Anthropocene 22 (2018) 10–19

Contents lists available at ScienceDirect

Anthropocene

journal homepage: www.elsevier.com/locate/ancene

Viewpoint

Critical transition in critical zone of intensively managed landscapes

Praveen Kumar[a,b,*], Phong V.V. Le[a,c], A.N. Thanos Papanicolaou[d], Bruce L. Rhoads[e], Alison M. Anders[f], Andrew Stumpf[g], Christopher G. Wilson[d], E. Arthur Bettis III[h], Neal Blair[i], Adam S. Ward[j], Timothy Filley[k], Henry Lin[l,m], Laura Keefer[g], Donald A. Keefer[g], Yu-Feng Lin[g], Marian Muste[n], Todd V. Royer[j], Efi Foufoula-Georgiou[o], Patrick Belmont[p]

[a] *Department of Civil & Environmental Engineering, University of Illinois at Urbana-Champaign, Urbana, IL 61801, USA*
[b] *Department of Atmospheric Sciences, University of Illinois at Urbana Champaign, Urbana, IL 61801, USA*
[c] *Faculty of Hydrology Meteorology and Oceanography, Vietnam National University, Hanoi, Vietnam*
[d] *Department of Civil & Environmental Engineering, University of Tennessee, Knoxville, TN 37996, USA*
[e] *Department of Geography and Geographic Information Science, University of Illinois at Urbana Champaign, Urbana, IL 61801, USA*
[f] *Department of Geology, University of Illinois at Urbana Champaign, Urbana, IL 61801, USA*
[g] *Prairie Research Institute, University of Illinois at Urbana Champaign, Urbana, IL 61801, USA*
[h] *Earth and Environmental Sciences, University of Iowa, Iowa City, IA 52242, USA*
[i] *Department of Earth and Planetary Sciences, Northwestern University, Evanston, IL 60208, USA*
[j] *School of Public and Environmental Affairs, Indiana University, Bloomington, IN 47405, USA*
[k] *Department of Earth, Atmospheric, and Planetary Sciences, Purdue University, West Lafayette, IN 47907, USA*
[l] *Department of Ecosystem Sci. & Management, Pennsylvania State University, University Park, PA 16802, USA*
[m] *State Key Laboratory of Loess and Quaternary Geology, Institute of Earth Environment, Chinese Academy of Sciences, Xi'an, China*
[n] *IIHR – Hydroscience & Engineering, University of Iowa, Iowa City, IA 52242, USA*
[o] *Department of Civil & Environmental Engineering, University of California, Irvine, CA 92697, USA*
[p] *Department of Watershed Sciences, Utah State University, Logan, UT 84322, USA*

ARTICLE INFO

Article history:
Received 25 July 2017
Received in revised form 21 March 2018
Accepted 5 April 2018
Available online 6 April 2018

Keywords:
Critical transition
Critical zone
Agricultural intensification
Anthropogenic modification
Complex systems

ABSTRACT

Expansion and intensification of managed landscapes for agriculture have resulted in severe unintended global impacts, including degradation of arable land and eutrophication of receiving water bodies. Modern agricultural practices rely on significant direct and indirect human energy inputs through farm machinery and chemical use, respectively, which have created imbalances between increased rates of biogeochemical processes related to production and background rates of natural processes. We articulate how these imbalances have cascaded through the deep inter-dependencies between carbon, soil, water, nutrient and ecological processes, resulting in a critical transition of the critical zone and creating emergent inter-dependencies and co-evolutionary trajectories. Understanding of these novel organizations and function of the critical zone is vital for developing sustainable agricultural practices and environmental stewardship.

1. Introduction

Humans have added considerable energy, directly and indirectly, to agricultural landscapes in an effort to overcome biogeochemical natural rate limits to maximize food production (Richardson and Kumar, 2017). Direct energy inputs include operation of farm machinery and modification of landscape topography such as terracing and filling to facilitate agricultural operations; managing water through channelization, periodic ditching, and installation of tile drainage; and pumping for irrigation. Indirect inputs include the embodied energy used for producing and distributing fertilizers and other chemicals such as pesticides. Although beneficial for short-term gains in productivity, these energy inputs are not synergistic with natural biogeochemical and geomorphic processes, and cause imbalances in natural rates of environmental metabolism, that is the natural ability to absorb the waste is lower than enhanced rate at which they are now produced. These imbalances often result in accumulation or efflux of biochemical constituents, and other

* Corresponding author at: Department of Civil & Environmental Engineering, University of Illinois at Urbana-Champaign, Urbana, IL 61801, USA.
E-mail address: kumar1@illinois.edu (P. Kumar).

https://doi.org/10.1016/j.ancene.2018.04.002

material fluxes such as sediment, in the environment, and alter the natural variability of fluxes of water, carbon, nutrients, and sediment, and affect other material changes that adversely impact ecosystem services and environmental quality (Richardson and Kumar, 2017).

While the impacts of unassimilated waste become evident when its accumulation exceeds the environmental carrying capacity, the displacement of these impacts in space and time (Kumar, 2013) also alters the heterogeneity of the landscape. The rate of these changes is on the order of decades, instead of centuries to millennia under natural conditions. This rapid evolution in landscape organization and function arises from a critical transition (Scheffer, 2009) resulting from and maintained by sustained anthropogenic inputs. This non-stationarity is similar to that of other human-induced trends such as changes in temperature, rainfall, and atmospheric CO_2 concentration associated with the climate system (Milly et al., 2008). These trends confound our ability to anticipate unintended consequences of human actions, and therefore to plan adequately to reduce risk and ensure sustainability of ecosystem services.

The goal of this paper is to provide scientific arguments in support of a proposed hypothesis that in the Midwestern U.S. critical transition from industrial agricultural activities have changed the landscape from primarily a transformation-dominated system characterized by long residence times of water, carbon, and nutrients, to a transport-dominated system characterized by rapid movement of water, sediment, carbon, and nutrients across land surfaces and through rivers and streams into receiving water bodies. We refer to this as the *transformer-to-transporter hypothesis*. Critical transition refers to the rapid or abrupt change in the dynamical behavior of the system, in this case the balance shifting from a transformer dominated to a transporter dominated functioning of the landscape, that is not easily reversible. We elucidate the causes and consequences of human-induced reorganization of landscape heterogeneity by casting current conditions within the context of the evolutionary trajectory of the landscape (Fig. 1). These lessons learned and concepts can be transferred to other areas influenced by intensive agriculture practices, and specifically to glaciated landscapes (Supplementary Fig. S1). We do not aim to provide a description of the progression of human impact in the Midwestern U.S., which happened quite rapidly over a period of only a few decades (Trimble, 2013). Our goal is to understand the functioning of the present day landscape in the context of its dynamics prior to the widespread adoption of industrial agricultural practices. In particular we show how cascade of human induced changes propagate through the interdependencies between the different carbon, water, nitrogen, soil, and ecological processes.

2. Legacy landscape

The modern agricultural landscape of the upper U.S. Midwest has been established within the context of the long-term evolutionary trajectory of the pre-agricultural landscape. This trajectory consists of a glacial legacy that has produced an edaphically, pedologically, and biologically enriched Critical Zone that supports a highly productive landscape (Fig. S1). The vertical and lateral structure of the critical zone in these landscapes reflects the legacy of at least 11 glacial/interglacial cycles over the past 2.6 million years (Fig. 2). During this period, advances of ice eroded uplands and filled valleys (Kemmis et al., 1992); and subglacial, englacial and supraglacial erosion, along with deformation and deposition processes produced spatially complex sediment sequences with varying hydrologic, geochemical and physical properties. Meltwater streams carved valleys and deposited thick layers of sand and gravel that today serve as aquifers. Across most of the region, the bedrock surface is buried beneath 10–100 m of unconsolidated and unlithified glacial sediment (Fig. 2) with significant lateral and vertical variation in its properties (Mickelson and Colgan, 2003). Late-glacial eolian activity covered glacial deposits with loess that decreases in thickness downwind from major glacial drainage ways from tens of meters to a few meters (Kemmis et al., 1992; Bettis et al., 2003). Across these landscapes, the loess buries clayey paleosols formed in weathered glacial till and outwash. In regions where the loess is less than 10 m thick, weathered till is exposed along steeper slopes. Outcrops of shale, sandstone, and carbonate bedrock occur where streams have incised deeply through the covers of Quaternary deposits. The resulting soil patterns, therefore, reflect regional and local distributions of loess, aeolian sand, glacial till and outwash, lacustrine sediments, paleosols, bedrock, and alluvium.

Prior to agricultural development, natural drainage networks in areas glaciated during the Late Pleistocene were poorly integrated with extensive wetlands characterized by poorly-drained soils (Patterson et al., 2003). Weathering profiles in these regions are relatively thin and usually grade to unweathered fine-grained glacial till within 5 m of the land surface, although deeper penetration of the weathering front through fracture networks may be present (Bettis, 2007). In parts of the Midwest last glaciated during the Middle to Early Pleistocene (ca. 0.5–2.6 Ma) landscapes are dissected by well-integrated drainage networks (Rovey and McLouth, 2015). In many recently glaciated areas, the present landscape greatly deviates from the preglacial and earlier Pleistocene topography. Glacial processes reorganized drainage networks and moved major drainage divides across central North America (Flint, 1949). Many of the region's modern rivers reflect the history of glaciation. For example, most have longer and lower-gradient profiles than before glaciation, some are underfit to their valleys, and all owe their sand and gravel resources and alluvial aquifer characteristics to a glacial heritage (Bettis et al., 2008). Within the most recently glaciated areas, rivers may not be in equilibrium with valley profiles (Yan et al., 2017) and channel adjustments are occurring through migrating knickpoints, bed incision, and bank failure (Belmont et al., 2011).

The soil continuum in the upper Midwest formed since the last glaciation reflects the cumulative effects of interactions between climate-change driven co-evolution of soil, vegetation and other biota over the underlying geology (Lin et al., 2011). As the climate warmed and interglacial atmospheric circulation patterns were established, the paraglacial bioclimatic zones (Supplementary Fig. S2) shifted northward (Fig. 1). The deglaciated areas were first colonized by pioneer plants with low nitrogen requirements, followed by arrival of nitrogen fixing plants and fungi, then hardy plants and trees, and finally a conifer forest to mixed conifer and deciduous forest. As temperate deciduous forest spread northward at rates of 100–300 m/y during the early Holocene, prairie and savanna expanded eastward as the Prairie Peninsula, reaching its maximum extent about 6000 years after the last mid-continent glaciation (Pielou, 1991).

Before the settlement of European immigrants two centuries ago, tall grass prairie with dispersed wetlands covered upland areas, while woodlands lined drainageways (Transeau, 1935). Higher water holding capacity of clay and fine sediment in the glacial till supported a rich variety of vegetation, while lower water holding capacity in the glacial outwash regions supported the establishment of conifer and broadleaf trees. Spatial and temporal patterns in precipitation and temperature further shaped the landscape, its soils, and the dominant native vegetation in the upper Midwest (Baker et al., 1996). Coincident with plant and microbial colonization were insects and animals that exploited the developing fertile lands but with exceptions of certain slow moving ecosystem engineers extirpated by the glacial advance, like

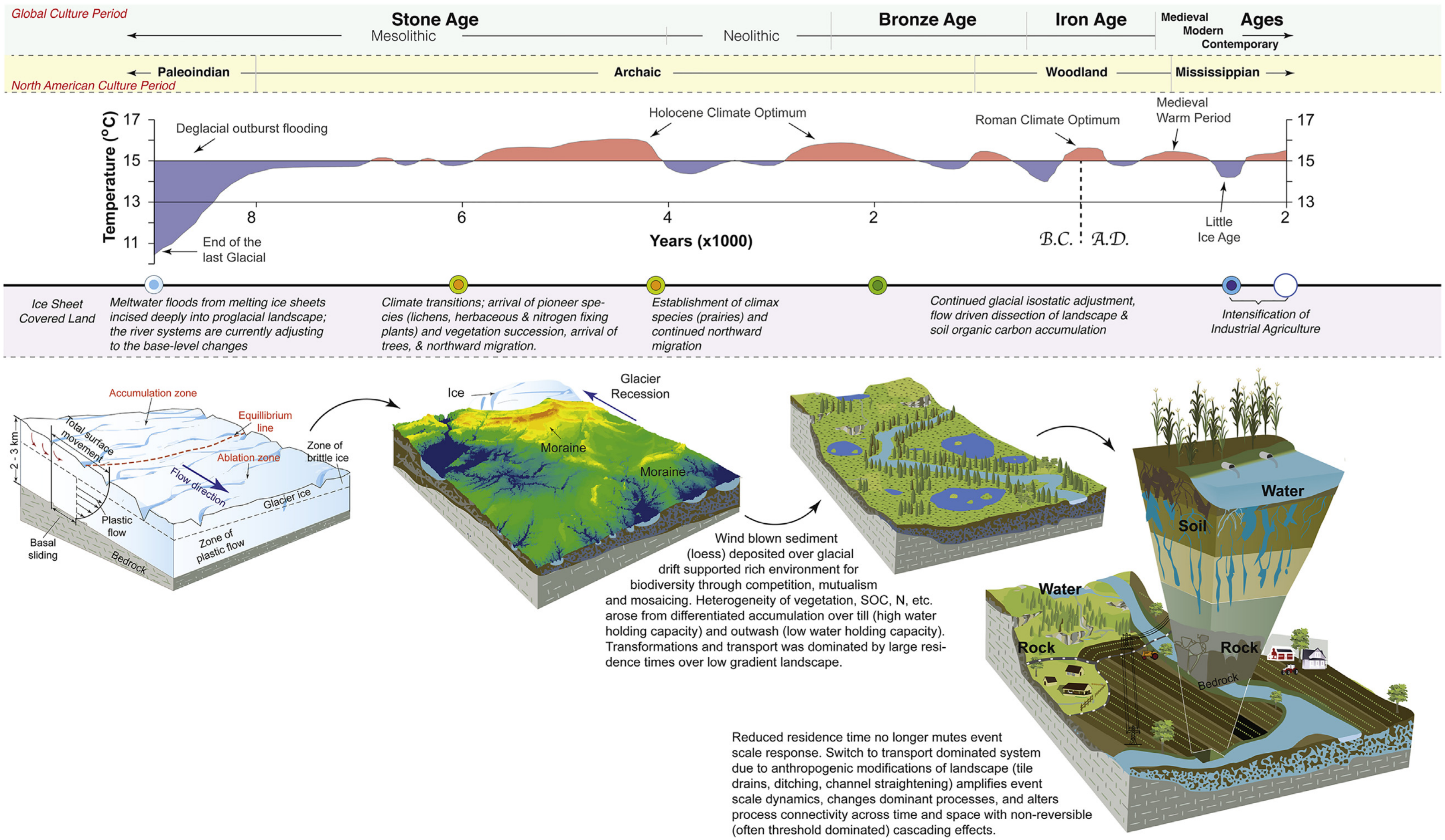

Fig. 1. Evolution of the glaciated agricultural landscape since last glacial maximum to present, illustrated in the context of human history. After the ice receded, the region went through a period of rapid climate transitions. Prairies and wetlands were the climax ecosystems. Settlement by European immigrants exploited the organic rich soil for agriculture by draining the land through extension and straightening of streams and through installation of subsurface tile drain networks connected to the stream network.

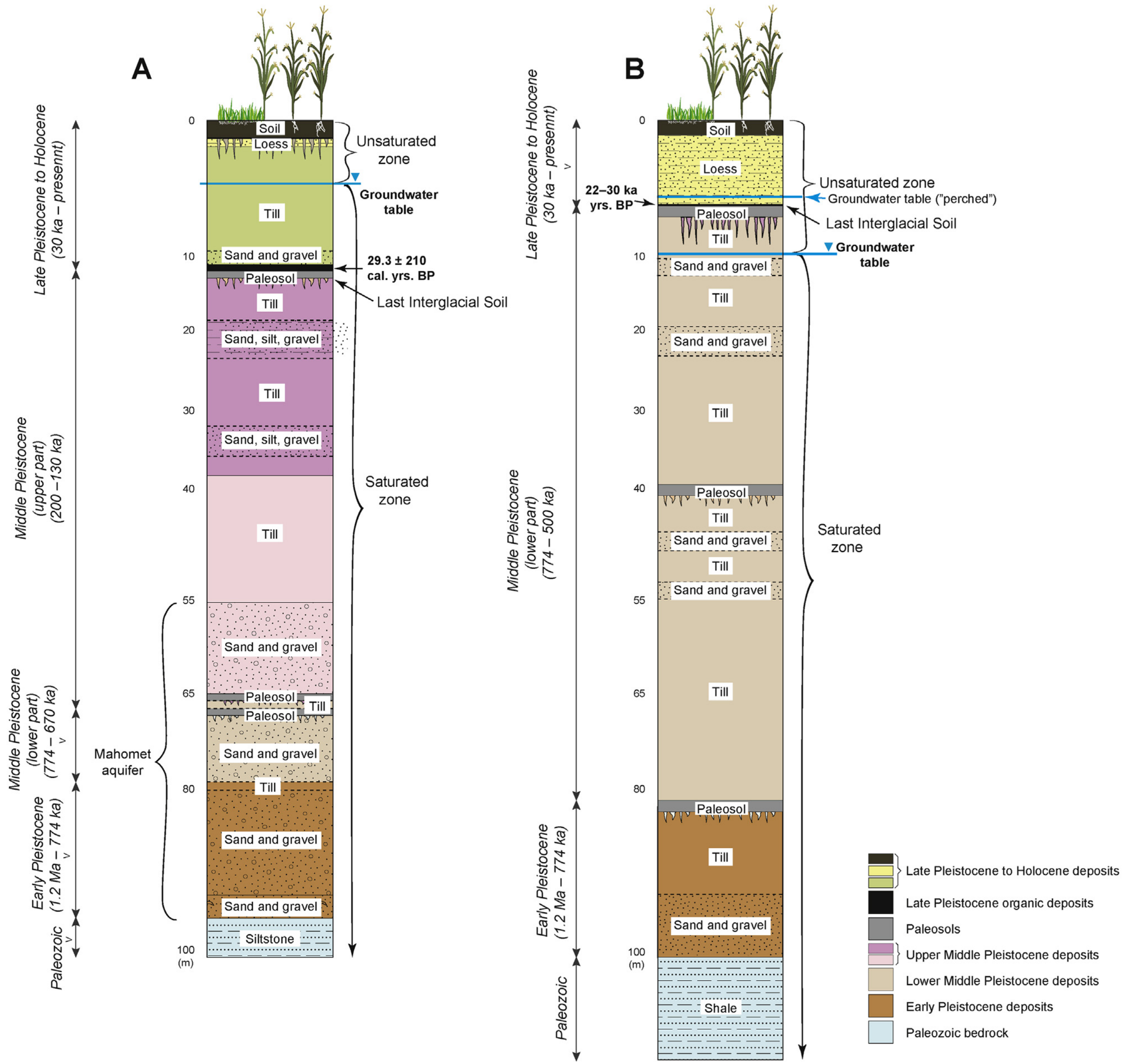

Fig. 2. Illustration of the vertical structure of the Critical Zone in intensively managed, glaciated landscapes of the upper Midwest, USA. Examples based on soil cores from the Upper Sangamon River Basin in central Illinois (A), and Clear Creek Watershed in eastern Iowa (B). The structure of the Critical Zone in this glaciated landscape developed through several episodes of deposition and erosion over the past 2.6 million years. Variations in glacial history and proximity to large valleys and loess sources dramatically influence the surface topography and vertical structure across the region. Areas glaciated during the Late Pleistocene, such as the Upper Sangamon Basin, have lower-relief landscapes formed in slightly weathered glacial deposits with thin to no loess cover that bury older, more weathered glacial deposits and glacial aquifers (sand and gravel). Areas last glaciated during the Middle Pleistocene, such as Clear Creek Watershed, have landscapes with greater relief formed in slightly weathered Late Pleistocene loess deposits that mantle weathered Middle and Early Pleistocene glacial till and glacial aquifers (sand and gravel). The

earthworms, which would otherwise act to vertically mix plant litter and mineral soil (Reynolds, 1994).

Moderate to low gravitational gradients, and prevalence of internally-drained basins in the glacially sculpted topography promoted low rates of runoff and long residence times of surface and sub-surface water. Further, moderate to low solar energy in the temperate latitudes and high soil-moisture supported slow biogeochemical transformations in the grassland and forest-grassland transition biomes. Together, these conditions led to the development of nutrient and carbon rich mollisols through accumulation of organic material in the form of plant roots and litter, which formed the basis for highly productive agricultural soils (Fig. 1).

3. Anthropogenic landscape modification

The transformation of the organically rich productive landscape for agriculture largely facilitated the expansion of European settlement throughout the region during the 19th and 20th centuries. Thick prairie sod required the development of the steel-bladed plow to cut through thick dense root zones for tillage (Rhoads and Herricks, 1996). To alleviate poor drainage and excessive wetness that impeded agricultural production, a vast network of artificial drainage (commonly called "tile drains") was installed underneath the fields and connected to the stream and drainage ditch network. Existing streams and prairie sloughs were enlarged, straightened, widened, and extended headward, increasing the density and efficiency of the drainage network (Rhoads et al., 2016). Today, channels are periodically dredged mechanically to maintain their conveyance capacity (Landwehr and Rhoads, 2003) and tile drains continue to be installed and maintained. The tile drains permanently lowered the water table, rapidly drained the rooting zone after storms, and reduced ponding of water in low lying areas (Hewes and Frandson, 1952). The reduction of moisture in the soil horizon facilitated the use and expansion of mechanized agriculture practices, and lengthened the growing period of crops by providing access to fields earlier in the spring planting season. It also created aerated root-zone conditions beneficial for seed germination and crop growth, and protected crops from excessive soil water (Skaggs et al., 1994). Over time, nutrient-related rate limits were surmounted through the use of synthetic fertilizers, enabled by the energy intensive Haber–Bosch process developed in the first half of 20th century (Smil, 2004). Together, these practices have overcome rate-limiting conditions for agriculture associated with moisture and energy needs. As a result of these changes, the organizational structure and function of the landscape is on an emergent trajectory. That is, the heterogeneous organization of soil structure and composition, carbon and nutrient distribution, and hotspots of biochemical transformations and sediment sources reflect an novel spatial organization under the stress and constraints of this intensive Anthropocene environment.

3.1. Carbon dynamics

Through a series of disruptions, intensive management has led to loss of organic C (carbon) from the soil, resulting in an increase in soil bulk density near the surface (David et al., 2009). First, the onset of agriculture and land conversion disrupted the stable rhizosphere of prairies and forests. Subsequently, increased vertical moisture movement and water flux from tile flow caused erosion (Papanicolaou et al., 2015a) and vertical translocation of both particulate and dissolved C (David et al., 2009). Saturated, and hence anaerobic conditions, which inhibited soil organic C decomposition, have been greatly reduced by land drainage. Additionally, subsurface tiles have allowed penetration of oxygen to deeper soil horizons. This increase in the duration of aerobic soil conditions now accelerates the breakdown of stored carbon at depth (Liu et al., 2012). As a result, deeper soil layers are becoming a source of C rather that a sink (Schmidt et al., 2011). Loss of C diminishes the capacity of biogeochemical cycling of other elements such as N (nitrogen) and P (phosphorus), and the formation of aggregates, thereby enhancing soil erodibility. This constitutes a positive feedback between carbon loss and erosion.

At the surface, tillage reduced soil organic C pools in the Midwest by 50% between 1907 and 1970 (Matson et al., 1997). This practice also profoundly affects soil organic C redistribution both vertically through the incorporation of residue within the soil profile, and laterally by disaggregating and exposing lighter, more carbon-enriched material to selective entrainment by flow (Papanicolaou et al., 2015a). Selective entrainment of small size fractions by surface runoff affects the enrichment ratio, i.e., the ratio of the concentration of C in the eroded sediment to that of the original soil, changing the availability of soil organic C (Wang et al., 2013). Net fluxes of fine sediment from the upslope, which are enhanced from agriculture, are enriched compared to downslope depositional areas, resulting in redistribution of C within the landscape. High enrichment ratios on upper portion of hillslopes are attributed to rain splash whereas concentrated flow is more important in the downslope region (Papanicolaou et al., 2015a).

Erosion rates have increased almost 10-fold over pre-disturbance levels (Hooke, 2000). Whether soil erosion is a net source or sink of C to the atmosphere is an issue marked by considerable controversy (Harden et al., 2008) and likely dependent on geography. The conclusion that soil erosion necessarily leads to a net flux of CO_2 to the atmosphere results from an incomplete accounting of the life cycle of the soil particles and their associated C (Harden et al., 2008). Subsurface soil exposed by the erosion of surface layers will accumulate organic C through vegetation growth, thus serving as a C-sink. Soil transported to aquatic environments, such as lacustrine and marine sediments, can adsorb organic C derived from primary production in the water column (Blair and Aller, 2012). For a large and complex system like the Mississippi River basin with its myriad sedimentary sinks, it is challenging to determine whether soil erosion is a net source or sink of C to the atmosphere.

Another disruption in C dynamics is caused by the suppression of the natural fire regime due to intensive agriculture and livestock management. Fire driven disturbance was an integral part of the prairie ecosystem (Collins and Wallace, 1990) in that it regulated nutrient cycling (Ojima et al., 1994) and populations of soil microbes (Lehmann et al., 2011), and provided C with long mean residence time to soil pools (Santin et al., 2015). Suppression of fire has substantially reduced the production rate of black carbon, which once contributed up to 20% of soil organic carbon (Singh et al., 2012; Santin et al., 2015). The mean residence time is estimated at centennial to even millennial timescales (Singh et al., 2012). These long residence times contrast sharply with other chemical components of the biomass material (Schmidt et al., 2011), thus its presence in soil is thought to be important for long-term soil organic C stabilization (Hernandez-Soriano et al., 2016).

Soil respiration, another important regulator of soil C, is a function of soil micro-climate, texture, prevalence of anaerobic conditions, and tillage. Tillage enhances C decomposition by breaking up soil aggregates and increasing mineralization (Reicosky et al., 2005). It also brings residue at the surface in contact with soil microbes, and aerates the soil (Reicosky et al., 2005), factors that enhance C decomposition. Under high N-fertilizer rates, fast-growing microbes that consume labile C can increase in abundance, replacing the slower growing microbes that breakdown the more complex and degraded C forms (Fierer et al., 2003). So while soil organic C has decreased dramatically, the loss of usable C may be even more severe. Changes to soil microbial substrate-use efficiency may have direct implications for the source of stabilized soil C (Cotrufo et al., 2013), given that soil microbial necromass is a substantial source of soil C (Liang et al., 2011).

3.2. Pedologic processes and erosion

The texture and organic matter content of soil, as well as the soil aggregate strength, architecture of pore network, and degree of compaction are repeatedly altered by farm machinery, tillage, and changes in vegetation cover during crop rotations (Stavi and Lal, 2011). Additionally, rainfall influences the soil composition and structure, and runoff induced and tillage-enhanced erosion selectively removes fine organic-enriched soil particles and aggregates (Abaci and Papanicolaou, 2009). Through these mechanisms, modern management practices have spatially altered hydropedologic characteristics across the landscape, including soil hydraulic conductivity (Papanicolaou et al., 2015a), thereby directly influencing runoff and soil erosion (Schoeneberger and Wysocki, 2005), and infield soil and in-stream water quality. In contrast to the surface increase in bulk density due to C loss, the vertical translocation of organic C by tile-induced moisture movement decreases the bulk density of deep soil and increases its water holding capacity (David et al., 2009). Together with changes in soil structure through the vertical column, the thermal regime and its variability, which in turn controls rates of weathering and microbial-driven redox processes (Gabriel and Kellman, 2014) has been changed.

Discrete erosional episodes, such as headcut migration, bank failures, landslides, and the development and enlargement of gullies, which are enhanced due to agriculture, can also deliver large pulses of sediment to streams (Papanicolaou et al., 2010). Not all sediment though leaves the hillslope. The majority of mobilized sediment is redeposited on floodplains, in riparian zones, or along roads and ditches. Across the upper Midwest, the introduction of intensive agriculture is accompanied by an order-of-magnitude increase in rates of floodplain deposition (Grimley et al., 2017). Spatial variability in floodplain deposition rates is influenced by loess thickness and landscape relief. Faster rates are observed in the high-relief, thick loess landscape of eastern Iowa and the Driftless Area of Wisconsin (Knox, 1987), than in the Illinois region (Grimley et al., 2017). Sediment stored in the floodplains may be remobilized in subsequent events through bank erosion or grazing in the floodplain (Neal and Anders, 2015). These punctuated sediment-delivery mechanisms add to the non-stationarity in the sediment flux of the landscape system. Conservation practices, such as grass waterways, can decrease the travel times, whereas intensification of tillage increases soil erosion and decreases sediment detention (Dermisis et al., 2010).

3.3. Moisture and transport dynamics

Flow through the tile networks is an important determinant of hydrologic response across scales from fields to watersheds. Event water often dominates the rising limb of the hydrographs from the tiles because of macropore flow and inflow of runoff into surface inlets in the field. During the falling limb, flow consists of a mix of event and pre-event water reflecting a more connected pore space in the soil matrix (Williams et al., 2016). Further, increase in water table depth and enhanced vertical transport of water through the soil profile, combined with rapid flow through drainage tiles, increases connectivity between distal points, resulting in flashier hydrographs and increased peak flows (Davis et al., 2014). These factors also increase baseflow and annual maximum flow, thereby altering the recession curves of hydrographs (Schilling and Helmers, 2008). Where relief is relatively high, the increase in erosional energy of streams induces head cut (Abaci and Papanicolaou, 2009) and knick point migration (Bressan et al., 2014), increased streambank (Sutarto et al., 2014) and bluff erosion (Day et al., 2013), enhancing sediment fluxes. Extension of channel networks by ditching further augments the availability of bed and bank areas as sediment sources (Abban et al., 2016). The removal of sinuosity and bedform complexity, the important physical determinants of hyporheic exchange in low-gradient alluvial systems, through straightening, dredging, infilling of gravel frameworks and clogging of streambed with fine sediment decreases hyporheic exchange (Gomez-Velez et al., 2015). With the removal of exchange flux, human management also inhibits ecological benefits such as denitrification, temperature regulation, and provision of physical habitat (Rhoads et al., 2003).

At the scale of catchments, the human modified drainage network results in rapid transport of water through the landscape. The headwater extension of channel network increases drainage density, reducing travel time through the watersheds by increasing hillslope and channel connectivity, and impacting both the hydrodynamic and geomorphologic dispersion that determine the stream-network scale hydrologic response (White et al., 2004). The relatively straight, unobstructed, trapezoidal drainage ditches are maintained to maximize conveyance of channelized flow (Rhoads and Herricks, 1996). Together, these direct and indirect impacts of human modification induce regime shifts in streamflow patterns across a range of time scales from daily to seasonal and annual (Foufoula-Georgiou et al., 2015).

3.4. Nutrient dynamics

Tillage alters the continuity of the macropore network near the surface, impacting both water and solute transport (Covino, 2017; Williams et al., 2016). Macropores also serve as biologically active sites (Jarvis, 2007) and the destruction of pores alters the redox potential of the landscape. Reduced transit times also limit the contact between solutes and biologically active subsurface locations, which reduces transformation and favors transport (export) from the hillslope and ultimately the catchment. Changes in (i) the quality of C due to land use change from prairie and wetland ecosystems to row crop agriculture (Wilson and Xenopoulos, 2009), and (ii) the vertical profile of C, nutrients, moisture and temperature, and their spatial heterogeneity, have altered biogeochemical dynamics across the landscape.

Synthetic fertilizers and fixation by leguminous crops are the largest inputs of N to the Critical Zone (Gentry et al., 2009). Recovery of fertilizer N by crops is typically $<50\%$ (Gentry et al., 2009), and N not taken up by plants is susceptible to loss through denitrification in soil or rapid runoff to streams or leaching to groundwater (Royer et al., 2006). Cation exchange of ammonium in the soil and the tile network determine the vertical distribution of age of nitrogen where the zone above the tile has younger nitrogen as a result of reduced accumulation due to the high N transport rates through the tile network. In contrast, the zone below the tile has high age indicating slow leaching into the groundwater system (Woo and Kumar, 2016, 2017). High loading of N often exceeds the assimilative capacity of streams and results in long transport distances for riverine dissolved inorganic N (Mulholland et al., 2008). Surface waters in intensively row-cropped landscapes are stoichiometrically imbalanced in molar ratios of N, P, and dissolved silica (Si) relative to the Redfield ratio (Downing et al., 2016) of 16:1:20 for N:P:Si. High riverine N:Si and P:Si ratios favor growth of non-siliceous algae (i.e., non-diatoms) which can promote blooms of harmful algal species, including toxin-producing cyanobacteria.

At the scale of river basins, a strong relationship exists between discharge and riverine nutrient loads, resulting in stationarity in annual flow-weighted nutrient concentrations (Basu et al., 2010). Although the rates and timing of key processes today are likely quite different from those prior to conversion to agriculture, these highly modified landscapes appear to have reorganized and achieved biogeochemical stationarity. However, recent evidence suggests that changes in N-use efficiency by crops, and to a lesser extent adoption of conservation practices, may be causing a transition to non-stationarity in N loads in some river basins (McIsaac et al., 2016).

3.5. Ecological processes

The midwestern U.S. has seen rapid conversion of native vegetation to agriculture, intensification of agriculture through mechanization, a shift from oats and small grains to soybeans (USDA, in press), and recently, expansion of corn acreage to support bioenergy crop production (Secchi et al., 2011). Incentives for bioenergy production are likely to lead to further conversion of land use from food/feed crops to grasses for second generation, lignocellulosic based bioenergy production (Hudiburg et al., 2016). Such changes have the potential to alter both the hydrology of soil by increasing transpiration (Le et al., 2011) and the biogeochemical composition of the soil (Woo et al., 2014).

During the growing season, terrestrial gross primary production in the upper U.S. Midwest is remarkably high (Guanter et al., 2014), although much of the fixed carbon is removed as grain. Over the long-term, a landscape dominated by corn-soybean rotation can be a carbon source due to the fraction of gross primary production removed as grain and the several months each year that fields are fallow (Dold et al., 2017). Freshwater ecosystems throughout the region, particularly streams and rivers, have been degraded by channel modifications to enhance drainage, sediment inputs, loss or modification of riparian vegetation, nutrient loading, and inputs of agro-chemicals (Allan, 2004; Blann et al., 2009). Biodiversity loss has been extensive (Ricciardi and Rasmussen, 1999) with significant impact to ecological functioning of freshwater ecosystems (Vaughn, 2010).

4. Critical transition and anthropogenically constrained self-organization

The Critical Zone of intensively managed agricultural landscapes is continually responding to a variety of anthropogenic drivers. Prior to significant anthropogenic influence, biotic and abiotic co-evolution occurred in a low water- and energy-gradient environment. However, new modes of self-organization are constrained by human infrastructure (e.g. the tile drainage network) and activities (e.g. agricultural practices, channel modification) that are targeted to achieve specific objectives of overcoming rate limits to maximize agricultural productivity. However, strong dependencies among various components induce self-organizing dynamics that impacts the spatial organization of different constituents, giving rise to emergent patterns of form and function. Therefore, we refer to this mode of behavior in the system, which is far from equilibrium and maintained in this condition by human energy inputs, as anthropogenically constrained self-organization.

The consequences of the critical transition in landscape organization and function from a transformation-dominated system to transport-dominated system are not trivial. This change in landscape dynamics alters rates of reaction and transport across the landscape for both fast response processes associated with weather time-scale phenomena and slow response processes associated with weathering and soil structure formation (Fig. 3). Event scale dynamics associated with weather phenomena now play a more prominent role in water and geochemical fluxes, resulting in significantly reduced residence times in the Critical Zone compared to those before the introduction of industrial agriculture. This change in the landscape to a system dominated by transport has led to emergent challenges such as hypoxia in distant receiving water bodies, and significant top soil loss in major parts of the Midwest (Sperow et al., 2003).

5. Conclusion

Along with direct human impact, influences driven by climate change have played, and are likely to play, an important role in the ongoing evolution of the Critical Zone. Vegetation acclimation to increased concentration of atmospheric CO_2 increases crop yield, leaf area index, above-ground dry matter, and also changes leaf nitrogen (Ainsworth and Long, 2005). However, elevated temperatures are predicted to reduce crop yields (Schauberger et al., 2017). The interplay between these competing influences of rising temperatures and CO_2, along with rainfall variability (Roque-Malo and Kumar, 2017), is likely to further shape carbon, water and nutrient dynamics across these landscapes as changes in extremes of rainfall and drought alter both soil and vegetation function through direct and indirect impact of moisture content, temperature persistence, wet/dry cycles, fluxes of nutrients (Davis et al., 2014) and trace gases, and surficial and stream transport of edaphic materials and organic matter.

To ensure the sustainability of these landscapes against loss of productivity (Baumhardt et al., 2015), we need to develop a framework of predictive understanding that weaves the complex network of processes, including human related activities, into an integrated whole, rather than pursue piecemeal or isolated strategies. Conservation or best management practices that leave more crop residue at the soil surface, or add more organic matter to the soil have the potential to substantially reduce the adverse impacts of human action in these landscapes. These practices also improve soil aggregation and porosity, so that soils are better aerated and hold more moisture (Papanicolaou et al., 2015b). Other practices, such as cover crops, conservation tillage, etc. can also reduced erosional loss and nutrient runoff. Furthermore, there is a need to consider a life cycle approach for evaluating the beneficial effects of management choices so as to incorporate the near- and long-term values of Critical Zone services (Richardson and Kumar, 2017). The integrated perspective offered here will enable the development of improved management practices that maintain or increase the societal and environmental value of these landscapes.

The transformer-to-transporter hypothesis is applicable to low gradient landscapes in temperate climates, such as those in North America and northern Europe, which have been sculpted through glaciation. The alternate hypothesis of conversion of landscapes that have historically functioned as transporters to transformer, by increasing residence time through human action, is also evident in certain parts of the world. For example, on the Chinese Loess Plateau, considerable effort has been devoted to reducing erosional soil loss from the highly dissected landscapes by terracing and reclaiming agricultural land on hilltops for revegetation (Chen et al., 2015) and filling valleys to create farmland for agriculture that are less prone to erosion. Nevertheless, such intensive management has to be supported through human infrastructure. In both situations, though, the landscape is maintained in far-from-equilibrium state through human infrastructure and periodic input of direct or embodied energy to achieve societal goals. Understanding how this human intervention induces imbalances across interrelated processes over a range of time scales is essential for developing effective sustainable approaches by realizing the often competing goals of economic value and environmental stewardship.

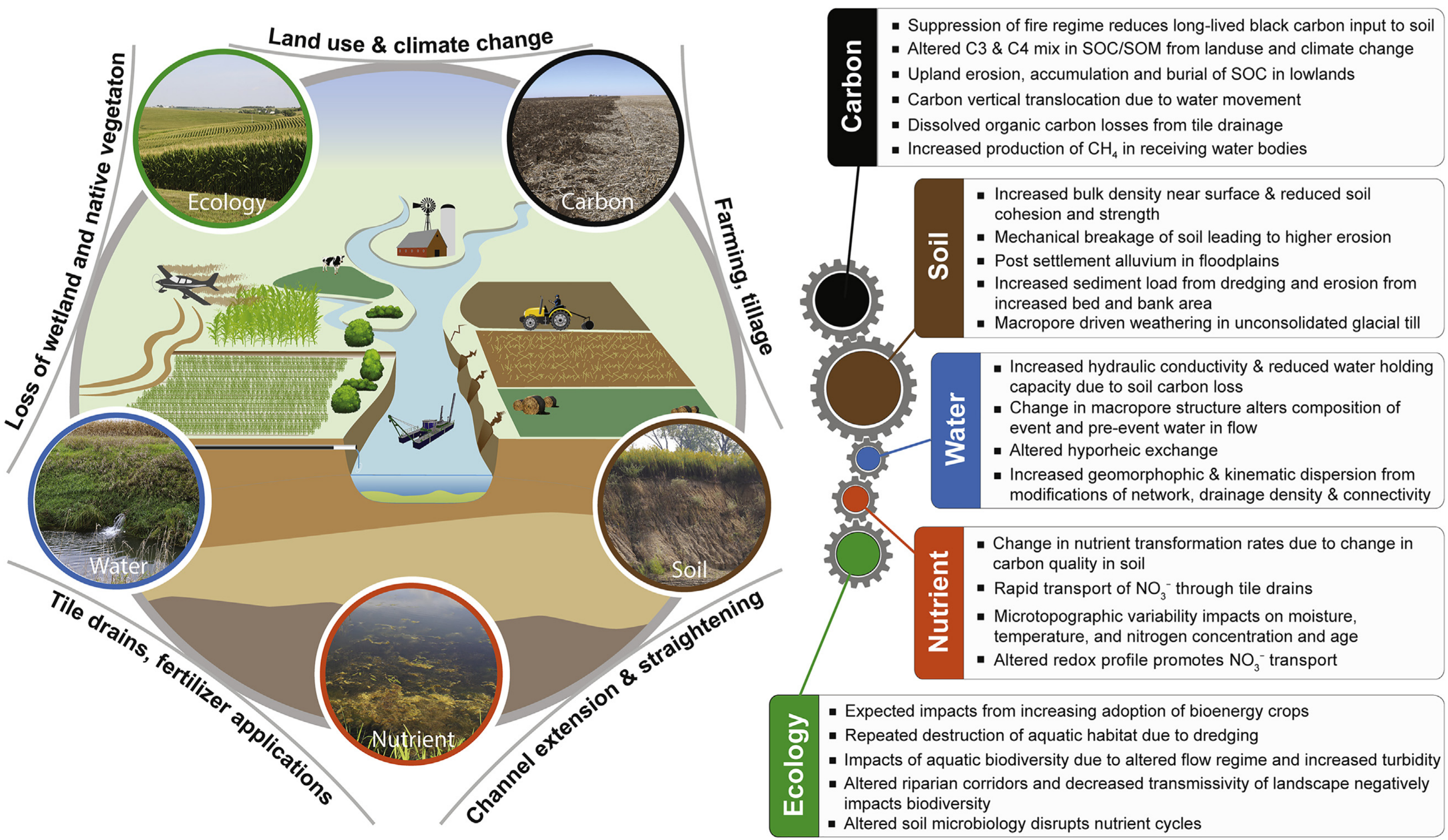

Fig. 3. Illustration (left) and summary (right) of how human modifications in intensively managed agricultural landscapes have impacted the inter-dependency between processes associated with carbon, soil, water, nutrients and ecology. The modifications include loss of wetlands and native vegetation for farming, tillage practices, channel extension and straightening, installation of tile drain network, application of fertilizers, and ongoing changes related to land use and climate. The gears indicate that changes in one component cascade to others across different time scales. Processes associated with water dynamics show the quickest response, therefore indicated as smallest gear. Processes associated with nutrient, ecology, carbon and soil dynamics exhibit progressively slower dynamics, roughly in that order.

Author contributions statement

This is a collaborative effort resulting from the National Science Foundation supported project on Critical Zone Observatory for Intensively Managed Landscapes. P.K. conceptualized the research presented here but numerous deliberative efforts shaped the details of this work and significant contributions were made by all co-authors through discussions and writing.

Acknowledgements

This work was supported by NSF Grant EAR-1331906 for the Critical Zone Observatory for Intensively Managed Landscapes (IML-CZO), a multi-institutional collaborative effort. Partial support from the following NSF grants are also acknowledged: CBET 1209402, ACI 1261582, EAR 1417444, ICER 1440315, ACI 1429699, DEB 1263559, EAR 1505309 and EAR 1748573. A grant from Joint Institute between Zhejiang University and University of Illinois at Urbana-Champaign, and collaborative effort with Institute of Earth Environment, Chinese Academy of Sciences, Xi'an, China are also acknowledged. Careful and critical edits to this submission by Charu G. Kumar and Allison Goodwell are gratefully acknowledged.

Art for Fig. 1 bottom right was created by Pamela Burroff-Murr, Jennifer Meek and Phong Le with input from Timothy Filley and Praveen Kumar. Composition of overall Fig. 1 was done by Phong Le and Praveen Kumar.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at https://doi.org/10.1016/j.ancene.2018.04.002.

References

Abaci, O., Papanicolaou, A.N.T., 2009. Long-term effects of management practices on water-driven soil erosion in an intense agricultural sub-watershed: monitoring and modelling. Hydrol. Process. 23, 2818–2837. doi:http://dx.doi.org/10.1002/hyp.7380.

Abban, B., et al., 2016. An enhanced Bayesian fingerprinting framework for studying sediment source dynamics in intensively managed landscapes. Water Resour. Res. 52, 4646–4673. doi:http://dx.doi.org/10.1002/2015WR018030.

Ainsworth, E.A., Long, S.P., 2005. What have we learned from 15 years of free-air CO_2 enrichment (FACE)? A meta-analytic review of the responses of photosynthesis, canopy properties and plant production to rising CO_2. New Phytol. 165, 351–372. doi:http://dx.doi.org/10.1111/j. 1469-8137.2004.01224.x.

Allan, J.D., 2004. Landscapes and riverscapes: the influence of land use on stream ecosystems. Annu. Rev. Ecol. Evol. Syst. 35, 257–284.

Baker, R.G., et al., 1996. Holocene paleoenvironments of Northeast Iowa. Ecol. Monogr. 66, 203–234.

Basu, N.B., et al., 2010. Nutrient loads exported from managed catchments reveal emergent biogeochemical stationarity. Geophys. Res. Lett. 37 doi:http://dx.doi.org/10.1029/2010GL045168 n/a–n/a.

Baumhardt, R.L., Stewart, B.A., Sainju, U.M., 2015. North American soil degradation: processes, practices, and mitigating strategies. Sustainability 7, 2936–2960. doi:http://dx.doi.org/10.3390/su7032936. http://www.mdpi.com/2071-1050/7/3/2936.

Belmont, P., et al., 2011. Large shift in source of fine sediment in the Upper Mississippi River. Environ. Sci. Technol. 45, 8804–8810. doi:http://dx.doi.org/10.1021/es2019109.

Bettis III, E.A., Muhs, D.R., Roberts, H.M., Wintle, A.G., 2003. Last Glacial loess in the conterminous {USA}. Quat. Sci. Rev. 22, 1907–1946. doi:http://dx.doi.org/10.1016/S0277-3791(03)00169-0.

Bettis III, E.A., Benn, D.W., Hajic, E.R., 2008. Landscape evolution, alluvial architecture, environmental history, and the archaeological record of the Upper Mississippi River Valley. Geomorphology 101, 362–377. doi:http://dx.doi.org/10.1016/j.geomorph.2008.05.030.

Bettis III, E., 2007. Paleosols and wind-blown sediments – weathering profiles. In: Elias, S.A. (Ed.), Encyclopedia of Quaternary Science. Elsevier, Oxford, pp. 2114–2125.

Blair, N.E., Aller, R.C., 2012. The fate of terrestrial organic carbon in the marine environment. Annu. Rev. Mar. Sci. 4, 401–423. doi:http://dx.doi.org/10.1146/annurev-marine-120709-142717.

Blann, K.L., Anderson, J.L., Sands, G.R., Vondracek, B., 2009. Effects of agricultural drainage on aquatic ecosystems: a review. Crit. Rev. Environ. Sci. Technol. 39, 909–1001. doi:http://dx.doi.org/10.1080/10643380801977966.

Bressan, F., Papanicolaou, A.N., Abban, B., 2014. A model for knickpoint migration in first- and second-order streams. Geophys. Res. Lett. 41, 4987–4996. doi:http://dx.doi.org/10.1002/2014GL060823.

Chen, Y., et al., 2015. Balancing green and grain trade. Nat. Geosci. 8, 739–741. doi:http://dx.doi.org/10.1038/ngeo2544.

Collins, S., Wallace, L., 1990. Fire in North American Tallgrass Prairies. University of Oklahoma Press, Norman, OK.

Cotrufo, M.F., Wallenstein, M.D., Boot, C.M., Denef, K., Paul, E., 2013. The microbial efficiency-matrix stabilization (MEMS) framework integrates plant litter decomposition with soil organic matter stabilization: do labile plant inputs form stable soil organic matter? Glob. Change Biol. 19, 988–995. doi:http://dx.doi.org/10.1111/gcb.12113.

Covino, T., 2017. Hydrologic connectivity as a framework for understanding biogeochemical flux through watersheds and along fluvial networks. Geomorphology 277, 133–144. doi:http://dx.doi.org/10.1016/j.geomorph.2016.09.030.

David, M.B., McIsaac, G.F., Darmody, R.G., Omonode, R.A., 2009. Long-term changes in Mollisol organic carbon and nitrogen. J. Environ. Qual. 38, 200–211. doi:http://dx.doi.org/10.2134/jeq2008.0132.

Davis, C.A., et al., 2014. Antecedent moisture controls on stream nitrate flux in an agricultural watershed. J. Environ. Qual. 43, 1494–1503. doi:http://dx.doi.org/10.2134/jeq2013.11.0438.

Day, S.S., Gran, K.B., Belmont, P., Wawrzyniec, T., 2013. Measuring bluff erosion Part 1: Terrestrial laser scanning methods for change detection. Earth Surf. Process. Landf. 38, 1055–1067. doi:http://dx.doi.org/10.1002/esp.3353.

Dermisis, D., Abaci, O., Thanos Papanicolaou, A.N., Wilson, C.G., 2010. Evaluating grassed waterway efficiency in southeastern Iowa using WEPP. Soil Use Manag. 26, 183–192. doi:http://dx.doi.org/10.1111/j. 1475-2743.2010.00257.x.

Dold, C., et al., 2017. Long-term carbon uptake of agro-ecosystems in the Midwest. Agric. For. Meteorol. 232, 128–140. doi:http://dx.doi.org/10.1016/j.agrformet.2016.07.012.

Downing, J.A., Cherrier, C.T., Fulweiler, R.W., 2016. Low ratios of silica to dissolved nitrogen supplied to rivers arise from agriculture not reservoirs. Ecol. Lett. 19, 1414–1418. doi:http://dx.doi.org/10.1111/ele.12689.

Fierer, N., Schimel, J.P., Holden, P.A., 2003. Variations in microbial community composition through two soil depth profiles. Soil Biol. Biochem. 35, 167–176. doi:http://dx.doi.org/10.1016/S0038-0717(02)00251-1.

Flint, R.F., 1949. Pleistocene drainage diversions in South Dakota. Geogr. Ann. 31, 56–74.

Foufoula-Georgiou, E., Takbiri, Z., Czuba, J.A., Schwenk, J., 2015. The change of nature and the nature of change in agricultural landscapes: hydrologic regime shifts modulate ecological transitions. Water Resour. Res. 51, 6649–6671. doi:http://dx.doi.org/10.1002/2015WR017637.

Gabriel, C.-E., Kellman, L., 2014. Investigating the role of moisture as an environmental constraint in the decomposition of shallow and deep mineral soil organic matter of a temperate coniferous soil. Soil Biol. Biochem. 68, 373–384. doi:http://dx.doi.org/10.1016/j.soilbio.2013.10.009.

Gentry, L.E., David, M.B., Below, F.E., Royer, T.V., McIsaac, G.F., 2009. Nitrogen mass balance of a tile-drained agricultural watershed in east-central Illinois. J. Environ. Qual. 38, 1841–1847. doi:http://dx.doi.org/10.2134/jeq2008.0406.

Gomez-Velez, J.D., Harvey, J.W., Cardenas, M.B., Kiel, B., 2015. Denitrification in the Mississippi River network controlled by flow through river bedforms. Nat. Geosci. 8, 941–945.

Grimley, D.A., et al., 2017. Using magnetic fly ash to identify post-settlement alluvium and its record of atmospheric pollution, central {USA}. Anthropocene doi:http://dx.doi.org/10.1016/j.ancene.2017.02.001.

Guanter, L., et al., 2014. Global and time-resolved monitoring of crop photosynthesis with chlorophyll fluorescence. Proc. Natl. Acad. Sci. 111, E1327–E1333. doi:http://dx.doi.org/10.1073/pnas.1320008111.

Harden, J.W., et al., 2008. Soil erosion: data say C sink. Science 320, 178–179. doi:http://dx.doi.org/10.1126/science.320.5873.178.

Hernandez-Soriano, M.C., et al., 2016. Long-term effect of biochar on the stabilization of recent carbon: soils with historical inputs of charcoal. GCB Bioenergy 8, 371–381. doi:http://dx.doi.org/10.1111/gcbb.12250.

Hewes, L., Frandson, P.E., 1952. Occupying the wet prairie: the role of artificial drainage in Story County, Iowa. Ann. Assoc. Am. Geogr. 42, 24–50.

Hooke, R.L., 2000. On the history of humans as geomorphic agents. Geology 28 (2000), 843–847. doi:http://dx.doi.org/10.1130/0091-7613.

Hudiburg, T.W., et al., 2016. Impacts of a 32-billion-gallon bioenergy landscape on land and fossil fuel use in the US. Nat. Energy 1, 15005 EP -.

Jarvis, N.J., 2007. A review of non-equilibrium water flow and solute transport in soil macropores: principles, controlling factors and consequences for water quality. Eur. J. Soil Sci. 58, 523–546. doi:http://dx.doi.org/10.1111/j. 1365-2389.2007.00915.x.

Kemmis, T., Bettis III, E., Hallberg, G., 1992. Quaternary Geology of Conklin Quarry: Iowa Geological Survey Guidebook Series No. 13. Iowa Department of Natural Resources.

Knox, J.C., 1987. Historical valley floor sedimentation in the Upper Mississippi Valley. Ann. Assoc. Am. Geogr. 77, 224–244. doi:http://dx.doi.org/10.1111/j.1467-8306.1987.tb00155.x.

Kumar, P., 2013. Hydrology: seasonal rain changes. Nat. Clim. Change 3, 783–784.

Landwehr, K., Rhoads, B.L., 2003. Depositional response of a headwater stream to channelization, East Central Illinois, USA. River Res. Appl. 19, 77–100. doi:http://dx.doi.org/10.1002/rra.699.

Le, P.V.V., Kumar, P., Drewry, D.T., 2011. Implications for the hydrologic cycle under climate change due to the expansion of bioenergy crops in the Midwestern United States. Proc. Natl. Acad. Sci. 108, 15085–15090. doi:http://dx.doi.org/10.1073/pnas.1107177108.

Lehmann, J., et al., 2011. Biochar effects on soil biota – a review. Soil Biol. Biochem. 43, 1812–1836. doi:http://dx.doi.org/10.1016/j.soilbio.2011.04.022.

Liang, C., Cheng, G., Wixon, D.L., Balser, T.C., 2011. An absorbing Markov chain approach to understanding the microbial role in soil carbon stabilization. Biogeochemistry 106, 303–309. doi:http://dx.doi.org/10.1007/s10533-010-9525-3.

Lin, H., Hopmans, J.W., Richter, D.D., 2011. Interdisciplinary sciences in a global network of critical zone observatories. Vadose Zone J. 10, 781–785. doi:http://dx.doi.org/10.2136/vzj2011.0084.

Liu, W., et al., 2012. Storage, patterns, and control of soil organic carbon and nitrogen in the northeastern margin of the Qinghai–Tibetan Plateau. Environ. Res. Lett. 7, 035401.

Matson, P.A., Parton, W.J., Power, A.G., Swift, M.J., 1997. Agricultural intensification and ecosystem properties. Science 277, 504–509. doi:http://dx.doi.org/10.1126/science.277.5325.504.

McIsaac, G.F., David, M.B., Gertner, G.Z., 2016. Illinois river nitrate-nitrogen concentrations and loads: long-term variation and association with watershed nitrogen inputs. J. Environ. Qual. 45, 1268–1275. doi:http://dx.doi.org/10.2134/jeq2015.10.0531.

Mickelson, D.M., Colgan, P.M., 2003. The southern Laurentide Ice Sheet. The Quaternary Period in the United States, vol. 1 of Developments in Quaternary Sciences. Elsevier, pp. 1–16.

Milly, P.C.D., et al., 2008. Stationarity is dead: whither water management? Science 319, 573–574. doi:http://dx.doi.org/10.1126/science.1151915.

Mulholland, P.J., et al., 2008. Stream denitrification across biomes and its response to anthropogenic nitrate loading. Nature 452, 202–205. doi:http://dx.doi.org/10.1038/nature06686.

Neal, C., Anders, A., 2015. Suspended sediment supply dominated by bank erosion in a low-gradient agricultural watershed, Wildcat Slough, Fisher, Illinois, United States. J. Soil Water Conserv. 70, 145–155. doi:http://dx.doi.org/10.2489/jswc.70.3.145.

Ojima, D.S., Schimel, D.S., Parton, W.J., Owensby, C.E., 1994. Long- and short-term effects of fire on nitrogen cycling in tallgrass prairie. Biogeochemistry 24, 67–84. doi:http://dx.doi.org/10.1007/BF02390180.

Papanicolaou, A.N., Sanford, J.T., Dermisis, D.C., Mancilla, G.A., 2010. A 1-D morphodynamic model for rill erosion. Water Resour. Res. 46 doi:http://dx.doi.org/10.1029/2009WR008486 n/a–n/a.

Papanicolaou, A.N.T., et al., 2015a. From soilscapes to landscapes: a landscape-oriented approach to simulate soil organic carbon dynamics in intensively managed landscapes. J. Geophys. Res.: Biogeosci. 120, 2375–2401. doi:http://dx.doi.org/10.1002/2015JG003078.

Papanicolaou, A.T.N., et al., 2015b. Spatial variability of saturated hydraulic conductivity at the hillslope scale: understanding the role of land management and erosional effect. Geoderma 243–244, 58–68. doi:http://dx.doi.org/10.1016/j.geoderma.2014.12.010.

Patterson, C., et al., 2003. Contrasting glacial landscapes created by ice lobes of the southern Laurentide Ice Sheet. Quaternary Geology of the United States, INQUA 135-153 .

Pielou, S.C., 1991. After the Ice Age: The Return of Life to Glaciated North America. University of Chicago Press.

Reicosky, D., Lindstrom, M., Schumacher, T., Lobb, D., Malo, D., 2005. Tillage-induced CO_2 loss across an eroded landscape. Soil Tillage Res. 81, 183–194. doi:http://dx.doi.org/10.1016/j.still.2004.09.007.

Reynolds, J.W., 1994. The distribution of the earthworms (Oligochaeta) of Indiana: a case for the Post Quaternary Introduction Theory for megadrile migration in North America. Megadrilogica 5, 13–32.

Rhoads, B.L., Herricks, E.E., 1996. Naturalization of headwater streams in Illinois. In: Brookes, A., Shields, F.D.JJr. (Eds.), River Channel Restoration: Guiding Principles for Sustainable Projects. John Wiley & Sons, Chichester, West Sussex, UK, pp. 331–367.

Rhoads, B.L., Schwartz, J.S., Porter, S., 2003. Stream geomorphology, bank vegetation, and three-dimensional habitat hydraulics for fish in midwestern agricultural streams. Water Resour. Res. 39 doi:http://dx.doi.org/10.1029/2003WR002294 n/a–n/a.

Rhoads, B.L., Lewis, Q.W., Andresen, W., 2016. Historical changes in channel network extent and channel planform in an intensively managed landscape: natural versus human-induced effects. Geomorphology 252, 17–31. doi:http://dx.doi.org/10.1016/j.geomorph.2015.04.021.

Ricciardi, A., Rasmussen, J.B., 1999. Extinction rates of North American Freshwater Fauna. Conserv. Biol. 13, 1220–1222. doi:http://dx.doi.org/10.1046/j. 1523-1739.1999.98380.x.

Richardson, M., Kumar, P., 2017. Critical zone services as environmental assessment criteria in intensively managed landscapes. Earth's Future 5, 617–632. doi:http://dx.doi.org/10.1002/2016EF000517.

Roque-Malo, S., Kumar, P., 2017. Patterns of change in high frequency precipitation variability over North America. Sci. Rep. 7, 10853. doi:http://dx.doi.org/10.1038/s41598-017-10827-8.

Rovey, C.W., McLouth, T., 2015. A new synthesis of pre-Illinoian till stratigraphy in the central United States: Iowa, Nebraska and Missouri. Quat. Sci. Rev. 126, 96–111. doi:http://dx.doi.org/10.1016/j.quascirev.2015.08.024.

Royer, T.V., David, M.B., Gentry, L.E., 2006. Timing of riverine export of nitrate and phosphorus from agricultural watersheds in Illinois: implications for reducing nutrient loading to the Mississippi River. Environ. Sci. Technol. 40, 4126–4131. doi:http://dx.doi.org/10.1021/es052573n.

Santin, C., Doerr, S.H., Preston, C.M., Gonzalez-Rodriguez, G., 2015. Pyrogenic organic matter production from wildfires: a missing sink in the global carbon cycle. Glob. Change Biol. 21, 1621–1633. doi:http://dx.doi.org/10.1111/gcb.12800.

Schauberger, B., et al., 2017. Consistent negative response of US crops to high temperatures in observations and crop models. Nat. Commun. 8, 13931.

Scheffer, M., 2009. Critical Transitions in Nature and Society. Princeton University Press.

Schilling, K.E., Helmers, M., 2008. Effects of subsurface drainage tiles on streamflow in Iowa agricultural watersheds: exploratory hydrograph analysis. Hydrol. Process. 22, 4497–4506. doi:http://dx.doi.org/10.1002/hyp.7052.

Schmidt, M.W.I., et al., 2011. Persistence of soil organic matter as an ecosystem property. Nature 478, 49–56. doi:http://dx.doi.org/10.1038/nature10386.

Schoeneberger, P.J., Wysocki, D.A., 2005. Hydrology of soils and deep regolith: a nexus between soil geography, ecosystems and land management. Geoderma 126, 117–128. doi:http://dx.doi.org/10.1016/j.geoderma.2004.11.010.

Secchi, S., Gassman, P.W., Jha, M., Kurkalova, L., Kling, C.L., 2011. Potential water quality changes due to corn expansion in the Upper Mississippi River Basin. Ecol. Appl. 21, 1068–1084. doi:http://dx.doi.org/10.1890/09-0619.1.

Singh, N., Abiven, S., Torn, M.S., Schmidt, M.W.I., 2012. Fire-derived organic carbon in soil turns over on a centennial scale. Biogeosciences 9, 2847–2857. doi:http://dx.doi.org/10.5194/bg-9-2847-2012.

Skaggs, R.W., Brevé, M.A., Gilliam, J.W., 1994. Hydrologic and water quality impacts of agricultural drainage. Crit. Rev. Environ. Sci. Technol. 24, 1–32. doi:http://dx.doi.org/10.1080/10643389409388459.

Smil, V., 2004. Enriching the Earth: Fritz Haber, Carl Bosch, and the Transformation of World Food Production. MIT Press.

Sperow, M., Eve, M., Paustian, K., 2003. Potential soil C sequestration on U.S. agricultural soils. Clim. Change 57, 319–339. doi:http://dx.doi.org/10.1023/A:1022888832630.

Stavi, I., Lal, R., 2011. Variability of soil physical quality in uneroded, eroded, and depositional cropland sites. Geomorphology 125, 85–91. doi:http://dx.doi.org/10.1016/j.geomorph.2010.09.006.

Sutarto, T., Papanicolaou, A.N.T., Wilson, C.G., Langendoen, E.J., 2014. Stability analysis of semicohesive streambanks with concepts: coupling field and laboratory investigations to quantify the onset of fluvial erosion and mass failure. J. Hydraul. Eng. 140 doi:http://dx.doi.org/10.1061/(ASCE)HY. 1943-7900.0000899.

Transeau, E.N., 1935. The prairie peninsula. Ecology 16, 423–437.

Trimble, S., 2013. Historical Agriculture and Soil Erosion in the Upper Mississippi Valley Hill Country. CRC Press.

USDA. United States Department of Agriculture, National Agricultural Statistics Service. https://www.nass.usda.gov/Publications/Trends_in_U.S._Agriculture/.

Vaughn, C.C., 2010. Biodiversity losses and ecosystem function in freshwaters: emerging conclusions and research directions. BioScience 60, 25. doi:http://dx.doi.org/10.1525/bio.2010.60.1.7.

Wang, Z., et al., 2013. Soil organic carbon mobilization by interrill erosion: insights from size fractions. J. Geophys. Res.: Earth Surf. 118, 348–360. doi:http://dx.doi.org/10.1029/2012JF002430.

White, A.B., Kumar, P., Saco, P.M., Rhoads, B.L., Yen, B.C., 2004. Hydrodynamic and geomorphologic dispersion: scale effects in the Illinois River Basin. J. Hydrol. 288, 237–257. doi:http://dx.doi.org/10.1016/j.jhydrol.2003.10.019.

Williams, M.R., King, K.W., Ford, W., Buda, A.R., Kennedy, C.D., 2016. Effect of tillage on macropore flow and phosphorus transport to tile drains. Water Resour. Res. 52, 2868–2882. doi:http://dx.doi.org/10.1002/2015WR017650.

Wilson, H.F., Xenopoulos, M.A., 2009. Effects of agricultural land use on the composition of fluvial dissolved organic matter. Nat. Geosci. 2, 37–41. doi:http://dx.doi.org/10.1038/ngeo391.

Woo, D.K., Kumar, P., 2016. Mean age distribution of inorganic soil-nitrogen. Water Resour. Res. 52, 5516–5536. doi:http://dx.doi.org/10.1002/2015WR017799.

Woo, D.K., Kumar, P., 2017. Role of micro-topographic variability on the distribution of inorganic soil-nitrogen age in intensively managed landscape. Water Resour. Res. 53, 8404–8422. doi:http://dx.doi.org/10.1002/2017WR021053.

Woo, D.K., Quijano, J.C., Kumar, P., Chaoka, S., Bernacchi, C.J., 2014. Threshold dynamics in soil carbon storage for bioenergy crops. Environ. Sci. Technol. 48, 12090–12098. doi:http://dx.doi.org/10.1021/es5023762.

Yan, Q., Iwasaki, T., Stumpf, A., Belmont, P., Parker, G., Kumar, P., 2017. Hydrogeomorphological differentiation between floodplains and terraces. Earth Surf. Process. Landf. doi:http://dx.doi.org/10.1002/esp.4234 n/a–n/a.

Critical Transition in Critical Zone of Intensively Managed Landscapes: Supplementary Information

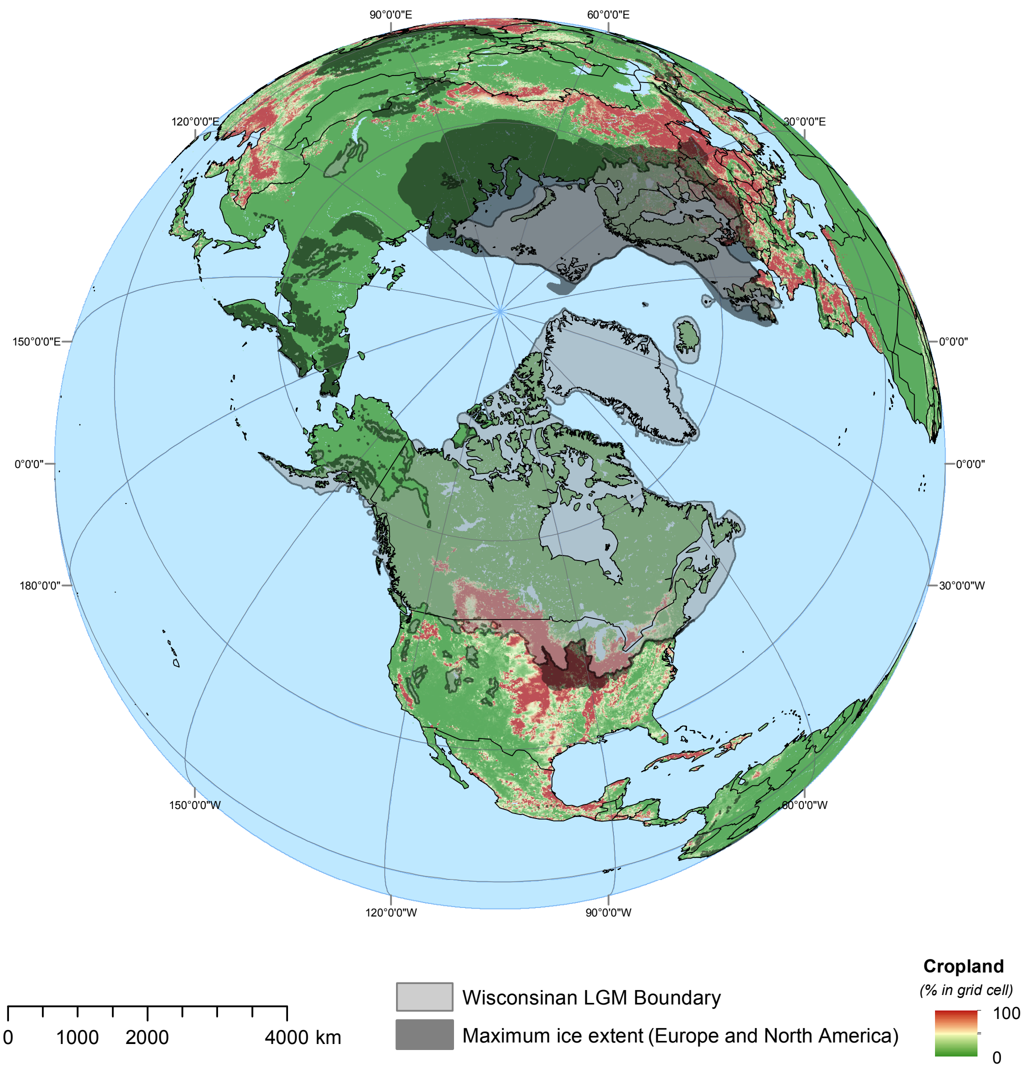

Figure S1. Illustration of the maximum glacial extent along with last glacial maximum (LGM) overlain on the modern day agricultural land. The colors indicate the intensity of agricultural activity identified as fraction of a 10×10 km^2 area under cropland[4]. Glacial extent data synthesized from a variety of sources[1–4] and map produced using Arcmap v. 10.4

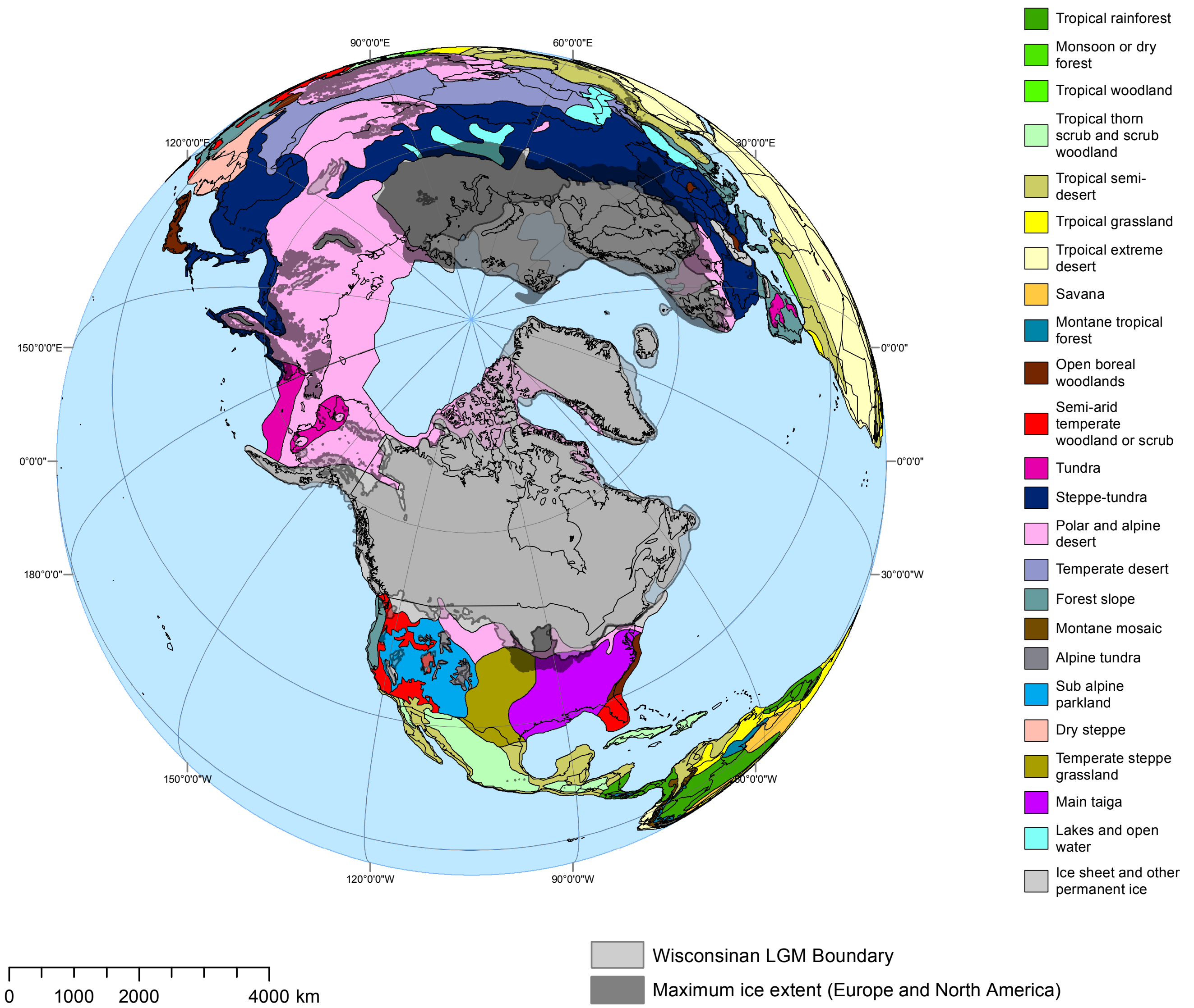

Figure S2. Illustration of the spatial extent of prevailing biomes at the time of last glacial maximum. Data from Internet Archeology[5].

References

1. Jürgen Ehlers, P. L. G. Quaternary glaciations extent and chronology part I: Europe. vol. 2(1) of *Developments in Quaternary Sciences*, 1–475 (Elsevier, 2004).
2. Fullerton, D. S., Bush, C. A. & Pennell, J. N. Map of surficial deposits and materials in the eastern and central United States (east of 102° West longitude). U.S. Geological Survey Geologic Investigations Series I-2789, 1–48 (United States Geological Survey, 2003).
3. Ehlers, J., Gibbard, P. L. & Hughes, P. D. In *Quaternary Glaciations - extent and chronology: a closer look*, vol. 15 of *Developments in Quaternary Sciences*, 1–1108 (Elsevier, 2004).
4. Ramankutty, N., Evan, A. T., Monfreda, C. & Foley, J. A. Farming the planet: 1. Geographic distribution of global agricultural lands in the year 2000. *Global Biogeochemical Cycles* **22**, n/a–n/a (2008). DOI 10.1029/2007GB002952.
5. Ray, N. & Adams, J. M. A GIS-based Vegetation Map of the World at the Last Glacial Maximum (25,000-15,000 BP). *Internet Archeology* (2001). URL `http://intarch.ac.uk/journal/issue11/rayadams_toc.html`.