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LUMINATE: linking agricultural land use, local water quality and Gulf of Mexico hypoxia

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Abstract

In this paper, we discuss the importance of developing integrated assessment models to support the design and implementation of policies to address water quality problems associated with agricultural pollution. We describe a new modelling system, LUMINATE, which links land use decisions made at the field scale in the Upper Mississippi, Ohio and Tennessee Basins through both environmental and hydrological components to downstream water quality effects and hypoxia in the Gulf of Mexico. This modelling system can be used to analyse detailed policy scenarios identifying the costs of the policies and their resulting benefits for improved local and regional water quality. We demonstrate the model's capabilities with a simple scenario where cover crops are incentivised with green payments over a large expanse of the watershed.

Keywords: integrate modelling, land use change, Gulf of Mexico Hypoxia, LUMINATE, water quality

JEL classification: Q51, Q52, Q57

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1. Introduction

Water quality problems remain ubiquitous around the world and in many locations are growing in severity. Nutrient enrichment of freshwater and coastal areas from a variety of human activities including agricultural runoff, sewage and industrial sources is the most prevalent type of water quality problem.[1] The delivery of nutrients, primarily nitrogen (N) and phosphorus (P), into waterways accelerates normal plant growth in a process called eutrophication. Eutrophication of water bodies results in a range of lost ecosystem services (Tilman *et al.*, 2001; Chislock *et al.*, 2013) whose values are increasingly understood and monetised (e.g. Knowler, Barbier and Strand, 2001; Massey, Newbold and Gentner, 2006; Polasky and Segerson, 2009; Barbier, 2012; Huang *et al.*, 2012). For example, eutrophication problems in local freshwater systems such as lakes and streams can include excess algae growth, changes in water clarity, odour and toxic algae blooms. In addition to these local effects, excess nutrients cause detrimental effects far downstream in the coastal areas, including the creation of oxygen-depleted (hypoxic) waters. In the last 50 years, hypoxia became a global problem with >415 areas across the world had been or are experiencing its symptoms. Among these areas, 169 have been identified as hypoxic zones, 233 as areas of concerns and 13 areas as recovered (Selman *et al.*, 2008).

Management of excess nutrient input into aquatic systems is a 'wicked' problem, incorporating a range of scientific and policy challenges (Turner, 2000; Elofsson, Folmer and Gren, 2003; Smith and Schindler, 2009; Chen and Hong, 2012; Rabotyagov *et al.*, 2014). In addition, the understanding of eutrophication impacts, its costs and benefits are both increasing and highly dependent on the particular system under study (Chen and Hong, 2012). A common theme and challenge is one of integration, both across space (from terrestrial ecosystems to freshwater and marine aquatic systems) and across disciplines.[2]

The impetus for integrated modelling has often been provided by public entities looking to develop policy responses to these problems, with the Baltic Sea Action Plan,[3] the Black Sea Ecosystem Recovery Project,[4] the environmental policies in the Mediterranean Sea (Karydis and Kitsiou, 2012), Gulf Hypoxia Action Plan (USEPA, 2001, 2008) and the Chesapeake Bay Program (USEPA, 2009) serving as notable examples in the developed world, with subsequent efforts in the developing countries to extend coastal management to upstream watersheds (Chen and Hong, 2012).[5] In almost all cases, explicit

1 http://www.un.org/waterforlifedecade/quality.shtml.

2 While the focus of this paper is agriculture and water quality, the recognition for better integration across ecosystems and disciplines has produced fruitful collaborations between economists and other scientists in other contexts as well (see, for example, the work of the Natural Capital Project: http://www.naturalcapitalproject.org/).

3 www.helcom.fi.

4 http://www.elmed-rostov.ru/Projects/BSERPper cent20Visitcard/.

5 http://water.epa.gov/type/watersheds/named/msbasin/upload/2008_8_28_msbasin_ghap2008_update082608.pdf.

recognition of the linkage between upstream landscapes and marine systems leads to the need to carefully consider the role of agricultural landscapes. This is especially true in the European Union and the United States, where agricultural sources are the most significant contributors to eutrophication;[6] however, wastewater from sewage and industry are viewed as the main pollutant contributors in Latin America, Asia and Africa (Selman *et al.*, 2008).

In the European Union, the Baltic Sea and the Black Sea ecosystems are two examples of agriculture impacting eutrophication. The increased agricultural activity that followed the inclusion of the Baltic countries in the EU also increased the amount of nutrients delivered to the Baltic Sea hence increasing the size of the eutrophic zone. On the other hand, the Black Sea ecosystem is currently identified as a system in recovery due to a decreased agricultural activity that followed the collapse of the former Soviet Union (Selman *et al.*, 2008, http://blacksea-education.ru/e2-1.shtml).

A striking example of the nexus between large-scale eutrophication problems with both detrimental local water quality effects and major downstream problems and agriculture occurs in the Corn Belt region in the central United States. The Upper Mississippi River Basin (UMRB) and Ohio-Tennessee River Basin (OTRB), which comprise most of the Corn Belt, each exhibit significant local water quality problems (USEPA, 2014), and contribute a significant share of nutrient loadings to the Northern Gulf of Mexico where excess nutrients result in a second-largest annual hypoxic zone in the world (USEPA-SAB, 2007). Since 1985, Rabalais and others have documented the annual extent of the seasonal hypoxic zone that forms in the Northern Gulf of Mexico (Turner, Rabalais and Justic, 2006; Rabalais *et al.*, 2010). Figure 1 shows the size and location of the watersheds in relation to the hypoxic zone.

Agricultural production has been a major land use activity for nearly 200 years in much of the UMRB and OTRB. Currently, the majority of US agricultural exports as well as the majority of the world's feed grain and soybean exports are produced in these two major Corn Belt watersheds. The current agricultural land use patterns in the UMRB and OTRB have resulted from the simultaneous decisions of over half a million individual landowners and agricultural producers over decades. These decisions have been influenced by crop profitability dictated in turn by economic conditions including commodity and energy policy. When scaled up to the basin level, the result has been a highly productive agricultural region, but with significant degradation to water quality and other ecosystem services.

Federal and state governments are actively looking for solutions and a number of conservation actions (abatement options) that can reduce nutrient losses from agriculture are available. However, to be effective (and even more so in order to be cost-effective) the cumulative and non-linear effects

6 For example, for the Baltic Sea, it has been found that the desired conditions cannot be achieved without reducing diffuse (non-point) sources (http://meeting.helcom.fi/c/document_library/get_file?p_l_id=16324&folderId=2062738&name=DLFE-52270.pdf, p. 15), while in the United States, the vast majority of nitrogen impairment is due to non-point sources (Ribaudo *et al.*, 2008).

Fig. 1. Location of the URMB and the OTRB within the Mississippi-Atchafalaya River Basin (MARB).

that occur, when abatement activities are adopted on individual farms, need to be accounted for as they are scaled up to the watershed level. For example, if multiple contiguous landowners install conservation practices along a river, this may have significantly more impact on downstream water quality than if the same number install those practices at non-contiguous locations. Likewise, nutrient loadings in some locations in the watershed have a much lower possibility of being transported to the Gulf than others depending on the features of the landscape and hydrology. It is further important to realise that nutrient fate and transport is not, in general, an exogenous physical process, but is in fact endogenous to water quality policy (for example, when a reconstructed wetland effectively traps and removes nutrients from upstream sources) (Randhir, Lee and Engel, 2000; Khanna *et al.*, 2003; Rabotyagov, Valcu and Kling, 2013; Shortle and Horan, 2013). Equally important is to recognise that the costs of adopting abatement actions vary by location as a function of soil characteristics, farming practices, weather patterns and a host of other location-specific features. An integrated modelling system that incorporates these non-linearities, scale effects and cost variability will be essential for good public policy.

In this study, we describe the development and implementation of an integrated assessment model for this large region of the central United States.

Our coupled natural and human system captures the key non-linearities in spatial processes associated with scaling up the impacts of individual decision-making at the agricultural field scale to the impact on local watersheds and the downstream hypoxic conditions. We begin with a brief literature review of integrated land use and water quality assessments for our study region. After describing an overview of the modelling system, we describe the components in more detail. In particular, we describe the development of an ecological production function relating nutrient loadings to the areal extent of Gulf hypoxia. We then present the results of a simple, but detailed, land use scenario where the effects of large-scale placement of cover crops across the UMRB and OTRB are simulated. This simple scenario is used to demonstrate the potential for the modelling system to inform policy as the simulated output includes predictions of the change in water quality throughout the large watersheds (including the local and downstream impacts), the impact on hypoxic zone conditions in the northern Gulf of Mexico, as well as the costs of adopting these practices. The results presented here highlight the value of such a modelling system for policy design and implementation. In the final section, next steps and additional discussion of the potential role of integrated assessment models are described.

2. Literature review

In general terms, the literature related to Northern Gulf of Mexico hypoxia and water quality problems in the UMRB and OTRB fall into two broad categories: (i) studies that focus entirely on the biophysical aspects of the processes contributing to the Northern Gulf of Mexico hypoxia and (ii) studies that attempt to integrate the socio-economic and the biophysical aspects of the water quality processes. In the first category, there are studies that focus on the modelling of Northern Gulf of Mexico hypoxia, but do not link to specific land use management strategies (Justić, Rabalais and Turner, 2002; Scavia *et al.*, 2003; Turner, Rabalais and Justic, 2006; Justíc *et al.*, 2007), studies that focus on estimating the nutrient loadings that ultimately reach the Gulf, but cannot perform scenarios relating specific land use changes to the hypoxic zone (Burkart and James, 1999; Donner *et al.*, 2002; Booth and Campbell, 2007; Alexander *et al.*, 2007; Broussard and Turner, 2009), and studies that estimate the nutrient loading and at the same time consider the impact of different land use scenarios (Johnes and Hearhwaite,1997; White *et al.*, 2014).

The 'integrating' modelling literature considers the impact of different land use scenarios on water quality by generally focusing on smaller watersheds rather than a large scale. These studies, although incomplete in one or more dimensions and not providing any direct linkage to Gulf of Mexico hypoxia, offer valuable insights and alternative policy options that have the potential to reduce the size of the hypoxic zone. In some cases, the complex relationship between runoff leaving the field and its impact on the overall water quality is simplified by estimating different proxy alternatives, e.g. delivery coefficients (e.g. Wainger *et al.*, 2013) or point systems (Rabotyagov, Valcu and Kling, 2013). The ultimate goal of these studies is to identify the cost-effective land use changes and to design the optimal economic incentives to reduce the runoff in the landscapes dominated by agriculture.

Similarly, within the 'integrating' literature, we can further identify several categories: studies that provide extensive reviews of the programmes that have been implemented, studies that compare the cost-effectiveness of different land use scenarios, studies that consider the optimal placement of the conservation practices or studies that consider the impact of different policies (i.e. biofuel crop production) on water quality.

Ribaudo *et al.* (2001) compare the cost-effectiveness of a fertiliser standard and wetland restorations as N reduction methods in the Mississippi basin. Doering *et al.* (2001) provide an economic analysis for reducing N on a large scale, but they rely on aggregate data that is not watershed based. Greenhalgh and Sauer (2003) present a comprehensive assessment of different cost-effective policy options to alleviate hypoxia, but with a focus on the local water quality issues. Wu and Tanaka (2005) estimate the costs of adopting three different conservation policies and a tax on the use of fertiliser in the UMRB. Secchi *et al.* (2007) outline a methodology to assess the economics costs and water quality benefits by considering a number of hypothetical land use scenarios for 13 watersheds in Iowa, one of the states with the highest contributions to the hypoxic zone. Jenkins *et al.* (2010) evaluate the benefits of wetland restoration in the southern part of the Mississippi corridor.

Rabotyagov *et al.* (2010) integrate a watershed based model with cost data and develop, via evolutionary algorithms, a trade-off frontier for the UMRB specifying the least cost of achieving different level of N and P reductions. For the same large watershed, Secchi *et al.* (2011) assess the water quality changes in the context of an increase in the corn acreage due to an ethanol policy.

3. Description of study regions

The UMRB drains 492,000 km^2 including large parts of Illinois, Iowa, Minnesota, Missouri and Wisconsin from Lake Itasca in Minnesota to just north of Cairo, Illinois, above the confluence with the Ohio River (Figure 1). The area is referred to as Region 07 by the US Geological Survey (USGS) at a '2-digit watershed' scale and is further composed of 131 USGS '8-digit watersheds' and 5,729 USGS '12-digit subbasins' (USGS, 2012a, 2012b). The average annual UMRB rainfall within the last four decades was 900 mm, ranging from 600 to 1,200 mm across the basin with values generally decreasing from east to west. Cropland consists mainly of corn–soybean rotations and occupies almost 50 per cent of the total UMRB area, with the majority of the land area consisting of slopes <5 per cent.

The OTRB covers $\sim$528,000 km^2 including large portions of seven states as shown in Figure 1. The OTRB is composed of 152 8-digit watersheds and 6,350 12-digit subwatersheds. The OTRB region received nearly 1,200 mm/year of average annual rainfall during the last 40 years. About half of the land cover in this basin is forested, with the majority of the remaining area planted in crops (20 per cent) or managed as permanent pasture/hay (15 per cent). Corn, soybean and wheat are the major crops grown (Santhi *et al.*, 2014). Compared with the UMRB, the OTRB's slopes are much steeper, especially in the forested Tennessee basin; however, cropland is mostly concentrated in the northwestern part of the region, which is characterised by mild topography.

According to USEPA SAB (2007), 43 per cent of the nitrate-N load and 26 per cent of the total phosphorus (TP) load delivered to the Gulf of Mexico came from the UMRB during 2001–2005, in spite of the fact that the UMRB drains only 15 per cent of the total Gulf of Mexico drainage area. The mean annual in-stream UMRB total nitrogen (nitrate-N plus organic nitrogen) and TP loads measured at Grafton, IL (Figure 3) are 500,000 and 30,000 t/year, respectively (USGS, 2013). Likewise, the OTRB exports roughly 500,000 t of N annually to the main Mississippi River channel, with close to 67 per cent in the form of nitrate-N. The annual average phosphorus loads measured at the OTRB outlet equal 48,000 t/year (USGS, 2013).

4. Overview of the LUMINATE system

Current water quality conditions in the UMRB and OTRB and the delivery of nutrients from these watersheds to the Gulf of Mexico results from the decisions of thousands of agricultural producers concerning crop choice, conservation practices and management choices. These decisions depend in turn on many factors including soil type, climate, expected market prices and policy variables such as subsidy levels. Figure 2 describes the basic components and model

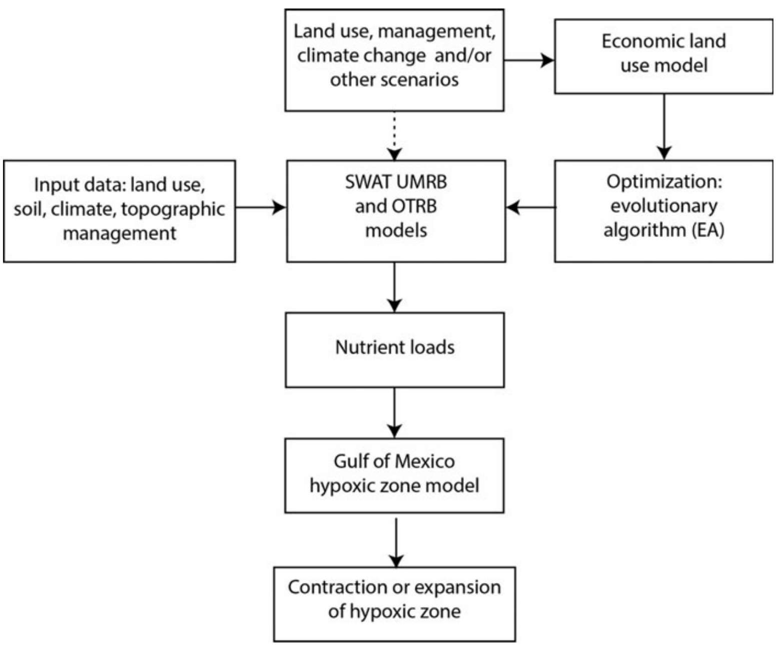

Fig. 2. Modelling schematic for LUMINATE.

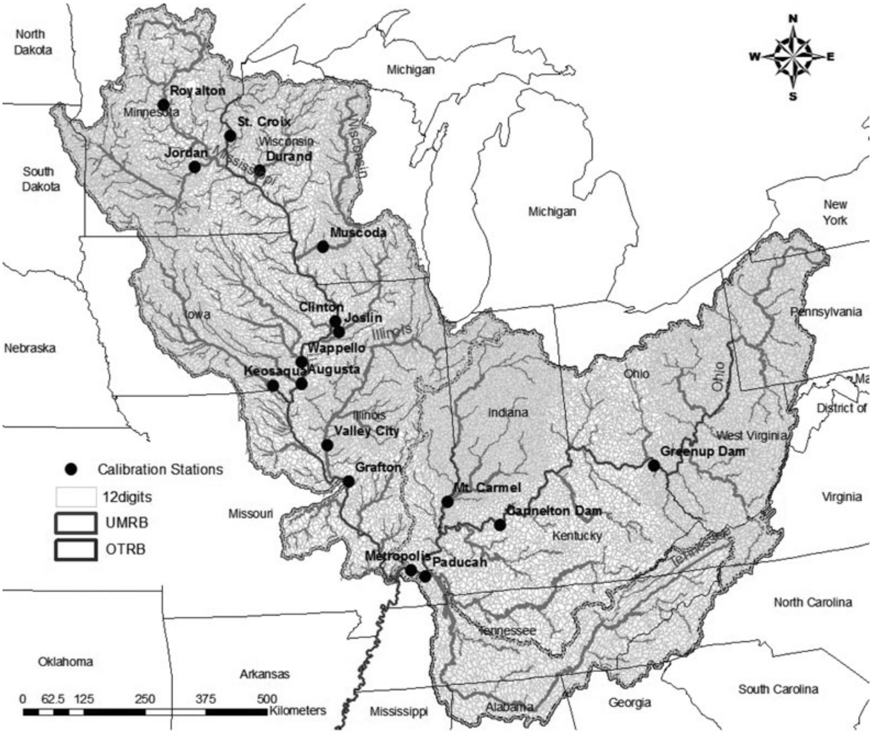

Fig. 3. Locations of the UMRB and OTRB within the United States, the main tributaries and monitoring site locations within each study region.

integration of the LUMINATE modelling system which is designed to capture the key features of this complex interaction between human decision-making, land uses across this broad landscape, and the resulting effects on water quality in the upstream watersheds as well as the Gulf of Mexico. As noted in the top-centre of the schematic, the primary purpose of the modelling system is to be able to undertake scenarios associated with changes in policy variables, weather conditions and market forces. The two key 'upstream' water quality models of this integrated system are identified in the box labelled 'SWAT UMRB and OTRB Models', which are fed and parameterised by a range of data inputs. The Soil and Water Assessment Tool (SWAT), a watershed-scale water quality model (Arnold *et al.*, 1998; Arnold and Forher, 2005; Gassman *et al.*, 2007), forms the core of the watershed water quality component of the modelling system proposed here and was used to construct the specific UMRB and OTRB models. As described below, the SWAT model of the region was developed to simulate a greater range of cropping systems and management practices.

Data and modelling representing economic drivers of land use are used in conjunction with scenario design to determine landscape configurations which are then fed into the SWAT modelling system. The output of this modelling is a spatially explicit prediction of nutrient loads and other water quality indicators throughout the basin. The final key model component of the overall modelling system is the hypoxic zone model, represented in the lower central box and which is described below.

An important feature of the modelling system built here is the ability to interface the model with an optimisation heuristic (evolutionary algorithm) for scenarios in which optimal landscape configurations to achieve a given environmental (or cost) goal are desired. While not necessary for running simple scenarios (such as the one presented in this paper), evolutionary algorithms are powerful tools for identifying and analysing trade-offs across a large range of cost and environmental objectives (see Vrugt and Robinson, 2007; Rabotyagov *et al.*, 2010) while employing the full suite of biophysical models, allowing for consistency of assumptions and a full coupling of cost-effective policy and natural science modelling. Regardless of whether the evolutionary algorithm is employed, the modelling system is also useful in evaluating specific policy scenarios.

Previous SWAT UMRB and OTRB applications relied on delineation approaches in which the subwatershed boundaries were aligned with 8-digit watershed boundaries (e.g. Rabotyagov *et al.*, 2010; Srinivasan, Zhang and Arnold, 2010; Secchi *et al.*, 2011; Santhi *et al.*, 2014). However, a greatly refined delineation scheme has been incorporated into the UMRB and OTRB SWAT models used in LUMINATE, which consist of using subwatershed boundaries that are coincident with the much smaller USGS 12-digit watersheds. This detailed subwatershed schematisation allows for improved linkages to climatic inputs, better representation of stream routing processes (and thus the non-linearities and potential endogeneities in nutrient fate and transport), and improved targeting of management practices and cropping systems on agricultural landscapes.

5. SWAT model description and water quality modelling system

SWAT is a conceptual, long-term continuous watershed-scale simulation model that is usually executed on a daily time step. Key components of the model include hydrology, plant growth, sediment loss and transport, nutrient transport and transformation, pesticide fate and transport, and effects of management practices. Typical applications of SWAT involve delineation of a watershed into subwatersheds, which are then further subdivided into subareas consisting of homogeneous land use, soil types and slope classes that are called hydrologic response units (HRUs). However, a 'dominant HRU approach' can also be used in which just the dominant land use, soil, land topography are used to define the complete characteristics of a given subwatershed, resulting in the subwatershed being equivalent to a single HRU. Flow and pollutant losses are generated at the HRU level in SWAT, aggregated to the subwatershed level and then routed through the stream system to the simulated watershed outlet. SWAT has been successfully applied in extensive applications worldwide for a wide range of watershed scales and environmental conditions (Gassman *et al.*, 2007; Tuppad *et al.*, 2011; Douglas-Mankin *et al.*, 2013; Gassman, Sadeghi and Srinivasan, 2014).

Topographic, soil, land use, climate, management and other key data are required for assembling and executing a SWAT simulation (Table 1). These data layers were assembled for the UMRB and OTRB models, and then overlaid on the subwatersheds for both regions, using the ArcGIS (version 10.1) SWAT (ArcSWAT) interface (SWAT, 2013). The dominant HRU approach described above was used for both regions such that each 12-digit subwatershed was equivalent to an HRU. Topographic characteristics such as slope and slope

Table 1. Key data layers incorporated in the UMRB and OTRB SWAT models

Data layer	Description of data layer	Primary data sources
Daily climate	Daily precipitation and maximum and minimum temperature; other data generated in model	NCDC-NOAA (2012)
Soil map/layer data	1 : 250,000 soil map; pertinent soil layer attributes included for each soil type	USDA-NRCS (1994; 2012b)
Major dams/ reservoirs	Key reservoirs on main channels of the Ohio and Mississippi Rivers and major tributaries	USACE (2012)
Topographic	30 m DEM data used to characterise slopes and slope lengths	USGS (2006)
Land use	Assignment of crop rotations or other landuse to each subwatershed; dominant rotations were 2-year sequences of corn and soybean	USDA-NASS (2012)
Point sources	N and P discharged from thousands of waste treatment plants and other point sources across the two study regions	Maupin and Ivahnenko (2011) and Robertson (2013)
Subsurface tile drainage	Installed at assumed depth of 1.2 m in poorly drained and relatively flat soils (<2 per cent slope)	Sugg (2007) and Neitsch *et al.* (2011)
Tillage practices	No till, mulch till, reduced till and conventional till practices represented as a function of tillage passes and residue cover and other parameters	Baker (2011) and Neitsch *et al.* (2011)
Fertiliser and manure	Nitrogen and phosphorus rates applied in inorganic fertiliser and manure; average rates used for landscapes located within each state	IPNI (2010)
Other conservation practices	Proxy approach used to represent terraces, contouring and other practices as a function of slope and slope length	USDA-NRCS (2012a), Arabi *et al.* (2008) and Neitsch *et al.* (2011)

length were incorporated for each subwatershed using a 30 m digital elevation model (DEM) USGS 2013). Crop rotations and other land use were incorporated by utilising 3 years (2008–2010) of the field-level USDA-NASS Cropland Data Layer datasets (USDA-NASS, 2012) with the 2001 National Land Cover Data (USGS, 2012b), similar to the approach used by Srinivasan, Zhang and Arnold (2010). Dominant soil types and soil layer characteristics were determined by using 1:250,000 State Soil Geographic (STATSGO) soil data (USDA-NRCS, 2012b). Major reservoirs are accounted for in both models based on data obtained from USACE (2012).

Management practices accounted for in the SWAT models (Table 1) included subsurface tile drains, tillage practices, fertiliser and manure application rates, and indirect accounting of other conservation practices such as terraces and contouring due to a lack of spatial estimates of such practices. The assignment of subsurface drainage tiles, which are key conduits of nitrate to surface waters, to specific subwatersheds was based on spatial distributions estimated by Sugg (2007) and soil/landscape properties. Tillage practices were imputed to specific subwatersheds by disaggregating distributions of tillage practices, which were compiled at the 8-digit watershed level by Baker (2011), to the 12-digit watershed level (while maintaining the overall 8-digit distributions to the extent possible). Nutrients applied to cropland ranged between 117–156 kg/ha N and 25–34 kg/ha P, respectively, with N applied only to corn, using state-level annual averages reported by IPNI (2010). The SWAT auto-fertilisation routine was used to simulate nutrient inputs to hay and pastureland. Point sources of N and P have also been inserted to the model based on data compiled by the US Geological Survey (USGS; Maupin and Ivahnenko, 2011; Robertson, 2013).

The UMRB and OTRB SWAT models were calibrated and validated for the 1981–2010 time period based on measured river flows, sediments and N and P loads at several locations along the rivers (Figure 3). Additional details about the development of the required data layers and management inputs for the UMRB and OTRB SWAT models as well as the calibration processes are described in Panagopoulos *et al.* (2014a, 2014b).

6. Hypoxia model, specification and coefficients

A part of the integrated assessment system is an 'ecological production function' representing an empirical relationship between spring nitrate-N and TP riverine loads and the areal extent of Gulf hypoxia (similar to the model described in Rabotyagov *et al.*, 2014). Ecological theory suggests that the extent of hypoxia should be a function of nutrient inputs. Thus, the 2008 Hypoxia Action Plan recommends a dual nitrogen and phosphorus reduction strategy. To date, statistical evidence on the impact of multiple nutrients on the size of the hypoxic zone has not been unanimous. Several studies have focused on the role of N including Scavia *et al.* (2003), Scavia and Donnelly (2007), Liu, Evans and Scavia (2010), and Scavia, Evans and Obenour (2013). The empirical corroboration of the importance of multiple nutrients has been elusive. Hypoxia formation is a complex, dynamic and heterogeneous process, potentially subject to impacts of winds, ocean currents and disruptive weather events (storms and hurricanes). Further, the history of nutrient enrichment of the Gulf may matter. Turner, Rabalais and Justic (2006) speak of 'ecosystem memory', where system attributes are carried from 1 year to the next, 'as would happen if organic matter accumulating in one year was metabolized over several years'.

Previous statistical models include Turner, Rabalais and Justic (2006), who postulated that nitrogen, phosphorus and silicate, as well as lags in nutrient inputs as short as 1 month before the July hypoxia measurement and as long as 3 years prior, would be significant in explaining hypoxic area variability. They found that nitrogen loads, as well as lagged phosphorus loads, were significant. However, the authors settled on a model having May N and a time trend as the regressors.

Greene, Lehrter and Hagy (2009) used backward and forward model selection procedures to choose the month of nitrogen and phosphorus concentrations and river flows to be included in the hypoxia model. May nitrogen concentrations, May river flows, and February phosphorus concentrations were found to be significant. They also found that a post-1993 dummy variable was significant, and interpreted that as potential evidence of a regime shift in the system (and the inclusion of the dummy made the impact of phosphorus non-significant).

For the LUMINATE hypoxia model, we build upon these studies and Rabotyagov *et al.* (2014) and estimate the area of the hypoxic as a production function relationship accounting for the role of nitrate-N and TP with the possibility of multi-year lags, cumulative nutrient impacts and the time series structure of the data. Nutrient loading data comes from the official USGS estimates (as described in Aulenbach *et al.*, 2007). The areal extent of hypoxia (in km^2) is postulated to depend on current and past loadings of nitrate-N (N_{t-i}, as $\log_{10}$ transform of May USGS estimates of mainstem Mississippi River loads) and current loading of TP (P_t, $\log_{10}$ transform of May USGS estimates). We test for the cumulative effects of nitrogen and phosphorus by adding 5-year cumulative loadings to the model. We explore the effect of hurricanes and unusual current conditions and their interactions with nitrogen and phosphorus. Specifically, we estimate

$$\begin{aligned}\text{HypoxicZone}_t = {} & \beta_{\text{intercept}} + \beta_{\text{hurricane}}\text{Hurricane}_t + \beta_{\text{current}}\text{Current}_t \\ & + \beta_{\text{hurrSN}}\text{Hurricane}_t \times \log 10(N\text{stock5}_t) + \beta_{\text{hurrSP}}\text{Hurricane}_t \\ & \times \log 10(P\text{stock5}_t) + \beta_{\text{N}} P_t + \sum_{i=1}^{5} \beta_{i,\text{N}} N_{t-i} + \beta_{\text{Nstock5}} N\text{Stock5}_t \\ & + \beta_{\text{Pstock5}} P\text{stock5}_t + \varepsilon_t,\end{aligned}$$

where $N_t = \log 10$ (May N load at time t) and $\text{P}_t = \log 10$ (May TP load at time t) and $N\text{stock5}_t$ $\sum_{i=0}^{4}$ USGS–May$-N_{t-i}$ and $P\text{stock5}_t = \sum_{i=0}^{4}$ USGS–

May$-P_{t-i}$. We estimate the model using hypoxia size measurements from 1985 to 2010 (excluding 1989 when lack of funding precluded data collection), and use the 2011–2013 observations to assess model performance. To conserve degrees of freedom available for error estimation, we use a polynomial distributed lag model (Greene, 2003), which constrains the lagged regressors' parameters to lie on a polynomial (linear form of lag distribution). That is, the error term is assumed to follow an autoregressive process $\varepsilon_t = u_t - \theta 1\ \varepsilon_{t-1} - \theta_2\ \varepsilon_{t-2}$, where $u_t \sim N(0, \sigma^2)$.[7]

An additional criterion for model specification different from the specification reported in Rabotyagov *et al.* (2014) is the ability of the SWAT modelling framework to simulate the inputs to the statistical hypoxia model. SWAT simulations do not include silicates, and, given that the impact of silicates is estimated to be small, silicates are not included in the specification. Meteorological variables considered by Forrest, Hetland and DiMarco (2012), including 'wind power' and the Sea Surface Temperature anomalies, are only available through 2010 and given that they are outside of the nutrient management control efforts, they are implicitly left in the residual, although their effects have been reported in Rabotyagov *et al.* (2014). Visually, the structural part of dependence of hypoxia on single year May N and P values can be shown as a 'hypoxia production function' (Figure 4).

7. Cover crop scenario

Winter cover crops, including rye, oats, winter wheat or other close grown crops, are increasingly being used in the Corn Belt region to maintain and improve the quality of soil resources, and mitigate export of sediment and nutrients from cropland landscapes (Tonitto, David and Drinkwater, 2006; Kaspar and Singer, 2011; Kovar *et al.*, 2011, 2012). In this study, the cover crop scenario consisted of planting rye within the typical 2-year rotations of corn and soybean or continuous corn, in which the rye cover crop was planted in the fall after corn or soybean harvest and then harvested shortly before planting of the following row crop in the spring.

The scenario was applied to virtually all of the cropland in both regions (slopes <5 per cent) that included tile-drained areas on flatter landscapes. These tile-drained landscapes are primary source areas of exported nitrate-N, which is a key water quality problem in regional stream systems (Schilling and Wolter, 2009; Jha *et al.*, 2010; Secchi *et al.*, 2011) and also a key contributor to the Gulf hypoxic zone. The implementation of the cover crop scenario provides an assessment of how much such widespread adoption of the practice can reduce nitrate losses from these landscapes as well as the impact of the practice on overall reduction of sediment, N and P. The cover crop scenario was implemented within the same 30-year period that the baseline testing was

7 Longer autoregressive lags in the error term were explored as well but no significant evidence for them was found. The estimated process satisfies the stationarity conditions as described in Greene (2003, p. 276). In addition, we tested and did not find significant evidence of heteroskedasticity.

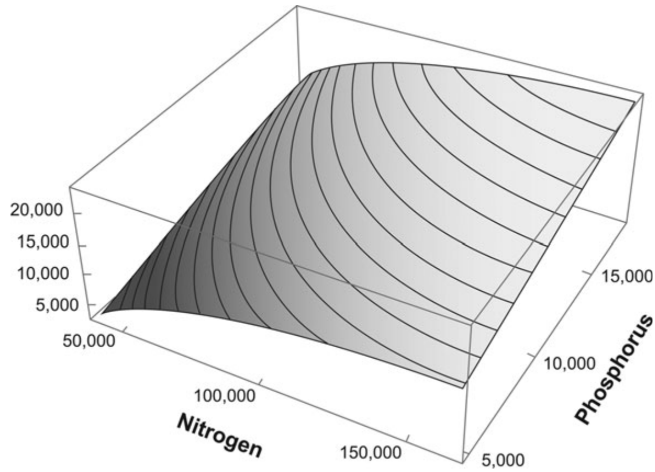

Fig. 4. Graphical representation of the empirical relationship between areal extent of hypoxia (*z*-axis, km^2), and N and P (*x* and *y* axes, metric tons). Relationship depicted for hypothetical May 2004 loadings (lagged loadings held at historical levels).

performed for, with all other model parameters (climate, soil and other management practices) unchanged.

8. Results and discussion

Table 2 summarises the SWAT results of the baseline and the cover crop implementation for both the UMRB and OTRB basins. The third row of the table indicates the area of cover crop implementation in the two basins, showing the large extent of affected agricultural areas in the Corn Belt. The cover crop within the traditional rotations resulted in reduced sediment, nitrate, N and P. As shown in Table 2, the reduction of all pollutants was greater in the UMRB due to the larger agricultural area where cover crops were established. Reductions were on the level of 20–25 per cent for N. The estimated P reduction was similar to the predicted sediment reduction due to the fact that phosphorus transport is closely tied to sediment movement.

Table 3 summarises the annual pollutant loads from both basins that enter the Mississippi River and clearly show that all of the annual pollutant loads decreased by ~20 per cent due to the widespread adoption of cover crops. Table 4 further shows annual N and P loading reductions on a unit land area basis. The N loadings were reduced by 5.06 and 7.97 kg/ha in the UMRB and OTRB, respectively, while the P loadings were reduced by 0.39 kg/ha in the UMRB and 0.72 kg/ha in the OTRB.[8] These impacts are comparable in relative magnitude with the overall N and P reductions reported in studies performed for the US Department of Agriculture Conservation Effects Assessment Project

8 The kg per ha values represent the average reductions.

Table 2. Mean annual (1981–2010) SWAT results at the outlets of UMRB and OTRB under the baseline and a cover crop scenario

	UMRB				OTRB			
Total crop area (ha)	23,000,000				10,130,000			
Crop area <5 per cent (ha)	21,900,000				10,050,000			
Pollutant	Baseline	Cover crop	Reduction	Reduction (per cent)	Baseline	Cover crop	Reduction	Reduction (per cent)
Flow (m^3/s)	4,095	4,064	31	0.75	7,716	7,681	35	0.45
Sediments (*t*)	29,008,718	21,344,125	766,4593	26.42	43,661,137	38,265,174	5,395,964	12.36
NO_3-N (*t*)	405,120	332,558	72,562	17.91	313,191	25,6206	56,985	18.19
TN (*t*)	536,361	425,508	110,853	20.67	434,581	35,4431	80,151	18.44
TP (*t*)	40,349	31,877	8,472	21.00	48,605	41,403	7,202	14.82

Table 3. Mean annual (1981–2010) pollutant loads entering the Mississippi River from the Corn Belt region (UMRB and OTRB) under the baseline and a cover crop scenario

	Mississippi			
Pollutant	Baseline	Cover crop	Reduction	Per cent reduction
Sediments (*t*)	72,669,856	59,609,299	13,060,557	17.97
$N0_3$-N (*t*)	718,311	588,764	129,547	18.03
TN (*t*)	970,942	779,939	191,004	19.7
TP (*t*)	88,954	73,280	15,674	17.6

Table 4. Overall N and P reductions and cost estimates for reduction of each kg of pollutant

	Per cent area treated	Reductions (kg/ha)	Cost ($/reduced kg)	Cost ($/reduced kg)
		UMRB		
Nitrogen	95	5.06	12.21	17.14
Phosphorus		0.39	159.96	223.95
Total costs ($mil)			1,355.20	1,897.30
		OTRB		
Nitrogen	99	7.97	7.74	10.85
Phosphorus		0.72	86.35	120.89
Total costs ($mil)			621.9	870.7
Cost cover crops			61.8$/ha ($25/ac)	$86.6 $/ha ($35/ac)

(CEAP) for the UMRB and OTRB.[9] However, our cover crops scenario differs from the CEAP scenarios in several aspects: (i) we consider a significantly larger treatment area, (ii) cover crops were not simulated in all of the CEAP scenarios and (iii) cover crops are just one of several conservation practices considered in the CEAP assessments. While a direct confirmation is not possible, these relative comparisons provide further support for our results.[10]

To provide some context for these analyses, assuming that the costs of cover crops adoption ranges from $61.8 to $86.6/ha ($25 to $35 per acre), we can

9 Depending on the scenario, the N reductions for the UMRB range between 6.5 and 42 per cent reductions, (USDA-NRCS, 2012b), while for the OTRB they range between 12 and 50 per cent (USDA-NRCS, 2011).

10 The CEAP reports consider five conservation scenarios: the treatment of critically undertreated acres (with a high need for conservation treatment) with water erosion control practices only, water erosion practices only, but now applied to a larger share of cropland (with high or moderate need for conservation treatment, nutrient management practices simulated *in addition to* water erosion control practices on acres with high conservation treatment need and nutrient management practices in addition to erosion control practices on acres with high or moderate conservation treatment need. The percentage area for high and moderate need is 60 per cent in the UMRB and 70 per cent in the OTRB. Cover crops is considered a water erosion conservation practice.

estimate that the cost of reducing a kilogram of N ranges between \$12.02 and \$17.10 for UMRB and between \$7.74 and \$10.88 for OTRB (Table 4).[11,12]

8.1. Hypoxia model results

The estimated statistical relationship between contemporaneous, lagged and stock levels of nutrients performs well relative to historical data and was able to predict the areal extent of the zone for 2011 and 2012 with greater precision than existing empirical models utilised by the USGS to create annual forecasts (Figure 5).[13]

Table 5 presents the coefficient estimates and fit statistics for the model. We find that the areal extent of hypoxia depends on N and P in fairly complex ways. As postulated in much of the existing literature, we do find that the areal extent of hypoxia (in km^2) depends on contemporaneous spring nitrogen (May nitrate-N). However, we also find that that spring nitrogen affects the hypoxia size with up to a 5-year lag, but the multiplicative cumulative effect is moderated somewhat (as shown by the negative coefficient on the 5-year sum of May N loadings). While nitrogen affects the size of hypoxia as both a flow and a stock, phosphorus (May P) affects the size of hypoxia as a stock (*P*stock5 significant and positive, logP not significant). Significant impact of lagged nutrients and the significance of nutrient stocks is consistent with the proposed 'ecosystem memory' conjecture for Gulf hypoxia (Turner, Rabalais and Justic, 2006), as well as the storage of nutrients upstream in drier years (Chen and Hong, 2012). Previously, it was suggested that the system response changed post-1993 and Turner, Rabalais and Justić (2012) found a positive time trend in a model of hypoxia response to nitrogen. The stock of phosphorus is highly correlated with time ($\rho = 0.71, P < 0.01$). Thus, observable variables (nutrients) which can impact hypoxia are included, instead of the time trend which proxies for accumulation of phosphorus and may include other unobservable factors.

We further find evidence for the effect of unusual current conditions and hurricanes (via their interactions with phosphorus stocks). The presence of unusual

11 The cost estimates for adopting cover crops are sparse and vary across location and type of crops. For example, Schipanski *et al.* (2014) estimate a cost of \$63.97/ha for adopting cover crops. Wainger *et al.* (2013) report a cost of \$76.6 per ha (\$32 per acre). Additionally, the Iowa Nutrient Strategy (IDALS, 2013) estimates a cost range \$71.6 to \$80.3 per ha (\$29 to \$32.5 per acre).

12 A similar analysis can be made for the cost of reducing a kilogram of P.

13 USGS publishes annual forecasts of the hypoxic zone after May nutrient estimates are released by USGS and before the Gulf hypoxia measurements are made later in the summer. The empirical relationship in LUMINATE was used to compare its out-of-sample predictions for 2011, 2012 and 2013 hypoxia with the forecasts released by USGS, which use two alternative models, one empirical (http://www.gulfhypoxia.net/news/default.asp?XMLFilename=201306191312.xml) and another one based on the Streeter-Phelps process with calibrated parameters (http://snre.umich.edu/scavia/wp-content/uploads/2013/06/2013-Gulf-of-Mexico-Hypoxic-Forecast_UM.pdf). The LUMINATE model was most precise for 2011 and 2012 and was second-best in 2013. The model we use also exhibits the lowest root mean square error of prediction for the 2011–2013 period (4,016 vs. 4,225 and 7,555 for the alternative models). While an improved process-level understanding of hypoxia formation remains needed, the empirical relationship used in LUMINATE appears to be on par or better than the existing models used for hypoxia policy evaluation.

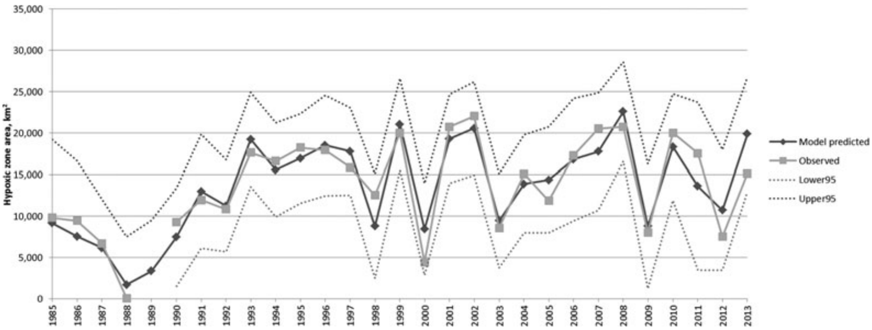

Fig. 5. Observed and model-predicted areal extent of Gulf of Mexico hypoxia, and 95 per cent confidence intervals for model prediction (2011–2013 observations were used to create out-of-sample predictions). The lower 95 per cent confidence interval bounds were estimated to be negative for 1985–1986 and 1988–1989 (lower bound for 1987 is 280.3 km^2, which is too small to be shown on the graph).

Table 5. Hypoxia model parameter estimates and fit statistics

Parameter estimates				
Variable description	Estimate	Standard error	*t*-value	*P*-value
Intercept	−670443	170396	−3.93	0.0017
Hurricane	164110	183151	0.90	0.3865
Currents	−13637	2776	−4.91	0.0003
Log10(current year *t* May N)	33892	9630	3.52	0.0038
Log10(year $t-1$ May N)	29368	8381	3.50	0.0039
Log10(year $t-2$ May N)	24844	7146	3.48	0.0041
Log10(year $t-3$ May N)	20321	5936	3.42	0.0045
Log10(year $t-4$ May N)	15797	4767	3.31	0.0056
Log10(year $t-5$ May N)	11273	3681	3.06	0.0091
log10P, year *t* TP	6803	4961	1.37	0.1935
*N*stock5	−0.0827	0.0336	−2.46	0.0285
*P*stock5	0.4424	0.0854	5.18	0.0002
hurrStockN5	283848	187917	1.51	0.1548
hurrStockP5	−371761	192792	−1.93	0.0759
AR1 (θ_1)	0.4043	0.1628	2.48	0.0274
AR2 (θ_2)	0.8117	0.1511	5.37	0.0001
Model fit				
Regression R^2		0.9503	Observations	25
Total R^2		0.9085	df error	13
SBC		485.372678	AIC	470.746168
Log likelihood		−223.37308		
Durbin–Watson		2.5251		

currents has a negative impact on the size of the hypoxic zone ($P < 0.01$). Hurricanes are a major disturbance and may affect current and future hypoxia formation in complicated ways. While hurricanes, by mixing the water column, may reduce the size of hypoxia, hurricanes may also be increasing oxygen demand from the sediments, so the net impact of a hurricane is an empirical question. Across hypoxia observations from 1985 to 2013, the average size of the hypoxic zone is 12,825 km^2 for non-hurricane years and 16,630 km^2 for years in which hurricanes were observed (a univariate *t*-test, assuming unequal variances, yields a *P*-value of 0.052).[14] In the model, the null hypothesis of no impact of hurricanes is rejected with $P < 0.001$. We find that the impact of hurricanes appears to be interacting with nutrients, especially phosphorus.

8.2. Cover crops scenario and estimated impact on Gulf hypoxia

The empirical model of Gulf hypoxia is estimated using USGS May nitrogen and phosphorus load estimates for the amount of nutrients reaching the Gulf of Mexico, while the cover crops scenario presented in this paper simulates load reductions at the outlets of the Upper Mississippi and at the OTRBs. Therefore, a link between the scenario and its impact on nutrient loads to the Gulf needs to be created. We follow a simple procedure and use (i) USGS-reported estimates for nutrient loadings originating from the UMRB and the OTRB and (ii) estimated delivery ratio in the mainstem of the Mississippi River for nitrogen and phosphorus from White *et al.* (2014).[15] Using this information, along with SWAT-simulated impact of cover crops on May N and P, we compute the relative reduction in the nutrient loads to the Gulf and use the adjusted nutrient values as inputs into the empirical model of the hypoxic zone. Keeping loads from other contributing basins (Missouri, Arkansas-White-Red and Lower Mississippi) constant, we assume that each nutrient's monthly load unit reduction contributes $d_j(1 - r_{ijt})$ units of reduction to the Gulf, where $j =$ N, P, $i =$ UMRB, OTRB and $t = 1981, \ldots, 2010$, and where r_{ijt} represents the SWAT model-simulated reductions in monthly nutrient loads. Weighing the USGS estimates of N and P loads from the UMRB and OTRB by these factors, we adjust the USGS estimates of N and P reaching the Gulf, and employ the final estimates in creating the counterfactual cover crops scenario. We assume a fixed delivery ratio of 0.87 for N and 0.9 for P, using estimates from White *et al.* (2014). Table 6 presents estimates of May N and P load reductions as simulated by SWAT.

Note that in comparison with mean annual N and P reductions (Table 2), simulated May mean N reduction is comparable with the annual values for the UMRB while quite smaller for the OTRB (Table 6). Impacts on P are simulated to be a relatively small reduction for the UMRB and a small net gain in

14 A non-parametric Mann–Whitney test (one-sided) yields a *P*-value of 0.07.

15 USGS estimates of nutrient flux at the outlet of UMRB (http://toxics.usgs.gov/hypoxia/mississippi/flux_ests/major_basins/MISS-THEB.html) and OTRB (http://toxics.usgs.gov/hypoxia/mississippi/flux_ests/major_basins/OH-GRAN.html) were used.

Table 6. Simulated percentage reductions in N and P, 1981–2010

	UMRB		OTRB	
Year	Per cent reduction in May N	Per cent reduction in May P	Per cent reduction in May N	Per cent reduction in May P
1981	38.8	14.4	36.1	−3.6
1982	0.2	8.3	−15.8	−7.1
1983	34.6	13.1	16.9	−3.9
1984	−34.4	3.7	−13.0	−5.1
1985	36.9	9.2	19.9	−0.3
1986	−1.5	8.0	−5.2	1.8
1987	19.7	8.3	15.7	−3.9
1988	−0.2	4.2	−4.3	−5.7
1989	28.3	11.0	31.8	5.7
1990	7.9	8.7	−2.2	3.1
1991	43.3	14.5	9.2	−6.9
1992	0.6	7.2	−4.9	−4.1
1993	40.0	21.3	11.8	−2.7
1994	−2.1	9.0	5.7	4.3
1995	40.5	10.8	27.2	1.3
1996	−10.0	2.4	−12.4	−2.1
1997	36.6	7.5	19.3	−1.7
1998	−11.4	5.7	−6.0	2.3
1999	41.4	9.9	11.6	−0.9
2000	12.0	15.4	2.5	−1.9
2001	40.7	10.8	32.1	−4.5
2002	8.3	9.4	−2.7	4.2
2003	30.8	14.3	18.3	0.7
2004	−18.3	4.3	−19.9	−4.4
2005	45.2	14.4	12.6	−1.4
2006	−4.0	13.2	−12.2	−2.7
2007	24.8	5.9	3.8	−5.3
2008	−7.4	4.4	−10.5	−5.0
2009	42.2	11.7	26.4	1.8
2010	−8.5	5.2	−1.5	1.9
Mean	15.8	9.5	6.3	−1.5
SD	22.8	4.3	15.7	3.5

monthly P values for OTRB (although the annual P reductions are significant in both basins). In the cover crops scenario, May nitrogen loads reaching the Gulf are reduced by 14.7 per cent, on average (SD = 24 per cent), while May phosphorus loads are reduced by 8.6 per cent, on average (SD = 6 per cent), over the period of 1981–2010.

Estimated expected hypoxia reductions are sizeable (mean annual reduction in the size of the zone is estimated to be 47 per cent), although the effectiveness of the scenario varies significantly over the simulation period, as would be

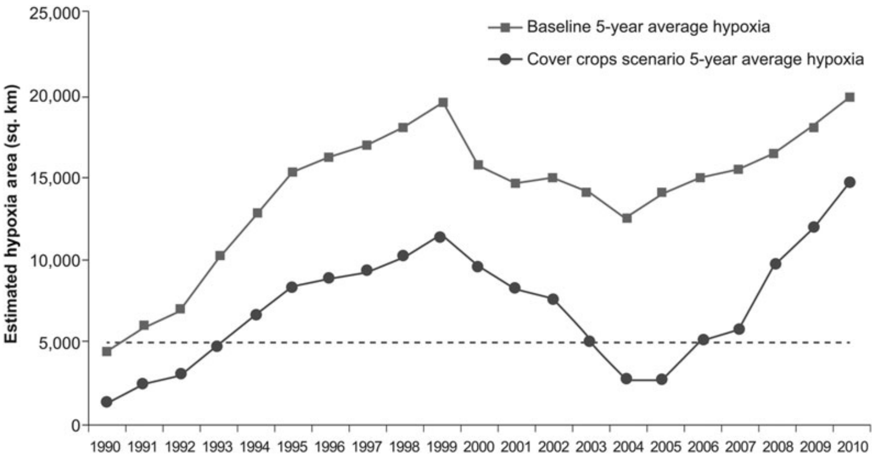

Fig. 6. The annual baseline and counterfactual hypoxia scenarios, 5-year mean hypoxia, 1990–2010 (structural component of the estimated relationship is used).

expected given variable reductions in spring nutrients simulated by SWAT. The national hypoxia Action Plan sets the target in terms of a 5-year running average of the hypoxic zone, which suggests that a smoothed relationship may be of interest to policymakers. Figure 6 presents the baseline scenario and the simulated impact of widespread application of cover crops on the 5-year average of the Gulf hypoxic zone. The Action Plan target of 5,000 km^2 is depicted by a dashed line. These results suggest that the cover crops scenario is likely not sufficient to reach the goal. The mean 5-year hypoxia is estimated to be 7,175 (SD = 3,590 km^2) and the hypoxia goal is reached in less than one-third of the 5-year periods between 1990 and 2010.

The results simulated by SWAT suggest that cover crops are generally effective in reducing N and P, especially in terms of reducing total annual nutrient loads. Spring nutrient load per cent reductions due to cover crops are simulated to be significantly smaller than annual reductions, which is reasonable given that cover crops are most effective at mitigating nutrient loss when traditional crops are not demanding nutrients. The reduction of pollutants at the local watershed level and specifically in densely cropped areas is much higher than the reduction at the outlet of the UMRB/OTRB which drains to the Gulf. This fact highlights the important benefits of pollution mitigation in intensively managed areas where local environmental problems exist and need remediation.

The establishment of rye as a winter cover crop is effective in reducing soil erosion, as well as sediment-bound and soluble forms of nutrients, and the results here are generally in agreement with estimated impacts reported by IDALS (2013). Combining cover crops with other conservation practices would be expected to be even more effective in reducing soil and nutrient pollutant losses. The findings on the impact of cover crops on Gulf hypoxia are consistent with the existing literature.

The 2001 Action Plan estimated that a 30 per cent reduction in the nitrogen load would likely be needed to achieve the hypoxia target. Subsequent research called for nitrogen load reductions closer to 35–45 per cent (Justić, Turner and Rabalais, 2003; Scavia *et al.*, 2003). Rabotyagov *et al.* (2014), using a similar empirical relationship, found that simultaneous 30 per cent reduction in spring N and P would have been sufficient to reduce the size of the hypoxic zone to <5,000 km.[2] The preliminary evidence based on our integrated modelling suggests that even a widespread adoption of cover crops, while likely having a fairly large impact in terms of hypoxia reduction, appears not sufficient in terms of reaching the national Gulf hypoxia goal.

9. Summary and final remarks

In this paper, we highlight the importance of developing integrated assessment models to support the design and implementation of policies to address water quality problems associated with agricultural pollution at multiple scales. We point to this emerging recognition and provide some examples of existing attempts at integrated modelling in the United States, Europe and Asia before turning our attention to large agriculture-dominated landscapes in the central US Corn Belt region. We describe a new modelling system, LUMINATE, which links land use decisions made at the field scale in the Upper Mississippi, Ohio and Tennessee Basins with local water quality effects and hypoxia in the Gulf of Mexico. We illustrate the model's capabilities with a simple scenario where cover crops are implemented over a large expanse of the watersheds. In future work, we plan to integrate the LUMINATE modelling system with evolutionary algorithms to find spatially optimised locations for the placement of cover crops and other conservation practices to achieve a wide range of environmental improvements. Optimisation using the same set of data and modelling assumptions used to develop the natural science policy basis can better serve to elucidate trade-offs, to highlight the potential gains from well-designed policies and to provide the basis for evaluating practical (if second-best) policies. Future integration with large-scale models of farmer behaviour, future climate scenarios and incorporation of other ecosystem service values (e.g. Wainger *et al.*, 2013) and terrestrial and marine models will be pursued. The systems which could be studied using integrated models of this sort are quite diverse in terms of their natural, economic and institutional characteristics, which promises plenty of opportunity for productive research by agricultural and resource economists.

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