP) yield of 1.70 kg/ha, which was estimated to result in roughly 15% of the annual TP load to the Gulf of Mexico during that time period (Schilling et al., 2020). Jones et al. (2018a) report that extremely high loads of nitrate were also exported from Iowa over the period of 1999 to 2016, which were equivalent to 29%, 45% and 55% of the total nitrate loads discharged to the entire MARB, UMRB and MRB regions during the study period. Dissolved inorganic and organic N exports from the MARB were found to increase by 140% and 53%, respectively, based on a simulation analysis performed for 1912 to 2014 (Tian et al., 2020). The discharge of nutrients from the outlet of the Mississippi River has been identified as a key underlying cause of the seasonal oxygen-depleted hypoxic zone that forms annually in the northern Gulf of Mexico (Donner and Scavia, 2007; David et al., 2010; Rabotyagov et al., 2014; Jones et al., 2018a; Rabalais and Turner, 2019; Tian et al., 2020).

David et al. (2010) found that the highest nitrate exports occurred from intensively row cropped and tile drained landscapes in Minnesota, Iowa, Illinois, Indiana and Ohio based on statistical modeling they performed for the MARB. Schilling et al. (2020) noted that 60% of TP is exported via dissolved P in tile drains from glaciated landscapes to streams in north central Iowa, and that the majority of TP transported to streams in western and southern Iowa occurs on eroded sediment from cropland or stream banks. Additional research conducted over 2000 to 2017 indicated that 31% of the riverine TP exported from Iowa is due to streambank erosion (Schilling et al., 2022). N applied in livestock manure to cropland landscapes was implicated as a key source of nitrate for some Iowa stream systems (Jones et al., 2019a; 2019b). Increased anthropogenic N inputs, via synthetic N fertilizer and atmospheric N deposition, were
primary drivers of the substantial increase in MARB total N export since the 1960s in the assessment reported by Tian et al. (2020). Booth and Campbell (2007) similarly determined that synthetic fertilizer and atmospheric N deposition were the primary sources of nitrate loading in the MARB, followed by livestock and municipal wastes. Donner and Scavia (2007) further point to climate as an important factor driving Corn Belt nitrate export, which was underscored by the need to reduce nitrate losses by 50% to 60% in “wet years” to achieve a Gulf hypoxic zone target size < 5,000 km².

The U.S. Department of Agriculture (USDA) administered roughly $110 billion between 1935 and 2010 through 32 different programs for conservation practices support, which was equal to almost $294 billion on the basis of 2009 dollars (Villarini et al., 2016). More recent federal support for conservation programs varied between $4.5 billion and $7.5 billion annually during 2000 to 2010 (USDA, 2011). Additional conservation practice cost share support has been provided by different states (e.g., Iowa), which invested almost $1.2 billion in support of terraces, grassed waterways, no till, Conservation Reserve Program (CRP) land and similar practices that were installed by 2007 (Feng et al., 2007). Much of the federal and state support has focused on cultural and structural soil conservation practices designed to mitigate sediment and sediment-bound nutrient losses, although support has expanded to include nutrient control, land set aside, wildlife habitat and other practices in more recent years (Sharpley et al., 2006; Feng et al., 2007; USDA, 2011; Villarini et al., 2016; Staver, 2020). Iowa and Minnesota are among states that have also provided funding for wetlands and other practices during the past two decades, that provide increased nutrient control, wildlife habitat and other associated benefits, through programs such as the Conservation Reserve Enhancement Program (CREP), Cleanwater Iowa, and Reinvest in Minnesota (RIM) (Mulla et al., 2008; IDALS, 2021; 2022; MBWSR, 2019).

Villarini et al. (2016) found that USDA conservation practice expenditures directly influenced declines in TSS and sediment load (SL) during 1937 to 2009 in the Raccoon River in west central Iowa. White et al. (2014) more broadly determined that agricultural conservation practices reduced nutrient load export by 20% from the MARB to the Gulf of Mexico on the basis of a simulation study. Implementation of conservation practices were found to reduce sediment and/or nutrient losses based on monitoring data
collected in several watershed studies, as reported in a review conducted by Mulla et al. (2008). However, the majority of the reviewed studies were conducted for relatively small watersheds and thus the results are not directly transferable to larger regions. A total of 94 studies were reviewed by Lintern et al. (2020), who found that roughly 60% of the studies reported reductions in watershed-level nutrient losses in response to adoption of conservation practices. But they further note that the majority of the “successful studies” were based on modeling assessments, that many factors such as nutrient lag and legacies are not well understood, and that the lack of historical success in controlling diffuse nutrient losses constitutes a “wicked problem.”

The extensive and pervasive MARB water quality problems underscore that the enormous investments in conservation programs and practices to date have generally not translated to successful control of diffuse nutrient pollutants from cropland and pasture landscapes across the region. Briske et al. (2017) further found that it was not possible to evaluate the impacts of USDA cost-shared rangeland conservation practices due to the fact that virtually no documentation has ever occurred regarding the efficacy of such practices. In addition, Suttles et al. (2021) cite multiple studies that collectively state: (1) projected global annual flood costs could increase from $157 billion to $535 billion in riverine urban areas; (2) populations impacted by floods globally could rise from 65 million in 2010 to 132 million in 2030; (3) U.S. agricultural nitrate pollution results in annual damages > $150 billion; and, (4) adoption of natural infrastructure (NI) could reduce U.S. flood and other climate related costs by nearly $250 billion, versus roughly double the cost for implementing equivalent grey infrastructure.

These composite challenges point to the need for a paradigm shift regarding the types and targeting of conservation practices in MARB agricultural production regions. Expanded installation of NI practices on cropland, riparian zones and other strategic landscapes could serve as a cornerstone of such a shift, due to the potential flood reduction and pollutant mitigation benefits these practices offer (Ozment et al., 2015; Hovis et al., 2021; Suttles et al., 2021). Several NI practices have been and/or currently are supported via various federal or state cost-share programs, such as various types of wetlands (IDALS, 2022; Roley et al., 2016; Thies, 2018; MBWSR, 2019; USDA, 2014, 2019; 2022a; 2022f), saturated
buffers (Thies, 2018; Jaynes et al., 2018; Bradbury et al., 2019), farm ponds (USDA, 2005; 2022a; Roley et al., 2016), two-stage ditches (Witter, 2013; Roley et al., 2016) and flood control interventions (USDA, 2014, 2019; 2022a). Extensive adoption across the MARB of the broader suite of existing NI practices (Ozment et al., 2015; Cunniff, 2019; Hovis et al., 2021; Suttles et al., 2021), in response to increased cost share support and other incentives, would provide the opportunity of magnifying the overall flood and diffuse pollution mitigation impacts these technologies offer.

Several previous modeling assessments have been performed for the MARB including flood analyses (Tavakoly et al., 2017), streamflow forecasting (Sivapragasam et al., 2014), effects of dams on streamflow (Masaki et al., 2017), nitrate export from cropland and/or other sources (Booth and Campbell, 2007; David et al., 2010; Tian et al., 2020), and the impacts of biofuel cropping systems (Ferin et al., 2021) or conservation practices (White et al., 2014) on water quality. However, evaluation of the potential impacts of widespread NI practice installation across the MARB is currently lacking in existing literature. Identification of robust modeling systems and other tools is a critical need to provide a basis for quantitative evaluation of expected flood and pollutant export reductions that would occur from widespread NI practice adoption in the MARB.

An extensive array of models has been developed that can be applied for various types of flood and/or ecohydrological (water quality) assessments. Available models represent a spectrum of simulation domains across applicable scales, functionality and portability. Many studies refer to “hydrological models” to describe models that are used for a wide range of hydrologic investigations including reviews by Singh and Woolhiser (2002) and Paul et al. (2021). However, this categorization includes models that are more accurately classified as ecohydrological models due to capabilities of simulating overland and/or in-stream pollutant transport, such as the USDA Soil and Water Assessment Tool (SWAT) (Arnold et al. 1998; 2012; 2021; Bieger et al., 2017; SWAT, 2022). There is also other overlap between various modeling categories in the literature, such as SWAT classified as a flood model (Kauffeldt et al., 2016).

Kauffeldt et al. (2016), Mosavi et al. (2018), Nkwunonwo et al. (2020), Chomba et al. (2021), Qi et al. (2021) and Hdeib et al. (2021) provide reviews of models or modeling systems that have been
applied for flood analyses, which include hydrologic, hydraulic/hydrodynamic, artificial neural networks (ANNs) and/or other modeling approaches. The U.S. Army Corp of Engineers (USACE) Hydrologic Engineering Center- Hydrologic Modeling System (HEC-HMS) and -River Analysis System (HEC-RAS) models comprise a widely used hydrologic and hydraulic modeling system that provides considerable flexibility for flood analyses (Feldman, 1981; 2003; USACE 2021a; 2022a; 2022b). The interface of the two models supports simulation of watershed-level rainfall-runoff with HEC-HMS and resulting flood inundation generated by HEC-RAS for an extensive range of environmental conditions (e.g., Knebl et al., 2005; Abdessamed and Abderrazak, 2019; Namara et al., 2021).

The LISFLOOD model was developed to replicate hydrological processes in large European rivers including simulation of flood forecasting and prevention, and drought assessments (Bates and De Roo, 2000; De Roo et al., 2000; Van Der Knijff et al., 2010; JRC, 2020). LISFLOOD-FP is a two-dimensional hydrodynamic model which is an extension of LISFLOOD that is specifically designed to simulate flood inundation (Bates and de Roo, 2000; Bates, 2022a; 2022b). Both LISFLOOD and LISFLOOD-FP have been applied for flood forecasting and other flood analyses for most of Western Europe within the European Flood Awareness System (EFAS) as described by Thielen et al. (2009), Smith et al. (2016), EFAS (2021), Brunner and Slater (2022), and Dottori et al. (2022). The Routing Application for Parallel Computation of Discharge (RAPID), AutoRAPID and AutoRoutemodels have been applied for flood inundation and related analyses for the MARB and/or other large basins in North America and Europe (David et al., 2016; Follum et al., 2017; Tavakoly et al., 2017).

Dozens of ecohydrological models have been developed to simulate terrestrial and/or in-stream transport of specific constituents or a broader suite of pollutants. Documentation of models that can simulate transformation, translocation and/or in-stream movement of specific pollutants have been compiled in reviews for sediment (Borrelli et al., 2021; Pandey et al., 2021; Ma et al., 2021), nutrients (Schoumans et al., 2009; Bouraoui and Grizzetti, 2014; Easton et al., 2017; Wang et al., 2020), pathogens (Coffey et al., 2007; Cho et al., 2016; Collender et al., 2016), pesticides (Quilbé et al., 2006; Mottes et al., 2014) and metals (Caruso et al., 2008). Other review studies describe ecohydrological models that can be
applied for multiple types of pollutants (Ambrose et al., 2009; Borah et al., 2019; Fu et al., 2019; Yuan et al., 2020). Example ecohydrological models that can be applied for both terrestrial and in-stream pollutant transport include the Hydrological Simulation Program-Fortran (HSPF) model (Bicknell et al., 1996; 2002; Donigian and Imhoff, 2005; Duda et al., 2012; USEPA, 2014), Agro-Integrated Biosphere Simulator (Agro-IBIS) model (Donner and Kucharik, 2008; Ferin et al., 2021), MIKE SHE (Système Hydrologique Européen) model (Refsgaard et al., 2010; Ma et al., 2016; DHI, 2022c; 2022d), SPAtially Referenced Regression On Watershed attributes (SPARROW) model (Smith et al., 1997; Preston et al., 2009; 2011; McLellan et al., 2015; USGS, 2019), SWAT (Arnold et al., 1998; 2012; Gassman et al., 2007; 2014) and successor SWAT+ versions (Bieger et al., 2017; Arnold et al., 2021; White et al., 2022).

The availability of these and other models leads to the question as to which modeling system can best support the assessment of NI practice impacts on flooding and water quality in the MARB. Multiple factors need to be considered when considering a modeling approach including: (1) the ability to account for upland and riparian NI practices; (2) the range of pollutants that the model can account for, especially from diffuse sources; (3) whether the model(s) can be obtained from public domain platforms or must be purchased as proprietary software; and, (4) the ability to configure the model(s) for the entire MARB. The primary objectives of this study are: (1) briefly describe MARB land use and U.S. Geological Survey (USGS) hydrologic unit codes (HUCs); (2) provide an overview of NI practices within the context of the broader realm of existing conservation practices; and, (3) assess the relative strengths and weaknesses of existing flood and ecohydrologic models for evaluating the potential of NI practices to mitigate the magnitude of flood and water quality levels across the MARB. This research also builds on insights obtained from precursor Geographic Information System (GIS) modeling, which describes an initial evaluation of optimal placement of seven NI practices for reducing flood volumes and nitrate levels across the MARB region (Schilling et al., 2022).
2. Study Region Description

The distribution of land use across the MARB drainage area located in the U.S. is shown in
Figure 1 and Table 2. The total area covered by the different land use types listed in Table 1 is 3,246,002
km², based on USDA Cropscape – Cropland Data Layer data (USDA, 2022e; Boryan et al., 2011). This
total area excludes 27,500 km² of MRB drainage area (Figure 1) located in southeast Alberta and southern
Saskatchewan (Ayers and James-Abra, 2006). The dominant MARB land use categories are (Table 2):
forest (1,051,986 km²); grassland/pasture/hay (951,426 km²); total cropland (796,449 km²); urban
(166,038 km²); wetland (147,660 km²); fallow/barren (67,619 km²); and, water (64,824 km²). Forest is
dominated by deciduous forest (496,632 km²), shrubland (311,647 km²) and evergreen forest (166,237
km²), and is distributed across large western, southern, eastern and north central MARB subregions.
Grassland and pasture covers much of the Great Plains Region in the MRB and AWRRS, with smaller
subareas of hay and pasture interspersed in other parts of the MARB. Wetlands are concentrated in the
northern UMRB and southern LMRB. Water bodies are also located in those two subregions, and in main
channels or reservoirs that form the Mississippi River or major tributaries.

Figure 1 highlights the primary production areas for corn, soybean and wheat/small grains, which
are the three major crops grown in the MARB on respective total land areas of 309,574 km², 275,808 km²
and 144,307 km² (Table 1). Corn and soybeans are both intensively grown in the expansive Corn Belt
region that stretches from the eastern MRB, across the UMRB to the western part of the ORB. Soybeans
are also produced in the Delta region that spans parts of the LRMB and AWRRS. Rice and cotton
production is interspersed in the Delta region among the intensive soybean landscapes (Figure 1 and
Table 1). Wheat/small grain production is dominated by spring wheat cropland in the northern Great
Plains region (northern MRB), and winter wheat cropping systems in the southern Great Plains region
(AWRRS) and to a lesser extent in the northern Great Plains region. Smaller patches of sorghum are
intermingled among winter wheat and corn-soybean production areas across in the Great Plains Region.

Delineation of stream networks in the MARB can be performed on the basis of standard HUCS,
which represent a hierarchy of basins or watersheds developed by collaborating federal agencies (USGS,
The four largest HUCs are categorized as regions, subregions, basins (accounting units) or subbasins (cataloging units) as described by USGS (2013, 2022b). Regions are identified with 2-digit codes, which include the six major basins that comprise the MARB (Figure 1). Subregion, basin and subbasin HUCs represent successively smaller drainage areas and are identified by respective 4-, 6- and 8-digit codes; the total number of these HUCs are listed in Table 3 for the MARB and each major basin. Two additional smaller levels have been formally incorporated in the HUC classification scheme: 10-digit watersheds and 12-digit subwatersheds (USGS, 2013). The total number of 12-digit HUCs is shown in Table 3 for the MARB and the six major basins. Additional refined 14- and 16-digit HUCs can also be delineated within the overall hierarchal scheme based on guidance presented in USGS (2013). The National Hydrography Dataset High Resolution (NHDPlus HR) dataa (USGS, 2022a) provides additional options for refined hydrologic assessments.

The HUC classification scheme is widely used for developing simulation applications of hydrology and pollutant transport as described in numerous studies (e.g., Kannan et al., 2008; 2019, Arnold et al., 1999; 2021; Panagopoulos et al., 2015; White et al., 2017; Sehgal and Sridhar, 2018; Chen et al., 2021). The different HUC levels are referred to simply as 2-, 4-, 6-, 8-, 10- and 12-digit watersheds for the remainder of this study. An overlay of 2-, 4- and 8-digit watersheds on major MARB stream segments is shown in Figure 2. A similar overlay provides a comparison of 12-digit watersheds versus 4-digit watersheds within the 2-digit UMRB (Figure 3). The maps and Table 3 reveal the relative density of the different hydrologic unit watersheds within each of the six major basins (2-digit watersheds) in the MARB. The relative ratios of 12-digit to 8-digit watersheds, which are two of the most commonly used HUCs in modeling and other ecohydrological analyses, range from a factor of 32 (LMRB) to 45 (Missouri River Basin). Overall, the total number of 12-digit watersheds is roughly 41 times greater than 8-digit watersheds for the entire MARB.
Figure 1. Land use map for the U.S. portion of the Mississippi-Atchafalaya River Basin (MARB) drainage area (Source: USDA, 2022e; Boryan, 2011).
Table 2. Distribution (total area) of major land use categories across the U.S. portions of the Mississippi-Atchafalaya River Basin (MARB) and six water resource sub-regions

<table>
<thead>
<tr>
<th>Region</th>
<th>Cornb</th>
<th>Soybeansb</th>
<th>Cotton</th>
<th>Rice</th>
<th>Sorghumb</th>
<th>Wheat &amp; other small grainsc</th>
<th>Grassland, pasture, &amp; hayd</th>
<th>Foreste</th>
<th>Urbanf</th>
<th>Fallow &amp; barreng</th>
<th>Waterh</th>
<th>Wetlandsb</th>
<th>Other cropsi</th>
</tr>
</thead>
<tbody>
<tr>
<td>MARB</td>
<td>309,574</td>
<td>275,808</td>
<td>14,975</td>
<td>9,332</td>
<td>21,503</td>
<td>144,307</td>
<td>951,426</td>
<td>1,051,986</td>
<td>166,038</td>
<td>67,619</td>
<td>64,824</td>
<td>147,660</td>
<td>20,950</td>
</tr>
<tr>
<td>Arkansas-White-Red</td>
<td>15,393</td>
<td>11,699</td>
<td>7,853</td>
<td>1,085</td>
<td>13,834</td>
<td>51,266</td>
<td>236,230</td>
<td>236,062</td>
<td>27,513</td>
<td>18,841</td>
<td>8,656</td>
<td>11,340</td>
<td>942</td>
</tr>
<tr>
<td>LMRB</td>
<td>9,855</td>
<td>33,565</td>
<td>6,490</td>
<td>8,237</td>
<td>73</td>
<td>1529</td>
<td>51,266</td>
<td>236,062</td>
<td>15,670</td>
<td>7,770</td>
<td>15,833</td>
<td>60,811</td>
<td>2,814</td>
</tr>
<tr>
<td>Missouri</td>
<td>114,329</td>
<td>88,286</td>
<td>5</td>
<td>1</td>
<td>7,476</td>
<td>83,050</td>
<td>560,31</td>
<td>337,542</td>
<td>36,854</td>
<td>37,076</td>
<td>18,204</td>
<td>25,221</td>
<td>14,651</td>
</tr>
<tr>
<td>Ohio</td>
<td>42,121</td>
<td>46,862</td>
<td>2</td>
<td>1</td>
<td>34</td>
<td>3,877</td>
<td>57,145</td>
<td>223,335</td>
<td>37,912</td>
<td>2,020</td>
<td>5,359</td>
<td>3,068</td>
<td>224</td>
</tr>
<tr>
<td>Tennessee</td>
<td>2,295</td>
<td>2,523</td>
<td>624</td>
<td>0</td>
<td>24</td>
<td>529</td>
<td>17,910</td>
<td>67,698</td>
<td>9,828</td>
<td>317</td>
<td>2,767</td>
<td>1,383</td>
<td>48</td>
</tr>
<tr>
<td>UMRB</td>
<td>12,558</td>
<td>92,874</td>
<td>1</td>
<td>9</td>
<td>62</td>
<td>4,056</td>
<td>59,756</td>
<td>107,709</td>
<td>38,262</td>
<td>1,595</td>
<td>14,005</td>
<td>45,837</td>
<td>2,272</td>
</tr>
</tbody>
</table>

aSources: USDA (2022e) and Boryan et al. (2011).
bCorn includes sweet corn, popcorn/orncorn and corn double-cropped with small grain crops; soybeans includes soybeans double-cropped with corn and small grain crops.
ccRepresents 17 different land use categories including winter wheat (78,012 km²), spring wheat (33,608 km²), winter wheat/soybean double-crop (8,630 km²), durum wheat (5,145 km²), barley (4,985 km²), oats (4,294 km²), rye (1,942 km²), triticale (1,743 km²), flaxseed (946 km²) and safflower (339 km²).
ndGrassland/pasture covers 833,849 km²; other land use includes non-alfalfa hay (70,035 km²), alfalfa (46,962 km²) and small areas of sod production and switchgrass.
eforest includes deciduous forest (496,632 km²), shrubland (311,647 km²), evergreen forest (166,237 km²), mixed forest (77,434 km²) and other minor land use with trees.
fIncludes developed/open space (97,488 km²), developed/low intensity (45,738 km²), developed/moderate intensity (16,535 km²) and developed/high intensity (6,278 km²).
gDominated by fallow (58,595 km²) with a smaller amount of barren land area (9,024 km²).
hWater includes open water (63,115 km²) and aquaculture (1,662 km²); wetlands include woody wetlands (96,578 km²) and herbaceous wetlands (51,082 km²).
iRepresents 51 different land use categories including sunflower (4,852 km²), peas (3,782 km²), sugarcane (2,312 km²), dry beans (2,304 km²), lentils (1,691 km²), sugar beets (1,072 km²), potatoes (555 km²), herbs (469 km²), chick peas (443 km²), pecans (316 km²), peanuts (252 km²), clover/wild flowers (214 km²), sweet potatoes (151 km²), tobacco (99 km²), pumpkins (81 km²) and mint (46 km²).
Table 3. Total number of different Hydrologic Unit Code (HUC) watershed levels\textsuperscript{a} for the Mississippi-Atchafalaya River Basin (MARB) and six MARB Major Water Resource Regions

<table>
<thead>
<tr>
<th>Region\textsuperscript{a}</th>
<th>2-digit</th>
<th>4-digit</th>
<th>6-digit</th>
<th>8-digit</th>
<th>12-digit</th>
</tr>
</thead>
<tbody>
<tr>
<td>MARB</td>
<td>6</td>
<td>84</td>
<td>134</td>
<td>845</td>
<td>34,944</td>
</tr>
<tr>
<td>Arkansas-White-Red</td>
<td>1</td>
<td>14</td>
<td>25</td>
<td>173</td>
<td>6,494</td>
</tr>
<tr>
<td>LMRB</td>
<td>1</td>
<td>9</td>
<td>23</td>
<td>82</td>
<td>2,640</td>
</tr>
<tr>
<td>Missouri</td>
<td>1</td>
<td>29</td>
<td>43</td>
<td>306</td>
<td>13,725</td>
</tr>
<tr>
<td>Ohio</td>
<td>1</td>
<td>14</td>
<td>21</td>
<td>120</td>
<td>5,278</td>
</tr>
<tr>
<td>Tennessee</td>
<td>1</td>
<td>4</td>
<td>5</td>
<td>32</td>
<td>1,075</td>
</tr>
<tr>
<td>UMRB</td>
<td>1</td>
<td>14</td>
<td>17</td>
<td>132</td>
<td>5,732</td>
</tr>
</tbody>
</table>

\textsuperscript{a}The 2-digit watershed codes for the six major MARB basins are (USGS, 2022b; Wikipedia, 2021): Arkansas-White-Red (11), LMRB (08), Missouri (10), Ohio (05), Tennessee (06) and UMRB (07).

3. MARB Conservation Practices

Adoption of conservation practices has been widespread across the Corn Belt region and other MARB subregions as evidenced by the previously described USDA and state agency investments. However, quantifying the extent and locations of existing conservation practices is difficult due to a lack of detailed records that document exactly where supported conservation practices were installed or adopted, especially for rangeland landscapes (Briske et al., 2017). In addition, some conservation practices have been implemented by land owners and/or producers that did not receive any government cost share support. White et al. (2014) based their analysis of conservation practices on survey data collected internally by the USDA, using methods similar to those described by Atwood et al. (2009). However, those survey data were collected primarily to support the Conservation Effects Assessment Project (CEAP; Duriancik, 2008) and have not been publicly released by the USDA.
Figure 2. Delineation of 2-digit (water resource regions), 4-digit and 8-digit watersheds, with stream network and state boundaries, within the Mississippi-Atchafalaya River Basin (MARB).
Figure 3. Delineation of 4-digit and 12-digit watersheds, with stream network and state boundaries, within the Upper Mississippi River Basin (URMB).
Table 4 lists the estimated areal extent (km²) by state that has been documented for six conservation practices in the 12-state north central region: land set aside, cover crops, no-till, reduced-till (USDA, 2017a), and the combined installation of terraces and grassed waterways (Martins et al., 2021). Portions of these states exist outside of the MARB drainage area, including virtually all of Michigan and parts of North Dakota, Minnesota Wisconsin, Indiana and Ohio. However, the primary agricultural production areas in most of the north central region are located within the MRB, UMRB, ORB or AWRRS drainage areas (Figure 1). Thus, it is inferred that the majority of the conservation practices represented by the Table 4 data are installed or occurring on cropland landscapes within the boundaries of the MARB.

Combined no-till and reduced-till adoption rates range between 60% to 73% of existing cropland in the majority of the north central states (Table 4). The exceptions are Michigan, Minnesota, Missouri and Wisconsin—total no-till and reduced-till adoption rates range from 42% to 52% of available cropland in those four states. Installation of terraces or grassed waterways on agricultural land parcels varies considerably between the 12 states, ranging from 1,700 km² in Iowa to 200 km² in North Dakota (and 125 km² in Michigan) based on the digital mapping reported by Martins et al. (2021). Incorporation of winter cover crops in row crop systems is a relatively new phenomena in the 12-state region, which is reflected by the lower overall adoption levels in the 2017 USDA Agricultural Census (Table 4). However, usage of winter cover crops increased 50% nationally between 2012 and 2017 (USDA, 2017b), with acute increases occurring in several of the north central states that exceeded 100% (USDA, 2017a), revealing that acceptance of the practice is growing among landowners and producers located in the MARB.

USDA-sponsored land set aside programs (Table 4) were initiated in 1985 (USDA, 2021) and presently include the CRP (CRP; USDA, 2021), Wetland Reserve Program (WRP; USDA, 2014), Farmable Wetlands Program (FWP, USDA, 2022f) and Conservation Reserve Enhancement Program (CREP; USDA, 2022d). The highest total land conversions reported in the 2017 Agricultural Census as a function of CRP and related USDA programs occurred in the western Corn Belt region (Figure 1 and Table 4).
Table 4. Estimated areas of cropland managed with cover crops, no-till, reduced-till, terraces and grassed waterways for 12 U.S. north central states.

<table>
<thead>
<tr>
<th>State</th>
<th>Total cropland areaa</th>
<th>Land set asidea,b</th>
<th>Winter cover cropsa</th>
<th>No-tilla</th>
<th>Reduced-tilla</th>
<th>Terraces and grassed waterwaysc</th>
</tr>
</thead>
<tbody>
<tr>
<td>Illinois</td>
<td>97,137</td>
<td>3,449</td>
<td>2,866</td>
<td>26,191</td>
<td>38,260</td>
<td>980</td>
</tr>
<tr>
<td>Indiana</td>
<td>52,244</td>
<td>906</td>
<td>3,788</td>
<td>19,841</td>
<td>16,443</td>
<td>365</td>
</tr>
<tr>
<td>Iowa</td>
<td>107,428</td>
<td>6,802</td>
<td>3,938</td>
<td>33,169</td>
<td>41,005</td>
<td>1,697</td>
</tr>
<tr>
<td>Kansas</td>
<td>117,867</td>
<td>7,970</td>
<td>2,252</td>
<td>45,316</td>
<td>31,351</td>
<td>533</td>
</tr>
<tr>
<td>Michigan</td>
<td>32,069</td>
<td>522</td>
<td>2,724</td>
<td>6,339</td>
<td>9,296</td>
<td>106</td>
</tr>
<tr>
<td>Minnesota</td>
<td>88,167</td>
<td>4,344</td>
<td>2,344</td>
<td>4,416</td>
<td>33,245</td>
<td>490</td>
</tr>
<tr>
<td>Missouri</td>
<td>63,129</td>
<td>3,704</td>
<td>3,408</td>
<td>18,797</td>
<td>14,141</td>
<td>553</td>
</tr>
<tr>
<td>Nebraska</td>
<td>90,013</td>
<td>3,144</td>
<td>3,027</td>
<td>41,509</td>
<td>24,082</td>
<td>757</td>
</tr>
<tr>
<td>North Dakota</td>
<td>113,116</td>
<td>6,217</td>
<td>1,636</td>
<td>31,478</td>
<td>36,633</td>
<td>152</td>
</tr>
<tr>
<td>Ohio</td>
<td>44,356</td>
<td>1,015</td>
<td>2,905</td>
<td>17,275</td>
<td>12,564</td>
<td>330</td>
</tr>
<tr>
<td>South Dakota</td>
<td>80,182</td>
<td>3,993</td>
<td>1,140</td>
<td>30,983</td>
<td>17,403</td>
<td>464</td>
</tr>
<tr>
<td>Wisconsin</td>
<td>40,183</td>
<td>937</td>
<td>2,474</td>
<td>9,014</td>
<td>11,478</td>
<td>217</td>
</tr>
</tbody>
</table>

*aSource: (USDA, 2017a; 2017b).*

*bIncludes Land enrolled in USDA Conservation Reserve, Wetlands Reserve, Farmable Wetlands, or Conservation Reserve Enhancement Programs.*

*cSource: Figure 4 reported in Martins et al. (2021) and Martins, V. Personal communication. Department of Agricultural and Biological Engineering, Mississippi State University, Mississippi State, MS.*

The overall environmental impacts of these and other widely used conservation practices (e.g., contouring, strip cropping, filter strips, contour buffers, sediment basins) is of primary interest regarding the current condition of the MARB stream network. As noted previously, White et al. (2014) found that existing conservation practices resulted in a 20% reduction of nutrients delivered to the Gulf of Mexico from the MARB, as compared to a scenario that was void of all conservation practices across the region.
Other studies have reported reductions in nonpoint source pollutants in response to conservation practice implementation for specific watersheds (Mulla et al., 2008; Villarini et al., 2016; Lintern et al., 2020). These findings are reinforced by Yuan et al. (2022), who report summaries of six literature review studies that analyzed the sediment and nutrient reduction impacts of the following conservation and management practices: denitrifying bioreactors (Christianson et al., 2021a), crop rotations (Koropeckyj-Cox et al., 2021), cover crops (Christianson et al., 2021b), in-field nutrient management (Liu et al., 2021), filter strips (Douglas-Mankin et al., 2021) and constructed wetlands (Messer et al., 2021). Some key results summarized by Yuan et al. (2022) include: (1) cover crops and filter strips result in mean sediment load reductions exceeding 70%; (2) cover crops, constructed wetlands, denitrifying bioreactors, and selected crop rotation and nutrient management strategies can reduce mean nitrate loads in subsurface tile drains roughly 30% to 40%; (3) filter strips reduced mean total N, nitrate, TP and dissolved P loads in surface runoff 57%, 34%, 63% and 47%, respectively; and, (4) cover crops reduced mean dissolved P loads in subsurface tile drains by 29%.

The conservation practice trends and general effectiveness reported in studies cited here (and many other studies; e.g., IDALS, 2017) can yield an impression that significant progress has occurred towards mitigating nutrient pollution in the MARB drainage area. However, the previously cited monitoring studies underscore that nutrient discharge remains largely unabated, especially in the Corn Belt region. This reality is particularly striking for Iowa, which is characterized by some of the highest levels of conservation practice adoption in the Corn Belt region (e.g., Table 4) but continues to export extremely high levels of nitrate and TP (Jones et al., 2018a; 2018b; Schilling et al., 2020; 2022).

Current trends of increasing cover crop adoption in the MARB (USDA, 2017) should result in future water quality benefits, although extensive implementation across the majority of Corn Belt cropland may be needed to obtain quantifiable impacts (Kling et al., 2014). Incremental water quality gains could also be obtained with improved management of manure and fertilizer nutrients applied to selected cropland parcels, such as crop fields in high livestock density subregions in Iowa as postulated by Jones et al. (2019 a;b). Similarly, it would be expected that the vast implementation filter/buffer strips
that has occurred across the U.S. (almost 2 million km by 2002; see Loftus and Kraft, 2003) has resulted in reduced sediment and sediment-bound nutrients to stream systems throughout the MARB. However, filter and buffer strips are largely ineffective in intercepting nitrate and dissolved P transported in subsurface tile drains (Jaynes and Isenhart, 2014; 2019). Practices such as constructed wetlands and denitrifying bioreactors do provide direct reductions of tile-drain-transported nitrate (and also dissolved P in wetlands). However, the extent of installation is relatively limited in the MARB region to date (e.g., 99 completed and 12 initiated CREP wetlands in 37 north central Iowa counties in 2020. See “status map of current sites” at IDALS, 2022).

Overall, the collective conservation practices that have been implemented in the Corn Belt region to date have proved largely ineffective at mitigating ongoing nutrient exports to the MARB stream network and ultimately discharged to the Gulf of Mexico. This is primarily due to the inability of most of the established practices to capture nitrate and dissolved P, although elevated amounts of TP are also exported from some MARB subregions (Schilling et al., 2020; 2022). Current trends in cover crop adoption and other possible future developments could result in some reduced diffuse nutrient losses as noted above. However, strategic shifts in conservation management are clearly needed for realistic pursuit of state and/or MARB nutrient reduction goals (USEPA, 2008; IDALS, 2017). Beyond this, paradigm shifts are also needed to distribute conservation practices that can provide effective defense against flooding in both upland and riparian landscapes. Current land use shifts due to cover crop and land set-aside initiatives do provide some reduction in surface runoff. But the devastating outcomes of the 1993, 2019 and other recent Midwest floods confirm that more reliable flood control interventions are critically needed throughout the MARB.

5. Natural infrastructure (NI) practices

Suttles et al. (2021) define NI practices as: “Durable structural and/or native perennial vegetative measures embedded in a landscape or riverscape that are inspired and supported by nature, restore ecological processes, and deliver multiple environmental benefits to downstream communities.” The
primary objective of introducing NI technology in the MARB is to deliver ecosystem benefits, in the form of improved water quality and flood control, in rural upland landscapes and riparian floodplains/channels. Advancement of the NI practice paradigm supplants reliance on “control infrastructure” such as levees and reservoirs, and dredging, straightening and other stream hydromodifications. The NI vision also incorporates a holistic watershed approach, resulting in strategically targeted practices in sensitive locations throughout an interconnected stream network. The resulting impacts on a hypothetical watershed flow regime is depicted by the impacts on peak and overbank flows shown in Figure 4.

Figure 4. Net effect of introducing multiple NI practices in a watershed on peak and overbank flows (Source: Cunniff, 2019; See Acknowledgements).

The suite of NI practices proposed by Suttles et al. (2021) is listed in Table 5 with one addition, beaver dams. Considering flood reduction potential, these practices were selected due to attributes that provide: (1) reduced runoff generation; (2) greater water storage; and/or, (3) attenuated flow from upland landscapes to local watershed stream systems. The majority of the NI practices were also chosen due to proven ability to reduce N losses within upland or riparian landscapes (Suttles et al., 2021). The ensemble of NI practices proposed here (Table 5) differ considerably from NI practice sets described by Ozment et
<table>
<thead>
<tr>
<th>Landscape position and/or function</th>
<th>NI practice</th>
<th>Description of NI practice</th>
</tr>
</thead>
<tbody>
<tr>
<td>Upland crop fields, pastures or other agricultural landscapes</td>
<td>Depressional wetlands</td>
<td>Also sometimes described as “isolated” wetlands. Most are shallow with depths &lt;1 meter and a median size of 0.16 ha, but they can be as large as several hundred hectares (Figure 2A).</td>
</tr>
<tr>
<td>Upland crop fields, pastures or other agricultural landscapes</td>
<td>Conversion of cropland to native forest or grasses</td>
<td>Land use change that converts cropland to forest or grass vegetation (Figures 2B &amp; 2C).</td>
</tr>
<tr>
<td>Upland crop fields, pastures or other agricultural landscapes</td>
<td>Runoff attenuation features (RAFs)</td>
<td>Used primarily in the UK and Europe, RAFs intercept overland flow and temporarily store water behind small “leaky” dams over a period of 4–24 hours (Figure 2D).</td>
</tr>
<tr>
<td>Upland crop fields, pastures or other agricultural landscapes</td>
<td>Farm ponds</td>
<td>Impoundments that are intended for long term storage of water for livestock and/or fishing. They can be constructed with an embankment/berm or dugout out of the earth to fill with water (Figure 2E).</td>
</tr>
<tr>
<td>Upland crop fields, pastures or other agricultural landscapes</td>
<td>Water and sediment control basins (WASCOBs)</td>
<td>Intercept surface runoff along minor slopes or flow paths; discharge water via outlet; reduce sediment and sediment-bound nutrients contained in runoff (Figure 2F).</td>
</tr>
<tr>
<td>Interception of tile drains in edge-of-field or riparian areas</td>
<td>Constructed wetlands</td>
<td>Wetlands built by creating an embankment to intercept tile drain effluent or other artificial drainage; usually much larger than depressional wetlands and intercept much larger drainage area (Figure 2G).</td>
</tr>
<tr>
<td>Interception of tile drains in edge-of-field or riparian areas</td>
<td>Saturated buffers</td>
<td>Intercept tile drains in riparian buffers; constructed with perforated distribution pipe to spread drainage water laterally across buffer subsurfaces to promote denitrification (Figure 2H; right panel).</td>
</tr>
<tr>
<td>Riparian areas, small streams or drainage ditches</td>
<td>Vegetated ditches</td>
<td>Drainage ditches planted with grasses/vegetation to reduce sediment/nutrient pollution (Figure 2I).</td>
</tr>
<tr>
<td>Riparian areas, small streams or drainage ditches</td>
<td>Stream restoration</td>
<td>Restoring the geomorphic structure of the stream by raising the stream bed, installing meanders to a channelized stream, re-grading the stream channel, reconnecting oxbows, etc. (Figure 2J).</td>
</tr>
<tr>
<td>Riparian areas, small streams or drainage ditches</td>
<td>Two-stage ditches</td>
<td>Modified drainage ditches with two “benches” on both channel sides that function as floodplains. Tile drains empty onto the constructed floodplains rather than directly into the ditch (Figure 2K).</td>
</tr>
<tr>
<td>Riparian areas, small streams or drainage ditches</td>
<td>Riparian forest buffers</td>
<td>Forested areas (natural/re-established) between streams/rivers and agricultural land (Figure 2L).</td>
</tr>
<tr>
<td>Riparian areas, small streams or drainage ditches</td>
<td>Vegetated buffers</td>
<td>Vegetated areas (natural/re-established without forest) established between streams/rivers and agricultural land (Figure 2H; left panel).</td>
</tr>
<tr>
<td>Riparian areas, Floodplains and channels aligned with larger streams and rivers</td>
<td>Floodplain restoration, including oxbow restoration</td>
<td>Reconnecting main channel with floodplain to allow for periodic inundation. Can result from levee modifications (levee removal, breaching or setback) or in areas without levees. Floodplain restoration includes reconnection of river flows to previously disconnected wetlands (oxbow restoration) and/or planting of native vegetation (e.g., forested floodplain restoration) (Figures 2M and 2N).</td>
</tr>
<tr>
<td>Riparian areas, Floodplains and channels aligned with larger streams and rivers</td>
<td>Beaver dams</td>
<td>Provide multiple ecosystem services including increased water storage, enhanced floodplain connectivity, increased temporary flood storage and more riparian vegetation species (Figure 2O).</td>
</tr>
</tbody>
</table>

*Information adapted from Table 3 in Suttles et al. (2021).*
al. (2015), Cunniff (2019) and Hovis et al. (2021). One primary difference is that Ozment et al. (2015) and Cunniff (2019) describe development of NI solutions for urban and coastal areas, in addition to agricultural production regions. Hovis et al. (2021) further propose no-till, hardpan breakup and cover crops as possible agricultural NI practices, due to reductions in surface runoff that occur in response to these interventions (Cunniff (2019) also suggests cover crops as a NI practice). However, these in-field practices are not considered in the possible NI strategies here (Table 5) due to their temporal occurrence in typical agricultural production systems.

Brief descriptions are provided for each intervention that comprise the NI practice ensemble (Table 5). Examples of each NI practice are also shown in Figures 5a-o. The suite of NI practices is divided into four general categories that reflect both landscape position and function: (1) in-field practices or land use modifications implemented in upland or other agricultural landscapes; (2) practices that capture tile drainage effluent at the edge of crop fields or within riparian zones; (3) alterations of small streams or drainage ditches; and, (4) floodplain or channel modifications of larger streams and rivers.

Upland landscape practices include land use conversions to woodland or perennial vegetation, depressional wetlands, farm ponds and runoff attenuation features (RAFs; Nicholson et al., 2020). RAFs have been predominantly used in Europe but can be replicated via similar interventions in MARB watersheds, such as the implementation of water and sediment control basins (WASCOBs; Benning and Craft, 2018; Gupta et al., 2018; USDA, 2022c). In general, these upland interventions are designed to slow and/or capture surface runoff for varying periods of time, which can range from hours to days (and even longer for ponds and some types of wetlands). Suttles et al. (2021) provide tabulated data based on several studies that show the N mitigation capabilities of most of these upland practices, which is further confirmed by additional studies (e.g., Hey and Philippi, 1995; Mitsch et al., 2001; Schilling and Spooner, 2006; Kvítek et al., 2009; Karki et al., 2018; Chen et al., 2020; Brunet et al., 2021). Suttles et al. (2021) also note that RAFs typically provide limited nutrient reduction, although structures such as WASCOBs can effectively remove sediment-bound nutrients (Benning and Craft, 2018).
5. Examples of natural infrastructure practices in agricultural landscapes. (A) Depressional wetlands (photo credit: USGS, David Mushet). (B) Conversion of crop land to native vegetation—forest (photo credit: USFS). (C) Conversion of crop land to native vegetation—grasses (photo credit: USDOE. (D) Runoff attenuation features (photo credit: Nick Barbers, Newcastle University). (E) Farm ponds (photo credit: USDA-NRCS). (F) Water and sediment control basins (Sources: Benning and Craft, 2018; Morrison SWCD, 2022; SCRCA, 2018). (G) Constructed or engineered wetlands (photo credit: Matthew Helmers, Iowa State Univ.). (H) Saturated buffers; conventional on left & saturated on right (photo credit: USDA). See Acknowledgments Section for further credits regarding Figures 5a to 5h.
Riparian areas, small streams or drainage ditches

Figure 5. continued: (I) Vegetated ditches (photo credit: USDA-ARS). (J) Stream restoration before and after (photo credit: USEPA). (K) Two-stage ditches (photo credit: USDA-NRCS). (L) Riparian forest buffer (photo credit: USFS). (M) Floodplain restoration (photo credit: USGS). (N) Oxbow restoration before and after (Wilke, 2022). (O) Beaver dams (Sources: BWW, 2022; DLT, 2019). See Acknowledgments Section for further credits regarding Figures 5i to 5o.
Subsurface tile drains are a key technology supporting crop production in MARB subregions characterized by saturated and/or less permeable soils (Figure 6). However, extensive research has revealed that tile drain systems are also very efficient conduits of nitrate, dissolved P and particulate P, to affected MARB stream systems (Mitsch et al., 2001; 2006; Sprague et al., 2011; Kleinman et al., 2015; Jones, et al., 2018b; Brendel et al., 2019). Constructed wetlands and saturated buffers (Table 5) are two structural practices that have been developed to intercept tile drain effluent at field edges, stream corridors or other strategic locations within a distinct drainage area (Suttles et al., 2021). Constructed wetlands provide storage of excess runoff and effective reductions of both N and P transported in tile drains (Mitsch et al., 2001; 2006; Messer et al., 2021). Saturated buffers provide limited water storage but consistently mitigate nitrate exported via tile drains in a range of Corn Belt conditions (Christianson et al., 2016; Jaynes and Isenhart, 2014; 2019; Groh et al., 2020; Jaynes et al., 2018; Streeter and Schilling, 2021). Denitrifying bioreactors also effectively reduce nitrate levels in tile drain effluent (Christianson et al., 2016; 2021) but generally cost considerably more than saturated buffers based on the bioreactor cost range presented by Christianson et al. (2021) versus saturated buffer costs (Brooks and Jaynes (2017); Jaynes et al., 2018). Most other vegetative interventions (Helmers et al., 2008), such as filter strips or vegetated buffers (Jaynes and Isenhart, 2014; 2019), are not classified as NI practices here due to the inability to capture tile drain effluent or other limitations that undermine removal of soluble nutrients.

Christianson et al. (2016) describe alternative open-ditch systems that provide multiple ecosystem benefits relative to standard ditches: (1) increased N retention in ditch vegetation; (2) lower peak flows during flood periods and higher flows during low-flow conditions; and, (3) greater biological diversity and intensity. Vegetated ditches and two-stage ditches (Table 5) are specific types of open-ditch systems that can provide both flood reduction and water quality benefits between cropland landscapes and stream systems. Two-stage ditches feature a trapezoidal channel design (USDA, 2007) with a narrow main channel bounded by lower vegetated benches (Figure 7), which results in a concentration of low flows in the main channel and higher flows covering the vegetated benches. Two-stage ditches have been found to effectively remove nitrate based on studies conducted in the eastern Corn Belt region (Kalcic et
Figure 6. Distribution of tile drainage in the Corn Belt and other MARB subregions (Source: Valayamkunnath et al., 2020).
Stream restorations (Table 5) focus on reducing peak flows and channel erosion by re-establishing stream geomorphic structures and/or repairing channels with rock rip-rap or other materials. Such restoration efforts have resulted in reduced nutrient losses in some stream segments where the interventions were performed (Filoso and Palmer, 2011; Thompson et al., 2004). Craig et al. (2008) state that stream restoration efforts should be focused on relatively small streams (1st to 3rd order) that receive high N loads during low to moderate flow conditions for the highest opportunities of N removal. Establishment of forested buffer systems was a key component of many riparian management strategies that were included in nearly 3,000 stream restoration projects conducted the Chesapeake Bay watershed (Hassett et al., 2005). Reviews conducted by Mayer et al. (2007) and Sweeney and Newbold (2014) show that forested buffers manifest strong denitrification characteristics for a range of conditions across several dozen sites in the U.S. and Europe. However, tile drain effluent discharge would bypass most forest buffer configurations.

The final category (Table 5) is focused on larger restorations that support intermittent inundation of floodplains, including levee modifications (removal, etc.), reconnection to oxbows or other wetlands, establishment of forest or other vegetation on floodplains and introduction of beaver dams. Recent
evaluations have shown that restored oxbows can provide substantial reductions in nitrate loads discharged by tile drains in north central Iowa (Schilling et al., 2018; 2019). Mondal and Patel (2018) further discuss different floodplain restoration strategies and review successes and problems that resulted from restoration efforts undertaken in several European watersheds.

Another natural intervention that could offer considerable flood control and possibly also water quality benefits is beaver dams. Hey and Philippi (1995) note UMRB population estimates of 10 to 40 million beavers prior to European settlement, and that UMRB hydrology was hugely altered due to the historical elimination of the majority of beavers and associated dam construction. The decline in beaver population and resulting hydrological shifts has been further magnified across the entire North American Continent (Brown and Fouty, 2011). Puttock et al. (2021) found that beaver dams constructed at four sites in England (representing 2nd to 4th stream orders) resulted in reductions of total and peak streamflows in response to >1,000 storm events, and that stream “flashiness” had begun declining at each site. Ronnquist and Westbrook (2021) describe considerable variation in beaver dam structures and hydrological functionality for 66 sites located in the Alberta, Canada Rocky Mountains and point to the need to develop modeling capabilities to account for the hydrologic effects of beaver dams. Beck (2022) reports that beaver dams constructed in Iowa consist of distinct structural organization consisting of grass, sediment, logs rocks and odd debris, exhibit minimal porosity, and hold considerable amounts of water in back-up pools (even in drought conditions). Beck (2022) also notes that beavers manifest considerably different behavior in channelized streams versus more natural stream systems.

Many of the NI practices listed in Table 5 are already supported in government cost-share programs, or could be supported with some modifications to current programs. This includes land use shifts from row crop to perennial vegetation or forests, restoration of upland or riparian wetlands, installation of constructed wetlands, saturated buffers, farm ponds, two-stage ditches, riparian buffers and some types of floodplain restoration. Expansion of support options for NI practices could provide greater opportunities to introduce these interventions in upland cropland and riparian landscapes, resulting in enhanced ecohydrological benefits in MARB subwatersheds.
Research focused on the development, implementation and impacts of NI-type interventions has increased dramatically since the start of the 21st century, especially during the past decade (da Silva and Wheeler, 2017; Ying et al., 2021). The vast majority of studies refer to such interventions as “green infrastructure,” followed by ecological, natural and blue infrastructure nomenclature (da Silva and Wheeler, 2017). Research describing the use of models for NI strategies is dominated by urban applications as reported in reviews by Ahiablame et al. (2012), Jayasooriya and Ng (2014), Kaykhosravi et al. (2018), Lisenbee et al. (2021) Kumar et al. (2021) and Qi et al. (2021). Kumar et al. (2021) further document a publication rate surge similar to the broader NI literature based on their review of model applications for assessing mitigation of flooding, droughts, heatwaves, landslides, and storm surges and coastal erosion impacts as a function of NI interventions. Existing reviews also include overviews of accounting for Low Impact Development (LID) practices (also referred to as “green infrastructure” or simply urban BMPS), such as green roof, bioretention systems, rain gardens, and bioswales, in an urban context (Hunt et al., 2009; Ahiablame et al., 2012; Kaykhosravi et al., 2018; Golden and Hoghooghi, 2018; Lisenbee et al., 2021; Qi et al., 2021). Golden and Hoghooghi (2018) note that LID practices have been evaluated extensively at the local scale, and stressed that expanded watershed-scale assessments are needed to more effectively evaluate the flood and water quality impacts of various urban BMPS.

Ecohydrological models have been applied to evaluate urban BMPS at the watershed scale including SWAT (Castelli et al., 2017; Seo et al., 2017; Pamungkas and Purwitaningsih, 2019; Avellaneda and Jefferson, 2020), HSPF (Martinez and Lopez, 2019; Job et al., 2020; Kim and Ryu, 2020) and MIKE SHE (Christensen and A.R. Schmidt, 2008; Locatelli et al., 2017). Ecohydrological model applications can also be performed for various agricultural- or rural-related conservation practices that can be classified as NI practices, such as wetlands, ponds, land use change, and/or filter and buffer strips (as described in various model documentation and literature sources; e.g., Bicknell et al., 1996; 2002; Graham and Butts, 2005; Gassman et al., 2007; Neitsch et al., 2011; Arnold et al., 2012; Evenson et al., 2018; DHI, 2021; USACE, 2022a; CARD, 2022). These existing BMP capabilities provide a foundation...
for extending ecohydrological model applications for more extensive assessments of broader suites of NI practices for MARB agricultural and rural conditions, depending on the flexibility and overall strengths of a given model. Similar attributes govern the portability of flood models for evaluation of riparian NI interventions across the MARB. In-depth assessments follow regarding the abilities of specific flood and ecohydrological models to accurately represent different NI practices for MARB conditions.

6.1. Assessment of selected flood models/modeling packages for NI practice evaluation

Table 6 lists key attributes for ten flood-related models or modeling packages, nine of which have shared development histories: (1) Auto-RAPID, RAPID and AutoRoute (David et al., 2016; Follum et al., 2017; Tavakoly et al., 2017; Follum et al., 2020; Follum, 2022); (2) HEC-HMS and HEC-RAS (Feldman, 1981; 2003; USACE 2021a; 2022a; 2022b); (3) LISFLOOD (Bates and De Roo, 2000; De Roo et al., 2000; Van Der Knijff et al., 2010; JRC, 2020) and LISFLOOD-FP (Hoch et al., 2019; Shustikova et al., 2020; Sosa et al., 2020; Bates, 2022a); and, (4) MIKE Hydro-River / MIKE Flood (DHI, 2017; 2022a; 2022b). The other reviewed model is GIS Flood Tool (GFT; Verdin et al., 2016; Lawrence et al., 2019).

The majority of these selected models are available in the public domain and feature open source codes. The main exceptions are the MIKE Hydro-River and MIKE Flood packages marketed by DHI (Danish Hydraulic Institute) located in Denmark. An option to access the AutoRoute source code has also not been identified, although Follum (2022) indicates that it may be obtainable from the USACE via request. The AutoRAPID, AutoRoute, LISFLOOD and LISFLOOD-LP models have either already been applied directly to the MARB or have been used to perform other continental-scale flood analyses, confirming portability to the scale of the MARB system. Development of MARB-level simulation frameworks may be possible with the other models (Table 6) but large-scale applications have not been found in the literature to date. All of the models (or combination of models) are capable of simulating flood routing, depth and inundation and can be coupled with ecohydrological models (interface capabilities have not been confirmed for the GIS Flood Tool model). None of the flood models are designed to account for upland NI practice or other upland functions; however, all of the models appear
Table 6. Selected flood models that could be applied within a MARB NI Practice modeling system.

<table>
<thead>
<tr>
<th>Model or Modeling system</th>
<th>Public domain</th>
<th>Open source code</th>
<th>MARB Scale</th>
<th>Flood routing, depth &amp; inundation</th>
<th>Upland NI practices</th>
<th>Riparian NI practices</th>
<th>Water quality</th>
<th>Dam/levee breaks</th>
<th>Interfaces with other models</th>
<th>Internet download location</th>
</tr>
</thead>
<tbody>
<tr>
<td>RAPID / Auto-RAPID / AutoRoute</td>
<td>yes</td>
<td>no&lt;sup&gt;a&lt;/sup&gt;</td>
<td>yes</td>
<td>yes</td>
<td>no</td>
<td>limited</td>
<td>no</td>
<td>limited</td>
<td>yes</td>
<td><a href="https://github.com/c-h-david/rapid">https://github.com/c-h-david/rapid</a></td>
</tr>
<tr>
<td>GIS Flood Tool</td>
<td>yes</td>
<td>yes</td>
<td>no</td>
<td>yes</td>
<td>no</td>
<td>limited</td>
<td>no</td>
<td>limited</td>
<td>NC&lt;sup&gt;b&lt;/sup&gt;</td>
<td><a href="https://code.usgs.gov/gft/python-gis-flood-tool">https://code.usgs.gov/gft/python-gis-flood-tool</a></td>
</tr>
<tr>
<td>HEC-HMS / HEC-RAS</td>
<td>yes</td>
<td>no</td>
<td>no</td>
<td>yes&lt;sup&gt;c&lt;/sup&gt;</td>
<td>no</td>
<td>partial</td>
<td>partial</td>
<td>yes</td>
<td>yes</td>
<td><a href="https://www.hec.usace.army.mil/software/">https://www.hec.usace.army.mil/software/</a></td>
</tr>
<tr>
<td>LISFLOOD / LISFLOOD-FP</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>yes&lt;sup&gt;c&lt;/sup&gt;</td>
<td>no</td>
<td>limited</td>
<td>no</td>
<td>no</td>
<td>yes</td>
<td><a href="https://ec-jrc.github.io/lisflood-model/1_1_introduction_LISFLOOD/">https://ec-jrc.github.io/lisflood-model/1_1_introduction_LISFLOOD/</a></td>
</tr>
<tr>
<td>MIKE-Hydro River / MIKE-Flood</td>
<td>no</td>
<td>no</td>
<td>no</td>
<td>yes</td>
<td>partial</td>
<td>partial&lt;sup&gt;d&lt;/sup&gt;</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td><a href="https://www.mikepoweredbydhi.com/">https://www.mikepoweredbydhi.com/</a></td>
</tr>
</tbody>
</table>

<sup>a</sup>Follum (2022) indicates that it may be possible to obtain source codes for RAPID, Auto-RAPID and/or AutoRoute from the U.S. Army Crop of Engineers by request.

<sup>b</sup>NC = not clear; interfaces of the GIS Flood Tool with other models have not been found in the literature to date.

<sup>c</sup>Flood routing, depth and inundation can be simulated in the HEC-RAS and LISFLOOD-FP models, respectively.

<sup>d</sup>The MIKE-Hydro River riparian NI practices capabilities may be relatively extensive although documentation to confirm this is not available.
capable of representing some aspects of riparian NI practices, although this may be very rudimentary for some of the models. HEC-RAS and the DHI MIKE packages are the only flood models evaluated here that can perform water quality assessments. Those two models, along with LISFLOOD and LISFLOOD-LP, can be used to simulate various scenarios of dam or levee breaks (or vice versa). Website locations containing software, source code and/or information about the flood models are also provided in Table 6.

### 6.1.1. RAPID, AutoRAPID and AutoRoute

The RAPID model was developed by David et al. (2011; 2016) to simulate propagation of water flow waves in river systems using streamflow data measured by the USGS at approximately 20,000 gauges that are linked to the NHDPlus HR (USGS, 2022a) for the conterminous U.S. RAPID simulates discharge at the outlet of every stream reach included in the model structure, which is automatically optimized using the NHDPlus gauge data (David et al., 2011; 2016). Surface runoff to a RAPID stream network must be simulated by an external hydrological or ecohydrological model, such as the following example interfaces: (1) the Noah Land Surface Model with Multiparameterization Options (Noah-MP; Niu et al. 2011) for the 26 000 km² combined Guadalupe and San Antonio River Basins in south central Texas (David et al., 2011); and, (2) the Variable Infiltration Capacity (VIC) Macroscale Hydrologic Model (Lianing et al., 1994; UW, 2021) for the entire MARB (Tavakoly et al., 2021). RAPID also has to be coupled with AutoRoute or similar model to generate flood inundation maps for a simulated domain (Follum et al., 2017; Follum, 2022).

AutoRoute was originally developed by the USACE in 2011 to assess route vulnerability of military units confronted by flooding over extensive regions in different parts of the world (Follum, 2013). The software is designed to quickly and accurately estimate flood extent as a function of Digital Elevation Model (DEM) data. AutoRoute has been used extensively outside of the U.S. to support U.S. military movements including in Afghanistan (Follum, 2022), and can be interfaced with mobility models to assess the risk of flooding to military vehicles (Follum, 2013). The accuracy of AutoRoute is
dependent on the availability of reliable DEM data and the results can degrade if a less accurate
topographic layer is used in the model.

The combined RAPID and AutoRoute modeling system is referred to as AutoRAPID, which
supports relatively quick estimates of streamflow and flood inundation across large regions using high
resolution DEM data (Follum et al., 2017). Flood inundation maps are created as a function of peak flow
processing in the AutoRoute component of AutoRAPID. AutoRAPID was initially tested for major floods
that occurred in 2008 in the Midwest and 2011 in the LRMB, followed by an application for the entire
MARB during 2005 to 2014 (Follum et al., 2017). The authors state that AutoRoute generated > 130
million “cross sections” that were aggregated across 39 “tiles” for the MARB simulation. The resulting
processing of the flood depth map required 20 minutes for each tile and an additional 15 minutes to
translate the tile flood raster files into shapefiles. The most accurate AutoRAPID flood inundation maps
were obtained for MARB subregions characterized by: (1) more accurate peak flows estimated by
RAPID; (2) moderate to high topographic relief; and, (3) lack of dams and/or backwater effects.

Improved accounting of MARB dams and reservoirs has since been established in RAPID (Tavakoly et
al., 2021).

Follum (2022) noted that the existing AutoRAPID MARB model may be accessible for users that
want to apply it for research or other purposes (further investigation is needed to confirm this). He also
noted that the riparian NI practice and dam/levee scenario capabilities of RAPID/AutoRoute/AutoRAPID
are very limited (Table 6) and that applications of the RAPID/AutoRoute/AutoRAPID modeling tools can
be conducted within the Streamflow Prediction Tool software package (Snow, 2016; Snow et al., 2016).

6.1.2. HEC-HMS and HEC-RAS

HEC-HMS, HEC-RAS and related packages (USACE, 2022b) represent continuous USACE
water resource focused software development that began in the early 1960s at the USACE Hydrologic
Engineering Center (Feldman, 1981; 2003). Early analysis tools were very simple and were often
configured for just one aspect of the overall hydrologic balance such as the Basin Rainfall and Snowmelt
Computation program (Feldman, 1981). The first “HEC models” were the HEC-1 Flood Hydrograph Package (USACE, 1981) and HEC-2 Water Surface Profile program, which were released in 1968.

Feldman (2003) notes that there were 56 different programs listed in HEC’s Computer Program Catalog—nearly 40 of those are shown in table format by Feldman (1981) including HEC-1, -2, -3, -4 and -5. The development of HEC-HMS is linked to the earlier HEC models, was well established by the start of the twenty-first century (Feldman, 2003) and now represents the fourth major release of the model (USACE, 2022b). The initial release of HEC-RAS (version 1.0) occurred in 1995, which has been followed by nearly 15 additional major releases (USACE, 2021a; 2021b).

HEC-HMS is designed to compute surface runoff and baseflow in response to precipitation and snowmelt for a simulated watershed, followed by subsequent channel and stream network routing. Users have the option of selecting from several different infiltration/runoff and routing methods to simulate those aspects of the overall hydrologic balance (USACE, 2021a). The model can also account for water impoundments such as reservoirs, lakes and ponds, simulated pumping from detention ponds and transport of sediment through a watershed system (USACE, 2021a). Hundreds of existing HEC-HMS studies including coupled HEC-HMS/-RAS investigations can be accessed in the literature that describe applications performed in different global subregions (e.g., Knebl et al., 2005; Meenu et al., 2013; Ibrahim-Bathis and Ahmed, 2016; Doll et al., 2020; Abdessamed and Abderrazak, 2019; Ismail et al., 2020; Tang et al., 2020a; 2020b; Namara et al., 2021).

HEC-RAS is a very established hydraulic model that can be used to simulate variations in streamflows, inundation of floodplains and tracking of selected water quality indicators (USACE, 2021a; 2021b; 2022b). Variations in streamflow can be simulated for the following conditions: (1) steady streamflow through a single stream reach, system of channels or watershed system, including the effects of stream junctures, bridges and other structures; and, (2) separate or combined unsteady flow simulations through a complete system of channels, floodplains and alluvial fans. Additional features incorporated in the unsteady streamflow component include automated calibration options, assessments of dam breaks, levee breaches and levee overtopping, and simulation of pumping stations. Current versions of HEC-RAS
can also simulate: (1) sediment transport and “movable boundary conditions” resulting from scouring and
deposition; (2) water temperature fluctuations; and, (3) transformation and transport of selected N and P
constituents, algae, dissolved oxygen and carbonaceous biological oxygen demand. HEC-RAS has also
been applied extensively worldwide including for the following examples: unsteady state flows (Knebl et
al., 2005), two-dimensional flow representation (Costabile et al., 2020; Zeiger and Hubbart, 2021), flood
inundation (Afshari et al., 2018), flood warning systems (Nguyen et al., 2020), and interfaces with HEC-
HMS (Tang et al., 2020a; 2020b) and SWAT (Tadesse and Dai, 2019; Baldan et al., 2021).

Studies describing applications of HEC-HMS and/or HEC-RAS for assessments of watershed-
scale NI-practice impacts are relatively sparse in available literature. Doll et al. (2020) describe the
impacts of evaluating the 80 km² Stoney Creek watershed in North Carolina with the following NI
intervention scenarios in HEC-HMS: (1) storage/retention based on installing small berms in the upper
reaches of the drainage area; (2) converting 30% of the agricultural land to forest; and, (3) a combined
strategy of storage/retention and land conversion. They further assessed the effects of floodplain
restoration and road crossing modification/removal for a small 8 km² using HEC-RAS. Tang et al.
(2020a: 2020b) evaluated the impacts of wetlands on streamflow using a coupled HEC-HMS/-RAS
modeling approach for the 692 km² Cypress Creek watershed located in Houston, TX—the wetlands were
simulated as reservoirs in HEC-HMS. Additional searching would be required to identify whether other
HEC-HMS and/or HEC-RAS studies exist that report applications that could be considered representative
of NI practice implementation.

One major limitation of HEC-HMS is that it does not account for subsurface tile drainage
(Klinger, 2014), which has been implemented in much of the MARB (Figure 6) and is a key pathway of
nitrate and TP transport as previously discussed. Klinger (2014) used a proxy method for tile drainage
effects in order to accurately account for hydrologic balance and streamflow characteristics representative
of the Raccoon River Watershed in west central Iowa. However, this weakness essentially eliminates
consideration of HEC-HMS as a tool that could mechanistically account for NI practices such as
constructed wetlands and saturated buffers (Table 5 and Figure 5). Beyond this, the HEC-HMS and HEC-
RAS user manuals (USACE, 2021a; 2021b; 2022a) provide very little guidance regarding simulation of land use, wetlands, buffers and other management approaches that would be considered NI practices, which is generally reflected in the large HEC-HMS/-RAS literature realm.

### 6.1.3. GIS Flood Tool

GFT was developed to provide “reconnaissance-level flood inundation mapping” for regions of the world that lack data and resources to create flood maps, with support from the U.S. Agency for International Development’s Office of U.S. Foreign Disaster Assistance (Verdin et al., 2016). It has also been utilized for rapid production of flood inundation maps as part of the USGS Flood Inundation Mapping Program. GFT flood inundation maps are created (Verdin et al., 2016) via an extension tool in the ArcGIS platform (ESRI, 2022) by: (1) computing the effects of Manning equation effects on steady streamflow within open channels using pre-processed DEM data; and, (2) user-supplied discharge data that is used to compute flood depth via a depth-discharge relationship. The software has been tested at several sites in the U.S. and in Nambia (Verdin et al., 2016; Lawrence et al., 2019). Results showed accurate replication of 1-D hydraulic model results for 75% of the test sites, with lower accuracy for locations with low relief or channels with very gradual water flow divides (Verdin et al., 2016).

Lawrence et al. (2019) describe an analysis of riparian wetland impacts on economic damage estimates of different levels of flood intensity. However, GFT was not used to directly simulate wetland position and dynamics as part of the study. In general, the GFT approach is similar to the AutoRAPID approach and is similarly constrained by weaknesses of not being able to robustly represent riparian NI practices.

### 6.1.4. LISFLOOD and LISFLOOD-FP

The LISFLOOD model (Bates and De Roo, 2000; De Roo et al., 2000; Van Der Knijff et al., 2010; Sutanto et al., 2020; JRC, 2020) was derived from the predecessor Limburg Soil Erosion Model (LISEM; de Roo et al., 1998) and was developed at the European Joint Research Centre to support operational flood forecasting (Smith et al., 2016; Bates, 2022b). The LISFLOOD model features the
following processes: (1) rainfall-runoff driven by precipitation, temperature and other climatic inputs; (2) hydrological balance calculations on a daily or sub-daily basis for each grid cell representing the terrestrial surface of a simulated watershed; and, (3) accounting for various hydrological processes including snowmelt, runoff, infiltration, soil moisture, base flow, groundwater storage and evapotranspiration (Sutanto et al., 2020). As noted previously, LISFLOOD has been adopted by the EFAS (EFAS, 2021) to conduct various flood forecasting analyses, such as EFAS threshold assessments (low, medium, high and severe) and flood alerts or warning (Thielen et al., 2009; Smith et al., 2016). It has also been used to forecast severity of drought across the European Union (Sutanto et al., 2020) and for flood forecasting across the African continent (Thiemig et al., 2015). However, LISFLOOD was structured such that it could not be used directly to support flood inundation assessments. As stated by De Roo et al. (2000): “Inundation extent will be simulated by extrapolating predicted water levels onto a DEM or LISFLOOD will be linked to an existing two-dimensional or three-dimensional model for detailed floodplain routing.”

The LISFLOOD-FP model is a distinctly different model relative to LISFLOOD (Bates, 2022b) and is specifically designed to simulate channel hydraulics and floodplain inundation, via integration with high resolution DEM data (Bates and de Roo, 2000; Bates, 2022a; 2022b). A storage cell technique was incorporated in LISFLOOD-FP to simulate flood inundation that was simpler than one- and two-dimensional hydraulic models used at the time such as HEC-RAS and MIKE11 (now superseded by MIKE-Hydro, Table 6) as described in a tabulated comparison by Bates and de Roo (2000). Enhanced capabilities have since been introduced in LISFLOOD-LP including two-dimensional hydrodynamic flows and “dynamic propagation of flood waves over fluvial, coastal and estuarine floodplains” (Bates et al., 2013; Bates, 2022a), and assessment of two-dimensional levee breach or overtopping scenarios (Shustikova et al., 2020). Dottori et al. (2022) interfaced LISFLOOD-FP with LISFLOOD to create European river hazard flood maps; and, LISFLOOD was used to generate the river discharge data, which was input to LISFLOOD-LP to determine the extent of flood inundation. LISFLOOD-FP has also been interfaced with VIC and RAPID to generate estimates of flood inundation for the 15,186 km² Upper
Krishna River Basin located in southern India (Nandi and Reddy, 2022), underscoring the versatility of the model.

It is clear that LISFLOOD and LISFLOOD-FP would have to be executed in a coupled modeling system to simulate any drainage system including the MARB, either together or with other appropriate models. In contrast, the ability of LISFLOOD and LISFLOOD-LP to account for upland or riparian NI interventions is difficult to discern from available literature. It is possible that pond, wetlands and vegetated buffers and/or filter strips can be accommodated in landscapes that are simulated in the two models. However, no information has been discovered so far to confirm this.

6.1.5. MIKE-Hydro River and MIKE-Flood

MIKE-Hydro River and MIKE-Flood are two of the wide array of products currently marketed by DHI (DHI, 2022e). The origin of current “MIKE models” can be traced to the earlier SHE model development (Refsgaard et al., 2010), which was initiated as a collaborative research effort in 1976 between DHI, Institute of Hydrology (United Kingdom) and French consulting company SOGREAH. The French partnership was later transferred to Laboratoire d’Hydraulique de France in 1990. The research eventually split into development of two different ecohydrological models, with DHI continuing work on MIKE SHE versus a team located at the University of Newcastle, United Kingdom who focused on the development of the SHETRAN model (Refsgaard et al., 2010; Bathurst and O’Connell, 1992). One interesting artifact of the SHE- and MIKE-related modeling history is that MIKE is not an acronym but rather honors Michael B. Abbott (Refsgaard, 2021), who pioneered the original development of most of the SHE software systems including MIKE SHE (J. Hydroinformatics, 2020).

MIKE-Hydro River is the current version of the predecessor MIKE11 model (DHI, 2017; 2022a), which was part of DHI’s fourth-generation modelling systems (J. Hydroinformatics, 2020). Key MIKE-Hydro River simulation strengths include flood inundation and other analyses, determination of flood mitigation strategies, accounting for canal systems, reservoirs and other operational structures, wetland hydrology and water quality dynamics, wetland restoration, sediment transport in rivers and river
morphology changes (DHI, 2017; 2022a). MIKE-Hydro River can simulate rainfall-runoff across
watershed landscapes and route flow through stream networks and into other water bodies (DHI, 2022a).
However, DHI (2022a) further notes that MIKE-Hydro River can be interfaced with MIKE SHE to
perform expanded analyses of land use change, and hydrologic evaluation of overland flow, soil water
and groundwater processes. Two examples of MIKE-Hydro River applications reported in the literature
are: (1) hydrodynamic simulation of a stream system in Queensland, Australia that is influenced by floods
and tidal conditions (Jahandideh-Tehrani et al., 2020); and, (2) streamflow simulation and accompanying
flood risk assessment for a watershed located on the Island of Crete, Greece (Goumas et al., 2022).
MIKE-Hydro River can also be executed in the broader MIKE FLOOD software package, which
DHI (2022b) states can “model any flood problem - whether it involves rivers, floodplains, flooding in
streets, drainage networks, coastal areas, dams, levee and dike breaches, or any combination of these.”
The MIKE FLOOD package also allows interfaces with “MIKE+ for collection systems and MIKE 21 for
2D surface flow” (DHI, 2022b). In general, the DHI MIKE products provide an impressive system of
models that likely could meet many of the NI practice simulation goals although there is some lack of
clarity regarding the full NI practice capabilities of the DHI models, particularly regarding MIKE SHE
(see discussion in Section 6.2.3). In addition, continental-scale applications of different MIKE models are
not readily apparent in available literature. Beyond these issues, DHI model product costs and the
inability to access and modify source codes are serious impediments regarding application of MIKE-
River and the MIKE Flood package for MARB NI practice scenarios.

6.2. Assessment of selected ecohydrological models for NI practice evaluation
Table 7 lists the attributes of seven selected ecohydrological models that could potentially be
considered for application within a MARB modeling system. HSPF, MIKE SHE and SPARROW are
independent models that are not inter-related with any of the other models (Table 7). Agro-IBIS is often
executed as an independent model; however, it has also been coupled with the Terrestrial Hydrology
Model with Biogeochemistry (THMB; Coe, 2000) software, which is the most relevant application mode
for the MARB NI practice modeling system. The Agricultural Policy / Environmental eXtender (APEX) model (Williams et al., 2008; Gassman et al., 2010; Wang et al., 2012; Steglich et al., 2019) is closely
<table>
<thead>
<tr>
<th>Model or Modeling system</th>
<th>Public domain</th>
<th>Open source code</th>
<th>MARB Scale</th>
<th>Flood depth &amp; inundation</th>
<th>Upland NI practices</th>
<th>Riparian NI practices</th>
<th>Water quality</th>
<th>Dam/levee breaks</th>
<th>Interfaces with other models</th>
<th>Internet download location</th>
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<td>yes</td>
<td>no</td>
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<td>yes</td>
<td>no</td>
<td>yes</td>
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<tr>
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<td>yes</td>
<td>no</td>
<td>no</td>
<td>extensive</td>
<td>partial</td>
<td>yes</td>
<td>no</td>
<td>yes</td>
<td><a href="https://epicapex.tamu.edu/about/apex/">https://epicapex.tamu.edu/about/apex/</a></td>
</tr>
<tr>
<td>HSPF</td>
<td>yes</td>
<td>no(^b)</td>
<td>no</td>
<td>no</td>
<td>partial</td>
<td>partial</td>
<td>yes</td>
<td>no</td>
<td>yes</td>
<td><a href="https://www.epa.gov/ceam/hydrological-simulation-program-fortran-hspf">https://www.epa.gov/ceam/hydrological-simulation-program-fortran-hspf</a></td>
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<tr>
<td>MIKE SHE</td>
<td>no</td>
<td>no</td>
<td>no</td>
<td>yes</td>
<td>partial</td>
<td>partial</td>
<td>yes</td>
<td>no</td>
<td>yes</td>
<td><a href="https://www.mikepoweredbydhi.com/">https://www.mikepoweredbydhi.com/</a></td>
</tr>
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<td>SPARROW</td>
<td>yes</td>
<td>yes</td>
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<td>no</td>
<td>limited</td>
<td>limited</td>
<td>yes</td>
<td>no</td>
<td>yes</td>
<td>[<a href="https://code.usgs.gov/water/stats/rsparr">https://code.usgs.gov/water/stats/rsparr</a> ow](<a href="https://code.usgs.gov/water/stats/rsparr">https://code.usgs.gov/water/stats/rsparr</a> ow)</td>
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<tr>
<td>SWAT+</td>
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<td>partial</td>
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<td>no</td>
<td>yes</td>
<td><a href="https://swat.tamu.edu/software/">https://swat.tamu.edu/software/</a></td>
</tr>
</tbody>
</table>

\(^a\) The source code currently provided for Agro-IBIS at the corresponding URL is for a predecessor IBIS model rather than Agro-IBIS; the source code available for THMB at that link is likely also out-of-date.

\(^b\) The HSPF executable can be obtained via Better Assessment Science Integrating Point and Non-point Sources (BASINS) software download options provided at the associated URL; there is no indication that the source code can be downloaded at that website.

\(^c\) A version of SWAT is also included in the BASINS software described in footnote b.
related to both the SWAT and SWAT+ models (Table 7). SWAT represents model codes through SWAT version 2012 (SWAT2012) as documented by Gassman and Wang (2015), versus the more recent SWAT+ code (Beiger et al., 2017). These three models are discussed together due to the developmental and application inter-relationships that exist between them.

All of the ecohydrological models are available in the public domain and most provide access to the corresponding source codes (Table 7). However, the source codes provided for Agro-IBIS and THMB at UW (2022) appear to be out-of-date (e.g., model description, source code and related files are presented for the predecessor IBIS model rather than Agro-IBIS). It is also not clear if the source code for HSPF can be accessed. Six of the models have been applied for the entire MARB: Agro-IBIS coupled with THMB (Donner and Kucharik, 2008; Ferin et al., 2021), SPARROW (Alexander et al., 2008), APEX coupled with SWAT (White et al., 2014), and SWAT+, which is being implemented within the National Agroecosystem Model (NAM) to simulate the entire conterminous U.S. (Arnold et al., 2021; White et al., 2022). HSPF has been applied extensively for the 166,000 km² Chesapeake Bay (Shenk et al., 2012) and could likely be scaled-up for MARB-level simulations. It is also possible that MIKE SHE could be ported to the MARB level, although large-scale applications have not been confirmed for the model. MIKE SHE is the only ecological model reviewed here that can simulate flood depth/inundation. All of the models can account for water quality impacts and can be interfaced with other models. In contrast, none of the models are capable of simulating dam/levee break or overtopping scenarios. Most of the models can account for at least some upland NI practices, and to a lesser extent riparian NI practices. SPARROW is a uniquely structured regression-based model, among the other models listed in Table 7, that does not directly simulate NI practices (or other agricultural management practices). The effects of NI interventions are instead accounted for by coefficient adjustments used in the regression equations that comprise the model structure (McLellan et al., 2015; García et al., 2016; Miller et al., 2020) as discussed below.
6.2.1. Agro-IBIS and THMB

The Agro-IBIS model is a terrestrial model that consists of the following major modules (Kucharik, 2003): atmosphere (climate), land surface, vegetation dynamics, natural vegetation and crop phenology, soil carbon and N cycling, crop management and solute transport. The model accounts for managed agroecosystems as a function of crop phenology (including hybrid selection) and management practices such as fertilizer, tillage and irrigation. It also features a detailed carbon and N cycling submodel that incorporates decomposition of soil litter and organic matter, soil respiration, N mineralization, fixation and other N transformation processes. Similarly, considerable detail is also built into the vegetation/crop physiology/phenology components of the model including photosynthesis, crop respiration, stomatal conductance, budburst and senescence. Agro-IBIS has been applied for evaluating crop growth/yield, carbon cycling and related analyses on multiple continents (e.g., Kucharik, 2003; Sun et al., 2017; Amuti et al., 2018; Mykleby et al., 2017; Moreira et al., 2018).

Agro-IBIS has been interfaced with the Terrestrial Hydrology Model with Biogeochemistry (THMB; Coe, 2000) to simulate N cycling and transport for the entire MARB in response to different biofuel scenarios (Donner and Kucharik, 2008; Ferin et al., 2021). The two studies generated similar dissolved inorganic nitrogen (DIN) loading estimates to the Gulf of Mexico, which included relatively strong simulated replication of observed DIN discharge at the Mississippi River outlet. However, Ferin et al. (2021) state that they did not account for subsurface tile drains in their study—their conviction is that this did not undermine the accuracy of their Agro-IBIS N leaching estimates and subsequent N transport simulation with THMB. But this is a major limitation when considering NI practices designed to capture tile drain effluent such as constructed wetlands and saturated buffers. It appears that land use shifts, depressional wetlands, farm ponds and filter strips can be accounted for in the coupled Agro-IBIS/ THMB models. However, the current dearth of documentation for the two models hinders full evaluation of which NI practices can be realistically simulated in this combined modeling approach.

6.2.2. HSPF
The origins of HSPF are rooted in several predecessor models including the Stanford Watershed Model (SWM), Agricultural Runoff Management (ARM), Nonpoint Source (NPS) model, and Hydrocomp Simulation Program (HSP) models (Donigian and Imhoff, 2005). Initial development of SWM occurred between 1959 and 1966 (Crawford and Burges, 2004), establishing it as one of the first hydrologic models ever developed (Donigian and Imhoff, 2005). Donigian and Imhoff (2005) further note that: (1) SWM eventually evolved into HSP by the early 1970s; (2) the evolution occurred concurrently with the development of the USEPA-sponsored ARM and NPS models; and, (3) continued support from the USEPA eventually resulted in the development of HSPF in the 1970s. The primary functionality in the HSP, ARM and NPS were merged into HSPF. Earlier HSPF documentation is available on-line for versions 11.0 and 12.0 which were released in 1997 and 2001, respectively (Bicknell et al., 1996; 2002; USEPA, 2014). The HSPF executable is currently provided in downloads of the USEPA Better Assessment Science Integrating Point and Non-point Sources (BASINS) software, along with other models and data including an earlier version of SWAT (Duda et al., 2012; USEPA, 2019). Updated documentation for executing HSPF within basins is provided by USEPA (2019).

The structure of HSPF is organized into four main modules (Donigian and Imhoff, 2005): (1) PERLND, which accounts for the hydrological balance and pollutant cycling/transport for pervious land areas; (2) IMPLND, which accounts for the hydrological balance and solids transport associated with impervious land areas; (3) RCHRES, which simulates flow hydraulics and pollutant transport for river reaches and well-mixed reservoirs; and, (4) BMP, which simulates hydrologic and pollutant effects due to implementation of management practices. The model provides the ability to simulate runoff and pollutant discharge from different land surfaces and subsequently route the flow and pollutants through a stream network for whatever simulation period is of interest to the user. Duda et al. (2012) provide a detailed summary of various terrestrial and in-stream processes that can be simulated in HSPF. The stream channel flow processes are described in “hydraulic terms” (Bicknell et al., 1996; 2002; Duda et al., 2012); however, it appears this does not include the ability to simulate flood depth and inundation as confirmed by Kourgialas and Karatzas (2014) who interfaced HSPF with MIKE11 to simulate flood impacts for a
watershed in Greece. HSPF has been applied in hundreds of other studies including calibration/validation analyses (Duda et al., 2012), climate change analysis (Dudula and Randhir, 2016), simulation of urban streams (Yazdi et al., 2019) and assessment of BMPs (Shenk et al., 2012).

Bicknell et al. (2002) provide documentation regarding various management practices that can be simulated in HSPF including planting/harvesting dates, fertilizer applications and irrigation and the user manual also provides guidance on simulating wetlands and reservoirs. The latter infers that farm ponds can likely be simulated in the model. However, Duda et al. (2012) clarify that HSPF cannot represent tile drains in cropland landscapes, which nullifies the model for consideration of constructed wetland, saturated buffer, and similar scenarios. Duda et al. (2012) also note that “selected agricultural conditions, such as crop rotations and certain BMPs, are difficult to represent explicitly,” which further indicates that HSPF would manifest multiple weaknesses regarding replication of pollutant discharge from cropland landscapes in the Corn Belt region and other MARB agricultural subregions. However, Palace et al. (1998) describe a suite of agricultural and urban BMPs that were simulated with HSPF for the Chesapeake Bay including CRP, forest/grass buffers, pond/wetland systems, conservation tillage, stream restoration (non-tidal), nutrient management and cover crops. This indicates that representation of a range of NI and other agricultural practices can be configured in HSPF, although there is a lack of clarity as to how these practices would be simulated based on user manuals and other documents encountered so far.

6.2.3. MIKE SHE

MIKE SHE developmental history is described in Section 6.1.5. and thus is not provided here. MIKE SHE simulates the complete hydrologic cycle including overland flow, groundwater flow, channel flow and interactions between the flow pathways (Graham and Butts, 2005). The hydrologic processes can be simulated at varying spatial detail and process complexity based on a grid delineation of the simulated watershed (Graham and Butts, 2005). Current versions offer different options for simulating surface runoff, soil infiltration and river channel flow that reflect different levels of complexity depending on the needs of the user, including two-dimensional flow simulation for some processes (DHI, 2022c).
MIKE SHE can also simulate floodplain management, floodplain inundation, two-way flow interaction with rivers, flooding influenced by groundwater, irrigation applications and drought assessments (DHI, 2022c; 2022d). MIKE SHE simulates nutrient cycling and transport, sediment loss and transport, and pesticide fate and transport, and supports integrated solute movement between surface sources and subsurface water (DHI, 2022c; 2022d). The option to interface MIKE SHE with MIKE ECO Lab provides the ability to simulate “water quality and ecological studies for coastal waters, rivers, lakes, wetlands, reservoirs, estuaries, groundwater and the sea” (DHI, 2022f). Ma et al. (2016) provide an in-depth overview of the various aspects of applying MIKE SHE, within the overall DHI MIKE platform, for environmental and ecological analyses.

A large body of MIKE SHE related literature exists such as the following example applications: interface with MIKE URBAN for an urbanized watershed in Australia (Locatelli et al., 2017), assessment of flood impacts in Oklahoma (Graham and Butts, 2005), water balance testing via coupled analysis with MIKE 11 for a lowland watershed in Germany (Waseem et al., 2020), assessment of lowland wet grassland in the United Kingdom using a coupled MIKE SHE/MIKE 11 modeling approach (Thompson et al., 2004) and replication of subsurface tile drainage flows in Iowa (Zhou et al., 2013). MIKE SHE is clearly a versatile modeling tool that can be interfaced with several other DHI software such MIKE-Hydro, MIKE URBAN and MIKE ECO Lab, that should be able to account for several of the NI practices listed in Table 4. However, specific guidance on how to simulate BMPs including the NI intervention strategies of interest here is quite limited in spite of extensive documentation that exists for the model. Ma et al. (2016) also discuss that MIKE SHE is a very complex tool to learn and operate, and that a team of professionals/scientists can be required to support ongoing applications. And, as noted, previously for MIKE-Hydro and MIKE FLOOD, the reality of expensive, proprietary software is a major constraint towards adopting MIKE SHE for a MARB modeling system.

6.2.4. SPARROW
SPARROW is a regression model method developed by USGS scientists in 1997 to estimate movement of TN and TP in stream systems for the conterminous U.S., as a function of spatially-linked pollution sources, land surface characteristics and stream channel attributes (Smith et al., 1997). The simulated stream network for the original SPARROW model was comprised of 60,000 reaches with a combined total distance of nearly one million km (Smith et al., 1997). The calibrated TN and TP models accounted for several pollutant sources: point, fertilizer and manure applications on cropland, combined urban, forest and rangeland, and N atmospheric deposition. Land surface factors incorporated in the model included soil permeability, slope, stream density, percent of total area defined as wetlands or irrigated cropland, long-term mean ambient air temperature and precipitation, and depth of irrigation water. In-stream losses of pollutant mass were then computed based on streamflow, travel time and the presence of a reservoir in a reach. Variation on the regression model inputs have occurred in subsequent studies. For example, Alexander et al. (2008) estimated MARB TN/TP discharge as a function of 10/8 pollutant sources, 6/5 climatic and landscape characteristics, and nutrient losses in streams and reservoirs. Other example regression model variations/developments reported in the extensive SPARROW literature are shown in Table 1.

The studies described by McLellen et al. (2015), García et al. (2016) and Miller et al. (2020) listed in Table 8 reveal how SPARROW can be used to represent aggregated conservation practice/NI structure impacts on nutrient transport for specific subwatersheds incorporated in the model structure (e.g., 12-digit watersheds). From this perspective, SPARROW could be used to evaluate any combination of the NI practices listed in Table 4 if aggregated flood and water quality impact estimates were known for a set of practices installed at the 12-digit watershed level or whatever subwatershed delineation was being used in the model structure. However, this would require rough cumulative impact guesses for a suite of practices implemented in every subwatershed and it is also not possible to evaluate the effects of targeting NI interventions for vulnerable landscape locations. One intriguing potential use of SPARROW Table 8. Example SPARROW applications that included part or all of the MARB.
<table>
<thead>
<tr>
<th>Study</th>
<th>Region</th>
<th>Constituents</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Preston et al., 2011</td>
<td>Six USGS NAWQA regions</td>
<td>TN; TP</td>
<td>Synthesis of 12 independently calibrated models (separate models for TN and TP)</td>
</tr>
<tr>
<td>Robertson and Saad, 2021</td>
<td>MARB</td>
<td>TN; TP</td>
<td>Uncertainty analysis of TN &amp; TP estimates; MARB model structure with 25,000 8-digit watersheds</td>
</tr>
<tr>
<td>Shih et al., 2010</td>
<td>Conterminous U.S.</td>
<td>TOC(^a)</td>
<td>Land use and in-stream effects on total organic carbon transport in the U.S. stream network</td>
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<tr>
<td>McLellan et al., 2015</td>
<td>Corn Belt (ORB &amp; UMRB)</td>
<td>TN</td>
<td>Effects of fertilizer applications and conservation practice interventions on Corn Belt TN export</td>
</tr>
<tr>
<td>Garcia et al., 2016</td>
<td>UMRB</td>
<td>TN; TP</td>
<td>CEAP conservation practice data interfaced with SPARROW to estimate UMRB nutrient loadings</td>
</tr>
<tr>
<td>Miller et al., 2020</td>
<td>Chesapeake Bay</td>
<td>TN</td>
<td>Export of TN to the Chesapeake Bay in response to nine source reduction and land use scenarios</td>
</tr>
</tbody>
</table>

\(^a\)TOC = total organic carbon

within an overall MARB modeling system is to serve in a verification role of other model output. For example, APEX-SWAT modeling system TN and TP outputs were compared versus SPARROW outputs for 848 MARB 8-digit watersheds (White et al., 2014).

**6.2.5. APEX, SWAT and SWAT+**

The APEX, SWAT and SWAT+ models are closely inter-related and share a common genesis in the development of single-storm-event hydrology models that commenced in 1965 at the co-located USDA Agricultural Research Service (ARS) and Texas A&M University (Texas A&M AgriLife) laboratories in Temple, TX (Williams et al., 2008). APEX is the direct successor to the previously developed Environmental Policy Integrated Climate (EPIC) field-scale ecohydrological model (Williams et al., 1984; 1989; Williams, 1990) and is designed to support multi-field, whole farm and small watershed assessments (Gassman et al., 2010; Steglich et al., 2019). The origin of SWAT (Arnold et al., 1992) traces to the interface of: (1) Simulator for Water Resources in Rural Basins (SWRRB) ecohydrological model (Arnold et al., 1987; Williams et al., 1985; 2008), the direct predecessor model; and, (2) the Routing of Outputs to the Outlet (ROTO) model (Arnold et al., 1995; Williams et al., 2008),
which supported routing of water and pollutants through large stream systems. The first major release of
SWAT version 94.2 (SWAT94.2) incorporated the EPIC crop/plant growth component, flexibility to
insert multiple hydrologic response units (HRUs) in a subwatershed and other improvements (Gassman and Wang, 2005). Development of the model has continued virtually unabated since that first release, spanning eight major versions and dozens of interim releases through SWAT2012 (Gassman and Wang, 2015). Current SWAT+ versions reflect a dramatically restructured code that provides numerous improvements including enhanced spatial and functional representation of cropping systems, management practices and conservation interventions (Beiger et al., 2017; Bailey et al., 2020a; 2020b; 2021; Arnold et al., 2021; Gassman et al., 2022; White et al., 2022; Čerkasova et al., 2022).

6.2.5.1. APEX

APEX consists of 12 primary components including climatic inputs, hydrologic balance, crop/plant growth and yield, pollutant (pesticide and nutrients) cycling/fate and transport, sediment loss and transport, soil organic carbon (SOC) cycling and transport, and routing of flow and pollutants from delineated subareas (Gassman et al., 2010; Steglich et al., 2019). An extensive suite of management and conservation practices can be simulated in APEX including fertilizer and livestock manure applications, irrigation (sprinkler, drip, or furrow), winter cover crops, subsurface tile drainage, impoundments (reservoirs, ponds, wetlands, and rice paddies), filter/buffer strips, grassed waterways, and rotational grazing of livestock (Gassman et al., 2010; 2022; Waidler et al., 2011). The crop/plant growth component provides considerable flexibility, allowing simulation of virtually any mix of row crops, small grains and/or legumes, rice grown in paddies, various tree, fruit and vegetable crops, and simultaneous growth of up to ten crops and/or other plants (Gassman et al., 2010). An APEX “test version” exists that includes saturated buffer, denitrifying bioreactor and drainage water management (Christianson et al., 2016) modules; these modules would need to be ported to the main code to become operational for APEX users (Jeong, 2022). APEX also serves as the underlying model for the Nutrient Tracking Tool, which is an online portal that allows users to evaluate the water quality effects of different nutrient application rates/timing, cropping systems, and other management/conservation practices for U.S. cropland.
landscapes (Saleh et al., 2011; 2022; Moriasi et al., 2016). The portal provides considerable flexibility for evaluating specific fields or farms but is not designed to support development and implementation of large-scale modeling systems.

APEX has been applied in dozens of studies in the U.S. and in other countries. Gassman et al. (2010) describe APEX assessments of manure and fertilizer application impacts on nutrient losses for watersheds dominated by livestock production in Texas and Iowa. They also summarize hydrologic and pollutant calibration/validation results for several APEX studies that were reported in the literature at that time. Additional review of appropriate APEX calibration and validation methods, and selected model results, are reported by Wang et al. (2012). Yin et al. (2009) describe surface runoff and sediment losses for three research plots characterized by different land use treatments located in China. APEX was also successfully tested for rice paddy dynamics in South Korea (Choi et al., 2017) and replication of perennial grassland vegetation in Kansas (Zilverberg et al., 2017). APEX was interfaced with SWAT in the MARB CEAP assessment (Santhi et al., 2014a; 2014b; White et al., 2014) as follows: (1) APEX was used to estimate flow and pollutant discharge from cropland and other agricultural sources, including accounting for conservation practices identified in supporting CEAP surveys; (2) the agricultural outputs were integrated with flow and pollutants discharged from urban, forest and other non-agricultural land sources, which were simulated with SWAT; and, (3) the resulting combined flows and pollutants were routed through the MARB stream system to the Gulf of Mexico. The interface of APEX and SWAT proved effective for simulating the MARB (and other basins in the conterminous U.S.), which facilitated the use of the more robust APEX options for representing agricultural management and conservation practices. However, adoption of both models resulted in increased complexity, expertise and costs needed to simulate the overall MARB system, which are major limitations of the approach.

6.2.5.2. SWAT
SWAT is comprised of several components (similar to APEX) including generated and/or measured climate inputs, hydrologic balance and routing, soil and stream temperature, crop/plant growth and yield, nutrient/pesticide/bacteria fate, cycling and transport, and land management (Neitsch et al., 2011; Arnold et al., 2012). A simulated watershed is divided into multiple subwatersheds in SWAT, which are then further subdivided into HRUs that consist of homogeneous land use, management, and soil characteristics that are not spatially identified within the respective subwatershed. Flow is generated via multiple pathways at the HRU level including surface, lateral, groundwater and tile drains (if tile drains are present). HRU-level pollutant export can also occur through one or more pathways, depending on the pollutant. Resultant flows and pollutants are aggregated at the subwatershed level and then routed through the stream network to the simulated watershed outlet. A wide range of conservation, NI and LID practices can be simulated in SWAT including interventions described by Waidler et al. (2011) and Marval et al. (2022). Development of additional conservation practice options in SWAT has mostly ceased due to code developmental efforts shifting to SWAT+, but simulation of other practices may still emerge in older SWAT2012 codes including a saturated buffer module currently being investigated (Raj, 2021).

The voluminous SWAT literature spans several thousand studies (CARD, 2022; Upadhyay et al., 2022) and continues to expand daily with applications reported worldwide. SWAT has been applied across the globe to assess an extensive array of water resource problems for areas ranging from < 1 km² to entire continents as documented by CARD (2022) and various general, regional and topical reviews (e.g., Gassman et al. 2007; 2014; 2022; Bressiani et al., 2015; Krysanova and White, 2015, Tan et al., 2019; 2021; Brighenti et al., 2019; Wang et al., 2019; Akoko et al., 2021). SWAT has been applied for major MARB water resource regions including the MRB (Rosenberg et al., 1999; Mehta et al., 2011; 2016; Daggupati et al., 2016; Chen et al., 2021), ORB (Santhi et al., 2008; Panagopoulos et al., 2015a; Demissie et al., 2017; Du et al., 2018; Rajib et al., 2020), Corn Belt region (Kling et al., 2014; Panagopoulos et al., 2015b; 2017); AWRRS (Rosenberg et al., 1999; Santhi et al., 2008; Baskaran et al, 2010; Jager et al., 2014), Lower Mississippi River (Ha et al., 2018; Xu et al., 2019) and UMRB (over 40 studies, see Chen et al., 2020). SWAT served in an integrating role for the previously described CEAP MARB study, in
which APEX was used to generate pollutant loads for agricultural landscapes (White et al., 2014); and, as noted before, the coupled modeling system proved quite effective at assessing the impacts of different conservation practices but requires considerable extra resources to support two different key models. Development of a MARB SWAT+ modeling system has been initiated which will rectify these limitations as discussed below.

6.2.5.3 SWAT+

Most of the SWAT+ primary components and algorithms mirror previous SWAT structure. However, SWAT+ features several key enhancements that introduce considerable simulation flexibility including (Beiger et al., 2017): (1) an additional “landscape unit” (LSU) structure that allows for distinct spatial configurations of floodplain and upland units, and potentially additional LSUs depending on choice of delineation process; (2) configuring one or more HRUs, within a given LSU, that are typically not spatially identified (but can be); and, (3) optional route flow and pollutants between HRUs/LSUs, which allows routing to occur within a subwatershed. SWAT+ also features a modular structure that facilitates potential code modifications, and streamlines standard processes such as inputs and outputs (Bieger et al., 2017). Decision tables are used that provide improved flexibility to set up field management inputs such as planting, harvesting, fertilization and tillage, and other processes including simulation of irrigation and reservoirs (Arnold et al., 2018). New modules have also been developed for SWAT+, including tile drain and groundwater submodels (Bailey et al., 2020a; 2020b; 2021), or are under development (e.g., rice paddy module described by Gassman et al. (2022)).

The development of NAM provides a platform to apply SWAT+ for the entire MARB at refined areal resolutions as small as individual crop fields (Arnold et al., 2021; White et al., 2022; Čerkasova et al., 2022). The NAM system is being configured for the contiguous U.S., which features a robust structure that incorporates 4,880,000 agricultural fields and 2,250,000 non-agricultural HRUs (Arnold et al., 2021). These refined spatial units are further overlayed by 2-, 4-, 6-, 8- and 12-digit watersheds, allowing analyses to be performed at multiple spatial scales. Planting and harvesting, tillage, fertilizer, irrigation and other management practices, coupled with structural conservation practices (Martins et al.,
2021), other conservation practices and subsurface tile drainage are also accounted for on cropland
landscapes (Arnold et al., 2021). The original tile drainage layer described by Arnold et al. (2021) has
since been updated with the improved spatial representation of tile drains reported by Valayamkunnath et
al. (2020) and shown in Figure 6. There are two important limitations regarding the initial configuration
of NAM for the MARB and other U.S. regions (Arnold, 2022): (1) routing is not currently being
simulated within subwatersheds—instead, HRU-level outputs are aggregated to the 12-digit watershed
level for further routing in the stream system; and, (2) riparian LSUs have not currently configured in the
NAM structure. These SWAT+ functions would need to be activated in a future phase of NAM
development if needed for a MARB modeling system.

Calibration of the NAM system has initially focused on the hydrologic balance, streamflow and
crop yield components of the system (e.g., Čerkasova et al., 2022). Uncalibrated nitrate loads estimated in
NAM for the MARB region were used for a GIS model evaluation of NI practices (Schilling et al., 2022).
The nitrate loads were reviewed by the NAM modeling team and other co-authors of this study to ensure
that expected areas of highest loads (e.g., intensely cropped tile drained areas) were replicated by the
modeling system. The GIS model NI practice assessment can be updated in the future with fully
calibrated and validated NAM nitrate loads.

7. Summary of ecohydrological model capabilities to simulate specific NI practices

Exact determination of NI practice simulation capabilities for the models reviewed here (and
many similar models) is challenging, due to a lack of explicit information in model documentation or
published literature. In addition, there is a dearth of literature available that provides comparisons of BMP
simulation capabilities between different models. Xie et al. (2015) compiled a chart comparing
agricultural BMP simulation capabilities between the AGricultural Non-Point Source Pollution (AGNPS)
model (USDA, 2022b; Young et al., 1989; Shresta et al., 2019), Annualized AGRicultural Non-Point
Source Pollution (AnnAGNPS) model (USDA, 2022b; Polyakov et al., 2007; Zhang et al., 2020), and
HSPF and SWAT (Figure 8). Most of the practices included in Figure 8 would not be considered NI
practices except for ponds and perhaps stream channel stabilization (SCS; BC, 2022; GeoSolutions, 2022), sediment basins (MCPA, 2022b; QG, 2022) and grade stabilization structures (GSS; MPCA, 2022a; USDA, 2017c). SWAT was the only model among the four compared that was considered capable of simulating all of the BMPs (Figure 8).

Figure 8. Selected agricultural BMPs which can be simulated by (GSS = Grade stabilization structure; SCS = Stream channel stabilization); black squares indicate that the model can simulate the respective BMP while white squares indicate the opposite (Source: Xie et al., 2015; see Acknowledgments).
<table>
<thead>
<tr>
<th>Model or Modeling system</th>
<th>Land use change</th>
<th>Depress. Wetlands</th>
<th>Farm ponds</th>
<th>Constructed wetlands</th>
<th>RAfs/ WASCOBs</th>
<th>Filter strips</th>
<th>Vegetated ditches</th>
<th>Two-stage ditches</th>
<th>Vegetated buffers</th>
<th>Saturated buffers</th>
<th>Riparian forest buffers</th>
<th>Restored oxbows</th>
<th>Riparian wetlands</th>
<th>Beaver dams</th>
</tr>
</thead>
<tbody>
<tr>
<td>Agro-IHIS / THMB &lt;sup&gt;a&lt;/sup&gt;</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>no</td>
<td>no</td>
<td>no</td>
<td>no</td>
<td>no</td>
<td>no</td>
<td>no</td>
<td>no</td>
<td>no</td>
<td>no</td>
<td>no</td>
</tr>
<tr>
<td>APEX &lt;sup&gt;b&lt;/sup&gt;</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>no</td>
<td>yes</td>
<td>prototype</td>
<td>no</td>
<td>no</td>
<td>no</td>
<td>no</td>
</tr>
<tr>
<td>HSPE &lt;sup&gt;c&lt;/sup&gt;</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>no</td>
<td>NC &lt;sup&gt;d&lt;/sup&gt;</td>
<td>yes</td>
<td>NC &lt;sup&gt;d&lt;/sup&gt;</td>
<td>no</td>
<td>NC &lt;sup&gt;d&lt;/sup&gt;</td>
<td>no</td>
<td>NC</td>
<td>no</td>
<td>no</td>
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</tr>
<tr>
<td>HEC-HMS &lt;sup&gt;e&lt;/sup&gt;</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>no</td>
<td>no</td>
<td>yes</td>
<td>no</td>
<td>no</td>
<td>no</td>
<td>no</td>
<td>NC &lt;sup&gt;d&lt;/sup&gt;</td>
<td>no</td>
<td>no</td>
<td>no</td>
</tr>
<tr>
<td>MIKE SHE &lt;sup&gt;f&lt;/sup&gt;</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>NC &lt;sup&gt;d&lt;/sup&gt;</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>no</td>
<td>yes</td>
<td>no</td>
<td>NC &lt;sup&gt;d&lt;/sup&gt;</td>
<td>NC &lt;sup&gt;d&lt;/sup&gt;</td>
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</tr>
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<td>no</td>
<td>no</td>
<td>no</td>
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<td>no</td>
<td>no</td>
<td>no</td>
<td>no</td>
<td>no</td>
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</tr>
<tr>
<td>SWAT &lt;sup&gt;h&lt;/sup&gt;</td>
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<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>no</td>
<td>prototype</td>
<td>no</td>
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<td>no</td>
</tr>
<tr>
<td>SWAT &lt;sup&gt;+&lt;/sup&gt;</td>
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<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>prototype</td>
<td>partial</td>
<td>partial</td>
<td>partial</td>
<td>partial</td>
<td>partial</td>
</tr>
</tbody>
</table>

<sup>a</sup>Sources: Ferin et al. (2021); Donner and Kucharik (2008); VanLoocke (2022).
<sup>b</sup>Sources: Gassman et al. (2010); Waidler (2011); Steglich et al. (2019); Jeong (2022)
<sup>c</sup>Sources: Palace et al. (1998); Bicknell et al. (2002); Duda et al. (2012); Xie et al. (2015)
<sup>d</sup>NC= not clear.
<sup>e</sup>Source: USACE (2022a)
<sup>f</sup>Sources: Thompson et al. (2004); Graham and Butts (2005); DHI (2022a)
<sup>g</sup>Sources: Smith et al. (1997); Alexander et al. (2008; 2012); Robertson and Saad (2021)
<sup>h</sup>Sources: Arnold et al. (2022); Gassman et al. (2007; 2022); Waidler et al. (2011); Xie et al. (2015)
The intent of Table 9 is identifying the ability of each model to “physically represent” the effects of each practice, as opposed to relying on more indirect or proxy methods. Physically based modeling is sometimes referred to as deterministic (Croton and Barry, 2001; Jeon et al., 2018; Maleki Tirabadi et al, 2022) or mechanistic (Menzies Pluer et al., 2019; Tang et al., 2019; Bhanja and Wang, 2021) modeling, although the exact meaning of these terms varies widely across disciplines and model structures. For example, SWAT is typically described as a physically based model (Neitsch et al., 2002; 2011), although the structure of the model is more accurately described as a fusion of physical and empirical functions (Gassman and Wang, 2014; Schilling et al., 2019) or as a partly-deterministic model (Camargos et al., 2018). Multiple empirical routines are used in SWAT including: (1) the runoff curve number (RCN) method (Neitsch et al., 2002; 2011; Williams et al., 2012) for partitioning rainfall between surface runoff and soil infiltration; (2) the Modified Universal Soil Loss Equation (MUSLE; Neitsch et al., 2002; 2011; Williams and Berndt, 1977), which estimates water erosion in response to storm events; and, (3) the representation of terraces and contouring on cropland landscapes based on relationships reported by Wischmeier and Smith (1978). Shao et al. (2013) introduced new modules into a modified version of SWAT2009 that support a considerably enhanced physically based representation of terraces, relative to the method used in the standard SWAT codes. However, these improved modules are currently not accessible by the broader SWAT user community, which reflects the reality of many other modified SWAT models reported in the literature (CARD, 2022).

Proxy methods are a common approach for representing various BMPs including NI practices for some of the models listed in Table 9. HSPF features a BMP evaluation (BMPRAC) module that represents the impacts of a specific BMP by applying “removal fractions” to each pollutant constituent that is being simulated (Bicknell et al., 1996; 2002), which are based on research findings for different environmental conditions (Xie et al., 2015). Xie et al. (2015) note that this approach can result in very inaccurate results; however, they further state that HSPF can simulate some BMPs in a more direct manner. SPARROW can effectively only be used to represent BMPs including NI practices in an aggregated proxy approach as previously described for studies conducted by McLellen et al. (2015), Garcia et al. (2016) and Miller et al. (2020). This weakness of SPARROW could also serve as a strength, in the context of supporting screening assessments of NI practice impacts in the MARB region by interfacing field- or watershed-scale NI practice impacts generated by a different model with SPARROW. Christopher et al. (2017) provide an additional example of a proxy simulation approach, in which the impacts of two-stage ditches (Figures 5k and 7) on nitrate and TP removal were evaluated in SWAT for the 2,736 km² River Raisin (located in southeast Michigan), on the basis of empirical data collected for two-stage ditches at 19 sites in Indiana, Michigan and Ohio.
Overall, SWAT+ emerges as the strongest option for evaluating NI practices in the MARB based on the model’s ability to represent the majority of NI practices listed in Table 9 and the developer’s interest in building capacity into the model to represent the full suite of NI practices (Arnold, 2022). The adoption of SWAT+ is especially viable in the context of the ongoing development of the NAM system (Arnold et al., 2021; White et al., 2022; Čerkasova et al., 2022), which will provide refined spatial and functional modeling capabilities for the entire MARB. SWAT+ also builds on the previous foundation of SWAT applications including extensive evaluation of agricultural BMP and cropping system practices. SWAT+ offers enhanced spatial representation of NI practice locations beyond the original SWAT capabilities, which would support configurations of NI practices on specific riparian, hillslope and other landscapes. The SWAT+ platform also builds on other SWAT modeling legacies including: (1) the ability of users to modify the source code to support specific research investigations; and, (2) interfacing SWAT with dozens of other models including multiple studies that report interfaces between SWAT and HEC-RAS (Table 10). Adoption of SWAT+ also provides the possibility of incorporating relevant modifications performed by previous SWAT users such as the terrace algorithms developed by Shao et al. (2013) and improved representation of wetland and/or pothole functions (e.g., Evenson et al., 2018; Qi et al., 2019; Muhammad et al., 2019; Rajib et al., 2020).

Conclusion

Extraordinary interventions are needed to overcome the extensive flooding and water quality problems that continue to manifest within the MARB. The MR&T levee system has resulted in successful flood control in recent decades in the LMRB. However, hundreds of levees have failed in other MARB subregions as a result of recent floods, and the extensive LMRB levee system and other factors are linked to the considerable loss of coastal wetlands and habitat areas in southern Louisiana. Water quality degradation continues essentially unabated across much of the MARB, especially in the Corn Belt region. Massive investment in a wide range of cropland conservation practices and other interventions has unfortunately yielded only minor improvement at best in MARB stream system water quality.

Widespread adoption of the NI practices described in this study (Table 5; Figure 5) provides the potential to establish greater progress in mitigating both flood impacts and improving in-stream water quality in MARB waterways, especially if those practices are embraced in a more aggressive manner within federal and state conservation practice strategies. NI interventions can be implemented within multiple landscapes in a given agricultural stream system including upland crop fields and pastures, edge-of-field locations, drainage ditches, and riparian areas and floodplains. The optimum configuration of NI practices will vary between different watersheds depending on topography, cropping systems, non-agricultural land use, climate and other factors. The GIS modeling performed by Schilling et al. (2022)
Table 10. Studies describing applications of interfaces between SWAT and HEC-RAS

<table>
<thead>
<tr>
<th>Source</th>
<th>Country (subregion)</th>
<th>Watershed (km²)</th>
<th>Focus of study</th>
</tr>
</thead>
<tbody>
<tr>
<td>Baldan et al. (2021)</td>
<td>Austria (north central)</td>
<td>Aist River (650)</td>
<td>Impact of natural water retention measures on watershed hydrology, sediment loss, reach hydraulics and fine bed material deposits</td>
</tr>
<tr>
<td>Chinnasamy et al. (2018)</td>
<td>India (States of Uttar Pradesh &amp; Uttarakhand)</td>
<td>Ramganga River (19,000)</td>
<td>Impacts of managed aquifer recharge scenarios on groundwater recharge by interfacing SWAT, HEC-RAS and MODFLOW</td>
</tr>
<tr>
<td>Cotugno et al. (2021)</td>
<td>Sierra Leone (Freetown metropolitan area)</td>
<td>Seven urban rivers in Freetown area</td>
<td>Estimate peak discharge and flood extents of urban rivers in ungauged watersheds for current and future land use scenarios</td>
</tr>
<tr>
<td>Gallay et al. (2021)</td>
<td>Slovakia (central)</td>
<td>Čierny Hron River (2.92)</td>
<td>Assessed extent of floods as a function of different rainfall levels and current land use versus total conversion to forest</td>
</tr>
<tr>
<td>Jähnig et al. (2012)</td>
<td>Germany (northern)</td>
<td>Kielstau River (50)</td>
<td>Linked hydrological, hydraulic and species distribution models to predict the habitat suitability for benthic invertebrates in a riverine system.</td>
</tr>
<tr>
<td>Javaheri &amp; Babbar-Sebens (2014)</td>
<td>United States (northwest of Indianapolis, Indiana)</td>
<td>Eagle Creek (419)</td>
<td>Estimated peak flows, flood inundation areas and velocity maps in response to wetland implementation designed for reducing flood hazards</td>
</tr>
<tr>
<td>Jha &amp; Afreen (2020)</td>
<td>United States (Kansas City, Missouri area)</td>
<td>Blue River (658.9)</td>
<td>Identified flood risk zones and threatened critical infrastructure in flood zones through inundation mapping</td>
</tr>
<tr>
<td>Lôi et al. (2013)</td>
<td>Vietnam (Quảng Nam province)</td>
<td>Vu Gia River (10,350)</td>
<td>Determined location of areas with high, moderate and low flood risk in downstream floodplains, and provides early flood warning system</td>
</tr>
<tr>
<td>Marcinkowski &amp; Miroslaw-Świątek (2020)</td>
<td>Poland (northeast)</td>
<td>Upper Narew River (4,231)</td>
<td>Assess the impact of projected climate change on average flow conditions in connecting stream channels in the river system</td>
</tr>
<tr>
<td>Peng et al. (2017)</td>
<td>China (Huailing County, Shaanxi Province)</td>
<td>Jianzhuangchuan River (342.78)</td>
<td>Estimation of the range of debris flow deposition</td>
</tr>
<tr>
<td>Song et al. (2014)</td>
<td>Germany (State of Schleswig-Holstein)</td>
<td>Upper Stör River (468)</td>
<td>Analysis of the effects of river power and flood power</td>
</tr>
<tr>
<td>Song et al. (2015)</td>
<td>Germany (State of Schleswig-Holstein)</td>
<td>Upper Stör River (468)</td>
<td>Conducted quantification and comparison of both channelized and floodplain sediment processes</td>
</tr>
<tr>
<td>Tadesse &amp; Dai (2019)</td>
<td>Ethiopia (west central)</td>
<td>Awash River</td>
<td>Estimated sediment load reaching the Koka Dam Reservoir</td>
</tr>
<tr>
<td>Taraky et al. (2021)</td>
<td>Afghanistan &amp; Pakistan</td>
<td>Kabul River (86,870)</td>
<td>Investigated flood risk in response to 30 climate and dam management scenarios to assess opportunities for transboundary water management</td>
</tr>
<tr>
<td>Tiepolo et al. (2021)</td>
<td>Niger (southwest)</td>
<td>Five rural settlements</td>
<td>Assessed the risk of pluvial floods in five rural settlements</td>
</tr>
<tr>
<td>Xue et al. (2021)</td>
<td>China (Shandong Province)</td>
<td>Xiaoqing River</td>
<td>Simulated ammonia nitrogen (NH₄-N) and chemical oxygen demand (COD) concentrations both low and high flows</td>
</tr>
</tbody>
</table>
provides preliminary insights regarding ideal placement of WASCOBs, depressional wetlands, nitrate-removal wetlands, riparian buffers, and floodplains levees and land use change in MARB subregions.

Dozens of flood and ecohydrological models have been created worldwide in recent decades that have been designed for different scales, conditions and purposes. Several of the more widely used and potentially most applicable models for MARB flood and/or water quality analyses were reviewed here (Tables 6 and 7). Some of the reviewed models are characterized by relatively weak capabilities regarding representation of NI practices (e.g., HEC-HMS and AGRO-IBIS). HEC-HMS coupled with HEC-RAS (USACE, 2022b) would enable accounting of some riparian NI practice effects and provide strong assessment of flood analyses (Table 6), but would be limited by the inability to simulate upland, edge-of-field and other NI practices (Table 5; Figure 5). The MIKE system of models (DHI, 2022e) could potentially be used to depict many of the NI practices (Table 5; Figure 5) coupled with in-depth flood simulations. However, the full range of NI practice representation is not clear in available literature (e.g., Table 9) and the proprietary status of the MIKE software is a major constraint when considering application for a MARB modeling system.

The APEX, SWAT and SWAT+ models developed at the co-located USDA and Texas A&M AgriLife laboratories in Temple, TX (Williams et al., 2008) show the strongest overall abilities to accurately replicate the NI practice impacts (Table 9). The SWAT+ model is particularly advantageous, due to the incorporation of previous APEX and SWAT modeling strengths and the current development of the NAM system (Arnold et al., 2021; White et al., 2022; Čerkasova et al., 2022). The utility of SWAT+ is further underscored by the relative ease of interfacing it with other models, such as previous studies that report model cascades between SWAT and HEC-RAS (Table 10). This attribute provides the potential to expand an adapted NAM modeling system to not only capture MARB NI practice flood and water quality impacts, but also account for more detailed flood analyses if needed. It is recommended that the SWAT+/NAM modeling system be adopted for future MARB NI practice simulations. Further development will be required to implement the full suite of NI practices discussed here (Tables 5, 9; Figure 5). This can be achieved via close collaboration with the SWAT+/NAM developers and the required investment needed to incorporate physical representation of the full suite of NI practices in future MARB applications.

Acknowledgments
The following individuals or organizations are acknowledged for granting permission to use specific figures or photos: (1) Figure 4: Natalie Snider. 2022. Personal communication. Environmental Defense Fund, Washington, DC, (2) Figures 5a, 5b, 5c, 5d, 5e, 5g, 5i, 5j, 5k, 5l and 5m adapted from Suttles et al. (2021), which were published under the terms and conditions of the Creative Commons Attribution license (http://creativecommons.org/licenses/by/4.0/), (3) Figure 5f (first photo on the left), courtesy of Kurt Blume, USDA-NRCS (Benning and Craft, 2018), (4) Figure 5f (photo in the middle), Morrison SWCD (https://morrisonswcd.org/about-us/contact-us/); (5) Figure 5n (both photos), Karen Wilke, Director, Iowa Freshwater Specialist & Boone River Project. Nature Conservancy, (6) Figure 5o (left photo), Beavers: Wetlands & Wildlife, Dolgeville, NY; (6) Figure 5n (right photo), Sarah Mowry, Outreach Director, Deschutes Land Trust, Bend, OR; (7) Figure 7, Christianson et al., 2016; (8) Figure 8, which was published under the terms and conditions of the Creative Commons Attribution license (http://creativecommons.org/licenses/by/4.0/).

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