A transition management framework to stimulate a circular phosphorus system

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Abstract

As the global population is projected to increase by two billion people by 2050, so will the demand for phosphorus (P), an essential nutrient for all living organisms and a major driver of eutrophication. To sustainably meet these challenges, we apply the conceptual framework of transition management (TM) to demonstrate how the trajectory of the current linear P use system could be strategically shifted toward a more circular P system. We present US case studies to examine P transitions management in intensive agriculture, wastewater disposal, and food waste management. Our goal is twofold. By first understanding past transitions in P management in the USA, we can build upon these insights for future management. This can then be applied to other global regions such as developing countries to bypass stages of transition as they intensify agriculture, incorporate sewers into cities, and expand waste management, to avoid becoming entrenched in unsustainable P management. We suggest how spaces for experimentation and collaboration can be created, how and which actor networks can be mobilized, and what action strategies and policies can be recommended to accelerate their transition to P sustainability. Our case studies show that while substantial improvements have been made, the transition toward a circular economy of P is far from complete. Our findings point to the value of utilizing TM for future progress in the US Development of TM frameworks for managing P in other regions of the world may enable them to achieve sustainable P development faster and more effectively than the USA.

Keywords Circular economy · Phosphorus · Transition management · Transition pathways

1 Introduction

As the global population is projected to increase by two billion people by 2050 (Food & Agriculture Organization of the United Nations, 2017), so will the demand for and potential environmental implications of phosphorus (P; Hunter et al., 2017). P is an essential constituent of all life forms, yet its management poses a complex trilemma (Obersteiner

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et al., 2013). The first dimension of this trilemma arises from the principal role of P as a non-substitutable input for crops, and hence critical for food and biofuel security. The second dimension relates to how P as fertilizer is applied to soil; if applied improperly, P fertilizers can leach or run off the land and become a pollutant, leading to eutrophication of water bodies (Kleinman et al., 2011). The third dimension relates to the finite nature of P reserves and their spatially disproportionate distribution (Jasinski, 2017). These three dimensions of P extraction, use, and disposal are generally separated in time and space, and involve different actors who have attempted so far to address these challenges within silos as a linear system. Thus, for instance, the P price hike in 2008 precipitated a global food crisis, which seen in the broader context of growing geopolitical instability surrounding P reserves in Morocco, heightened awareness about the long-term vulnerability of our food systems to P scarcity (Jacobs et al., 2017). Yet attempts to address these P scarcity and food insecurity challenges over the last decade have remained fragmented and disconnected from contemporaneous incidents of eutrophication due to inefficient use of P on soils, which increasingly threaten our aquatic ecosystems and public health (Brooks et al., 2016; Hoppe et al., 2016; Pick, 2016; Shen et al., 2019). To address this societal problem, we need sustainable solutions to improving P use efficiency along the whole food chain, and specifically within agricultural systems since farming is the main driver of the global P cycle (Nedelciu et al., 2020; Nesme & Withers, 2016; Sarvajayakesavalu et al., 2018).

As our experience with the COVID-19 pandemic has revealed, we must take heed of early warning signs—as these call for rapid coordinated action across sectors—before critical thresholds are crossed. These disruptive hazards cause P-deficient regions to be more vulnerable when the global or regional food trade is interrupted, which can be catastrophic for regions already impacted by food insecurity (Laborde et al., 2020; Nedelciu et al., 2020; WFP, 2020). Our paper is motivated by the urgent imperative for P sustainability through the transformation of the current—mostly linear P system—to a more circular system, where there is greater coordination among the actors and where P is recycled and maintained within regional systems for beneficial reuse. However, transforming the current linear P system into a more circular P system is a wicked problem that cuts across social, ecological, and technological domains; involves actors with competing interests and values; and leads to large-scale, long-lasting, and uncertain effects (Geels, 2005a; Jacobs et al., 2017; Shen et al., 2019). Wicked or persistent problems of this nature (Rittel & Weber, 1973) can be traced back to systemic failures that arise from locked-in flaws and path dependencies (Rotmans, 2005; Rotmans et al., 2001).

Resolving systemic failures requires a fundamental restructuring of society. Transitions are “processes of structural change in societal (sub-) systems” that emerge when the dominant structures are put under pressure either by external changes or through a process of endogenous innovation (Loorbach, 2010, p. 166). Over the past 200 years, key transitions include the technological advancement from sailing ships to steamships (Geels, 2002), from cesspools to modern sewer systems (Geels, 2006), and from horse carriages to automobiles (Geels, 2005b). These transitions have followed different trajectories under varied contexts and have been systematically studied within a growing literature on “transition pathways” (TP), with the objective of providing guidance for the wicked problems that we currently face.

In studying these transitions, Rotmans (2005) observed that not all of these transitions have been in the direction of enhancing sustainability. Most of these past transitions have taken a long time, often decades, to unfold, while most of the problems we face currently are of a more urgent nature. This realization has spurred research on new forms of governance, broadly categorized as transition management (TM) or transition governance. Given

alternative TPs that may be plausible, the term “management” used here is derived from an explicitly normative orientation of sustainability: defining *where* we want to be, *what ought* to be done, and *how* it should be done (Hoppe et al., 2016; Nelson & Vucetich, 2012; Sarvajayakesavalu et al., 2018; Shen et al., 2019). This is in contrast to much sustainability research, which simply points to the need for change and offers prescriptive policies (Dieleman et al., 2019; Ross & Omelon, 2018).

The goal of this paper is to synthesize and apply concepts and learnings from the emerging field of TM to systematically explore (and make specific recommendations on) how to shift the trajectory of the current linear P system to a more sustainable, circular P system. Although past research—stretching back to more than a decade—has drawn attention to the broken P cycle (Cordell & White, 2014; Cordell et al., 2009; Elser & Bennett, 2011) and identified alternative transition pathways (Jacobs et al., 2017), past research offers limited evidence-based guidance on how this transformation can be purposively governed over a reasonable time frame to avert an imminent P crisis. To this end, the specific objectives of the paper are to: (i) apply concepts from the TM literature to past transitions in US P management to develop a first of its kind conceptual framework for transition management of linear to circular P system that integrates across different parts of the P cycle and across different levels (landscape, regime and niche); (ii) apply this framework to explore future P transitions; and finally, (iii) assess how future transitions to a more sustainable P system can be governed using the TM framework. By first understanding past transitions in P management in the USA, we can build upon these insights for future management. Keeping in mind that the global imbalances in P stocks requires differentiated actions (Nesme & Withers, 2016), these insights might then be applied to other global regions, such as developing countries, to bypass stages of transition as they intensify agriculture, incorporate sewers into cities, and expand waste management, to avoid becoming entrenched in unsustainable P management. Based on the literature and our team’s association with several action research fora for P sustainability, we suggest how spaces for experimentation and collaboration can be created, how and which actor networks can be mobilized, and what action strategies and policies can be recommended to implement change at a somewhat accelerated pace, given the urgency of the P sustainability problem.

2 Conceptual framework for transition management

In this section, we develop a conceptual framework for the TM of the P system. Addressing the question of *how* the transition of a complex system can be accomplished, TM focuses on producing broad innovation networks including businesses, government, scientists, and citizens that have shared visions and agendas for social transformation and developing action strategies to actualize this vision and constantly monitor and learn from it.

We begin by reviewing the different concepts and perspectives from the TM literature. This includes the multi-level perspective that underlies the TM framework. We then present the TM framework itself, focusing on its dual, but closely related, functions as: (1) an analytic framework to examine past transition pathways (TPs) in order to provide insights for navigating future transitions and (2) a governance approach that outlines a deliberate process to initiate and promote societal transitions toward sustainability. We then critically review the potential and the limitations of the standard TM framework for sustainability transition of the P system and discuss how it needs to be adapted building on learnings

from two US-based case studies of P transitions in an agricultural system and urban wastewater system.

2.1 Multi-level perspective

Given the complexity of societal transformations, TM encapsulates a multi-level perspective (MLP) consisting of three independent levels: micro, meso, and macro. The *micro*-level is composed of niches, a small core of innovative actors or network of key players such as engaged community citizens, political officials, or researchers aligning themselves with a new design or demonstration. The *meso*-level is constituted by regimes that are characterized by shared cognitive rules in a community (e.g., problem agendas, search heuristics, guiding principles) that contribute to its stability (Geels, 2006). The *macro*-level refers to the landscape, which is an exogenous environment comprising of the macroeconomics, deep cultural patterns, and macro-political development that are beyond the direct influence of niche and regime actors (Geels & Schot, 2007).

Geels and Schot (2007) use the MLP to explain how transitions happen through interactions between processes at these three levels. For example, starting at the niche level (micro-level), innovators through their experiments and learning processes may build up internal momentum and garner support from powerful groups to change underlying rules or find new markets or induce behavioral or cultural changes (i.e., regime-level changes). Or it may be that changes at the landscape level (macro-level) brought about by exogenous factors can create pressure on the regime, which may destabilize the regime creating windows of opportunity for niche innovations. It is the alignment of these processes, across and between the different levels, which breaks through the inherent stability of regimes and leads to transitions.

2.2 Transition pathways (TPs)

Much of the thinking on societal change has been based on linear cause-and-effect relationships that can be managed to lead to predictable, and often desirable change. More complex environmental problems facing us today cannot be solved with this same linear thinking, but instead require the unraveling of nonlinear patterns of interdependencies and dynamic feedback loops at multiple levels within a system (Rotmans & Loorbach, 2009). Such problems defy reductionist approaches based solely or predominantly on technology or markets referred to as “a silver bullet” (Funtowicz & Ravetz, 1993). Both modern industrialized and developing societies are confronted with such complex and unstructured problems (e.g., healthcare, climate change, agriculture, energy, mobility) for which long-term solution strategies need to be developed at the societal level (Loorbach, 2010). Given the possibility of multiple equilibria, the focus of complexity literature has been to show how technological biases, institutional barriers, and path dependencies may lock a system into a suboptimal or undesired equilibrium, similar to what has been exhibited in the current P management system (Weber & Rohrer, 2012; Hoppe et al., 2016; Sarvajayakesavalu et al., 2018; El Bilali, 2020; Nedelciu et al., 2020).

Despite these tendencies for lock-ins, there are examples in history of how societies have transitioned from one stable system to another. The objective of the TP literature is to analyze these transitions in a systematic way using insights from complexity theory. “Pathways” are the sequence of events or timing of interventions that lead to a specific process of change, given the multiple possibilities. An understanding of these “pathways” from

the past provides greater insight into the dynamics of a complex adaptive societal system, which hopefully could offer a basis for thinking forward into the feasibility of directing and influencing it toward a desirable state, resulting in “intentionally driven” change or transition management.

Geels and Schot (2007) proposed a typology to systematically understand the different types of TPs that have occurred over history. In Table 1, we have coded the main characteristics of these TPs. We provide one example drawn from some well-known historical transitions and a second example specifically related to P transition, which we also elaborate on in later sections. This typology is built on four basic characteristics. The first characteristic is the type of environmental change that triggers the transition, which can be distinguished by the frequency, speed, scope, and amplitude. The second characteristic is defined by the main actors at the niche, regime, or landscape levels. The third characteristic relates to the timing of multi-level interactions. For example, as Geels and Schot (2007) point out, if landscape pressure occurs at a time when niche innovations are not yet fully developed, the transition path will be different than when they are fully developed and can take advantage of the opening of windows of opportunity. The final characteristic refers to the type of interactions among actors, which could be competitive, reinforcing, or synergistic.

2.3 Transition management (TM)

The concept of TPs leads to the idea of TM, also referred to as transition governance. This governance approach recognizes complexity and uncertainty, and is sometimes referred to as “co-evolutionary management,” with focus on adjusting, adapting, and influencing (Rotmans, 2003). TM concentrates on influencing persistent societal problems. However, unlike classical management, the assumption is that there is not necessarily full control and management of these problems, but more the organization of a joint searching and learning process, adjusting and adapting in response to this learning. TM is not directly focused on a specific solution as an end product, but is more process-based, and thus explorative and design-oriented.

Although there is not one right way to investigate transitions (El Bilali, 2020), the TM framework that we use is adapted from Rotmans and Loorbach (2009), which conceptualizes four stages in a TM cycle (Fig. 1). The first stage deals with identifying the challenge and envisioning guiding principles for the needed transition. Following from the envisioning process, transition experiments are initiated by niche actors. Rotmans and Loorbach (2009) characterize these experiments as “high risk experiments with a social learning objective.” As the niche attracts enough interest, it may mobilize additional actors, leading to the development of coalitions and interest groups. Discussions, debate, and movement among all involved actors fosters the development of partnerships among the network of actors, resulting in the establishment of demonstration projects, trials by municipalities, and tax incentives, which represent regime changes for the adoption of innovative technologies in the third stage. In the final stage of this iterative cycle, short- and long-term monitoring and evaluation is critical to quantify the effectiveness of the transition process, which then feeds back to further adoption and development, influencing the wider landscape dimension.

While the TM framework—and the MLP approach it encapsulates—has been widely cited and applied to examine a wide range of sustainability transitions, recent research has also directed attention to some of its limitations. For example, Papachristos et al. (2013) emphasized the need to move beyond a single system analysis of transitions to include

Table 1 Basic characteristics of different types of transition pathways (TPs)

Type of transition pathway	Type of change that triggers transition	Main actors	Timing of interactions and Type of interactions	Examples
Technological substitution	Specific shock: very low frequency but high speed and high amplitude	Incumbent firms versus new firms	Newcomers develop novelities which lie dormant until market shocks make new technologies profitable. Competitive relationship between incumbent and new firms	(1) British transition from sailing ships to steamships (Geels, 2002) (2) Transition to chemical P fertilizers
Transformation	Disruptive change: low frequency and low speed initially, but picks momentum, high amplitude changes in one dimension	Regime actors and outside actors (e.g., social movements, scientific community)	Outside actors exert pressure; niche innovations not fully developed yet; regime actors respond through changing regime rules (e.g., guidelines on nutrient management)	(1) Dutch transition from cesspools to sewer systems (Geels, 2006) (2) Eutrophication of lakes and harmful algal blooms (HABs)
Reconfiguration	Initial trigger is same as in Transformation TP above, but it spreads more widely as symbiotic innovations take place, leading to changes in system architecture	Regime actors and suppliers	Initial innovations are developed in niches, but these innovations have symbiotic relationship with other components of the system. This may create space for sequences of component innovations, which over time add up to major reconfigurations and changes in the regime's basic architecture	(1) American transition from traditional factories to mass production. Here transition is not caused by a single technological breakthrough but by multiple symbiotic technologies (Geels, 2002) (2) Significant improvements in agricultural machinery technology in US agriculture led to symbiotic increases in row crop planting efficiency, which in turn was associated with increases in crop nutrient uptake and crop yields

Table 1 (continued)

Type of transition pathway	Type of change that triggers transition	Main actors	Timing of interactions and Type of interactions	Examples
Dealignment and realignment	Avalanche change: low frequency, high speed and amplitude, change in multiple dimensions	New niche actors	The regime experiences major internal problems, collapses, erodes and de-aligns. Incumbents lose faith in the potential of the regime to respond. Multiple novelties emerge. Given political, economic, cultural and infrastructural trends, one of these novelties may become dominant and the system then realigns	<p>(1) Dealignment of horse-based transportation leading to novelties like electric trams and automobiles, with eventual domination of automobiles in the USA (Geels, 2005b)</p> <p>(2) Hypothetical scenario of major disruption in supply of mined P, which results in some of the innovative approaches to P capture developed in niches to become dominant</p>

Fig. 1 Stages within the transition management cycle (adapted and modified from Rotmans & Loorbach, 2009; Geels & Schot, 2007)

multi-system interactions. There is also an increasing recognition of the need to address the politics of transition (Kok et al., 2021; Meadowcroft, 2009) and to ensure proposed transitions are (socially) just (Newell & Mulvaney, 2013) and are framed in ways that go beyond the more apolitical and managerial origins of the MLP (Goldthau & Sovacool, 2012). Recent scholarship has used these broader framings of the MLP to explore socio-technical transitions in a wide array of sustainability fields, such as energy (Lin et al., 2019; Zhang et al., 2020), solid waste (Gaeta et al., 2021), and bioeconomic transformation (Kuckertz et al., 2020). However, to the best of our knowledge the TM framework has not been systematically applied to examine P sustainability transitions (El Bilali, 2020). Our discussion in this paper is primarily based on Rotmans and Loorbach (2009), although we have adapted their approach as relevant, and address some of the limitations pointed above, specifically related to the need to consider multi-system interactions and the politics of transition (Hoppe et al., 2016; El Bilali, 2020).

3 A conceptual framework for phosphorus sustainability transitions

The TM framework discussed above has been developed in the context of a range of socio-technical systems. However, we argue that the P system can best be conceptualized as—what has recently come to be recognized as—a social, ecological and technological system (SETs) (Markolf et al., 2018). We show in this subsection, through several examples and two US-based case studies, the limitations of the existing TM literature because of its missing or inadequate treatment of the ecological dimension, and the interactions of this ecological dimension with technological and social dimensions. In our conceptualization of P system as SETs, P moves in time and space through different stages, each of which is characterized by complex and interconnected social, ecological, and technological dynamics. In the current system, P moves in a linear fashion from phosphate rock mining, to production of fertilizer and P supplements for animal feed, to use of these products in growing crops and animals, to harvesting crops and livestock slaughter, to food production, to human food consumption, and finally, to waste production and disposal. In this current system, there are inefficiencies in P use, very little P is recycled, and significant environmental losses and

adverse environmental impacts occur (Baker et al., 2016; Cordell & White, 2014; MacDonald et al., 2012; Scholz & Wellmer, 2015). P recovery from waste streams is limited primarily to research projects and trials rather than full-scale commercialization (Bradford-Hartke et al., 2012; Williams, 1999).

This suggests that the conceptual framework for TM of the P system needs to capture not only the interactions among the different levels (i.e., niche, regime, and landscape) but also the interactions among the different stages of the P cycle, in order to transform the current linear P system to a circular system. Figure 2, adapted from Geels (2002), outlines our first generation, conceptual framework as applied to the case of transitioning from a linear P system to a circular P system. Using the SETs lens, the P regime, depicted by the hexagon in Fig. 2, is multi-dimensional—including science and technology, ecology, economic markets, cultural influence, policy, and infrastructure—all of which play a role in the transition process.

Our conceptual framework is informed both by these general concepts and a ground-up empirical understanding, based on case studies of P transition pathways in different SETs—specifically P use efficiency in an agricultural system and P recycling in an urban system, as described in the next two subsections. These two SETs were selected because agro-ecosystems are the major consumer of P and urban ecosystems are major sites for P recycling. Both of these SETs have witnessed several transitions, by which studying could provide important lessons for the future. We applied a case study design because it allows for using multiple sources of evidence within the natural context with the goal of understanding in depth a specified phenomenon (Creswell & Creswell, 2018). According to Yin (2009), case study research is best applied when the research addresses descriptive or explanatory questions: i.e., what happened, how, and why? Specifically, we address two

Fig. 2 Multi-level perspective on transition from a linear to a circular phosphorus (P) system, adapted from Geels (2002)

questions using TPs and TM: (1) how have past transitions in P management occurred, using case studies of agricultural P fertilization and P recovery from urban waste streams, and (2) how does this knowledge inform future P transitions especially the transition from a linear P system to a circular P system?

3.1 Transitions in US agricultural crop phosphorus use efficiency

As the major consumer of P, the agricultural sector is key to understanding P sustainability, as it often dominates P stocks and flows (Rothwell et al., 2020). In this section, we examine changes in agricultural P use over the past 70 years, driven by increased pressure for efficient crop production, technological advancements, and niche innovations. For simplicity, we focus on a single metric—P use efficiency, given its salience in the technology and policy literature—and examine how “interannual fluctuations in P use efficiency (depicted in Fig. 3) can be used to delineate four specific TPs in P use in agriculture.

Efficiency analyses are key for the sustainable management of resources (Scholz & Wellmer, 2015). P use efficiency for an agricultural cropping system can be quantified as the deliberate P outputs (i.e., P harvested with crops) from a defined system boundary divided by the P inputs (i.e., manure, by-products, or conventional fertilizer) into a defined system boundary (Dibb et al., 2003; Fixen et al., 2010; Peterson et al., 2017). We consider it partial P use efficiency because it does not consider the transfer of P through the environment (i.e., leaching, erosion, or runoff). When the system is in balance and P inputs are equivalent to P outputs, P use efficiency is equal to 1.0 (i.e., 100% efficiency). In general, the larger the P use efficiency, the more efficient the system is at utilizing the available P.

A relevant agricultural crop-based case study that exemplifies four specific TPs is corn P use efficiency in Minnesota, which consistently ranks within the top five US states for total agricultural production and sales. We focus on corn, as it dominates the US landscape—its largest producer and exporter. As a response to population growth, global corn

Fig. 3 Conventional phosphorus (P) fertilizer use efficiency for harvested corn in Minnesota depicted as a bold solid line and the phosphate fertilizer price index depicted as a black dashed line (USDA, 2015; Jasinski, 2017). Four periods where an agricultural transition shift occurred (AgPT1, AgPT2, AgPT3 and AgPT4) are labeled and approximated using a gray dashed line. The Y1 axis represents partial P use efficiency, defined as the ratio of P removal by corn harvested divided by conventional P fertilizer inputs. The Y2 axis is the P price index, which is a measure of the relative price change over time with 2011 = 100

demand will continue to increase to meet food, livestock feed, and industrial needs (Tigchelaar et al., 2019). The trend of conventional P fertilizer use efficiency for corn in Minnesota is plotted to illustrate the different TPs (Fig. 3). The plotted data illustrate the trends in the P fertilizer price index and P use efficiency from the mid-1960s to the present, using actual corn data for Minnesota. To calculate the annual P use efficiency, the harvested corn acreage and yield (USDA, 2015) data were multiplied by the P_2O_5 nutrient removal rate (0.35 lb per bu; IPNI, 2014) to get the annual P output, which was divided by the total annual phosphate applied (USDA, 2015) or P input. P fertilizer price index was extracted from the USDA Economic Research Service Fertilizer Use and Price database (USDA, 2016). A P use efficiency of 1.0 reflects a cropping system where inputs are equivalent to a crop P removal; a P use less than 1.0 indicates buildup of soil P, and values greater than 1.0 indicate the drawdown of stored soil P. One limitation to note in this case study is that it is only considering conventional P fertilizer as an input since manure data is not available for the time period. However, by using the PUE from solely the conventional P fertilizer the TPs are more apparent because of the influence on conventional fertilizer costs. To achieve true P circularity in agriculture, the cropping systems must become integrated with the livestock systems to increase the recycling of manure P.

Agriculture P transition 1 (AgPT1) in Fig. 3 resulted from a postwar jump in food demand due to global population boom (i.e., demand-side landscape change), accompanied by the intensification of rock phosphate exploration in the 1950s and 1960s, which brought down the cost of phosphate fertilizers (i.e., supply-side landscape change [Van Kauwenbergh, 2010]). Regime actors included the fertilizer industry, land-grant university cooperative extension units, and agricultural producers. Most of the increased P demand was met by increased domestic rock P mining. University extension agents recommended fertilizer applications based on soil test P (STP) levels. Farmers were encouraged to add P even at STP levels beyond what crops needed and to adopt a “build-up management approach,” to avoid the risk of P limiting crop growth over the long term under complex and highly unpredictable growing conditions (Leikam et al., 2003). This dramatic behavior represents a specific shock, in TM parlance, which is a shift that occurs rarely, but rapidly and is high in intensity, but eventually dissipates with time. As a result, P use efficiency declined into the 1970s as P fertilizer application increased; peak commercial fertilizer consumption occurred in 1981 (Osteen et al., 2012). This kind of TP comes under the classification of *technological substitution* as shown in Table 1. In summary, the transition was triggered by landscape-level demand and supply factors at the time when niche innovations, in the form of chemical P fertilizers, had already been developed and tested but could not break through the stability of old regimes.

Agriculture P transition 2 (AgPT2) in Fig. 3 represents a *transformation pathway* (Table 1) resulting from a disruptive change due to emerging concerns associated with intensification of P, specifically eutrophication of surface waters. This concern arose as academic research during the 1960s and 1970s proved that P was a “limiting nutrient” that controlled eutrophication (Schindler, 1974). During the same period, the visual impacts of extreme eutrophication, with dead fish and algal scums, became embedded in the public psyche, captured in the phrase “Lake Erie is dead” (Rotman, 2017). Both of these landscape changes led the public to pressure political actors to support pollution reduction efforts, which in the agricultural arena became the promotion of agricultural “Best Management Practices” (BMPs). New knowledge on the need for improved fertilizer and manure management led to a reduction in fertilizer P inputs which resulted in an increase in P use efficiency during AgPT2, beginning in the early 1980s through the 1990s (Osteen et al., 2012; USDA, 2015). In addition, although niche innovation in the area of plant

genetics had been ongoing for several decades, there was rapid adoption of genetically engineered or modified seeds in the 1990s (Wang et al., 2015). In Minnesota, the use of genetically engineered seeds grew from less than 40% in 2000, to more than 90% in 2015. Adoption of these insect-resistant and herbicide tolerant seeds also reduced labor hours, enabling farmers to manage more cropland. These advancements led to improved P recovery in harvested corn to bring P use efficiency near 1.0, with recovery by soybeans above 1.0 (Osteen et al., 2012).

Agriculture P transition 3 (AgPT3) represents a *reconfiguration* pathway (Table 1) wherein initial innovations are developed in niches, but these innovations have symbiotic relationship with other components of the system, leading to major reconfigurations and changes in the regime's basic architecture. Beginning in AgPT2 and continuing into AgPT3 there were significant improvements in agricultural machinery technology. This led to increases in row crop planting efficiency by 25-fold, from 40 acres per day in 1970 to 945 acres per day in 2010 and harvesting efficiency increased from 4000 bushel per day in 1970 to 50,000 bushel per day in 2010, a 13-fold increase (MacDonald et al., 2013). As producers began to reduce their P inputs, planting technology was advancing, which encouraged improved productivity and crop nutrient uptake as crop yields increased (International Plant Nutrition Institute, 2015). The P use efficiency during AgPT3 trended above 1.0 as harvested crop P exceeded P fertilizer inputs.

Agriculture P transition 4 (AgPT4) was triggered by a specific, global shock in 2007, as the price of rock phosphate increased due to the amplified international agricultural demand and supply reduction, which can be seen in the resulting phosphate fertilizer (P_2O_5) price index, depicted as the Y2 axis (Jasinski, 2017; Fig. 3). The P price index is a measure of the relative price change over time, with 2011 equal to 100. This incentivized broader adoption of the technological innovations that had been underway throughout AgPT3. During this same time, niche actors adopted precision fertilization technology. The fertilizer industry began to develop and adopt the "4R Approach to Nutrient Stewardship," defining the "right source" of nutrients, applied at the "right rate," "right time," and in the "right place" to promote and encourage a consistent message on the responsible management of agricultural nutrients to ensure alignment with economic, social, and environmental goals (Bruulsema et al., 2009). Many farmers reduced their fertilizer inputs and improved manure P crediting to offset the rapid increase in cost while maintaining P use efficiency above 1.0 (Fig. 3). Yet, less than 40% of the manure P in the USA is recovered for crop land application (IPNI, 2014; Spiegel et al., 2020). This transition also represents a *technological substitution* pathway (Table 1), and is similar in terms of its underlying mechanisms to AgPT1, which was also triggered by a specific shock, albeit in the opposite direction, specifically, an increase in rock P prices as opposed to decrease in AgPT1.

The data compiled for this corn-based, state-level case study represent just one example that is consistent with observations in crop P use efficiency studies at other scales and regions of the world (Bruulsema et al., 2019; Scholz & Wellmer, 2015). The transition shifts occur at varying times due to scale-dependent niche actors, environmental factors, and access to innovation, but indicate that in many regions of the world, agriculture is operating close to, or exceeding 100% P use efficiency. At some point in time, the Law of Marginal Diminishing Returns will impact the efficiency. For example, in Africa and Eastern Europe, the high P use efficiency means that the soil in these countries has been depleted or "mined" due to very low fertilizer input and very low yields, whereas, in tropical regions with highly weathered soils, high soil P fixation capacity has resulted in low P plant availability. In particular, Brazil's expansion of agricultural row crop production since the 1980s, has required P input rates to exceed outputs more than twofold (Bruulsema et al., 2019).

However, as soil P accumulation continues, farmers in these tropical regions may also be able to reduce their P inputs as they adopt new management practices and intensify their production to achieve higher P use efficiency (Withers et al., 2018; Bruulsema et al., 2019).

Transition shifts make P use efficiency a tool that requires a long-term perspective and global thinking beyond a one-dimensional assessment (Scholz & Wellmer, 2015). For example, when manure P is considered, it can provide a buffer against conventional P fertilizer shortages and price fluctuations, potentially reducing the vulnerability of livestock intense areas to shocks related to global P availability (Rothwell et al., 2020). Eventually, the P shocks on grain producing areas would be felt by the livestock areas, particularly those heavily reliant on imported feeds that are dependent on conventional P fertilizer (Rothwell et al., 2020). To become truly sustainable, research and policy development must be motivated to balance the requirements for both food security and environmental safety, globally. This will require improved P use efficiency and reduced environmental P losses through the adoption of technological innovation, recycling manure P, and other resources with policy support and motivated actors (Shen et al., 2019; Vitousek & Liu, 2019). Until manure P becomes more globally recognized as a resource rather than a waste, livestock production systems will be a barrier to a circular P system (Vitousek & Liu, 2019).

3.2 Transitions in US urban phosphorus recycling

The largest P fluxes moving through cities are food and human excreta. In the USA, per capita P consumption is about 0.8 kg P/capita per year (Fissore et al., 2011) all of which is excreted, about half as urine (Egle et al., 2015) and the remainder as feces. In the USA, human excreta is either treated by onsite wastewater treatment and disposal systems (i.e., septic systems) or centralized wastewater treatment plants. In addition to food consumed, annual consumer-level food waste in the USA is 124 kg/capita (Buzby & Hyman, 2012) and likely has a higher P content than consumed food, averaging around 0.6% P for several key food waste streams (Baker et al., 2016; Fung et al., 2019), which translates to 0.74 kg P/capita per year. Some food waste is disposed to in-sink grinders or composting, but most US food waste enters the municipal solid waste stream either directly or as ash from incinerated solid waste. A small fraction is collected separately as “source-separated solid waste,” which is mostly composted for landscape use.

P transitions from the middle of the twentieth century to the present altered the amount of P in both treated effluent and biosolids (Fig. 4, with calculation basis in supplemental documents). In the 1960s, about 40% of municipal sewage received no treatment or

Fig. 4 Representative transition pattern in urban wastewater discharge from a typical sewage treatment plant, incorporating technological and regulatory changes since the 1970s

“primary” treatment intended to remove solids and disinfect wastewater. About half of the P in primary sewage is converted to biosolids (“sludge”). The first urban P transition (UPT1) occurred in the early 1970s, when Congress passed the 1972 Clean Water Act (CWA). This law was motivated by the landscape change of widespread and expanding municipal and industrial pollution. The CWA established a new regulatory framework for sewage treatment and provided federal funding to enable cities to modernize sewage treatment, reducing the number of people served by “less than secondary” treatment from 66 million in 1968 to 4 million in 2012; conversely, over the same period, the number of people receiving advanced sewage treatment increased from 4 to 128 million (EPA, 2016). UPT1 also included a technological substitution, replacing P-based detergents which contributed about half of the P to wastewater in 1970, with other formulations.

Between 1970 and 1998, 27 states also banned the use of P-based detergents (Litke, 1999). Production of sewage biosolids P remained relatively stable, balanced between the effect of improved sewage treatment (increasing biosolids P) and detergent P bans (decreasing biosolids P) (Fig. 4). The metal content of biosolids was also reduced, primarily due to industrial pre-treatment requirements, increasing the usability of biosolids for agricultural application (National Research Council, 2002).

Urban P transition 2 (UPT2) was encouraged through the adoption of enhanced biological P removal (EBPR), a technological innovation used sometimes in conjunction with iron flocculation, and a complement to nitrogen removal plants (Bott et al., 2012). This advancement began in the early 1990s and has become a common implementation for wastewater treatment plants discharging to impaired water resources (Bunce et al., 2018). Large effluent P reductions have occurred to achieve compliance in response to regulatory action in the Midwestern USA and other parts of the country where landscape pressures have been high. For example, a 75% reduction in P loads from wastewater treatment plants plus combined sewers entering the Chesapeake Bay (CBP, 2016) and a 56% reduction in P loads over 10 years from Minnesota’s wastewater treatment plants (MPCA, 2017) were both driven by regulatory pressure to reduce eutrophication. However, there are many other locations where P in wastewater effluent is still too high and detrimental to water quality (Carey & Migliaccio, 2009).

Urban P transition 3 (UPT3) is evolving, driven largely by conservation goals for water and nutrients, moving the USA toward a more circular P economy. There are four primary niche areas facilitating this transition. First, there has been a renewed interest in water reuse on a much larger scale over the past two decades, motivated primarily by the need for additional water, but also to reduce effluent discharge to surface waters (National Research Council, 2002; Garcia-Cuerva et al., 2016). In rural areas surrounded by agriculture, recycled wastewater used for crops also has a P fertilization benefit. Second, there are several technological niche activities seeking to recover P from wastewater streams for fertilizer (technological substitution in Table 1) including precipitation of P from wastewater to produce recoverable mineral forms, such as struvite (Le Corre et al., 2009; Massey et al., 2010; Peng et al., 2018) and to use incinerated biosolids ash for agriculture, either directly (Bierman & Rosen, 1994) or from extraction of P from ash and concentration of P into mineral forms. Third, urine diversion has more recently become a niche activity, recognizing that urine is a small volume (<2 L/day-capita) with high P content (ca 600 mg P/L). Urine diversion requires the use of waterless urinals and urine-diverting toilets, which means P can be more efficiently extracted from urine than from wastewater (O’Neal & Boyer, 2013; Jagtap & Boyer, 2020). Urine diversion may be especially valuable to reduce new P loads to wastewater treatment plants, allowing them to increase connections from households on septic systems or new buildings without overloading the plants (Wood et al.,

2016). Projections by van Puijenbroek et al. (2019), based on historical global patterns of sewerage development, indicate that under five development scenarios in which urban populations grow from 2010 to 2050 from 23 to 45%, sewerage grows from 25 to 77%, levels of wastewater treatment increase, and P discharges in wastewater effluent will increase from 11 to 68%.

Outside the realm of wastewater, there is an emerging niche for utilizing nutrients from food waste, which is about one-third of the US food supply. From a P perspective, food waste is rich in P (0.6%, Fung et al., 2019), yet most food waste is disposed of in landfills or incinerated in mixed waste. The beginning of a regime change regarding food was largely influenced by a paper monetizing the value of food waste (Buzby & Hyman, 2012), which catalyzed the EPA and USDA to initiate a Food Waste Challenge to advance recovery of food waste at the federal level. Approximately half of the states have also enacted state-level policies to encourage food waste recovery (www.refed.com/tools/food-waste-policy-finder/). Utilizing the nutrient content of food waste requires moving to source-separated organics waste collection, which is now a niche activity in the USA; as of 2012, only 5% of post-consumer food waste is recycled (Gunder & Bloom, 2017). The challenge and niche activities surrounding source-separated organics waste collection is similar to the challenge and niche activities surrounding urine source separation for P recovery. Although most recycled post-consume food waste is used for compost, the conversion to commercial animal feed is an emerging niche area (Fung et al., 2019). The primary motivation of food waste conversion to animal feed is to provide an inexpensive source of energy (calories) and protein to farm animals; P and other minerals are a minor benefit.

A major barrier to UPT3 is the logistics of delivering recovered P products to farms and other end users (Landry & Boyer, 2016). Most urban wastes have far lower P content than commercial P fertilizer, may contain contaminants, and the site of generation (cities) is often far away from the farms that need it (Sampat et al., 2018). Yet the recycling and recovery of P from waste streams is essential to the sustainability of global P management (Schneider et al., 2019; Withers et al., 2015). The solution to potential contaminants in sewage is often source reduction, especially for industrial contaminants, as has been done with cadmium in the USA (National Research Council, 2002). To varying degrees (depending on the waste), P from waste streams can be concentrated by water removal, incineration, or extraction of P, but with this technology comes additional barriers of both cost and compliance (Barquet et al., 2020). One limited solution to the complex logistics for delivery of recovered P products is to use the P locally in urban agriculture (Sanyé-Mengual et al., 2012). This would decrease the transportation costs for both the recovered P and urban agricultural products. Recovered P fertilizer products can serve an important niche with agronomic and environmental benefits; however, research is still needed to address the consistency across cropping systems and economic feasibility (Schneider et al., 2019). Without affordable and innovative technology, overcoming the barriers to achieving UPT3 may not be feasible (Nesme & Withers, 2016).

4 TM to a sustainable P regime

We analyzed P transitions that have occurred in agricultural and urban waste subsystems, and identified the challenges that remain in both sectors. In this section, we build on the insights gained from this analysis and apply the TM framework (Fig. 1), to explore a more deliberate and directed process of developing long-term sustainability visions linked to

desired societal transitions to implement change at a somewhat accelerated pace. Currently, there is no public institution responsible for governing global P resources. The success of any regulatory framework requires transboundary action that incorporates resource demand and multi-scale planning across existing and emerging directives to establish a global framework (Reitzel et al., 2019). Most sustainability challenges we face, specifically, the P management challenge, require urgent action; however, existing regimes of policy and institutional structures pose barriers and are resistant to change. We provide examples of ongoing initiatives and some forward-looking ideas for each stage of the TM cycle as applied to the case of P sustainability transition.

4.1 Challenge identification and niche opportunity

As described in the previous sections, P management transitions in the USA have generally been incremental. This raises two questions: (1) what landscape pressures could trigger a system reconfiguration? and (2) what might a system reconfiguration look like? Two potential landscape pressures that could trigger a major regime shift in P management include interruptions in phosphate rock production and increased occurrence of harmful algal blooms (HABs; Reitzel et al., 2019).

The spatial distribution of phosphate rock across the global landscape is not homogeneous. Although the exact quantity is not transparent, it is estimated that 90% of reserves are concentrated in six countries: Morocco and Western Sahara, China, Algeria, Syria, South Africa, and Jordan. Although worldwide reserves should last at least 260 years, projections have indicated that US reserves would be exhausted in 40 years if P use in the USA continues at the current rate (Jasinski, 2017). This spatial disproportionality could contribute to landscape pressures since P use demand is expected to grow globally, driven by increasing population, increasing per capita wealth, and hence increasing meat consumption (Barquet et al., 2020; Sampat et al., 2018; Van Vuuren, 2010). Since there is no substitute for P and it is essential for modern, intensified food production systems, an international supply interruption for a prolonged period could be catastrophic. This landscape pressure could be the impetus needed to propel P recovery through the adoption of advanced technology such as urine diversion or food waste recycling, from niche activity to a major component of the regime. The current COVID-19 pandemic illustrates how global trade, complex supply chains, and even consumer behaviors (Laborde et al., 2020) can be disrupted overnight providing a plausible scenario on how a major disruption in phosphate rock supply could occur. For example, due to phosphate plant lockdowns at the onset of the pandemic, phosphate fertilizer production rates fell to around 20–30% of the total capacity in Hubei (Marlow, 2020), China's lead producing region for diammonium phosphate and monoammonium phosphate (Shen et al., 2019).

Large-scale surface water eutrophication could also trigger a regime shift in P management, focused on reducing P in agricultural runoff and wastewater effluents. The number of dead zones in estuaries has increased from 50 sites in 1960 to over 400 sites by the early 2000s (Diaz & Rosenberg, 2008), and planetary standards for eutrophication of freshwaters by P have already been surpassed (Carpenter & Bennett, 2011). In 2017, the US National Oceanic and Atmospheric Administration measured the Gulf of Mexico's dead zone at its largest recorded size reaching over 14,000 km. Going beyond scientific circles, widespread public attention to this problem is drawn whenever crisis levels are reached in lakes that supply water to several major cities, notably Wuxi, China, supplied by Lake Taihu (Deng et al., 2014; Qin et al., 2010) and Toledo, Ohio, supplied by Lake Erie (Michalak

et al., 2013). The formation of harmful algal blooms is expected to increase with warmer spring and summer temperatures (Wheeling, 2019). This suggests that there are indicators of potential disruptive events on the horizon that provide a motivation to start envisioning a new, sustainable P regime.

China's P fertilizer industry grew rapidly between 1980–2005, and so did the detrimental impact to the land (Shen et al., 2019). As China became the world's largest producer and consumer of P fertilizers, the Ministry of Agriculture of China implemented the *Action Plan for the Zero Increase of Fertilizer Use by 2020*. Since enacted in 2015, fertilizer use has decreased and projections expect the trend will continue (Ji et al., 2020). To do so while meeting P demands, Shen et al. (2019) present a conceptual model to systematically address sustainable P management in the upstream and downstream sectors of agriculture from mineral extraction to food consumption which calls upon strong environmental protection and P resources management policies, as well as strong agricultural science and technology innovation and service policies.

An alternative example of an ongoing voluntary-based approach designed to bring about such a transformation is the 4R Nutrient Stewardship initiative implemented in the Western Lake Erie Basin (WLEB). In 2011, WLEB experienced above average spring precipitation events, which contributed to high P loss from the agricultural fields into the basin, contributing to a HAB three times larger than the next largest bloom previously on record (International Joint Commission, 2014). In 2014, a HAB in WLEB resulted in the City of Toledo issuing a "Do not Drink" alert for several days leaving over 400,000 residents without drinking water. This disruption increased the awareness of the issue and presented a niche opportunity. The agricultural industry recognized improved nutrient management as one piece of a more comprehensive approach to address this critical P issue within the basin, and proactively created the WLEB Nutrient Stewardship Advisory Committee to engage a diverse group of stakeholders in a coordinated discussion (Vollmer-Sanders et al., 2016).

4.2 Coalition development and actor mobilization

The onset of an epidemic or crisis gets knowledge into the public and builds awareness quickly because of the urgency. For example, the labor shortages, export restrictions, and consumer demands spurred by the COVID-19 health crisis rapidly shifted into a global food crisis leaving the most vulnerable at risk (Laborde et al., 2020). In order to respond to a stressor before it reaches crisis levels, the development of a coalition is critical to mobilize actors from all sectors to build upon the momentum necessary for a regime shift. Niche innovation adoption at scale requires that all key actors have a seat at the discussion table so that there could be informed debates and consensus building. Funding is often critical to establish collaborative efforts, and to promote research and development prior to the onset of a P crisis.

Currently, several P sustainability initiatives or platforms around the world are mobilizing niche and regime actors. These include the previously mentioned US WLEB Nutrient Stewardship Advisory Committee, Phosphorus Futures in Australia, the European Sustainable Phosphorus Platform, and the Sustainable Phosphorus Alliance in North America (Jacobs et al., 2017). Each of these examples bring together stakeholders and researchers from different sectors to share knowledge and experience, and generate or propose solutions. P cannot be viewed as solely a national agricultural or local environmental issue; coordination must consider the complex nature of the global P cycle and the diverse stakeholders involved (Reitzel et al., 2019). Long-term solutions

must remove the fragmentation between sectors and link management scales in order to embrace a sustainable system approach.

4.3 Project implementation and increased technological adoption

Once the challenge area has been defined and actors mobilized, the next step is to collectively design the transition agenda. The agenda is developed based on a shared understanding of the persistence of the problem, the necessity of a transition or radical change, and a set of guiding principles for the envisioned transition.

Increasing P sustainability and reducing vulnerability to P shocks will require connectivity between sectors, circularization, and minimization of P loss throughout the entire P production system (Rothwell et al., 2020). As a society, we must also focus on reducing the P demand and creating a more balanced distribution of P sources and sinks. This incorporates recapturing and recycling the P once it is in the food system, which is dependent upon an alignment among policies, economic capacity, local awareness, and cultural acceptance (Elser & Bennett, 2011; Reitzel et al., 2019; Withers et al., 2018). Within the USA, nutrient recovery technologies can help to extend nutrient recycling from local to national levels; however, the potential value of manure as a P input also depends on overcoming potential liabilities. More specifically, technologies to improve scraping, storing, hauling, and land spreading will need to improve to reduce the concerns regarding odors, pathogens, and pests (Spiegel et al., 2020). Nutrient redistribution is beyond just the scope of individual farmers and ranchers. It requires innovative and logistical solutions at the societal scale.

In the 4R Nutrient Stewardship initiative example discussed earlier, an Advisory Committee was established, consisting of a diverse set of stakeholders. This committee determined that a transparent program should be developed and led by the entities most trusted by farmers (Vollmer-Sanders et al., 2016). As a result, a voluntary certification program was launched in WLEB in 2014, and within four years the certification program extended to over one million hectares of the basin, covering approximately 40% of the farmland. In 2020, the program expanded outside of the USA, leading to the development of the Global Nutrient Stewardship Certification Council. The success of this project is an example of the transition management process; however, to achieve a truly circular system, this type of initiative must expand beyond cropping systems to fully engage the livestock industry.

4.4 Impact or effectiveness monitoring and evaluation

Given the complexity of the P system and the associated uncertainties, monitoring and evaluation (M&E) is a critical part of the learning process of transitions. In the TM framework, a distinction is made between “monitoring the transition process itself and monitoring transition management” (Rotmans & Loorbach, 2009: p. 193). The former involves monitoring physical changes in the system such as water quality sampling or tracking P fluxes through the system, while also looking out for niche developments and seeds of change. The latter involves monitoring the behavior of actors and the transition agenda itself. This M&E process must be transparent and reflexive to stimulate social learning; otherwise, actors will not be able to learn from mistakes and achievements and ongoing niche development may cease, losing focus on sustainable P management.

5 Conclusion

P management is not as salient in public debates as the management of other resources such as water and energy; however, it is critical for long-term human survival and ecosystem health that we transition from our current P regime toward a more sustainable one. We framed the challenge of transforming the current linear P system into a more circular P system as a wicked problem that cuts across social, ecological, and technological domains, and can be traced back to systemic failures and path dependencies. As such, a new conceptual framework is needed to strategically inform and guide efforts toward sustainable P management.

Ultimately, we aim to illustrate how use-inspired research in sustainability science can contribute toward effective action, leading to sustainability transitions using the critical issue of P management as the focal point. Our case studies show that US transitions in P management have been slow with limited interactions between actors and across levels or sectors, and most occurring nearly a century after basic sewerage of cities. As demonstrated by the example of Minnesota conventional corn farming (among the most intensive agricultural systems in the world), use efficiency of conventional P fertilizer has increased greatly, primarily due to improved crop genetics, planting technology, and improved fertilizer management.

Throughout the globe, increasing populations, increasing urbanization, and increasing per capita wealth (resulting in increasing meat consumption) will increase P demand. At the same time, sewerage and sewage treatment of global cities will increase. Our analysis of patterns emerging from past transitions in US agriculture and urban P subsystems contributes toward a deeper understanding of the dynamics of complex adaptive systems and provides insights into the opportunities, barriers, and conditions under which it is possible to manage future P transitions at an accelerated pace. Although global P imbalances requires differentiated actions, by understanding past transitions in US P management, we can apply these insights to other global regions, such as developing countries, to bypass stages of transition and to avoid becoming entrenched in unsustainable P management. Clearly, following the historical pattern of the USA has not yet lead to sustainable P management, and we suggest that adoption of locally tailored P-TM programs could lead to more rapid progress toward P sustainability and reduction of downstream eutrophication, quite possibly leapfrogging the lengthily US P transition.

Without a major landscape pressure, adopting a TM approach is critical if we want to shift our society's P management toward a more sustainable system and resilient future. By embracing this TM framework, we foster niche innovations, mobilize actors and networks conducting and starting new initiatives or experiments, and support enhanced M&E toward a sustainable P regime.

Supplemental calculation for P in sewage

The calculations for Fig. 4 were based on the following information:

1. The P concentration in raw sewage declined in raw wastewater from 11 mg/L to 5 mg/L following detergent P bans in the 1970s (Litke, 1999),

2. The -per capita P in raw wastewater after P ban was 1.0 in 1970s (Gakstatter et al., 1978), increasing by 12% in 2010, corresponding to increased food consumption (Baker et al., 2016).
3. P loads were calculated using an average wastewater flow of 420 L/day (Gakstatter et al., 1978).
4. Based on Metcalf and Eddy (1991), P removal rates were 18% in 1970 to 1990) and then increased to 90% in 2000 and 95% in 2010, reflecting improved P removal processes.

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Availability of data and material Original data files are available through inquiry to the corresponding author.

Declarations

Conflict of interest The authors declare that they have no conflict of interest.

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