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Governing Environmental and Economic Flows in Regional Food Systems

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ABSTRACT

Globalization, specialization, and intensification have transformed the global food system, generating material flows and impacts that span multiple scales and levels, presenting novel governance challenges. Many argue for a transition toward a sustainable food system, although the scope and specific goals are fiercely contested. Theory and method is needed to evaluate competing normative claims and build legitimacy.

In this dissertation Vermont serves as a case study to investigate how environmental and economic flows impact regional governance, focusing on efforts to manage agricultural phosphorus to achieve water quality goals. A material flow account is developed to estimate phosphorus flows embedded in commodities flowing in and out of Vermont's agricultural system from 1925-2012. The results indicate a net imbalance of phosphorus flows for the entire period, leading to the accumulation of legacy phosphorus in soils that constitutes a long-term threat to water quality. Agricultural intensification and land cover change during this period led to increased phosphorus use efficiency, livestock density, and dependency on imported feed, the largest source of phosphorus entering Vermont since the 1980s.

The evidence of persistent imbalance calls into question the effectiveness of current nonpoint source pollution policy. A critical investigation of nutrient management planning policy reveals several shortcomings: pasture is frequently excluded; many phosphorus flows that cross the farm-gate are not captured; critical information on soil phosphorus levels and runoff risk is not collected in a manner that facilitates regional governance. The integration of nutrient management plans and mass-balances is proposed as an alternative approach that can increase accountability, encourage efficiency, and facilitate management and governance, albeit within constraints imposed by Vermont's position in a globalized market for agricultural commodities.

The empirical and policy analysis is complemented by a theoretical investigation that starts from the observation that a sustainability transition inevitably entails tradeoffs amongst competing normative goals. Navigating these tradeoffs is complicated by mismatch between the reach of governance institutions and the spatial and temporal dimensions of the challenges they face. This investigation contributes to understanding how legitimacy and consensus are constructed in the context of competing normative claims and multi-level governance. It considers deliberative democracy as a means for evaluating normative claims and arriving at a shared, legitimate basis for social action. An instrumental perspective on deliberation is contrasted with a deeper notion that sees deliberation as constitutive of sustainability at a local-to-global level. A conclusion grounds this analysis by drawing out the ways in which deliberation can inform Vermont's efforts to govern its agriculture, water quality, and economic development, sowing the seeds for a sustainability transition.

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CHAPTER 1: INTRODUCTION

The nature and pattern of production, consumption, and exchange have changed considerably since the days of the classical political economists. Globalization and the evolution of capitalism have transformed the ways in which humans relate to each other and satisfy basic needs and desires. Today, it is normal for a Swede to consume bananas in the dead of winter. A business consultant in New York may end the workday by sending a draft document to a colleague in Bangalore so that it is ready for presentation at the start of the Manhattan business day. Finance and services dominate the "advanced" economies; London, Dubai, and Singapore have become playgrounds for the elite.

And yet, the commodity remains a fundamental unit of analysis, especially for investigators interested in understanding the material underbelly of social life. The commodity is the economic manifestation of society's metabolism: we survive by producing, trading, and consuming commodities, whether in their raw or refined ("value-added") form. Commodities are a benchmark currency in the exchange of matter and energy between humans and the ecosystems we inhabit. Hence, this investigation centers on an analysis of commodities.

1.1 Dissertation Overview

In this dissertation, I am concerned with the ways in which the production, consumption, and exchange of agricultural commodities impact the environment, human livelihoods, and governance at multiple levels and scales (Cash et al., 2006; Young, 2002). More broadly, I am interested in understanding how major transformations in human society – globalization, urbanization, modernization – manifest themselves physically.

Such a multi-dimensional problem (or problématique) demands a suitably multi-faceted approach; therefore, I combine a mixed methods empirical case study with social-theoretical analysis. I take a transdisciplinary approach, drawing from a diverse literature and both quantitative and qualitative empirical data to form a rich understanding of the case. Among other perspectives, I apply a geographic as well as an "ecological" political economic lens to the case study. From geography, I draw upon critical analysis of space and place to understand the ways in which the modern, globalized economy dis-embeds space from its place-based context. From political economy, I capitalize on the rich history of studying economics as part of social theory (Milonakis and Fine, 2009).

The case I investigate is that of Vermont's agricultural sector and its role in the eutrophication of Lake Champlain, a large body of freshwater shared by the states of Vermont and New York as well the Canadian Province of Quebec (see Figure 1). Lake Champlain has suffered from episodic eutrophication since the 1970s (Smeltzer et al., 2012); millions of dollars have been spent to study and "fix" the problem, with little visible success (Osherenko, 2013). At the heart lies Vermont's dairy-dominated agricultural industry, which imports feed and fertilizer and exports milk and other commodities. Contained in these commodity flows is phosphorus (P), which is both a critical, non-renewable input for agriculture as well as a driver of eutrophication (Cordell and White, 2014; Jarvie et al., 2015; Kleinman et al., 2015; Smil, 2000). Reducing P runoff into Lake Champlain is of paramount importance for improving water quality (U.S. EPA, 2015).

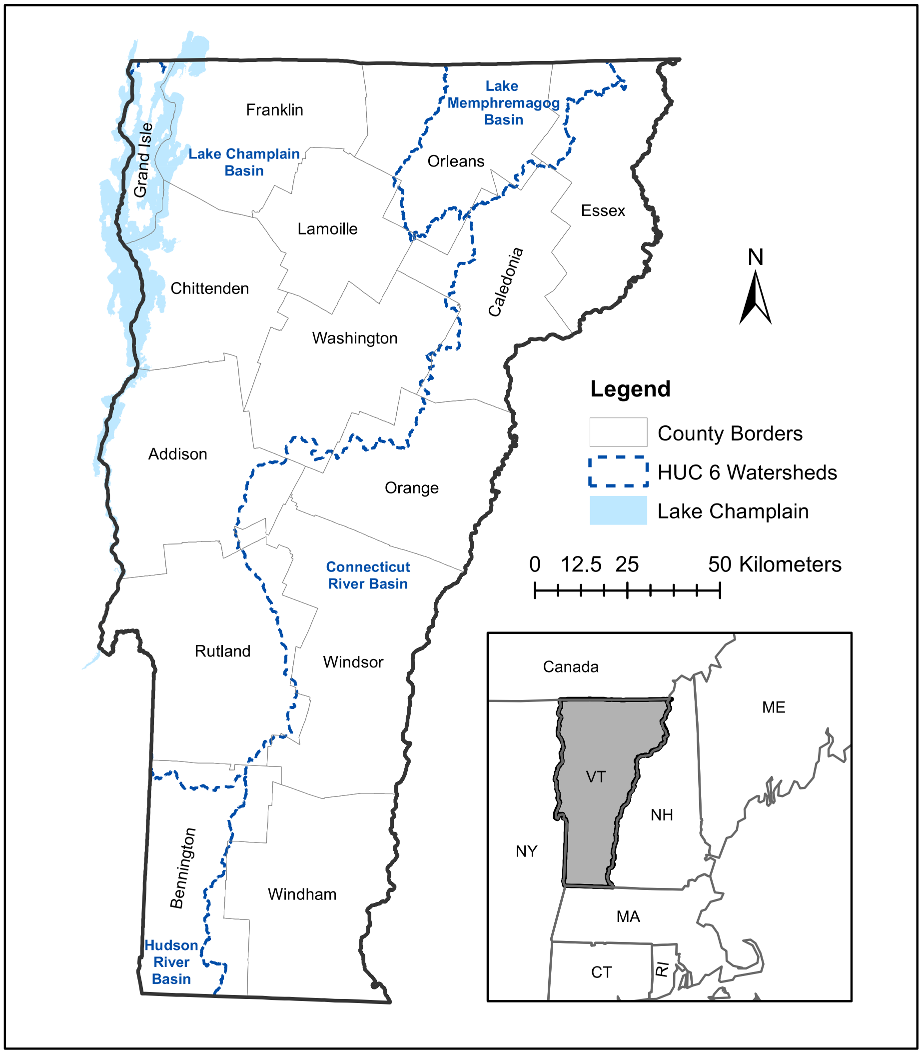

Figure 1: Map of Vermont indicating watersheds, Lake Champlain, and county borders.

Much of the discussion of the eutrophication of Lake Champlain has been framed, understandably, as a problem of poor water quality resulting from point and nonpoint sources of nutrient pollution (Lake Champlain Basin Program, 2015; U.S. EPA, 2015). This framing reflects the language of the Clean Water Act, which establishes permit requirements for point sources but defers to states to develop nonpoint source programs, most of which have historically been voluntary (Shortle et al., 2012). Proposed solutions emphasize environmental management and pollution abatement enforced as part of the State of Vermont's regulation of its waters (State of Vermont, 2016). This discourse, while technically reasonable, masks important structural complexities and dynamics that underlie the problem. My investigation is intended to reveal some of these features, situating them within a broader debate about the political, economic, and environmental dimensions of trade, food systems, and sustainability.

By starting with a lake – a tangible thing – I aim to root a complex and occasionally abstract study of globalization, food systems, commodity markets, biogeochemistry, rural landscapes, and livelihoods in a specific place. I make the case that the challenges facing Vermont typify the difficult choices facing many regions around the world that strive to balance competing economic, environmental, and social ends in a globalizing world. This investigation reveals the challenge of constructing legitimacy and grounds for action in a multi-level, multi-scale world where normative conflicts abound.

1.2. Research Context and Motivation

The State of Vermont (VT), which occupies approximately 56% of the Lake Champlain drainage basin, is under regulatory pressure from the United States (US) Environmental Protection Agency (EPA) to make progress in improving water quality in Lake Champlain (U.S. EPA, 2015). The EPA has developed a set of Total Maximum Daily Load (TMDL) limits on the amount of P that can enter each of the Lake's 13 segments; these limits are allocated to different sources (agricultural runoff, stormwater runoff, forested land runoff, wastewater treatment plant discharges, etc.) and jurisdictions (Vermont, New York, and Quebec). In response to a recent update of the TMDL, the State of Vermont established a plan of action to achieve reductions in P runoff (State of Vermont, 2016). Modeling results suggest that even with implementation of the plan it will be difficult to achieve the State's water quality standards in every segment of the Lake (U.S. EPA, 2015, 2013).

Much of the anthropogenic (cultural) nutrient pollution entering Lake Champlain in VT is attributed to farms (U.S. EPA, 2015). Runoff and leaching from farm fields and barnyards transports P into streams and eventually the Lake, where it stimulates primary production and leads to algae blooms, oxygen depletion, and fish kills (Heisler et al., 2008; Sterner, 2008). In some instances, blooms are dominated by species of cyanobacteria (blue-green algae) that are toxic and threaten human health (ibid.). The impacts of eutrophication are not trivial; Lake Champlain is a popular place for recreation, a regional tourism attraction, an important habitat for many species, and a source of freshwater for adjacent communities (Lake Champlain Basin Program, 2015). There are strong ecological, economic, and social reasons to reduce P pollution and improve water quality.

Limiting the runoff of phosphorus (P) into Lake Champlain to achieve the TMDL will require effort by wastewater treatment plant managers, road maintenance crews, developers and property owners, forestry operators, farmers, and others. This diffusion of responsibility (or "shared burden") could hypothetically mitigate political conflict; instead, it has pitted sector against sector, with farmers at the center of the conflict. For farmers, P is a critical input, essential for plant and animal nutrition and therefore integral to their business. Limiting P use could have economic repercussions.

This dissertation engages with a set of complex questions and interrelated phenomena that can be described as a problématique. This problématique is typically framed as merely a problem: the management of nonpoint source pollution which degrades water quality. This reductionist framing fails to engage with the ways in which the problem intertwines with many other contentious social, economic, and environmental issues. In exploring, identifying, describing, and modeling some of these connections, this research will form a richer picture of the policy context, including tradeoffs, potential points for intervention, and the possible consequences of policy intervention. More generally, this research will contribute to an understanding of the ways in which trade has transformed the economic and environmental policy landscape, with important consequences for democracy and social welfare.

The problématique can be illustrated by delving into some of the factors that drive poor water quality. For example, most of VT's dairy output, and therefore most agricultural output (Vermont Dairy Promotion Council, 2014a), is produced using modern confinement and semi-confinement systems that are capital and input-intensive (Winsten et al., 2010). This has allowed farmers to intensify and, at the same time, partially "dis-embed" or "decouple" themselves from the landscape, in some cases adding more cows than can be sustained by a farmer's land base (Naylor et al., 2005). The feed needed to sustain these larger and more productive herds is purchased from farms in the state as well as grain dealers who import it from elsewhere (Ghebremichael and Watzin, 2011; Soberon et al., 2015). The import of feed represents an inflow of P that is consumed by livestock and excreted as manure. The balance of inflows and outflows determines whether VT's agricultural soils accumulate or draw down their P reserves. Most agricultural regions in the US have run persistent P surpluses for decades (Jarvie et al., 2015; MacDonald et al., 2011; MacDonald and Bennett, 2009). The legacy P that accumulates in soil can exacerbate the likelihood of runoff losses (Haygarth et al., 2014; Rowe et al., 2016; Sharpley et al., 2014). Managing legacy P requires management of the P balance, something that lies outside current regulatory constructs.

Most of the milk (~85%) produced in VT is exported as either fluid milk or value-added products (e.g. cheese) (Parsons, 2010). From an economic perspective, the State is a major exporter, although its modest contribution to the national and global milk supply makes it a price taker not a price maker. The ability of farmers to pass the environmental costs of production on to consumers is limited; costs are borne by the farmers, local citizens/taxpayers, and the environment. This situation occurs in many commodity-exporting regions. If regulation is too onerous, farmers argue that they will go out of business because the break-even price rises too much. Even as rural areas depopulate, farming remains a major source of jobs for rural communities (USDA-ERS, 2016). The dual goals expressed by VT citizens – clean water and a vibrant working landscape – may be difficult to meet simultaneously.

As one delves deeper into the problématique, it becomes clear that the ability of VT policymakers to regulate its lands and waters is shaped in part by forces that lie outside its borders. Milk supply, grain prices, trade policy, changing diets, demographic shifts, new technology, and other factors shape the nature and viability of VT's farms and, in turn, the effects on its land and waters. This socio-economic complexity is superimposed upon a heterogeneous biophysical setting, where soils, slopes, hydrology and shifting land cover can make the P contribution of one field very different from another (Vadas et al., 2015, 2005). I seek to shed light on key relationships in this complex picture, at the heart of which lies tradeoffs among competing environmental, social, and economic ends. Devising governance institutions to navigate these tradeoffs is a fundamental challenge of the Anthropocene (Cash et al., 2006; Gupta et al., 2013; Liu et al., 2015).

1.3. Research Scope

This dissertation is organized in “paper format,” with three distinct parts: part one (literature review and theory); part two (empirical case study and policy analysis); part three (synthesis and conclusion). Parts one and two are written for publication as book chapters or in peer-reviewed journals.

Part one (Chapters 2 and 3) is a review and critique of ecological economics through the lens of deliberative social and political theory, which focuses on the potential for public deliberation to evaluate and achieve consensus regarding competing normative claims. This is the central challenge in applying ecological economics to real world problems: how to make decisions when facing tradeoffs among competing goals. By investigating how deliberative social and political theory can inform ecological economics, I advance the academic discourse while laying the groundwork for the synthesis and conclusion. Like much social theory, I combine analytical and normative content, seeking not only a map of the current situation but also a path forward (Callinicos, 2007).

In part two (Chapters 4 and 5), I investigate the case of water quality degradation in Lake Champlain. I ask first how the changing nature of Vermont’s agricultural economy (from 1925 to 2012) has impacted the flow of phosphorus in and out of the state. This research use material flow analysis (MFA) to calculate a P mass-balance, estimate legacy P accumulation, and situate these trends within the context of technological and structural change in the agricultural sector. This analysis highlights specific lacuna in VT's regulatory infrastructure; I use a critical analysis of existing nutrient management planning policy to point to potential interventions that can increase accountability for nutrient flows, facilitate farm-level management and regional-level governance, and mitigate legacy P.

The case study in part two provides an empirical basis that bolsters the social-theoretical argument of part one, demonstrating the limits of a techno-rational approach to problem-solving. It reframes problem(s) as problématique, demanding an approach to policy that recognizes the essentially political nature of the tradeoffs at hand as well as the challenge of environmental governance mismatch in a globalized economy. Finally, the case study research illustrates the ways in which economic globalization has outpaced the social and political structures that have, in the modern era, acted as constraints. This helps explain the Anthropocene while raising troubling questions about the ability of existing social and political institutions to manage its negative repercussions. In part three (Chapter 6), I attempt a synthesis, emphasizing policy implications and pointing to ways in which deliberative approaches can support a sustainability transition in VT's food system.

CHAPTER 2: ECOLOGICAL ECONOMICS, MODERNITY, AND THE ANTHROPOCENE CHALLENGE

2.1 Introduction: the Anthropocene Discourse and Ecological Economics

The life-support systems of earth are in decline (MEA, 2005; Pachauri et al., 2014). Humans have wrought massive change, disrupting biogeochemical cycles, transforming landscapes, eradicating species, and polluting the land, water, and air. This has led some scientists to propose a new geological epoch, the Anthropocene, that recognizes humans as a significant force in transforming the biotic and abiotic components of the earth system (Crutzen, 2002; Steffen et al., 2007). This discourse reflects widespread consensus in the scientific community that humans are affecting the earth system in ever more powerful ways, with some impacts even surpassing planetary boundaries that define a "safe operating space" for society (Rockström et al., 2009; W Steffen et al., 2015). Urgent calls have emerged demanding research and action to address myriad inter-related ecological and social challenges (Biermann et al., 2012; Guerry et al., 2015; Kates et al., 2000; Palsson et al., 2013; Steffen et al., 2011).

Determining exactly when and if the Anthropocene began remains contentious (Steffen et al., 2007; Zalasiewicz et al., 2015). Some critics have challenged whether the term is correct, in part because vast historical and present-day inequities in wealth, resource use, and pollution production make some more responsible than others for our present predicament (Malm and Hornborg, 2014; Moore, 2017; Rickards, 2015). Yet, the competing discourses recognize that global interdependencies and “teleconnections” bind people and nature in increasingly complex assemblages (Liu et al., 2015). The Anthropocene is an era marked by global-scale, fossil-fueled interconnectedness of peoples, industries, and ecosystems (Fischer-Kowalski and Anand, 2001; Liu et al., 2015; Young et al., 2006). Globalization has amplified the effects of humans on nature while rendering the governance landscape more complex (Biermann and Pattberg, 2012).

Upon first glance, the Anthropocene discourse presents little new content for ecological economics. A central concept in ecological economics since its earliest days is that human activities drive change that may surpass biophysical limits at multiple scales. Indeed, the recognition of humans as a geological force undermines the notion that economic, social, and biophysical systems can be treated as isolated fields of inquiry, affirming a central tenet of ecological economics. Some argue that disciplines like economics must be re-grounded in the political and the material, becoming once again servants of larger social goals (Brown and Timmerman, 2015; Milonakis and Fine, 2009; Palsson et al., 2013). The Anthropocene demands an understanding of the human economy as situated in its social and biophysical context, a central premise of ecological economics (Costanza et al., 2012; Daly, 1974; Daly et al., 1989; Spash, 2012).

The Anthropocene discourse is thus a vindication of the essential problem frame of ecological economics. However, despite success in analyzing and critically describing an economic system degrading its containing and sustaining ecosystem, ecological economics has struggled to turn critique into action, or praxis. This, we argue, is partly due to the failure of ecological economists to engage with political and social theory capable of bridging multiple scales and levels. The Anthropocene discourse highlights the fundamental interconnectedness of human and natural systems at local-to-global levels, raising important questions about normative legitimacy and social action.

In this paper, we critically review ecological economics in light of the Anthropocene discourse, pointing to social and political challenges that have not been well-addressed in ecological economics, in part due to limited engagement with the long-standing debates around modernity and post-modernity. Our contention is that the absence of a coherent social and political theory (or set of theories) and corresponding models for governance and transformation undermines the ability of ecological economics to contribute to social change. Addressing this shortcoming presents new opportunities for theoretical development, analysis, and collaboration while simultaneously laying the groundwork for action.

To this end, in Section 2 we review key concepts in ecological economics, elaborating its inherently normative underpinnings as science oriented toward praxis, built upon an ontology of economy-within-society-within-nature. In Section 3 we explore two defining challenges facing ecological economics in the Anthropocene epoch, drawing out the social and political implications for ecological economics praxis. In Section 4 we critically examine how ecological economics has engaged with these questions in the past, highlighting tensions and elisions in the literature. Finally, in Section 5 we point to the potential for deliberative social and political theory to serve as a foundation for ecological economics in the Anthropocene epoch.

2.2 Ecological Economics as a Normative Discipline

Ecological economics emerged as a direct challenge to the neoclassical direction and momentum within mainstream economics, attempting to reclaim the biophysical and moral areas of inquiry in economics (Røpke, 2005, 2004).[1] The impetus was the perception of an environmental crisis, attributed in part to economic activity which was perceived as both a biophysical and a social process, meaning economic growth could not continue forever, due to absolute limits (scarcity) imposed by the finite earth system (Daly, 1968; Georgescu-Roegen, 1975). The theoretical underpinnings of mainstream economics, it followed, was built on a series of utopian assumptions, capable of achieving miraculous efficiency so long as nature and society could be subsumed within a perfectly functioning market framework. This was in stark contrast to heterodox traditions that, in some instances, gave greater attention to the often fraught, political relationships connecting nature, society, and the economy (Milonakis and Fine, 2009).

Indeed, following the marginalist revolution in economics, land and natural resources were steadily downplayed or even omitted as distinct factors of production with their own particular properties (Daly and Stiglitz, 1997; Hubacek and Giljum, 2003; Parks and Gowdy, 2013). The distributional effects of markets, once a primary concern

[1] We use the term mainstream economics as synonymous with orthodox economics, which is predominately, but not exclusively, neoclassical in outlook. We do not include heterodox schools (institutional economics, evolutionary economics, etc.) under the rubric of mainstream economics. Ecological economists have adopted ideas, assumptions, approaches, etc. from many different schools of economics.

of economists as moral philosophers, were left to others. Whole categories of labor – such as the unpaid work of caring for elders or raising children – were ignored (Power, 2004). Previously broad notions of welfare and utility were equated instead with narrow exchange value. This constrained the field of inquiry and narrowed the realm of economic explanation; mainstream economics became the science of the market, with anything outside the market deemed an externality that must be commodified for inclusion within economic analysis and theory, or alternatively left to other disciplines (Milonakis and Fine, 2009). The shift from political economy to economics led, in part, to the emergence of new disciplines such as economic history and sociology, which picked up areas of study cast off by economics (ibid.).

Ecological economics, in contrast to mainstream economics, starts from the normative premise that the economy should be studied in relation to the environment in which it is situated. The economy is viewed as "embedded" within the larger social sphere – fluxes of matter and energy mark the exchange between society and nature, abetted by social institutions such as markets. The starting point for analysis is an earth system with finite resources and fixed solar income, where economic activity is both a social and a biophysical process (Daly, 1974; Georgescu-Roegen, 1975; Gowdy and Erickson, 2005). This represents a profound contrast with mainstream economics, substantially enlarging the field of study to encompass questions often left to other disciplines like ecology, sociology, political science, and philosophy. The domain is expanded, not through methodological imperialism, but through transdisciplinary scholarship that is purposely pluralistic. At the same time, the notion that economics can be an objective, positivist science is roundly rejected in favor of science oriented towards praxis (Söderbaum, 1999). Ecological economics is thus a normative discipline, motivated by a pre-analytic vision that perceives an environmental crisis with economic roots, which it seeks to both understand and avert.

The notion that the economy is subsumed within society ("economics as social theory") is one shared by many heterodox schools of economics (Milonakis and Fine, 2009). Yet, most heterodox schools fail to seriously engage with the embeddedness of the broader social system within nature, which distinguishes ecological economics. Along these lines, ecological economists[2] have written extensively about the magnitude of human reliance on the earth as an ultimate means to satisfy human needs and desires. This analysis has led many ecological economists to question the sufficiency of the price mechanism and innovation as means for allowing humanity to avoid environmental collapse (Ayres et al., 2001; Daly, 1974; Ekins et al., 2003; Farley, 2012). This argument is bolstered by the ubiquity of market failure, the complexity of ecosystems, and the challenge of valuing non-market goods (Farley, 2012, 2008). This normative argument

[2] We acknowledge that ecological economics as a discipline is far from monolithic, with multiple schools of thought and, in some cases, conflicting ontological and epistemological positions. In this paper, ideas are typically linked to individual thinkers, although on occasion general statements are made; these generalizations should be treated as the necessary simplifications that they are. Furthermore, many individuals described as ecological economists may self-associate with other transdisciplines (e.g., industrial ecology, sustainability science, political ecology, etc.).

runs counter to much work in mainstream economics and, more broadly, the neoliberal worldview of free markets, deregulation, and privatization that dominates many political debates.

On the question of moral ends, ecological economists have rejected the notion that economic growth is, in itself, an important social goal (Daly, 1974; Daly et al., 1989; Kallis et al., 2012). Even as a means for achieving social goals, growth is not a panacea; beyond certain thresholds, growth does not necessarily represent an improvement in net welfare or wellbeing (Easterlin, 1995; Max-Neef, 1995). More broadly, market exchange provides only part of that which contributes to human wellbeing (Gowdy, 2007; Muraca, 2012; Amartya K. Sen, 1999). Proposals to remedy this conflation of wealth with wellbeing have ranged from philosophical contributions – for example proposing concepts of sufficiency (Lamberton, 2005) or flourishing through a non-instrumental relationship with nature (Brown, 2007) – to broader indices for measuring welfare, such as the Index of Sustainable Economic Welfare (ISEW) (Cobb and Daly, 1989) and the Genuine Progress Indicator (GPI) (Lawn, 2003).

Contrary to neoclassical purists, the explicit normative stance of ecological economics also embraces concern for fair or just distribution of economic goods and services (Daly, 1992; Gowdy and Erickson, 2005; Spash, 2012). Given the critique of the possibility and desirability of continued material growth, attention to distribution is particularly pertinent, as growth cannot be relied upon to increase total utility (Daly, 1974; Kallis et al., 2012; Muraca, 2012). Indeed, insofar as economics can play a policy role beyond designing conditions for market efficiency, questions of distribution must be broached (Bromley, 1990). Distribution has normative and instrumental aspects, since relative inequality can undermine wellbeing and prompt a desire for increased material consumption (Easterlin, 2003, 2001).

Ecological economics rejects the notion that economics can be a value-free, objective science. Mainstream economics brings with it a host of assumptions – many unstated – about human behavior, economic exchange, the role of government, humanity's relationship with nature, and the ultimate goals of human society (Bromley, 1990; Gintis, 2000; Söderbaum, 1999; Spash, 2012). These assumptions have tremendous import for how we structure institutions, frame problems, and conceive solutions. As Bromley (1990: 104) states, "the persistent belief that economists who advocate efficiency are being objective scientists is simply wrong."

As a normative discipline, ecological economics is both descriptive (e.g., economic activity has material consequences) and prescriptive (e.g., material consequences should be minimized for the sake of humans and the rest of nature) (Faber et al., 2002). Underlying the prescriptions are normative claims (X is good, Y is bad) and models of how social change takes place. It follows that ecological economics must engage with political questions, namely how to deliberate and decide about competing normative propositions in a way that renders action socially legitimate. Similarly, the implicit models of social change merit inspection and critique. Absent this, normative positions and social models are embraced unconsciously (and uncritically), which is exactly the accusation leveled by ecological economists against the mainstream. Spash (2012: 45) captures this when describing an ongoing project to align ontology with method, aimed at the "avoidance of holding totally contradictory positions simultaneously."

To date, ecological economics has given less attention to the social and political, as opposed to the economic and ecological. This, despite early promise: Røpke (2005: 271) recounts how, for a period in the late 1980s to early 1990s, socio-economists were interested in contributing to the emerging ecological economics because of the shared "idea that the economy is embedded in society and culture and that this should influence the analysis of environmental issues." Yet, as Sneddon, Howarth, & Norgaard (2006 :261) note "the vast majority of articles in the journal Ecological Economics do not address the social and ecological implications of power relations." Similarly, Spash (2011) calls for a "social ecological economics" that would take a more socially and politically-grounded, heterodox approach to understanding the relationship between nature and the economy. This echoes M'Gonigle (1999: 12), who more than a decade earlier called for "situating the field of ecological economics within a larger ecological political economy."

This relative neglect of the social and political, we argue, is connected with the difficulty in achieving praxis, and so represents a critical area of inquiry for ecological economics. It also helps explain ongoing tension within the literature, in that competing normative claims and social theories are left unexposed and unanalyzed. This unstructured pluralism arises, in part, because ecological economics has not established a coherent identity around a set of ontological and epistemological claims that can shape normative content (Spash, 2012). Nor has ecological economics grounded itself in a specific social or political theory or set of theories that provide a structure for eliciting, understanding, and arbitrating among competing normative propositions to enable collective decision-making and action. The lack of a clear ontology and epistemology leads to ambiguity about how ecological economists would answer important normative questions, for example:

- In the ecological economy, are universal values assumed? If so, what are they?
- What is the role of democracy? Liberty?
- What theory (or theories) of justice serve as a basis for distributing the earth's resources?
- Do justice claims extend to past or future generations? What about nature?
- What role is there for nation-states in an interconnected, whole earth system in which many individual and local decisions have global consequences?
- Is global consensus needed regarding the need to respect ecological limits or can it be assumed?
- To what extent can humans steer or navigate toward sustainability, given the complexity and unpredictability of social-ecological systems?

These questions can have markedly different answers depending on the scale of analysis and whether one draws from modern, post-modern, or other social theories. Each is profoundly political. Seeking answers to these questions, we argue, is a necessary step in the evolution of ecological economics, especially if it is to help navigate the treacherous waters of the Anthropocene.

2.3 Defining Challenges for Political Economy in the Anthropocene

The Anthropocene label aptly renders the scope of the governance challenge: coordinating the decisions and actions of a “superorganism” species with the power to shape a planet, yet without a central nervous system operating under a single command. Rising to this governance challenge will require (1) negotiating the constraints imposed by biophysical reality and the multiple constructed social realities characteristic of human society; and (2) addressing the complexity, uncertainty, and conflict that arise as one moves across multiple levels and scales. These challenges are broad, representing central objects of study for the social sciences. In this section, we argue that they are of critical import to ecological economics, specifically as ecological economics seeks to move from theory and analysis to praxis, supporting the transition toward a new human-earth relationship.

2.3.1. Modern and Postmodern Perspectives

Ecological economics departs from a diagnosis, informed by research in ecology, environmental science, geography, and other disciplines, that there is an ecological crisis that threatens human society. But this diagnosis is not universally shared; questions emerge, for example, about which society or societies will be threatened, how broadly the impacts will be felt, and whether or not this is a threat, an opportunity, or a trifling affair. This reflects a fundamental tension in the ecological economics paradigm: with the tools of science, we can gain some partial form of understanding of the world, yet interpretation of the meaning, significance, and implications of the empirical results is inevitably a subjective, contestable process. We can monitor the population of an endangered species and observe that the population is declining. Yet, it does not immediately follow that we should do something about it. If we can agree that action is necessary, the question then becomes which action(s) to take. These are not scientific questions, although science can certainly inform the process.

For a normative discipline like ecological economics, the problem is two-fold: first, how to transcend or otherwise resolve competing disciplinary perspectives and methodologies in the process of producing transdisciplinary knowledge (science); second, how to create or enable social and political processes that can interpret and, if necessary, act upon the knowledge. In the first case, it may require natural and social scientists to collaborate to create models and theory that build upon and inform one another in a way that provides a basis for action (Liu et al., 2015; Palsson et al., 2013).

It requires at least some acceptance that there is a biophysical world that exists independent of humans and is describable and knowable, at least in part, through the tools of contemporary science (Spash, 2012). This may seem intuitively obvious, but disciplinary specialization and other forces have led to a major divide between the world of social theory and that of applied and natural sciences. As Callinicos (2007: 306) notes, social theory as a discourse "would benefit from a dialogue with a naturalistic conception of the world which recognizes the continuities between both the physical and social worlds … but which does not suppress or ignore the discontinuities between them." Conversely, appreciation for the diversity of social thought, norms, and cognitive biases may help natural scientists understand why simply communicating science does not necessarily lead to rational outcomes (Failing et al., 2007; Kahan, 2010).

The difficulty in bridging the natural and social sciences becomes especially acute when the focus is on turning ideas into action, or praxis, where the positive and normative collide. The crux of *democratic* praxis lies in establishing the legitimacy and validity of competing ideas and their underlying normative propositions to achieve agreement on a basis for social (or communicative) action. This, we argue, cuts to the heart of the modern/postmodern debate in the sciences and humanities, and as such requires further elaboration.

The debate around modernity and its variants (late modernity, anti-modernity, reflexive modernity, postmodernity, etc.) has been ongoing for decades. The debate revolves around competing social theories that seek to describe society and explain the way in which it is evolving, with the concomitant changes in discourses, social relations, structures, institutions, and the individual self. These social theories are tied – explicitly or implicitly – with economic, political, aesthetic, philosophical, linguistic, and other discourses and have the potential to offer great explanatory power. The value of social theory, in our perspective, is that it draws out the connections between empirical and normative content in evolutionary (or historical) context. Indeed, as Callinicos (2007: 5) notes: "social theories… tend to weave together normative and analytical dimensions," and hence "some consideration of the relationship between social theories and political ideologies is unavoidable."

In its simplest (and most superficial) form, modernity is a discourse that describes both an historical age and a set of norms, attitudes, institutions, and practices that emerged in the European Enlightenment and were subsequently developed in various places around the world (Callinicos, 2007). Modernity is deeply connected to the rise of capitalism, the scientific method, the discourse of progress, the dominance of the "West," and the differentiation of society into political, economic, and other spheres (Smart, 1990). Modernity emerged from and replaced a pre-modern period where legitimacy was secured through appeals to an external source of validity – for example religion or unquestioned cultural traditions passed on from generation to generation (Habermas, 1990). In the modern era, legitimacy and validity are created directly by individuals and society through the application of reason, a capacity deemed inherent to all humans.[3]

Modernity thus offered a break from traditions that some viewed as oppressive and stifling. Rather than appeal to some external and unquestioned source for validation (e.g., God), modernity provided individuals the liberation and freedom to create a world of their own, knowable through the application of rationality. In place of injustices caused by seemingly arbitrary forces of oppression (e.g., the feudal Lord), modernity offered "universal" rights and privileges, as well as the power over one's own labor (Callinicos, 2007). The technical mastery unleashed with the rise of modernity also led to the accrual of great power for modern societies.

The optimistic, liberational narrative of modernity has, rightly, met with widespread critique, especially as modern societies have interacted with others in their spread to the farthest reaches of the globe. This criticism has focused on the oppressive

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[3] As Jürgen Habermas & Ben-Habib (1981: 5) note, "modernity lives on the experience of rebelling against all that is normative." The breakdown of traditional sources of legitimacy was revolutionary and gave rise to new forms of organization that can be considered the cornerstones of modernity in a more material and epochal sense. Capitalism was the outcome of the rationalization of the provision of goods for the satisfaction of the need to materially reproduce society (*ibid.*). Democracy provided a system by which the newly empowered individual could exercise their political right to self-determination. Industrialization (and the technology that drove it) allowed for unprecedented control over the forces of nature and its productive capacities. Science allowed for the rational enquiry into the basis for life and the universe (Habermas, 1971). Art was freed from the tyranny of realism and ancient aesthetics and modes of production (Habermas and Ben-Habib, 1981).

effects of modernity such as colonization justified through appeals to "civilizing" progress (Mignolo, 2007): science, technology, and rational administration employed for the extermination of social groups, as in the Holocaust (Horkheimer and Adorno, 2002); the constant social and cultural disruption and anxiety engendered by fast-paced change (Polanyi, 2001); and ecological devastation arising from humankind's ever-growing technical and administrative power (Beck, 1992). Uncritical modernists have dismissed these as aberrant, while critical modernists have tried to rescue the modernity project through theories of late or "second" modernity, e.g. reflexive modernization, communicative rationality, etc. (Beck, 1992; Beck and Grande, 2010; Habermas, 1987a). These competing theories and analyses have tried to make sense of the dominant discourse or narratives that guide and legitimate social action and organization. Others have rejected modernity altogether, proposing a return to pre-modern ways or alternatively describing the emergence of a new postmodern, postcolonial era (Escobar, 2004; Lyotard, 1984; Mignolo, 2007; Ophuls, 1997). Postmodernism is fundamentally pluralist, and thus evades easy characterization (Habermas and Ben-Habib, 1981; Smart, 1990). As famously described by Lyotard, (1984: XXIV), postmodernism is "incredulity toward metanarratives" and is counter-posed with modernity, which seeks to legitimate itself through appeal to grand meta-theoretical narratives such as "reason" and "progress." Postmodernism instead de-centers the rational subject, arguing that individuals are constituted through power-relations. The de-centering of the subject also represents the death of reason – humans no longer can be viewed as possessing an internal form of reason with which to ground their actions (Habermas, 1990). It follows that universal grand narratives serve simply to mask power. The emergence of postcolonial and subaltern studies reflects the postmodern critique of Western metanarratives (Mignolo, 2007). The triumph of postmodernism as social theory is that it lays bare the hegemonic power structures lying at the root of the metanarratives of liberation through Enlightenment, allowing the particular to emerge from the general.

The extensive debate around modernity and competing social theories has direct relevance to ecological economics in the Anthropocene epoch. The Anthropocene discourse is, arguably, an emergent phenomenon intimately tied to the dominant modern discourses of Western society, e.g. progress, technological innovation, growth, change. If modernity has engendered ecological impacts that threaten the (modern) human endeavor, is the solution to be found in more modernity, critical modernity, postmodernity, or something else entirely? At the risk of oversimplification, the debate hinges on whether and how humans can legitimate their actions as individuals and social groups, the discourses that emerge, and how they in turn give rise to particular social and biophysical phenomena.

In essence, ecological economics is a challenge to the dominant uncritical modern discourse, seeking to change society to avert an ecological crisis. This will require bridging the modern, rational world of science and administration and the contentious, diverse realms of social and political life. Praxis, if it is to be non-coercive and intentional, must build on a set of shared goals and norms, although these do not necessarily need to be comprehensive. Questions emerge such as: What are the collective goals for the transition to an ecological economy? On a zero-sum planet governed by the laws of thermodynamics, what tradeoffs are acceptable? Who wins and who loses? Why? How do multiple cultures and social groups interact in a way that creates a basis for legitimate action? Underlying various answers to these questions may be, to adapt Keynes, the ghost of some defunct social and political theorist. The Anthropocene demands that ecological economics bring these ghosts into the light and subject them to careful inspection.

2.3.2. Theory and Praxis Across Multiple Scales and Levels

The object of study of ecological economics – the economy-within-society-within-nature – encompasses the entire planet and contains within itself tremendous diversity. Societies, economies, and nature interact at multiple scales and level, creating complex networks and systems that present challenges for policy and systems integration (Cash et al., 2006; O. Young, 2002; O. R. Young et (Liu et al., 2015)al., 2006).

The rapid pace of change, growing global interconnectedness, and the complexity of social and biophysical feedbacks – all, incidentally, characteristics of late modernity – make a strict disciplinary focus on scale inappropriate and prone to blind spots (Cash et al., 2006). For example, much traditional political science scholarship has focused on human institutions at different levels (local, national, international), without necessarily addressing their interrelationships (Gibson et al., 2000; Young, 2002). This is challenged in practice. Institutions of central importance in modern political theory (e.g., the sovereign nation-state) are increasingly unable to manage and control a globalized economy comprised of transnational corporations and highly mobile financial capital able to traverse multiple levels and scales (Biermann et al., 2012; Biermann and Pattberg, 2012).

Alternatively, disciplinary focus on a specific sector, problem, place, or institution may mask important indirect effects. For example, conservation biologists have raised concerns over "leakage", where conservation efforts in one place lead to deforestation in another (Gan and McCarl, 2007; Liu et al., 2013). Revealing these "teleconnections" is a core focus of ecological economics, e.g. through work to use global input-output models to understand national and global environmental impacts resulting from economic production and trade (Turner et al., 2007; Wiedmann et al., 2007, 2013). Scale and level thus hold central importance as an analytical concept for understanding, characterizing, and managing the various social and ecological processes in the political economy of the Anthropocene.

Scale is also relevant, in a slightly different sense, when assessing alternatives pointed to as examples of potential "ecological economies," such as indigenous cultures, transition towns, degrowth experiments, etc. The scale challenge in this case is turning local action and initiative into concerted action sufficient to address planetary-scale problems. At the local level, communities may share a similar culture and values, facilitating collaboration and collective problem-solving. This shared sense of community becomes harder to sustain as problems become more remote and the affected polity becomes larger. This is a challenge for social and political theory, which have struggled to address the challenges of global governance on a diverse and rapidly changing planet. For some of the most intractable ecological problems, such as climate change, the problem is global and so solutions must be collective and comprehensive, giving intransigent local or regional actors disproportionate power (Klinsky and Dowlatabadi, 2009). The Anthropocene is rife with these local and global collective action problems (Ostrom, 2010; Ostrom et al., 1999).

In the Anthropocene, ecological economic praxis must bridge scales and levels with regard to the efficient, fair, and legitimate allocation of resources. This demands social and political theory that can be drawn upon to establish legitimacy around different normative propositions and resource distributions. Critical social and political theory may be useful in this task. For example, a triumph of the Enlightenment was the establishment of concepts such as universal rights and values. These Enlightenment ideals may provide a basis for global concepts of justice and a shared, humanist identity that can facilitate concerted action without being oppressive of all difference. They could provide for pluralism bounded within a globalized, interdependent human community. On a finite planet, thinking globally is an imperative, but should not be an excuse for ignoring the importance of the local. For ecological economics, a central question is thus what social and political theories can "do the work" of bridging local and global in all their myriad forms of interaction.

2.4 Competing Visions for Transitioning to an Ecological Economy

Ecological economics has, for several decades, engaged directly with the trends that underlie the Anthropocene discourse, connecting them in various ways to the functioning of our economic and social system. However, the pluralism of the discourse – initially intentional – has led to important elisions and tensions within the literature regarding underlying social and political theories and the levels at which they operate. Foundational social and political theory remain implicit in the ecological economics literature, much like in the neoclassical economics that it emerged to critique. We argue that this has prevented the necessary discussions regarding the advantages, drawbacks, and open questions of different visions of "sustainable" socio-ecological systems and the transformations themselves – the praxis.

The pioneering work of Herman Daly is a relevant starting point for critical analysis, as it has in many ways helped set the research agenda for ecological economics. Daly draws on the notion of an "ends-means spectrum" to capture the interplay between nature (means) and individual and social goals (ends) (Daly, 1980). In Daly's rich and engaging body of work, considerable attention is focused on connecting the notion of "ultimate and intermediate means" with the theory and practice of mainstream economics, proposing policies such as quota systems, cap-and-trade, and similar interventions as possible means for managing a steady-state economy (Daly, 1974, 1968, 1992, 1990, 1980; Daly et al., 1989; Daly and Farley, 2010). Attention is also given to "ultimate and intermediate ends," drawing upon the work of environmental ethicists, sociobiologists, and theologians to criticize the utilitarian ethics of mainstream economics and suggest less materialistic individual and social goals for humanity. This sense, Daly's critique of economics echoes arguments made by Aristotle (Kern, 1983). Underlying all appears to be an implicit assumption that, if presented with facts and a good argument, human societies have the capacity to take steps to right our relationship with the earth.

What is less apparent in Daly's work is a sustained engagement with the social and political tissue that connects means and ends. Specifically, how their interplay and organization combine to form a society (or societies) oriented toward a set of ends that can be achieved with the available means, and how power will be mobilized and directed to achieve this outcome. Also, no real attention is given to how conflicts between different ends may be resolved, which is a central concern of social and political theory (Callinicos, 2007; Giddens, 1995). Indeed, questions that are critical to ecological economic praxis were being engaged with in canonical works of sociology, economic history, and political science nearly a century before Daly's work appeared. The technocratic rationality that imbues much of Daly's work has been subjected to intense scrutiny by social theorists. In building a powerful critique of mainstream economics, Daly in some ways hews too closely to the subject of his derision (Pirgmaier, 2017).

To an extent, Daly's work is representative of how things remain in ecological economics: much attention has been dedicated to documenting the extent of human impact on the earth system and the limits that arise (ultimate and intermediate means), or to different valuation and modeling techniques for bridging ecology and mainstream economics. Spash (2011) rightly points out that continental scholars and certain others have devoted more time and effort to political and social questions of relevance to ecological economics, yet the work still builds on conflicting ontological, epistemological, and methodological foundations (Spash, 2012). In this vein, the "shallow" and "deep" variants of ecological economics proposed by Spash (2013) roughly equate with uncritical and critical perspectives on modernity. In the debate around "ultimate" ends, anti-modern and post-modern voices emerge, proposing eco-centric ethics (Brown, 2012, 2007) or critiquing Western science and technology as forces (or discourses) of oppression (Ophuls, 1997).

The challenge for ecological economics is to find a path through the minefield of competing social theories that modern and postmodern thought encapsulate, starting with coming to terms with the metanarrative of mainstream economics. The mainstream approach – including environmental economics – largely accepts the quasi-utilitarian ethics and rational actor model that underpin much neoclassical theory. A specific set of values and norms is thus imposed on the world under the guise of a "universal" deductive scientific methodology (Milonakis and Fine, 2009). This theory, especially as it co-evolves with the sociological phenomenon of neoliberalism, which promotes markets at the expense of alternative forms of social organization, is directly implicated in driving the trends that have led to the Anthropocene. The perpetuation of a utilitarian ethics, an instrumentally rational basis for social organization, and growth and market fetishism as ideology exemplifies the totalizing “modern” metanarratives so roundly criticized by critical theorists and postmodernists alike.

In the face of an uncritical acceptance of modernity, adopting a postmodern basis for an ecological political economy has some appeal. Modernity and the Enlightenment are, to some, inextricably linked with the economic liberalism, individualism, and self-centered greed that have led to the ecological crisis (Mignolo, 2007; Ophuls, 1997). According to this line of reasoning, one cannot retain the promise of the Enlightenment without accepting that capitalism and a destructive, extractive relationship toward nature will follow. Thus, a social and political theory rooted in the Enlightenment is fated to failure, as far as ecological economics’ goals are concerned.

Postmodernism rejects self-centered reason as a basis for legitimacy, arguing that this has been used to justify the oppression of minorities, indigenous peoples, and other marginalized populations, and cannot be meaningfully distinguished from brute power (Callinicos, 2007). Many of the indigenous lifeways exterminated by capitalism and the forces of the Enlightenment constituted entirely different relationships with nature, much more in line with the land ethic or nature worship proposed by environmental ethicists and some ecological economists (Eckersley, 1992; Ophuls, 1997). A new discourse is needed; indeed, many discourses are needed. Postmodernism embraces plurality, encouraging a flourishing of narratives, with the role of science simply that of uncovering new narratives (Lyotard, 1984).

As a critique, postmodernism is powerful. Yet, as a basis for theory that can explain and guide social change, it suffers from what Habermas calls “a performative contradiction” (Habermas, 1990). In destroying the false claims to legitimacy put forth by Enlightenment thinkers, postmodern social theory rejects all claims to legitimacy including those of science. What is left is a Nietzschean will-to-power, exactly what postmodernists declare lurks behind modernist metanarratives. Unbounded value pluralism and complete epistemological relativism provide no basis for praxis. Intellectual liberation comes at the expense of the ability to make claims on others and correct contemporary and historic injustices. It also undermines the notion that humanity may form and exercise a collective will to attempt to, in the case of the Anthropocene, avoid future suffering and protect our shared home. While superficially attractive, postmodern lines of thought provide an inadequate basis for ecological economics praxis, especially given the explicitly humanist normative vision motivating ecological economics.

An alternative for ecological economics is to build on critically modern social and political theory, preserving some of the important social and political advances of the Enlightenment while attempting to correct for the oppressive tendencies and growth fetishism incompatible with a finite, culturally diverse planet. This critically modern social and political theory will need to establish a means for establishing legitimacy that encompasses a role for science (biophysical reality) and room for a bounded form of value pluralism (constructed social reality).

2.5 Critical Social and Political Foundations for Ecological Economics

The Anthropocene is a time of great contradiction: humans have amassed tremendous power to control and subjugate nature, which, in turn, threatens the foundation upon which human society is built. At the same time, humanity is not a master of its barely-comprehended power. Choices made by particular individuals, social groups, and governments have the power to transform ecological systems locally and globally, both in isolation and in aggregate. Insofar as these transformations are considered a concern – which ecological economics takes as given – this demands some amount of coordinated social action to avert crisis.

The Anthropocene epoch represents, in our view, a monumental challenge: how do we construct a social and political order that makes room for the tremendous diversity of human culture and social life and yet also establish a common, legitimate basis for managing our individual and collective impacts on a shared home, planet earth? Furthermore, on what basis does this social and political order legitimate itself? If ecological economics is to move from analysis and critique to praxis, this challenge must be tackled head on.

In seeking to develop theory that can rise to the Anthropocene challenge, ecological economists may shed light on unresolved tensions within the transdiscipline, where discord about competing theories and methodologies may find its root in different approaches to the discourse of modernity. Ecological economics may be able to draw from important bodies of work in social and political theory while simultaneously advancing it. As noted by Milonakis & Fine (2009: 92), following the marginalist revolution, "not all of political economy has been retained." In fragmenting political economy into the disciplines of economics, sociology, economic history, etc., the unique role of land and natural resources in economic and social activity was forgotten: ecological economics can re-illuminate the connections.

Critically modern theory – in particular, deliberative democratic social and political theory – offers a potential platform on which ecological economics can build in its effort to spur praxis that responds to the challenge of the Anthropocene. Deliberative approaches establish social and political legitimacy through communication with the contention that humans have, for all our history as a species, used language and speech to establish intersubjective truths to guide collective action (Baber and Bartlett, 2005). This simple observation locates a basis for non-coercive (indeed, democratic) social and political action in inter-subjectively constituted "truths" or, more specifically, validations of normative claims (Habermas, 1987a). Through language humans have the power to shape a social reality that is perceived as (broadly) legitimate and yet presents considerable space for diversity of thought, opinion, and lifeways. This responds to the postmodern critique of rationality located in the self – a core component of modern thought – without abandoning the notion of reason altogether.

Deliberative democratic theory has been explored by ecological economists and degrowth scholars previously, often in specific technical contexts (Norgaard, 2007; Zografos and Howarth, 2010, 2008). For example, in response to flaws in contingent valuation methods, scholars have proposed "deliberative monetary valuation," which draws on this body of theory and the notion of constructed preferences more broadly (Lo and Spash, 2013; Sagoff, 1998; Spash, 2007; Wilson and Howarth, 2002). Similarly, in response to the challenges in making social or public choices, ecological economists and others have proposed deliberative, multi-criteria decision analysis processes (Gregory et al., 2005; Huang et al., 2011; Rauschmayer and Wittmer, 2006). The utility of a deliberative approach has been demonstrated in specific technical applications; however, the implications for the social-theoretical foundations of ecological economics have not been explored in nearly as much depth. Deliberative social theory could provide a strong alternative to the technocratic, managerial approach of an uncritical modernity. Deliberative democracy could be compatible with a degrowth or steady-state future (Deriu, 2012; Ott, 2012). Yet there are major theoretical challenges that require extensive work, both in "scaling up" from local-to-global and in according adequate primacy to the biophysical impacts of social and economic life. It is unclear whether the bounded or narrow forms of intersubjective truth that emerge from deliberative processes can sow the seeds for more radical, global transformation.

The Anthropocene discourse, while on the face of it no surprise for ecological economists, reveals unresolved challenges for ecological economics as a normative transdiscipline oriented toward praxis. In responding to this challenge, ecological economics would be well served by a more direct engagement with critical work emerging from social science disciplines that have – since the marginalist revolution –

engaged with questions of political and social change. This can inform new theoretical work that engages with questions of central relevance to achieving the goals of ecological economics in a globalized Anthropocene epoch. As a critical discipline, ecological economics has made the case that the current trajectory of human society is unsustainable. What is needed now is theory oriented toward supporting the transition to a new, ecological economy. This requires, more than anything, an ecological economics that gives deep and sustained attention to the social realm: in the ecological economics ontology, the social is what connects the economic and the ecological.

CHAPTER 3: DELIBERATION AND THE PROMISE OF A DEEPLY DEMOCRATIC SUSTAINABILITY TRANSITION

"Science is meaningless because it gives no answer to our question, the only question important for us: 'What shall we do and how shall we live?'"

- Tolstoy, quoted in Weber (1946)

3.1 Introduction

Growing concern about the extent of human impact on the planet, along with increasing acknowledgement of the deep interconnectedness of social and ecological systems, has led to the emergence of new academic endeavors that span traditional disciplinary boundaries in research that aims to contribute solutions to pressing problems. In this vein, conservation biology (Soulé, 1985), ecological economics (Gowdy and Erickson, 2005), sustainability science (Kates et al., 2001), agroecology (Méndez et al., 2013), political ecology (Robbins, 2012), and other "transdisciplines" aim to bridge theory and practice with a critical/normative framing that guides research. This framing can be simple, often relayed in pragmatic terms: the world's ecosystems are suffering; biodiversity is intrinsically and instrumentally valuable; we should do something to protect it. Conversely, the framing can be more reflexive, building on a constructivist critique of science.

A central challenge for normative science is recognizing and acknowledging the normative content guiding the descriptive, critical, and prescriptive aspects of research (Alvesson and Skoldberg, 2000). Some suggest that scientists draw a sharp line separating their research from advocacy or direct action (echoing Hume's Guillotine and a more positivist stance toward science) (Lackey, 2007); others see the two as intrinsically connected, for example in participatory action research (echoing hermeneutics) (Méndez et al., 2013). An underlying question is whether and how the normative content guiding research and advocacy can be justified or legitimated (Cooke, 2005). Is legitimacy conferred via an appeal to authority (e.g., the scientist as member of the scientific establishment), via an appeal to process (e.g., science as a product of peer review by credentialed experts), via an appeal to reason (e.g., prescriptive advice based on a reasoned review of the facts), or via some other claim? The line between scientist working in a discipline and activist participating in a social movement is both uncertain and contested.

Ecological economics exemplifies the challenges facing normative transdisciplinary science, and hence is the focus of our analysis. Emerging as a critique of the implicit values of orthodox economics – an unreflexive idolization of efficiency and growth – ecological economists have argued that these same values are, in part, to blame for the social and ecological crises facing the planet (Daly, 1974; Georgescu-Roegen, 1971; Jackson, 2009; Kallis et al., 2012). The starting point is an ontology that sees the economy as a subset of humanity's myriad social institutions, with the social sphere itself embedded in the broader ecosystem or environment (Costanza et al., 2012; Daly, 1980). In other words, human society cannot be materially dis-embedded from the rest of nature. Through considerable empirical and theoretical research, ecological economists and peers in related transdisciplines have used this shared ontology to challenge orthodox thinking that ignores or underemphasizes the human species' fundamental embeddedness in nature (Gowdy and Erickson, 2005). Like other normative disciplines, the critique is situated within a broader, sometimes implicit, appeal for a transition toward an alternative social arrangement (Söderbaum, 1999).

In ecological economics, one prominent framing of the alternative is that of Daly and Farley (2010): an ecological economy is one that achieves sustainable scale, just distribution, and efficient allocation. Setting aside whether this vision is representative of all of ecological economics, it does help illustrate a tension in the broader literature – can the value claims of transdisciplinary normative science be legitimated, and if so, how? In work agitating for a "sustainability transition," how does the vision for and content of a "sustainable society" garner social legitimacy? In justifying these goals, and ecological economics more broadly, Daly and Farley (2010) mix appeals to instrumental reason (recalling the "nature's benefits" or ecosystem services strand of the ecological economics literature) with more metaphysical arguments about intrinsic value and the "ultimate ends" that guide human life. Presumably, if one finds these claims compelling, they will embrace the arguments and recommendations; if not, one is free to choose another author of closer ideological alignment.

Our point is not to impugn the value of ecological economics or Daly and Farley's work in particular, nor is it to paint all normative scholarship aimed at a sustainability transition as theoretically ungrounded. It is to highlight an as yet unresolved question for normative science: how can competing value-claims be justified or legitimated so as to create some basis for collective action? What is the normative theory that underpins normative science? As Cooke (2005: 379) notes, "Critical social theories… (rely) on a number of important assumptions about human nature, society and history, above all, the assumptions that human beings are formed for the better or the worse by the social arrangements in which they are involved in their everyday lives and that these social arrangements are neither naturally necessary, nor divinely ordained nor historically inevitable." What, if anything, can ground social critique?

Rather than reflect some defect in the literature or the science, we argue in this paper that unresolved questions about normative legitimation reflect ongoing debates about modernity, truth, legitimacy, democracy, and collective action. We make the case that ecological economics has begun to embrace one possible answer to this tangled knot of questions – deliberative social and political theory – but only tentatively, in a problem-solving capacity. While the problem-solving utility of deliberative theory in ecological economics is indisputable, we make the case that a deeper, critical embrace could draw on the radical transformative potential of deliberation, which has been underrepresented in the ecological economics and, to a lesser extent, the governance literature. In this sense, we highlight proposed connections between deliberation and social-ecological transformation, which merit both empirical analysis and further theoretical development. These connections raise the possibility that deliberation can not only nurture a sustainability transition but also give rise to the transition in and of itself. We raise questions that constitute a research agenda for a deeply democratic sustainability transition. We conclude by considering the scope and reach of ecological economics as a normative, critical transdiscipline.

3.2 Normative Science and the Crisis of Late Modernity

The current social-ecological predicament – where humans have achieved tremendous power over nature and each other, yet with only weak control over the collective execution of this power – is very much a product of modernity. The rise of modernity in the West, manifested in the Enlightenment, the emergence and spread of capitalism, and a growing techno-scientific power or mastery over nature, saw the slow shedding of traditional forms of power and normative authority in favor of subject-centered reason (Callinicos, 2007; Habermas, 1990; Smart, 1990). Put simply, modernity saw reason and the individual triumph over the pre-modern in a process of "disenchantment," or shedding of traditional values and forms of social reproduction (Habermas and Ben-Habib, 1981). In the pre-modern, normative content finds legitimacy through widely-shared faith in an external source of validity (e.g., tradition, the church, the sovereign). In modernity, it is a human's innate capacity to reason (our consciousness) that provides the source of legitimacy. Hence, to some modernity represents liberation, the throwing off of the chains of traditional forms of power in favor of the universal possibility of the enlightened individual (Callinicos, 2007).

An optimistic take on the discourse of modernity sees the rapid advance of science and technology, enabled via the application of reason, as a source of growing power and control over our collective fate. There are parallels with contemporary environmental discourses such as eco-modernism (Asafu-Adjaye et al., 2015): the current ecological predicament can be resolved through the application of more reason ("getting the prices right") and technology, the product of that reason. The notion of a strong fact/value dichotomy is typical of (uncritical) modern thought (Putnam, 2002): facts are objectively visible and knowable to all through science and reason, whereas values are subjective, with vestiges of the mystical. The (neo)classical adage of mainstream economics – "*de gustibus non est disputandum*" – captures this elegantly: we can agree on external facts, instrumental reasons for action (e.g. efficiency), but our subjective valuations cannot be compared in a defensible manner (Robbins, 1932).

A pessimistic take on modernity views its liberational promise as false: old forms of power and domination have merely been replaced by new forms, masquerading as subject-centered reason (Horkheimer and Adorno, 2002). Reason and human consciousness are not sublime; they are constructed from a socially and historically infused muck. If modernity reveals anything, it is that all truth-claims are suspect, especially those claiming to be universal (Lyotard, 1984). Domination in the name of an absent god is replaced by secular forms of domination. In practice, the post- or anti-modern critique can take the form of an extreme skepticism bordering on complete relativism about truth claims: the truth is not knowable since there are many truths

(Callinicos, 2007). Any attempt to judge one subjective claim as superior to another is a form of domination (Habermas, 1990).

These two extreme interpretations set out the guideposts for our discussion of the challenge facing normative science. Cleary, the scientific pursuit of knowledge has furnished great power to understand and manipulate the world around us, arguably enabling the Anthropocene to emerge. Yet the postmodern critique is a necessary caution, because power does frequently don the guise of liberational discourse. Modernity is a phenomenon that emerged in a particular place and cultural context; while the discourse has spread throughout the world, sometimes at gunpoint, it has not obliterated all pre-modern or anti-modern societies and discourses. Values exist and matter, especially in the context of collective action – to agree on a shared vision or set of value claims ("what ought to be") is to agree on some form of social truth, however contingent.

The crisis of late modernity is that we have rejected external sources of normative legitimacy while also recognizing the limits of subject-centered reason. If our subjective decisions, enabled by science and technology, have begun to yield consequences that threaten the entire human endeavor, perhaps something should be done. The critical theory of the rationalization of society – where instrumental rationality has crowded out forms of value rationality, colonizing the lifeworld – echoes the ecological economics critique of an economic system that blindly pursues growth and efficiency, neglecting and sometimes undermining the values and systems central to our lived experience as humans. Per Habermas (1990: 355), "processes of monetarization and bureaucratization penetrate the core domains of cultural reproduction, social integration, and socialization." The extension of bureaucratic and economic systems into the social world alienates individuals from control over the lifeworld and in the end, threatens social integration. As Fergus & Rowney (2005: 25) explain, "non-economic social frameworks, institutions, and cultural traditions have less and less significance in the forming of society's structures." Indeed, "economic rationality has become so prevalent in our society that it is difficult to use language in everyday life without referring to the dictionary of economics (ibid.: 22)." This outcome is visible today, as political discourse repeatedly appeals to the logic of the marketplace to justify social positions, including the need to protect nature (Deriu, 2012).

At the same time, the archetypal political institution of modernity, the Westphalian nation-state, struggles to keep pace with economic and social forces that extend beyond the bounds of a traditional polity (Biermann and Pattberg, 2012; Swyngedouw, 2004). Goods, services, people, ideas, organisms routinely traverse distances that a century ago would have been unthinkable. The effects of these flows are multifarious, complicating governance and creating collective action problems that bedevil policymakers (Cash et al., 2006).

It is into this complex battlefield that the normative transdisciplines have wandered. Science and reason, however faulty or limited, have given us the ability to recognize and understand our conundrum, yet we struggle to extricate ourselves. The heart of the challenge is one of normative commitment: how can plural individuals and groups agree on a normative basis for collective action? How can value claims be evaluated and legitimized when both traditional sources of legitimacy and innate, subjective reason are suspect?

One potential answer is intersubjective reason. Rather than claim that legitimacy derives from consciousness or a pre-modern, external source, un-coerced communication can give rise to intersubjective forms of reason that can serve as the basis for collective action. This work of theory, prominently associated with Jürgen Habermas (1987a; 1987b), builds on the observation that when humans enter into communication with each other, they seek to create a shared understanding via deliberation that can coordinate social action to solve everyday problems, including making complex, value-imbued decisions. The speech act is therefore a commitment to seek a form of intersubjective truth, even regarding claims typically relegated to the subjective realm of values. The speech act may constitute the fundamental building block for constructing deliberative or "discursive" governance institutions that can navigate the complex landscape of late modernity (Dryzek, 2002). Communication becomes the angel heralding the promise of some form of cosmopolitan governance that can steer society through the complex landscape of later modernity.

Deliberative social and political theory has been widely embraced in the political science and governance literature, with deliberative democracy touted as a means for moving beyond some of the limitations of interest-group liberalism (Cohen, 1989; Dryzek, 2002; Parkinson and Mansbridge, 2012). This has included claims about environmental governance, including the potential for deliberative processes to facilitate the emergence of shared preferences and norms that can guide environmental policy and management at the level of traditional policymaking – local, regional, and national government – and in new domains of "earth system governance" (Baber and Bartlett, 2015, 2005). Deliberative theory has also been embraced, albeit on a more limited basis, in ecological economics and the sustainability transition literature.

3.3 Deliberation in Ecological Economics

Ecological economists have drawn on deliberative social and political theory since at least the 1980s. In a review, Zografos and Howarth (2008: 7) highlight two areas where deliberative theory and ecological economics have aligned: first, in the context of environmental decision-making and second, in a critique of the ability of a "growth-oriented, capitalist economy to genuinely integrate environmental goals in its operation." The first topic has been the focus of much of the literature, where deliberation has been used as *problem-solving theory* in the sense of Cox (1981). Deliberation has been proposed as a means for valuing non-market goods and services in the context of applied economic analysis. It has also been proposed as a means of "democratizing" decision-making via multi-criteria and structured decision-making processes. There remains tension between the democratic, process-focused use of deliberation in decision-making and the normative goals of ecological economics. This is captured in Robert Goodin's (1992: 168) oft-quoted remark that "to advocate democracy is to advocate procedures, to advocate environmentalism is to advocate substantive outcomes." Deliberative theory points to a means to deepen democracy, helping legitimate normative evaluation in the face of a daunting plurality of viewpoints and perspectives. It also provides a means to grapple in the public sphere with essentially contested concepts such as justice and wellbeing, both of which are central objects of concern in a sustainability transition. But since deliberation emphasizes the procedural, the outcomes of deliberation cannot be fore-ordained.

The second topic, which following Cox (1981) draws on deliberative social and political theory as *critical theory,* is a point of alignment albeit less developed in the ecological economics literature. One area in which deliberation has supported a (potentially) more radical critique of economic orthodoxy is in grounding alternative theories of development, such as the capabilities approach. In this section, we review the different applications of deliberative theory in ecological economics, touching upon the specific deliberative functions that they rely upon.

3.3.1 Deliberation and Valuing Non-Market Goods and Services

Mark Sagoff (1988) was an early proponent of the value of deliberative theory for ecological economics, calling for juristic deliberation to support the formation and negotiation of environmental and social values in collective decision-making. Sagoff's work started from a critique of welfare economics, specifically the use of cost-benefit analysis as a decision tool. Cost-benefit analysis has been frequently criticized in the ecological economics literature, for a multiplicity of reasons: preferences are incomplete, especially with regard to non-market goods (Gregory et al., 1993); contingent valuation and other methods give inconsistent, seemingly arbitrary results (Ackerman and Heinzerling, 2002); experimental evidence challenges welfarist assumptions about human psychology (Gowdy, 2007); treating all factors as commensurable (and measurable in units of money) does not reflect actual human values (Aldred, 2006; Martinez-Alier et al., 1998); cost-benefit analysis can be analyst-driven and opaque, undermining its legitimacy (Spash, 2007); and efficiency (as welfare maximization) is not a neutral nor universal goal (Bromley, 1990; Gowdy, 2007; Nyborg, 2014). This critique of cost-benefit analysis reveals a general concern in ecological economics that public decision-making may not adequately reflect the value that people do (or should) ascribe to nature and non-market social relations. It may also fail to consider the value ascribed by future generations and non-human nature (O'Neill, 2001). Deliberation has been proffered as a solution to specific problems within cost-benefit analysis as well as part of alternative approaches to public decision-making.

Many ecological economists have criticized the dominance of rational choice and revealed preference theory in economics, arguing, among other things, that they do not necessarily hold when applied to public goods and other aspects of the natural and social world that lie outside the market. Indeed, valuation may be approached from two perspectives: that of a consumer, emphasizing the contribution to an individual's personal utility or particular interests; that of a citizen, emphasizing the contribution to social utility or the general interest (Sagoff, 1998). As a consumer, a person may have no idea how much an endangered newt is worth, either in dollars or utils, since it is not a decision they have grappled with before. More generally, as a citizen that same person may not have much of a grasp of the value of an endangered newt for society or the laws designed to protect said newt. Following this line of reasoning, it is asking too much to ask someone to complete a willingness-to-pay survey or vote on the value of endangered newts without any further context. Their subjective preferences are not fully-formed, never mind complete, consistent, and transitive.

Deliberation has been proposed as a means toward better valuation by supporting preference formation (Bartkowski and Lienhoop, 2018; Kenter et al., 2016; Lo and Spash, 2013; Spash, 2007; Wilson and Howarth, 2002). In deliberative monetary valuation (DMV), participants are exposed to important scientific and policy context about an issue and are then asked to deliberate in small groups *prior* to making either an individual (via stated preference survey; subsequently aggregated) or group valuation (via consensus or voting) (Spash, 2007). The DMV design can vary, but the basic intent is to help participants developed better-informed preferences about the issues at hand. The public aspect of deliberation helps expose participants to different perspectives – technical, moral, etc. – that they may not have come across independently (Wilson and Howarth, 2002). The group discussion can highlight which claims have more public support and can withstand scrutiny, which may impact a participant's individual valuation; a consensus-based approach be expected to yield a different result than an aggregative approach (Howarth and Wilson, 2006). Decisions made on the basis of DMV will hopefully better reflect the reasoned individual (consumer) or public (citizen) values of the given alternatives, at least when compared to processes that lack a deliberative element.

3.3.2 Deliberation and Multi-Criteria Decision-Making

In ecological economics, there is a vast literature criticizing cost-benefit analysis *tout court*, arguing for example that tradeoffs among competing values are best understood as "weakly commensurable" and hence not directly reducible to monetary valuation (Ayres et al., 2001; Martinez-Alier et al., 1998). In some decisions, participants may hold lexicographic preferences, for example with regard to impacts on sacred sites. In situations such as these, solutions like DMV may be inadequate, necessitating alternative approaches to decision-making, e.g. multi-criteria, structured, or consensus-based approaches, all of which have been embraced in ecological economics (Ananda and Herath, 2009; Gowdy and Erickson, 2005; Hermans et al., 2007; Rauschmayer and Wittmer, 2006).

Multi-criteria approaches can be entirely analyst-driven and aggregative of individual preferences, eschewing deliberation. However, as with DMV, multi-criteria approaches can integrate deliberation in support of preference formation. Furthermore, deliberation can provide for legitimation, insofar as the parties affected by the decision participate in or are represented in the deliberative process (Cohen, 1989). Structured decision-making, which builds off a multi-attribute utility framework, integrates deliberation throughout the decision-making process, to support problem definition and understanding, preference formation, choice, and legitimation (Gregory et al., 2012).

In ecological economics, deliberation aligns with the transdiscipline's normative commitment to valuing non-scientific sources of knowledge, such as traditional ecological knowledge. This commitment has been captured in describing ecological economics as post-normal science (Funtowicz and Ravetz, 1994). Deliberation provides a forum in which non-scientific knowledge and expertise, as well as values, can be shared and evaluated (Failing et al., 2007; Martinez-Alier et al., 1998). Engaging parties to the decision to co-produce knowledge can contribute to the legitimation function of deliberation.

In multi-criteria approaches, as with cost-benefit analysis, a decision-rule can be established to select the optimal (or acceptable) alternatives; deliberative consensus is neither required nor prohibited. Consensus-oriented decision methods hew more closely to the Habermasian ideal, with the goal of the process to be the careful weighing and evaluating of claims such that a consensus choice will emerge from the process (Baber and Bartlett, 2015; O'Hara, 1996). This is the basis for the juristic model of decision-making, which has been explored widely in the governance literature (ibid.) but less frequently in ecological economics. For example, citizens' juries (also referred to as planning cells) have been used in local and regional settings to grapple with complex issues of public policy (Smith and Wales, 2000); they have also been proposed as a means of establishing international legal norms that can guide global environmental governance (Baber and Bartlett, 2015, 2009). While these efforts are clearly of relevance, they have not seen as much application in ecological economics *per se*.

3.3.3 Deliberation and Alternative Theories of Development

Zografos and Howarth (2008) argue that ecological economics and deliberative theory align in a shared critique of capitalist development and the ability of a capitalism to substantively incorporate environmental values. While there is certainly *potential* alignment, the broader sociological critique of capitalism and modern society – a central component of the critical theory branch of deliberative theory – has received scant attention in ecological economics. The most notable point of alignment is in attempts by ecological economists to decouple the notion of economic development as qualitative improvement from that of quantitative growth in biophysical throughput.

Ecological economists have extensively criticized the idolization of growth in mainstream economics. Growth is contrasted with development, re-centering the economic process on the production of "an immaterial flux of the enjoyment of life", to use the language of Georgescu-Roegen (1975: 353). There have been numerous proposals for what development could or should constitute in the sustainability transition, many of which propose specific outcomes or material endowments (Dearing et al., 2014; Lamberton, 2005; Lawn, 2003; Lehtonen, 2004; Max-Neef, 1995, 1991; Sneddon et al., 2006). Given the contested, socially constructed nature of development as a concept, these alternative proposals are subject to the same problems of normative evaluation and legitimation that deliberation has been used to "solve" in the context of public decision-making.

One alternative approach to defining development is the "capabilities approach," a theoretically rich attempt to describe the purpose of the economy that has been embraced to an extent within ecological economics. The capabilities approach is fundamentally deliberative, in that it relies on "public reasoning" as a means for justifying a subjective, particular notion of development via public discourse (Nussbaum, 2000; Sen, 2005; Amartya K. Sen, 1999). The capabilities approach stands in stark contrast to traditional welfarist concepts of utility maximization and its related utilitarian ethic (Muraca, 2012; Polishchuk and Rauschmayer, 2012; Rauschmayer and Leßmann, 2011). Sen has developed a robust critique of utilitarian ethics in welfare economics, broadly criticizing its indifference to distribution, its neglect of rights, freedoms, and other non-utility concerns, and its inability to factor in mental conditioning and subjective interpersonal comparisons (Sen, 2009; Amartya K. Sen, 1999; Amartya Kumar Sen, 1999). In contrast, the capabilities approach places emphasis on "the substantive freedoms – the capabilities – to choose to live a life one has reason to value (A. Sen 1999: 74)." Sen (ibid.) carefully distinguishes between the freedom to use capabilities (the procedural aspect) and the actual use made of capabilities (the outcome, or "functionings"), emphasizing that economics is typically concerned with outcomes and not possible choices (ibid.). The capabilities approach has some profound implications for the way in which the economy is structured and conceived, including:

- Correcting for the tendency of economists to focus almost exclusively on means (goods, resources) rather than the actual ability or opportunity to achieve desired ends (Polishchuk and Rauschmayer, 2012; Scerri, 2012);
- Rejecting the hedonic notion that simple happiness or pleasure is really the ultimate end of human action and existence, an ideological stance in common with ecological economics (Spash, 2012);
- Reemphasizing the importance of institutions in providing individuals with both specific capabilities (e.g., health, education) and the freedom to exercise those capabilities (e.g., via security, guarantee of basic right) (Lessmann and Rauschmayer, 2013);
- Providing an emancipatory promise – freedom to live a life one has reason to value – that allows for "bounded" value pluralism;
- Requiring reasoned consideration and, in some cases, public deliberation about individual choices and values; and
- Emphasizing the importance of maintaining spaces in society in which to discuss and reason about competing beliefs and norms.

This last point merits particular attention. At the heart of Sen's concept of freedom is the idea that one must use public reason to justify that which one values (Sen, 2009; Amartya K. Sen, 1999). This requires individuals to be able to legitimate claims about their freedoms and how they exercise them, thereby creating and contributing to social norms that govern society's choices (ibid.). This provides room for plural concepts of value and justice to coexist, insofar as they can be reckoned via deliberation.

The capabilities approach has been drawn on in ecological economics as means for understanding the relationship between nature and human development (Pelenc and Ballet, 2015; Polishchuk and Rauschmayer, 2012); to examine conflict (Griewald and Rauschmayer, 2014); in the context of intergenerational equity (Gutwald et al., 2014); and in its broader sense of providing a basis for understanding the "social" objectives of economic activity (Dodds, 1997; Howarth, 2007; Lehtonen, 2004; Muraca, 2012; Rauschmayer et al., 2015). In this sense, the capabilities approach has fruitfully been adapted by ecological economists in a problem-solving guise. However, there remains tension between the procedural focus on public reasoning that undergirds the capabilities approach and the normative goals of "sustainability" motivating ecological economics. And importantly, what Sen remains silent upon – which is essential for realizing the promise of the capabilities approach – is whether and how the larger political-economic edifice needs to change to create the vibrant public sphere so essential to the capabilities approach. Can creating a public sphere "capable" of adopting the capabilities approach as a model of development simultaneously incite a sustainability transition?

3.4 Deliberation as Catalyst for a Sustainability Transition

Ecological economics is motivated by general concern about the impacts of human society on the earth system and the feedbacks that this might engender. A vision of a "sustainable" future – the concept is contested, but generally involves lessening the environmental impact of human society – is contrasted with the present situation. In evaluating how to begin transitioning toward this sustainable vision, deliberative political and social theory have been helpful in solving problems related to valuation, decision-making, and grounding alternative concepts for human development. Given the open-endedness of deliberative processes, the question remains whether or not deliberation is a reliable partner in the sustainability transition. Perhaps, in the colorful phrasing of Georgescu-Roegen (1975: 379), "the destiny of man (sic) is to have a short, but fiery, exciting and extravagant life rather than a long, uneventful and vegetative existence." This "destiny" may emerge as the reasoned conclusion of open, un-coerced, discursively-representative deliberation. But there is reason to think otherwise.

We review several central functions of deliberation (preference formation, normative evaluation, and legitimation), drawing out potential feedbacks that could contribute to "sustainable" social-ecological transformation. For each of these functions, there are multiple approaches that fall on a spectrum of deliberative designs (Figure 1). We conclude by talking about the mechanisms by which deliberation can "scale-up" to create a more reflexive or deliberative society.

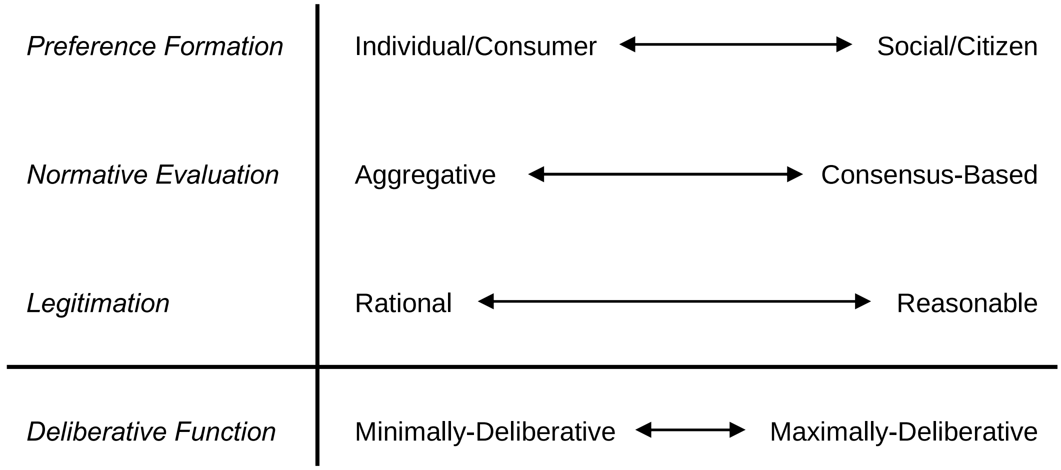

Figure 1. Deliberative functions and the range of potential approaches from minimally- to maximally-deliberative.

3.4.1 Deliberation and Preference Formation

Deliberation is explicitly intended to contribute to preference formation. In entering into deliberation, a participant is signaling a willingness to be exposed to new facts and perspectives – to listen and learn – that may change her/his preferences regarding an issue or topic. In its minimum sense, a passive process of exposure to information can help shape an individual's particular or "consumer" preferences, without demanding any engagement with broader questions of social or public concern (Figure 1). But deliberation is intended to go further, in that this passive process is complemented by an active process of discussion (communication) with other members of the deliberative community. This communicative process involves the sharing of different perspectives, the making of normative claims, and is ideally representative of the broader community affected by a particular decision or issue of concern. Representation can be at the level of the individual/group (following more traditional liberal theory) or the discourse (following more critical discursive theory).

Can this process of preference formation be conducive to a sustainability transition? Possibly, since deliberation can lead to more informed, "rational" decision-making, which involves exposure to and consideration of the multiple discourses – including environmental – pertaining to an issue.

For complex environmental and social issues that lie outside routine experience, deliberative settings provide for a public, social process of learning. This is one reason for the emergence of DMV – contingent valuation assumes people already have well-formed preferences. At a minimum, this process ensures that a participant's subjective preferences are formed in the light of prevailing facts and the complex array of private and public values that are intertwined with interpretations of those facts. In a discursively representative deliberative forum, discourses representing the interests of non-human nature, future generations, communicatively-limited humans, and distant but affected parties may be aired, challenging parochialism in preference formation (Dryzek, 2002; Eckersley, 2002). Given that many decisions have dimensions that extend across the bounds of a traditional polity (to which the term "citizen" may attach), discursive representation can provide deliberation with the more cosmopolitan framing it may merit. Discursive representation, combined with a commitment to listening and learning, provide deliberative pre-conditions that can at least ensure that relevant environmental claims are expressed and considered in preference formation.

3.4.2 Deliberation and Normative Evaluation

Deliberation in its more radical, critical theory sense, departs from the premise that normative claims can *and should* be evaluated or ranked, so as to enable legitimation and social critique. It is not simply that personal preferences may be shaped through communication, but also that the process of deliberation itself can allow for normative evaluation through public reasoning, or the giving and scrutinizing of claims. In this sense, participants in a deliberative process may inter-subjectively (via communication) agree on an ordering of claims, achieving partial or complete consensus.

The topic of consensus and normative evaluation exposes a divide in the deliberative literature. Baber & Bartlett (2005) delimit three camps: the "full liberalism" of Amy Gutmann, Dennis Thompson, and James Bohman; the mandatory discourse of Habermas; and, the normative pre-commitments of Rawls. To this we add a fourth: the evaluative "public reasoning" of Amartya Sen.

The full liberalism strand of deliberative theory aims to set the preconditions for public deliberation, without any particular constraints on the types of reasons that are given, other than that they are addressed to the public in such a way that they do not undermine the continuation of the deliberative process. Hence both particular and general claims are allowed; consensus is not mandated, and aggregative procedures can be used.

The aim is a fair process and an enriched debate that will, hopefully, help solve collective problems (Baber and Bartlett, 2005).

Rawls see the point of deliberation as limited to establishing “just institutions” within which individuals can pursue their personal interests. In this sense, deliberative consensus should be achieved on the basis of “public reasons,” which are reasons that all are willing to ascribe to – in this sense, any particular individual claims are excluded from deliberation. This more transcendental or idealist approach to deliberation and justice, while philosophically interesting and worthwhile, is of limited usefulness for making everyday decisions and navigating actually existing questions of injustice (the presumed subject of social critique) (Sen, 2009).

Slightly less exacting is the theory of Jürgen Habermas (1987a, 1987b), where deliberation can admit a broader array of topics yet is still intended to address matters of common concern (the general interest) (Habermas, 1991). The goal remains to achieve consensus via a process that explicitly aims to evaluate general normative claims, discarding those that cannot be deemed communicatively or inter-subjectively rational. Rationality is found in mutual acceptance of the validity claims of speech or discourse; validity claims pertain to matters of empirical (mind external) fact as well as normative subjects such as moral rightness, aesthetic value, etc. This mutual acceptance of normative claims relies in part on a shared lifeworld of the subjects of communication. The broader goal of deliberation is to establish normative legitimacy that can be used to critique and therefore reshape society, in a direct response to the crisis of modernity –

this invites parallels with the aim of the normative transdisciplines discussed earlier. However, the conditions he sets for deliberation are strict, representing a practical ideal perhaps more than a workable model.

Sen establishes a middle ground between the "full liberalism" and Habermasian models, in that he sees deliberation as necessary to the evaluation of normative claims yet allows for aggregation (rather than strict consensus) to determine the evaluative outcome (Sen, 2009). In acknowledging the deeply pluralistic nature of human society, he accepts that in many cases the results of deliberation followed by aggregation may yield only a partial ordering of normative claims ("we agree that B is manifestly unfair and should be stopped, yet we cannot determine whether A or C is better") (ibid.). This aggregative approach demands evaluation via deliberation but does not demand agreement on the reasons for the ranking, something that Habermas might demand as part of determining the rationality of validity claims. In this sense, Sen hews closely to Sunstein (1995) on the acceptability of "incompletely theorized agreements."

The idea of deliberation as a means for normative evaluation holds promise in the context of a sustainability transition, even if considered in the less exacting sense of "full liberalism" or Sen. It demands that claims be evaluated publicly, potentially privileging those that appeal to the general interest ("citizen" or social preferences) rather than the particular ("consumer" or individual preferences). As Dryzek (1987: 204) argues: "… the human life-support capacity of natural systems is the generalizable interest *par excellence*." This opens up space for consideration of an array of moral, ethical, and aesthetic claims that may, in non-deliberative settings, be trampled by power structures advancing instrumental claims. Environmental claims that are aired as part of a “discursively representative” deliberative process may receive more weight in the final decision, whether it is made via aggregation or consensus under more or less strict conditions. It is harder to sustain a claim in favor of your own particular interests in a public forum constituted of those who would potentially bear the costs of such a claim. This is the “constraint” that Sen imposes on his form of “development as freedom” – an individual can pursue their own interests only insofar that they can be publicly justified.

3.4.3 Deliberation and Legitimation

The connection between deliberation and the discourse of modernity lies most clearly in the notion of legitimation, the process whereby a course of action or discourse gains legitimacy. Legitimacy can be understood as occurring when something “is accepted as proper by those to whom it is supposed to apply” and is an essential aspect of democratic governance (Dryzek 2010: 21). Deliberation can be viewed as a legitimation process, in that deliberation seek to construct among the participants a shared understanding of the issues at hand and some agreement – full or partial – on the facts, preferences, and norms that are at play and their relative importance and validity. Deliberative legitimacy in an ideal sense demands the free and willing participation of all those affected by a decision, the recognition of deliberative capacities in others, as well as some alignment between the decision made as a result of deliberation and the corresponding actions taken (Cohen, 1989). This ideal form raises complicated questions about representation and scale, i.e. the ability to deliberate about decisions that affect large groups of people, non-human nature, or those lacking deliberative capacity (Dryzek, 2010; Parkinson, 2006). We return to the challenges of representation and scale later in this paper.

In the strictest sense, legitimacy derives from the achievement of reasoned consensus on issues of general concern. Particular interests – the subject of strategic communication, to use Habermasian language – can be pursued on the basis of instrumental rationality and hence do not demand deliberation. In this sense, Rawls distinguishes between claims that are rational and those that are reasonable (Rawls, 1993) (Figure 1). It may be rational for a hungry individual to eat all the food in the communal bowl, but if they are part of a community of hungry people this behavior may not be deemed reasonable according to the community's shared moral and ethical standards. For Habermas, reasonableness emerges through the intersubjective renewing of validity claims via communication. In a less strict sense, deliberation can be understood as the process of forming a shared understanding of the rational, the reasonable, and their relationship in a given context (Bartlett and Baber, 1999).

The legitimation function of deliberation can potentially both support and give rise to a sustainability transition. First, insofar as environmental concerns are privileged in deliberation (per the preceding arguments), the legitimacy-conferring capacity of deliberation can potentially render decisions more stable, heading off conflict and contestation that might otherwise arise if the same decision were made after a non-deliberative (and less legitimate) procedure. This is further enhanced by the goal of achieving decisions that are "reasonable," rather than simply (instrumentally) rational. Third, the deeply democratic intent of deliberative processes may confer longevity – a central goal of deliberative processes is to maintain the willingness of participants to continue deliberating (Cohen, 1989) – and engender skills and attitudes among participants that are, for lack of a better term, pro-social. In this sense, deliberation can in itself create deliberants that hold the democratic, tolerant "citizen" preferences that be necessary to incite a sustainability transition (Baber and Bartlett, 2015).

3.4.4 Deliberative Systems for Sustainability: Radicalizing the Public Sphere

Deliberative social and political theory fundamentally breaks with the subjective model of reason characteristic of uncritical modern thought. In locating reason in the intersubjective, there is acknowledgement that the fixed and unquestioned "subjective preferences" of individuals are in fact socially constructed, malleable, and subject to normative evaluation. This has parallels in ecological economics, where similar arguments mark a break with neoclassical theory and the idea of the impossibility of intersubjective utility comparisons. In deliberative political theory it marks a break with voting as "passive" preference aggregation and a potential escape from the impossibility of social choice (Dryzek and List, 2002). In critical theory it marks a break with postmodernism, "rescuing" the critical project by locating a universal basis for a normative critique of society (Habermas, 1990). In the most radical interpretation, deliberation can be constitutive of social transformation, in that deliberation reinvigorates the public sphere, allowing for the emergence of a collective critique of society that provides the basis for legitimate, transformative social action.

The shared basis for critique is, in most interpretations, constrained by the bounds of the group taking part in deliberation. This begs the question of "mini-publics and their macro consequences" Dryzek (2010: 155): how can deliberative processes, which under most designs involves tens to hundreds of people, scale up to the level of a "deliberative system?" As J Parkinson & Mansbridge (2012: 1) note, "no single forum, however ideally constituted, could possess deliberative capacity sufficient to legitimate most of the decisions and policies democracies adopt." Given that environmental (and many other social) concerns are complex and span multiple scales and levels (Cash et al., 2006), this question evades easy answer. At the level of the nation-state, the issue of representation and scale has been addressed by acknowledging that deliberation can (and should) happen at multiple sites, with the warning that this risks privileging existing sources of power (experts, legislatures, interest groups) that can mobilize deliberative capacity within the system or otherwise dominate the system (Parkinson and Mansbridge, 2012).

While the nation-state remains the principal vehicle for converting public opinion (emanating from deliberation and other channels) into administrative power, the importance of networked governance has been widely recognized in recent years

(Biermann and Pattberg, 2012; Dryzek, 2010). This further complicates deliberation in that the "demos" of the nation-state is no longer, so the process of turning deliberative opinion into administrative power becomes less clear. Dryzek (2010) suggests that governance networks can be analyzed in light of the standards of deliberative democracy (i.e., non-domination/non-coerciveness, reciprocity – the criteria vary by author). This offers some hope at least in terms of identifying deliberative democratic deficits. In some sense, networked governance may provide a vehicle for achieving the cosmopolitan vision of some deliberative democrats, in that they can build deliberative capacity and processes that span traditional polities. However, networked governance can exist in elite spaces that are hard to hold accountable via traditional democratic politics, complicating any rosy interpretation.

Given that many of the sustainability challenges documented by ecological economists and others are regional or even global in scope, a deliberative approach to governance demands consideration of how micro-level deliberative processes can work in a larger deliberative system. The deliberative functions and potential positive feedbacks for a sustainability transition outlined earlier can be viewed as operating at two levels: first, they serve a micro (political) function, helping clarify interests and evaluate claims in support of better decision-making; second, they serve a macro (social) function, cultivating pro-social behavior and democratic norms that may help invigorate the public sphere. The first (political) level has been covered extensively already; we now turn to the second (social) level by revisiting some of Habermas' earlier work on the public sphere and the crisis of modernity.

The fundamental challenge for normative transdisciplines such as ecological economics, as we described earlier in this paper, is legitimating normative claims so as to turn analysis and critique into social action. A sustainability transition involves motivating a collective response to decades of scientific warnings about the status of the planet's life-support systems. If one accepts the deliberative premise, this process of legitimation can happen via deliberative communication amongst the parties (or their representatives) affected by a given decision or concern. Translating the outcomes of legitimate deliberation into social action is the crux of praxis. Yet given the complex, interconnected – indeed, sometimes undefined – nature of many environmental problems, it is not conceivable (at least under present-day political circumstances) that a unique process of deliberation be arranged to address each and every collective decision. What is to be done?

Habermas points to a potential solution in a radicalized (or re-invigorated) public sphere (Habermas, 1991, 1987a). The public sphere is a space of un-coerced deliberation about collective problems and the general interests that pertain in solving them; it is the space where social critique can be formed and gain legitimacy. Habermas does not see the public sphere as necessarily existing in any physical space or institution (although he traces its origin to the bourgeois coffeehouses of 17th and 18th century Europe), but rather consisting of an inclusive process of communication that takes place in multiple locales and media, generating a form of public opinion that can challenge, critique, and ultimately shape the functioning of the political-economic superstructure (Habermas, 1991).

The micro-processes of deliberation – including those adapted by ecological economists – may both demand and cultivate the behaviors and social norms conducive to a thriving and effective public sphere. Insofar as deliberative processes demand and seek to promote inclusivity, reciprocity, open-mindedness, and public-mindedness (among other conditions), they may become “value-articulating institutions” (Vatn, 2005) that actually cultivate the types of deliberants conducive of a sustainability transition. In this capacity, the purported link between pro-social norms (which may be cultivated by deliberation) and pro-environmental behavior merits further exploration (Steg and Vlek, 2009; Turaga et al., 2010). The question then becomes whether the deliberative norms cultivated through micro-scale processes have the capacity to achieve sufficient support to trigger a “norm cascade” that leads to their broader acceptance in society (Sunstein, 1997). If this potential exists – that deliberation can become self-reinforcing (Grossmann et al., 2017) – it may provide support for the shift toward deliberative systems capable of grappling with regional and global-level environmental challenges. In this sense, deliberative processes could be generative of a more vibrant public sphere, cultivating the conditions needed for communicatively rational deliberation. Despite a daunting plurality of views, the expansion of deliberative processes could help create shared norms that transcend group/community bounds and can become quasi-universal or at least sufficient for normative evaluation at a supra-national level.

3.5 A Deeply Democratic Sustainability Transition: Some Open Questions

We have demonstrated that deliberation has been useful in solving problems in ecological economics, particularly around public decision-making. The second broad domain in which deliberation has been invoked is to ground an alternative theory of development such as the capabilities approach, which has the potential to integrate sustainability concerns through the demand that individual choices be justified via public reasoning, imposing bounds that can be deliberatively determined. This more radical and systemic use of deliberation demands a widespread institutionalization of deliberative processes akin to a "deliberative system." We have argued that the wider realization of a deliberative system offers the potential for feedbacks that may be conducive to a sustainability transition. In this sense, deliberation is *procedure* that may contribute to *substantive ends*.

The allegedly utopian promises of deliberative theory, combined with the strict conditions imposed by some of the earlier advocates (e.g. Habermas' Ideal Speech Act) have opened deliberative theory to extensive critique, especially with regard to the design of deliberative processes. The major criticisms have been well explored in other work: the problems of scale and representation (Dryzek, 2010; Parkinson and Mansbridge, 2012); the concerns of the difference democrats about inequality and suppression of rhetoric and other forms of politics (Dryzek, 2002); problems of legitimacy (Parkinson, 2006); problematic participants (Baber and Bartlett, 2005); and more. The response has been to introduce different forms of representation, allow for aggregation, permit forms of speech that violate some of the stricter standards of communicative rationality, and generally accommodate the ideal forms of deliberation to the messier realities of differentiated political life. The risk, of course, is that this accommodation merely reproduce the *status quo*.

In scaling up from the individual deliberative process to the broader deliberative system, many questions abound about the limits of the deliberative model. In this section we address a subset of critiques that are particularly salient to the purported connection between deliberation and the sustainability transition, which demands collective action from local-to-global.

Deliberative democracy is, in many ways, a revolutionary normative theory of democracy, in that it specifies a set of conditions that when realized constitute an authentically deliberative process (Fung, 2005). While Habermas locates the root of deliberative legitimacy in arguably universal speech processes, there is no universal law that existing power structures acquiesce to claims emanating from the public sphere, never mind actively strive to support deliberative governance processes. Even in nominal democracies, the notion that an individual can have little impact on the superstructure can create collective action problem discouraging deliberative engagement (Baber and Bartlett, 2005). Likewise, in a context of global governance, deliberative processes at the global scale may be conducted by elites, with little to no connection or legitimacy in the eyes of the governed. While Dryzek (2010) points to some potential for deliberation within authoritarian states, the development of a truly global, vibrant public sphere can expect to be constrained by the profound lack of democracy in many parts of the world. While philosophers and political theorists may sleep well with the idea that they have deduced a quasi-universal political theory of democracy, it may be the case that contentious politics – perhaps aided by deliberation – is needed to create the opening for forms of deliberative democracy to spread to the more authoritarian corners of the earth.

Even within more or less functional liberal democracies, group polarization stands as an important threat to the functioning of the public sphere and a more deliberative democracy (Sunstein, 2009). Group polarization tends to occur in deliberation amongst like-minded groups, where the result is to drive decisions "to extremes" rather than toward the center (ibid). This criticism, like many of deliberative processes, has been met with design solutions, for example using random deliberant selection or "discursive representation" to avoid homogeneity in formal deliberative settings (Baber and Bartlett, 2015; Dryzek, 2010). This begs the question of who designs the process, which is important in its own right. But group polarization is more problematic in the public sphere writ large: there is increasing evidence that the changing media and communications landscape is allowing people to self-segregate into groups that are likeminded, meaning that deliberation in the public sphere may in fact exhibit polarization tendencies (Sunstein, 2017). This may be compounded by actual physical segregation along ideological lines, although the evidence on partisan sorting, at least in the United States, is mixed (Mummolo and Nall, 2017). The potential for deliberative processes to support a broader radicalization of the public sphere conducive to a sustainability transition may be constrained by changing social conditions.

The actual conditions (including capability deprivations) facing many people around the world may constitute a barrier to participation in a vibrant public sphere. In terms of deliberative capacity, we do not refer to the ability of deliberants to communicate (i.e. we set aside questions about non-human nature, the disabled, etc.), nor about the particular structures of a political system that enable deliberation to function (Dryzek, 2010). We refer rather to achievement of the broader development conditions (or "minimum capabilities") conducive to active participation in a vibrant public sphere, regardless of the political context. In this vein, Nussbaum (2000) has proposed a minimum set of capabilities based on a conception of a good human life ("eudemonia") and the idea of humans as fundamentally social beings (Muraca, 2012). Is not so much that these are preconditions for deliberation *tout court* – clearly people can deliberate under even terrible conditions – but they constitute a minimum standard of development that, arguably, may be needed to sustain a broad-reaching deliberative system. These claims merit investigation, but clearly the effective turning of deliberative agreements into administrative action is challenging in the context of a failed state, deep authoritarianism, or famine conditions. To respect Sen's view that capabilities be determined solely through public reasoning, one could envision a structured, global deliberative process to specify a set of "minimum" universal capabilities, similar to the global norm formation advanced by Baber & Bartlett (2015).[4] For ecological economists, the challenge then becomes identifying efficient, fair, and legitimate means to provide the minimum capabilities while remaining within planetary boundaries.

Were the various barriers to a global, vibrant public sphere described earlier in this section removed, deliberative scholars must still respond to several concerns relevant to the sustainability transition. Deliberative processes are specifically designed to foster slow, careful reasoning and to provide space for the airing of many or even all views. Yet the ecological crises motivating many of the normative transdisciplines are urgent, demanding rapid and often major action. It remains unclear whether deliberative institutions or even a deliberative system is capable of generating the course correction needed to avoid the effects of transgressing critical ecological thresholds. As Baber and Bartlett (2005: 203) note, "markets, legal systems, bureaucracies, and other political institutions of all kinds have imperialistic tendencies, resist deliberative innovation, and are vested in the status quo." While deliberation may have the capacity to counter this resistance, the process may be, as the name implies, deliberate and slow. This may be, but the promise of greater legitimacy may help create durable decisions, weakening or undermining resistance borne of exclusion or a perceived lack of representation. At the

[4] It can be argued that many global normative commitments already exist (e.g. Universal Human Rights) or are implied (e.g., Millennium Development Goals), but it is not clear that the deliberative processes that led to these commitments meet the standards of "free and uncoerced deliberation."

expense of rapid decision-making, deliberation may provide more decisive decisions; legitimation may even confer strength to counter-hegemonic discourses, helping drive more substantial change.

Yet when faced with hard choices, it is unclear that the reasoned or consensus decision will be sufficient to the task. As Deriu (2012: 556) succinctly puts it: "… the paradox of democratic freedom is that... we are called on to use our own freedom to affirm the limits to that very freedom in the most radical way." There are likely some decisions that deliberation cannot resolve in a compelling fashion, demanding more aggregative procedures or contentious forms of politics.

Finally, there are open questions about the feasibility of truly representing deeply environmentalist, ecocentric discourses in deliberation. Deliberative theory has multiple points of origin: the critical theory of Habermas; Rawlsian public reason; and, the "full liberalism" of Bohman, Guttman, and Thompson (Baber and Bartlett, 2005). In the problem-solving sense, all three build on an arguably liberal humanist foundation. Deliberation seeks to respect and preserve the freedom and autonomy of the individual human, recognizing that each human exists within a pluralistic society lacking any universal normative consensus. The procedural norm of deliberation trumps any particular "comprehensive doctrine," to use Rawlsian terminology. This privileging of individuals engaging in procedure appears to be – and is often formulated as – an anthropocentric orientation for deliberative democracy (Eckersley, 1992).

In the interest of a sustainability transition, the question remains whether deliberative theory can support an eco-centric orientation (assuming such a thing is necessary) (Eckersley, 2002). The capabilities approach, grounded in deliberative theory, could support a sustainability transition in that it counters a materialistic notion of "freedom purchased on the market" with a richer concept of freedom that combines procedural and consequential elements and has the potential to be substantially dematerialized. As described earlier, deliberation may privilege normative claims that invoke the general interest, which may include recognition of the rights of other species, future generations, or reparation of past injustices, but this is not guaranteed. This social liberal approach leaves room for ecocentric values to emerge, but does not require them as a precondition (Bell, 2002; Sen, 2004).

Eckersley (2002) contrasts an "environmental pragmatist" approach to deliberation (roughly problem-solving as we have described it) with an "ecocentric" approach, arguing that both are needed in a functioning democracy. She notes that the strategic and rhetorical interventions of ecocentrist advocates may appear to run counter to pragmatic deliberation, but in fact are efforts to create fair and favorable conditions for true deliberation to occur (ibid.). In line with Dryzek's (2010) notion of "discursive representation" as the basis for legitimacy in deliberation, Eckersley (2002) notes that the interests and perspectives of non-humans may deserve representation even if they are not actually the concern of any individuals party to the deliberation.

Ecocentric values may not emerge from a particular deliberative process, nor are they obliged to in the liberal humanist sense of deliberation. But, if one invokes the socially transformative promise of deliberation as envisioned by the critical theorists (e.g., early Habermas), the hope for ecocentrism may lie in a radicalized public sphere. It is there where counter-hegemonic discourses (including ecocentrism) can emerge and achieve “discursive representation” in future deliberations.

3.6 The Promise and Peril of a Deeply Democratic Sustainability Transition

Ecological economics builds from the simple acknowledgement that the economy is a social institution embedded within society – a perspective embraced by heterodox economists since the era of classical political economy – and that society itself is embedded within nature. This perspective is valuable, illuminating connections and relationships between human society, economic activity, and the earth system that might otherwise be neglected. Yet there is also growing recognition that merely describing and analyzing the world so as to know it better is insufficient and that, following Marx, “the point is to change it.” The normative transdisciplines have stepped into this arena, often purposefully, resurrecting old questions about how to legitimate normative evaluation and social critique.

Deliberative theory offers a potential response, proposing that intersubjective communication – fostered via deliberative institutions and processes – can serve as the basis for legitimate normative evaluation. For those agitating for a sustainability transition, the question remains whether the purposeful integration of deeply democratic, deliberative processes is adequate to the perceived task. Can the radical transformative promise of deliberation be realized? Can deliberation create the positive feedback needed to transform the way an entire species – comprised of innumerable social groups with complex histories and contested aspirations – relates to its home and its fellow species? Will the outcome bear any familiarity to contemporary visions of a sustainable future?

The challenge for normative transdisciplinary science, insofar as it embraces a deliberative democratic approach, is accepting that scientists' normative claims can only be socially legitimated in a deeply democratic, unpredictable and open-ended process. Rigorous, peer-reviewed analysis that appears to indicate an impending crisis may, despite demands to "act with urgency," may yield little in the way of immediate or coordinated response. Deliberation in its problem-solving mode can help improve environmental decision-making and, in a more abstract sense, ground alternative theories of development. Yet deliberation as catalyst for a radicalized global public sphere capable of legitimating the normative goal of a sustainability transition… that remains a hope, with some rational basis, albeit one that may be difficult to empirically test.

The evidence is indeed strong that humans are destroying the planet's life-support systems. A deep embrace of deliberative theory in ecological economics would provide the radical underpinning for a just and legitimate sustainability transition, should we agree that a sustainability transition is normatively desirable. And that is the challenge with environmental concerns: while we may deeply believe that nature has intrinsic value and therefore deserves to be protected, the crisis of late modernity has revealed the weak grounds for legitimating *any* normative claims. Deliberation and the promise of legitimation via inter-subjective communication, holds forth the hope that the force of the better argument may yet hold sway.

CHAPTER 4: PHOSPHORUS FLOWS AND LEGACY ACCUMULATION IN AN ANIMAL-DOMINATED AGRICULTURAL REGION FROM 1925 TO 2012

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Abstract

Phosphorus (P) is a scarce but critical input for agriculture, yet its overuse can lead to water quality degradation. Most P applied as fertilizer and manure binds to soils, accumulating over time, constituting a legacy source with implications for mitigating nutrient pollution. To investigate how the flows and balance of P evolved over a period of rapidly changing technology, agricultural practices, and land cover, we modeled P flows in Vermont's dairy-dominated agricultural system at county- and state-levels from 1925 to 2012. An important dairy exporter, Vermont faces water quality challenges complicated by a mismatch between the scale of the market and that of policymaking, a common occurrence in export-oriented agricultural regions. Over the period analyzed, agricultural soils accumulated at > 1,000 tonnes of P annually, accruing a legacy stock > 230,000 tonnes. The peak surplus of 4,439 tonnes occurred in 1950, declining to 1,493 tonnes per annum in 2012. Legacy P accumulation at the state-level ranged from < 1 to > 16 kg ha^{-1}, depending on year and measurement method. The decline in total P surplus reflects an 82% decline in fertilizer use that was partly offset by an increase in animal feed imports, the largest source of P entering Vermont since 1982. Despite declining inputs, milk output doubled, evidence of increased P use efficiency. Simultaneously, animal unit density increased by > 250%, enabled by rising feed imports. While feed is

imported and milk exported, manure remains in Vermont; hence, Vermont soils continue to accrue legacy P at rates > 5 kg ha^{-1}, undermining efforts to reduce P runoff and achieve water quality targets. We discuss the governance, management, and policy implications, outlining opportunities to improve input accountability to address the persistent P imbalance. We highlight constraints facing regional policymakers due to increased embeddedness in commodity trade networks.

Keywords: Phosphorus, material flow analysis, nonpoint source pollution, agriculture, nutrient management, legacy P

Highlights:

- Vermont's farms accumulated > 1,000 tonnes of legacy P each year from 1925 to 2012.
- P surplus peaked in 1950, declining since, reflecting improved P use efficiency.
- Efficiency was offset by intensification, with livestock density highest in 2012.
- Since 1982, feed imports have been a larger source of P inputs than fertilizer.
- Regulatory accountability for feed inputs can help reduce imbalances and legacy P.

4.1 Introduction

For centuries, farmers have added phosphorus-rich amendments such as manure and guano to agricultural soils to boost productivity (Smil, 2000; Vitousek et al., 2010). Since the late 19th century, there has been rapid growth in the mining and application of phosphorus (P) as fertilizer, with production in 2016 higher than ever recorded (Cordell and White, 2014; Jasinski, 2016). Phosphorus is a scarce mineral resource with no substitutes in agriculture; hence, P stewardship is of interest to policymakers concerned about long-term food security (Cordell and White, 2014; Elser and Bennett, 2011; Jarvie et al., 2015; Van Vuuren et al., 2010).

While P is a crucial input for agriculture, P runoff can contribute to water quality degradation. Phosphorus availability often limits phytoplankton growth, so runoff can drive eutrophication, especially in freshwater systems (Sterner, 2008). Some argue that the cumulative global impacts of P runoff on water quality may be nearing, or even past, a threshold or "planetary boundary," beyond which we can expect disproportionately negative impacts (Carpenter and Bennett, 2011; Will Steffen et al., 2015). Concerns about the impacts of both water quality degradation and resource scarcity have led to calls for better P management (Jarvie et al., 2015; Kleinman et al., 2015; Sharpley et al., 2015). Recent scholarship has emphasized the importance of understanding long-term dynamics – including the accumulation of legacy P in agricultural soils – for sustainable P management (Haygarth et al., 2014; Jarvie et al., 2013; Motew et al., 2017; Rowe et al., 2016; Sattari et al., 2012; Sharpley et al., 2014).

Much of the P applied to farmland binds to soil, accumulating over time (Sharpley et al., 1996). Typically less than 25% of applied P is taken up directly by plants, although uptake rates can increase significantly over time (Syers et al., 2008). Another portion – usually 10% or less – is lost to runoff in particulate and dissolved forms (Bouwman et al., 2009; Hart et al., 2004; Johnes and Hodgkinson, 1998). Both farming practices and edaphic factors (e.g., soil type, slope, climate) control the fractionation and quantity of P lost from a given farm field (Gburek et al., 2002; Hart et al., 2004; Smil, 2000; Syers et al., 2008).

The accumulation of residual or legacy P – the fraction of P inputs that binds to agricultural soils and is neither lost to runoff nor taken up by plants in the short term – can have important, long-term consequences for receiving water bodies (Carpenter, 2005; Sharpley et al., 2014). Soil P levels have been shown to be positively correlated with runoff losses (Pautler and Sims, 2000; Pote et al., 1996; Sharpley et al., 1994; Wang et al., 2015). Soils may become P-saturated over time, exhausting the ability of the soil to absorb and retain P (Sharpley et al., 1996). The accumulation of legacy P can increase both particulate and dissolved runoff losses, due to the greater concentration of P in the soil (increasing the relative quantity of P lost per unit of eroded soil) and the depletion of available sites that can tightly bind P. The effects of legacy P on runoff can persist for decades unless action is taken to reduce soil P levels (Carpenter, 2005). Soil P exhibits a spatially heterogeneous distribution in agroecosystems at global (MacDonald et al., 2011; Schipanski and Bennett, 2012), regional (Jarvie et al., 2015), and farm scales (Page et al., 2005; Wall et al., 2013). In some regions, the combination of decades of widespread fertilizer use and the growing geographic concentration of livestock production has led to “hotspots” of P accumulation in the landscape, where runoff losses also tend to be higher (Gburek et al., 2002; Motew et al., 2017).

Confounding efforts to manage P runoff is the fact that agricultural commodities are some of the most widely traded goods on the planet, and the flows of goods can themselves involve large quantities of nutrients, including P (MacDonald et al., 2015). As animal agriculture has concentrated in specific regions and farms have grown in size, farmers have increasingly turned to commodity markets to secure feed for their livestock, in effect decoupling part of their animal operation from the land they actively own and/or cultivate (Naylor et al., 2005). Regions with dense concentrations of livestock operations are thus at increased risk of running regional-scale P surpluses due to trade-related P flows, where the amount of feed and fertilizer brought into a region exceeds the amount leaving as agricultural commodities (Ribaudo et al., 2003; Schipanski and Bennett, 2012). This creates policy challenges, as farms effectively become integrated into complicated, multi-region supply chains, with environmental costs and economic benefits accruing unevenly at different points in each supply chain. Export-oriented agricultural regions can experience scalar mismatch, where “the authority or jurisdiction of the management institution is not coterminous with the problem” (Cash et al., 2006, p. 4).

The State of Vermont in the northeastern United States (US) is a model system for understanding how changing agricultural practices, nutrient inputs, and land cover impact the regional P balance, with corresponding eutrophication impacts on water bodies including Lake Champlain, the sixth largest freshwater body in the US. We conducted a P material flow analysis (MFA, or substance flow analysis) to estimate the P balance for the agricultural sector in Vermont from 1925 to 2012 at state and county levels. Quantifying the mass balance of P in Vermont's agricultural system provides an important regional indicator of sustainability. If the balance is positive, legacy P will continue to build, undermining efforts to reduce P runoff; if the balance is negative, legacy P will decline. By analyzing P flows over such a long period, we demonstrate some of the impacts of changing agricultural regimes, land cover, and technology on the P balance and legacy P accumulation. We highlight the intensification of agriculture, the growing importance of imported animal feed as a source of P, and the trend toward decoupling of animal agriculture from the land base, drawing out the implications for nutrient management policy. We place this case study in its global context, emphasizing how governance mismatch shapes and constrains the range of potential solutions in export-oriented agricultural regions.

4.2 Agriculture and Phosphorus Flows in Vermont

Agriculture has been a major feature of Vermont's landscape for two centuries, becoming the dominant land use early in the 19th century when most forest was cleared to make way for livestock (Albers, 2002). By the start of the 20th century, dairy dominated Vermont agriculture; it remains dominant today, although the industry has changed (ibid). Small farms, often located in the hills that blanket much of the state, have nearly disappeared over the past century, hastened by changing technology (e.g., the movement from milk cans to bulk tanks in the 1960s) and competition from regions with lower production costs (ibid.). As a result, much farmland has been abandoned and forests now cover nearly 80% of Vermont (Homer et al., 2015) (Figure 1). Yet, during this same period, dairy production doubled, accounting today for approximately 80% of farmland (including pasture, hay, and corn land) and 60-70% of agricultural sales, with an estimated 85% of milk and dairy products exported from the state (Parsons, 2010; Vermont Dairy Promotion Council, 2014a). No other US state is so dominated by a single agricultural commodity (Parsons, 2010).

As agricultural land has returned to forest, farming has concentrated in the Lake Champlain and Lake Memphremagog basins, which contain Vermont's two largest water bodies as well as the state's most fertile soils (Figure 2). These basins are sub-watersheds of the St. Lawrence Basin, draining northward into Quebec. Both lakes are transboundary waters, both are subject to eutrophication, and both face Total Maximum Daily Load (TMDL) P limits, which mandate reductions in P loading. State and federal officials have been concerned about the impact of P runoff on eutrophication in Lake Champlain, by far the larger of the two water bodies, since at least the 1970s (Smeltzer et al., 2012). The largest source of P entering Lake Champlain is agriculture (U.S. EPA, 2015). Phosphorus is brought onto farms as fertilizer, animal feed (e.g., corn grain, soybean meal), and mineral supplements; some leaves as a marketable commodity, and the rest accumulates in the soil or runs off to water bodies.

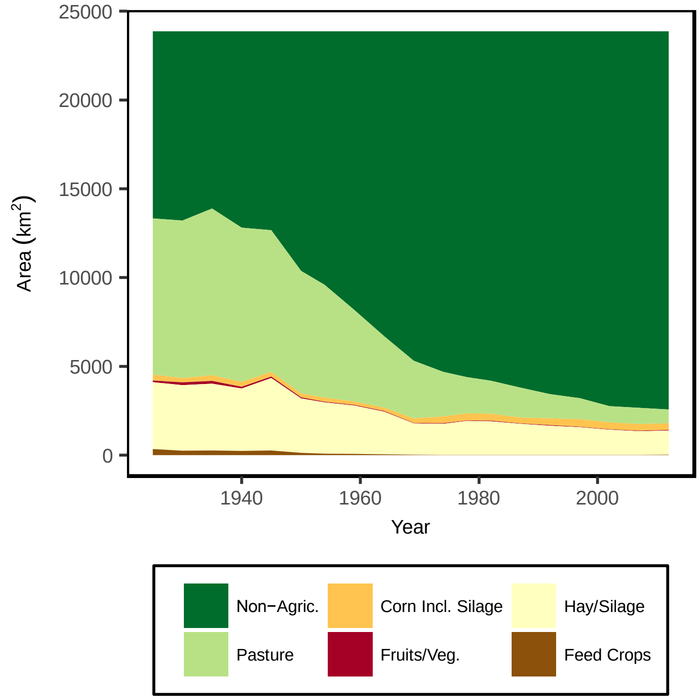

Figure 1. Changes in Vermont land cover from 1925 to 2012. "Non-Agricultural" includes forest, urban, and other land cover classes not actively cultivated or used as pasture. "Non-Agricultural" also includes land devoted to farm structures (barns, farmhouses, etc.) as well as forest that may be used for maple sugar collection.

Considerable effort, including tens of millions of dollars, has been expended to mitigate P pollution in the Lake Champlain Basin, albeit with limited success (Osherenko, 2013). Despite major reductions in P loading from wastewater treatment plants (an 80% decline since the 1970s), P concentrations in northeastern parts of Lake

Champlain increased between 1979 and 2009, while other sections remained stable or declined (Smeltzer et al., 2012).

Under the current Lake Champlain TMDL for Vermont, the agricultural sector must reduce its P contribution by 53% over the next few decades in order to meet state water quality standards (U.S. EPA, 2015). The TMDLs regulate the runoff of P into water bodies, however they do not directly regulate legacy P accumulating in agricultural soils. Yet, the quantity of legacy P is positively related to the likelihood of loss via erosion and leaching (Jarvie et al., 2013; Motew et al., 2017; Sharpley et al., 1996). Since the balance of P flows in the agricultural system determines the rate of accumulation (or drawdown) of P in soils, it is directly relevant to the achievement of the TMDL targets. This balance is presently unknown. Managing the balance will require addressing flows of agricultural P in and out of Vermont, connecting local environmental management with global commodity markets and national and international policy regimes.

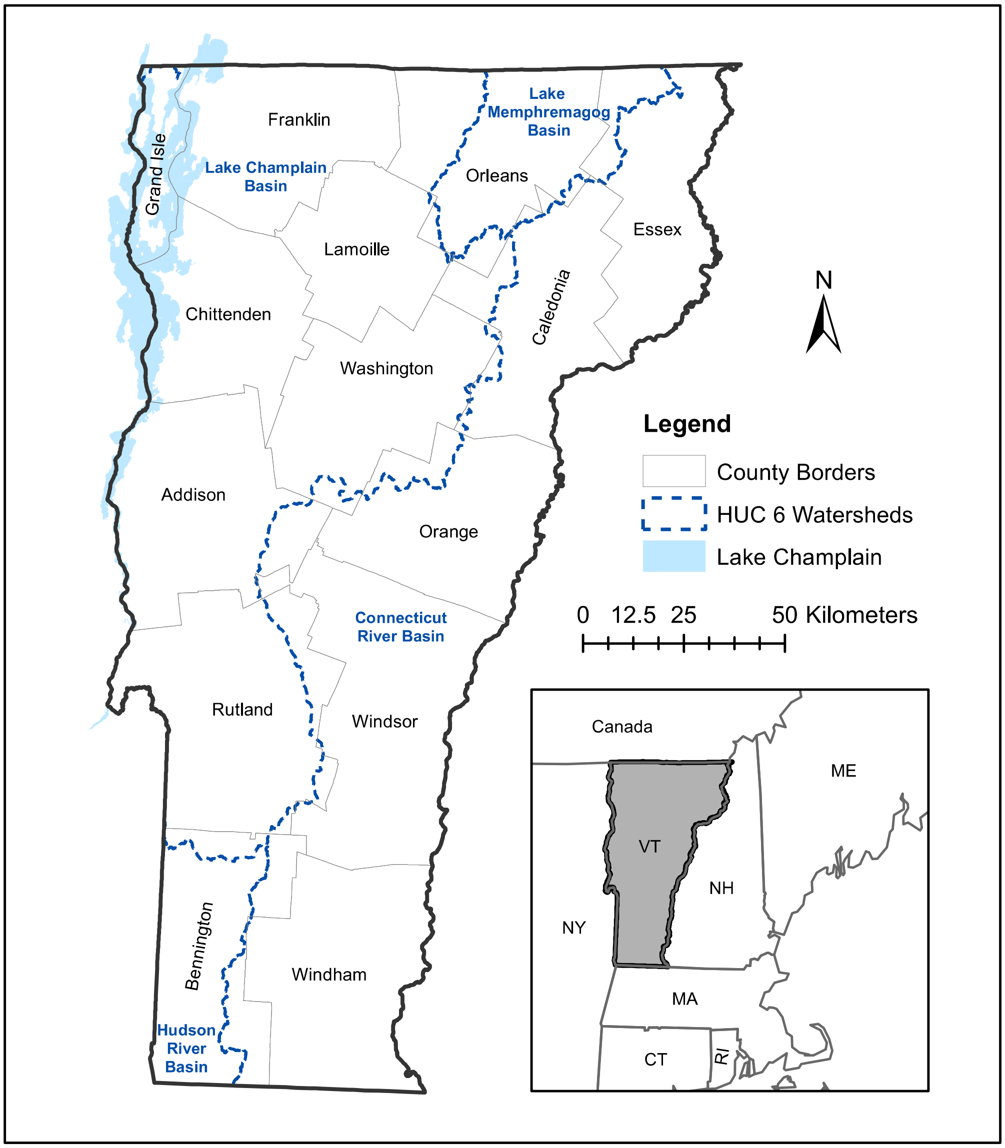

Figure 2. Map of Vermont. Counties and major (Hydrological Unit Code 6) watersheds are delimited.

4.3 Material and Methods

The Vermont MFA estimates P flows and balance (in units of elemental P) at the state- and county-level for each year in which the U.S. Census of Agriculture is published (i.e., every fourth or fifth year), starting in 1925 and ending in 2012. The MFA is a mass-balance "bookkeeping" model that tracks material flows within a defined system boundary (Fischer-Kowalski et al. 1998; Haberl et al. 2004). The method has been used to study P flows at the national and international level (Antikainen et al., 2005; Chen et al., 2008; Liu et al., 2008; MacDonald et al., 2012; Senthilkumar et al., 2012a). However, few studies assess P flows over multiple decades (except see Hale et al. (2013), MacDonald and Bennett (2009), and Schmid Neset et al. (2008)).

Most P MFAs in agriculture have used a soil surface balance approach (e.g., MacDonald et al. 2011; MacDonald and Bennett 2009; Withers et al. 2001), which tracks nutrients that enter the soil surface as fertilizer and manure and leave via crop removal (Oenema et al. 2003). We expand on this approach by including flows of P in runoff losses, imported feed, and exported animal products. The MFA includes two stocks and eleven flows that approximate the soil system P balance (i.e., legacy accumulation) while also capturing flows of policy relevance (Figure 3). Inflows consist of P that enters Vermont's farm system at the state or county-level (e.g., feed and fertilizer purchased and brought on-farm). Internal flows are those that occur within the farm system, typically between the two stocks. Outflows represent P that leaves Vermont's farm system (e.g., milk or meat transported to market, runoff from farm soils).

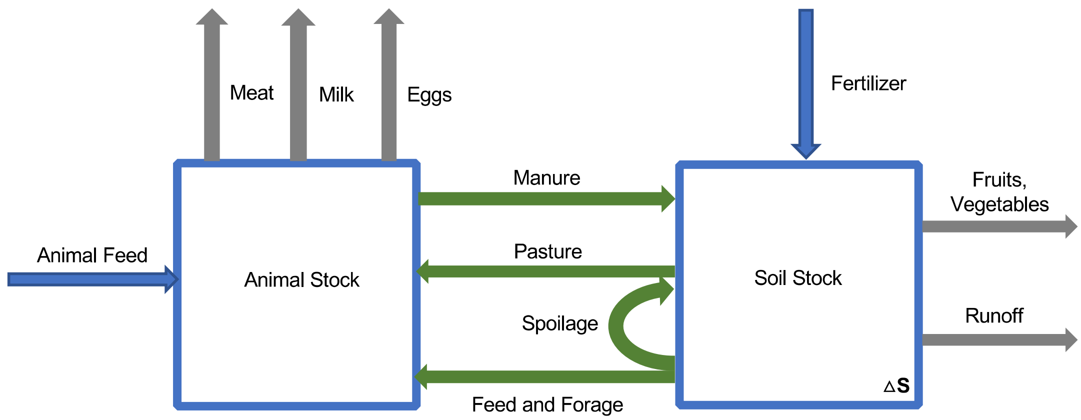

Figure 3: MFA System Diagram. Inflows into Vermont's agricultural system are indicated by blue arrows, internal flows by green, and outflows by grey. The soil stock can change due to storage or depletion.

Following Withers et al. (2001), we assume that the animal stock is at a steady-state with no change in the quantity of P stored in animals for a given year. The soil stock changes based on the net P-balance. Most P flows are estimated by multiplying a material quantity (e.g., bushels of wheat, pounds of milk) and a conversion factor representing the material's P-content (% P) (Table 1). The material flow data are mainly compiled from statistics reported in the *Census of Agriculture* (Census Bureau, 1925-1992; USDA-NASS, 1997-2012), the *Annual Agricultural Statistics* (USDA-NASS, 1936-2014), and the *Annual Statistical Bulletin – New England* (USDA-NASS, 2001-2013).[5] In some

[5] For each of these sources, we drew upon multiple years' reports to compile flow data. The year listed in the citation is the "nominal" year; for example, the Census of Agriculture for the nominal year 1925 was released in 1927.

cases, missing or undisclosed values were estimated using proxy variables or equations. A comprehensive summary of data sources, estimation procedures, and assumptions is presented in the Supplementary Information, Appendix A1.

Table 1. Stocks, flows, calculation methods, and sub-flows.

Flow	Type	Calculation Method	Sub-flows
Imported Animal Feed	Inflow	Calculated, see text	None
Fertilizer	Inflow	Reported or estimated, see text	None
Manure	Internal Flow	Milk cows: function of inventory and avg. milk production per cow All other livestock: animal count * manure production coefficient	Cattle on feed; beef cattle; milk cows; heifers and calves; steers; slaughter cattle; hogs and pigs; layers; broilers; tom and hen turkeys; sheep and lambs; goats; horses and ponies
Pasture	Internal Flow	Yield * conversion factor, with yield estimated as a function of "other tame hay" production	None
Feed and Forage	Internal Flow	Yield * conversion factor	Corn grain; wheat; oats; barley; soybeans; buckwheat; mixed grains; soybean hay; corn silage/corn grazed; sorghum hay; oat hay; alfalfa hay; small grain hay; other tame hay; wild hay; haylage, silage, and greenchop
Spoilage	Internal Flow	Constant	None
Milk	Outflow	Production * conversion factor	None
Meat	Outflow	Production * conversion factor	Beef; pork; lamb; chicken; turkey
Eggs	Outflow	Production * conversion factor	None
Fruits and Vegetables	Outflow	Estimated yield per acre * acreage * conversion factor	Apples; sweet corn; potatoes; other vegetables
Runoff	Outflow	Constant * (fertilizer + manure)	None
Animal Stock	Stock	Value = 0; assumed steady-state conditions	None
Soil Stock	Stock	Calculated as remainder	None

4.3.1 Inflows

Imports of animal feed are estimated for each county (i) and year (j) as a remainder, where:

- $Animal\ Feed\ Imports_{i,j} = [Animal\ Products_{i,j} + Manure_{i,j}] - Local\ Diet\ Inputs_{i,j}$

The sum of the counties is assumed to be equivalent to the state value for a given year. Animal products consist of milk, meat, and eggs. Manure includes all manure generated by livestock in Vermont. Local diet inputs are pasture and Vermont-grown feed and forage, less spoilage. All animal feed and forage grown in Vermont is assumed to be consumed in Vermont. Vermont is a net importer of feed and any exports would be compensated by imports, so our estimate is a minimum. We do not distinguish P imports in animal feed from those of mineral supplements mixed in the feed ration; we use the term “animal feed” to refer to both. Our approach to estimating feed imports is similar to Hale et al. (2013), although we cover a longer time period, account for more flows, and capture some technical change.

Fertilizer P use was estimated for the period 1925-1940 and derived from reported statistics for 1945-2012 (see Appendix A1). For 1925 and 1930, we multiplied the reported fertilizer tonnage used in Vermont by the average composition to obtain the quantity of P consumed. For 1935 and 1940, we adjusted five-year average P use data for Vermont (1935-1939 and 1940-1944) from the *Annual Agricultural Statistics* based on the proportion of each period’s tonnage applied in 1935 and 1940, respectively. For the period 1945-1982 we used data reported in the *Annual Agricultural Statistics*. Finally, we used county-level data from the American Association of Plant Food Control Officers for 1987-2012 (IPNI, 2012). For the period 1925-1982, we were estimated or obtained fertilizer use data at the state-level; proxy variables were used for county-level allocation.

Bedding, seeds, atmospheric deposition, and chemical weathering are excluded as insignificant; collectively they account for less than 1.5 kg P ha^{-1} y^{-1} (see Appendix A1). We exclude animal imports and exports since no data are available and these flows are expected to balance. Vermont has little year-to-year variation in animal stocks and is dominated by dairy, with only 1% of animal units comprising “cattle on feed,” i.e. cattle that may spend only part of their lifecycle in Vermont (USDA-NASS, 2012).

4.3.2 Internal Flows

In the MFA, feed and forage P flows directly into Vermont’s animal stock, with high rates of internal recycling through land application of manure. We estimate P uptake based on reported yields and conversion factors compiled from multiple sources (see Appendix A1, Table A1-6). A spoilage factor of 7.5% was applied to all feed and forage flows. Hale et al. (2013) apply a 10% spoilage factor, MacDonald et al. (2012) apply a 4% spoilage factor, and Antikainen et al. (2005) apply a 5% spoilage factor to hay and 10% to other feed stocks. Our value was selected to fall within this reported range. Spoilage is assumed to be disposed of on-farm, where it is typically mixed with manure and spread on fields.

We estimate pasture P uptake using the method reported in Conrad et al. (2016). The *Census of Agriculture* variable "other tame hay," which is the dominant hay type grown in Vermont, was used in combination with a defined harvest efficiency to approximate pasture production. This is different from most P MFAs, which use a constant uptake value (e.g., Kellogg et al., 2000; MacDonald et al., 2012). This approach helps control for inter-annual climate variability and changes in pasture management and quality, which are assumed to track hay lands. The P conversion factor for pasture is derived from the Dairy One Forage Lab (2017) and assumes an 80:20 mix of grass and legumes (Goslee, 2014).

We estimate the flow of P from animal manure to soil stocks using the animal inventory and a P excretion factor applied to each animal type (Appendix A1, Table A1-4). The P content of manure varies substantially based on animal age, life stage (e.g., lactating or not), breed, diet, and other factors (ASAE, 2005; Nennich et al., 2005; Rotz et al., 2002). The standard approach in most P MFAs is to use "as excreted" constants to convert animal inventory into a P flow, which is appropriate at the regional scale. However, during our study period, animal production practices and genetics changed substantially (Capper et al., 2009). This is especially true for milking cows, which dominate Vermont's animal herd (Figure 4). Milk production per cow increased by more than 400% between 1925 and 2012 (USDA-NASS, 1925-2012); at the same time, Holsteins increased from 40% of the dairy population to over 80% (Covington, 2013; USDA, 2008). From 1950 to 1997, the quantity of grain and concentrates fed to dairy cows in Vermont rose from 803 kg animal^{-1} y^{-1} to 2,635 kg animal^{-1} y^{-1} (USDA-NASS, 1936-2014). Hence, we adjusted the P excretion factor for milk cows to account for rising milk output. We used equation 22 from the ASAE 384.2 standard, which estimates P excretion (in grams animal^{-1} day^{-1}) based on milk output (in kg animal^{-1} day^{-1}) (ASAE, 2005):

- $P_{excreted} = (Milk_{output} * 0.773) + 46.015$

Finally, animal mortalities are excluded due to the steady-state assumption described earlier.

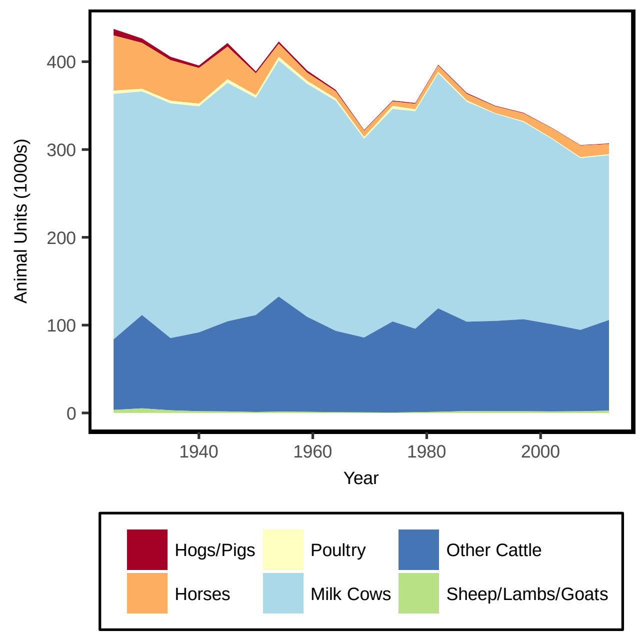

Figure 4. Change in the quantity and distribution of animal units in Vermont from 1925-2012. One animal unit is 1,000 lb of equivalent animal, so a 1,400 lb dairy cow equals 1.4 animal units. See Appendix A1, Table A1-5 for details.

4.3.3 Outflows

State-level reported egg, milk, and meat production data were allocated to Vermont's counties based on each county's share of the relevant animal herd (e.g., milk production was allocated based on the size of each county's milking herd), with P conversion factors derived from multiple sources (see Appendix A1, table A1-7). Fruit production in Vermont is dominated by apples (at least 80% of acreage in all years), for which yields are reported. Yields are unreported for most other fruits for most years, hence the P uptake rate per hectare for apples was applied to all hectares of orchard fruit land for all years. Berries were excluded as insignificant. Comprehensive reports of vegetable yields by crop were unavailable for most years, except for potatoes, dry edible beans, and sweet corn (acreage only). Hence, vegetable land area was divided into four pools: potatoes, dry edible beans, sweet corn, and other vegetables (see Appendix A1).

Runoff losses were estimated by multiplying total soil inputs (manure and fertilizer) by a loss factor (Rotz et al., 2002; Sattari et al., 2012). Sattari et al. (2012) apply a 10% loss factor in a global study, whereas Rotz et al. (2002) apply a 5% loss factor in a study of northeastern US dairies. We use a loss factor of 7.5%, the median value from these studies. This approach provides an approximation of runoff losses, but fails to factor in changes in the type of land cultivated (e.g., abandonment of marginal hillslopes), cultivation practices (e.g., no-till), and changing soil P levels, all of which influence P runoff rates yet are not captured in available data for much (or all) of the period analyzed. Similarly, extreme events (e.g., Hurricane Irene in 2011) can affect losses significantly (Yellen et al., 2014); our approach captures long-term trends but misses inter-annual variation. We conducted a sensitivity analysis (see Appendix A1) to compare the effect of using different loss factors and to evaluate results against estimates made using the process-based Soil and Water Assessment Tool (SWAT) for the Lake Champlain Basin (U.S. EPA, 2015), concluding that our approach was robust for the given purposes.

Manure export, while a potential pathway for P, is negligible in Vermont and was excluded.

4.3.4 Calculating P Use Efficiency, P Balance, and Total Accumulation

Phosphorus use efficiency (PUE) is a measure of how efficiently the agricultural system converts P inputs into P outputs. We calculated PUE as a ratio of outflows to inflows, where outflows consist of animal products (milk, meat, eggs) and crops for human consumption, and inflows consist of fertilizer and imported animal feed.

The P balance is calculated at the county- and state-levels as P inflows minus P outflows minus runoff. To assess the uncertainty of the estimated balance, we assigned uncertainty intervals to each variable based on reported statistical properties or expert judgement, allowing error to propagate through the calculations necessary to estimate the balance (see Appendix A1). This approach is based on Hedbrant and Sörme (2001) and Antikainen et al. (2005; 2008).

The balance is reported as a total and as surplus per unit area (kg P ha^{-1} y^{-1}). The surplus per unit area is a broad indicator of nutrient management and can be a useful benchmark for comparison (Cela et al., 2014; Öborn et al., 2003). However, surplus per unit area is reported inconsistently: some studies include only harvested cropland in the denominator, which captures the land most likely to receive fertilizer or recovered manure (MacDonald and Bennett, 2009); some include all cultivated land, including pasture (Cela et al., 2014); and some include harvested cropland and a portion of pasture, to correct for pasture that may be inaccessible to manure-spreading equipment (Kellogg et al., 2000). The inconsistency makes results difficult to compare and to interpret, hence we report all three.

Finally, following MacDonald and Bennett (2009), total P accumulation is estimated via linear interpolation between each *Census of Agriculture* to generate a surplus for each year.

4.3.5 Additional Analyses

We evaluated the relationship between P surplus per unit area and animal unit density, expecting a strong relationship. Because we used reported values for each time step and county in the model, we had to control for the effects of both time and geography. We used a linear, fixed-effects model (with time-demeaning) to control for these effects. The analysis was done in R using the plm package and the “within” transformation.

4.4 Results

Vermont's agricultural sector has consistently had a statewide annual P surplus of 1,000 tonnes or more, reaching a peak of 4,439 tonnes in 1950 (Figure 5). Since the early 1960s, the surplus has declined, reaching an estimated 1,493 tonnes in 2012; nevertheless, legacy P continues to accumulate, with Vermont agricultural soils accumulating nearly 240,000 tonnes of legacy P from 1925 to 2012 (Appendix A2, Figure A2-1). Underlying the declining surplus is a major reduction in fertilizer use. Fertilizer use peaked at 4,152 tonnes in 1950, declining to less than 800 tonnes in 2012 (Figure 5). Notably, feed imports increased, constituting the largest source of P entering Vermont's agricultural system since 1982. Feed imports currently contribute more than three times as much P to Vermont's agricultural system as fertilizer.

Phosphorus use efficiency (the ratio of P exports to P imports) reached an early peak of 0.34 in 1930, during the Great Depression (Figure 6a). The lowest PUE of 0.15 was observed in 1950, coincident with peak P surplus. By 2012, PUE had risen to 0.40, its highest level during the period analyzed, meaning that 40% of P imported into Vermont's agricultural system as fertilizer and feed subsequently leaves as food commodities. Underlying the increase in PUE and decrease in surplus P since 1950 is a general trend of intensification in Vermont's agricultural sector, illustrated by increasing crop and forage P uptake per hectare, animal unit (AU) density, and P exports per AU (Figure 6b-d).

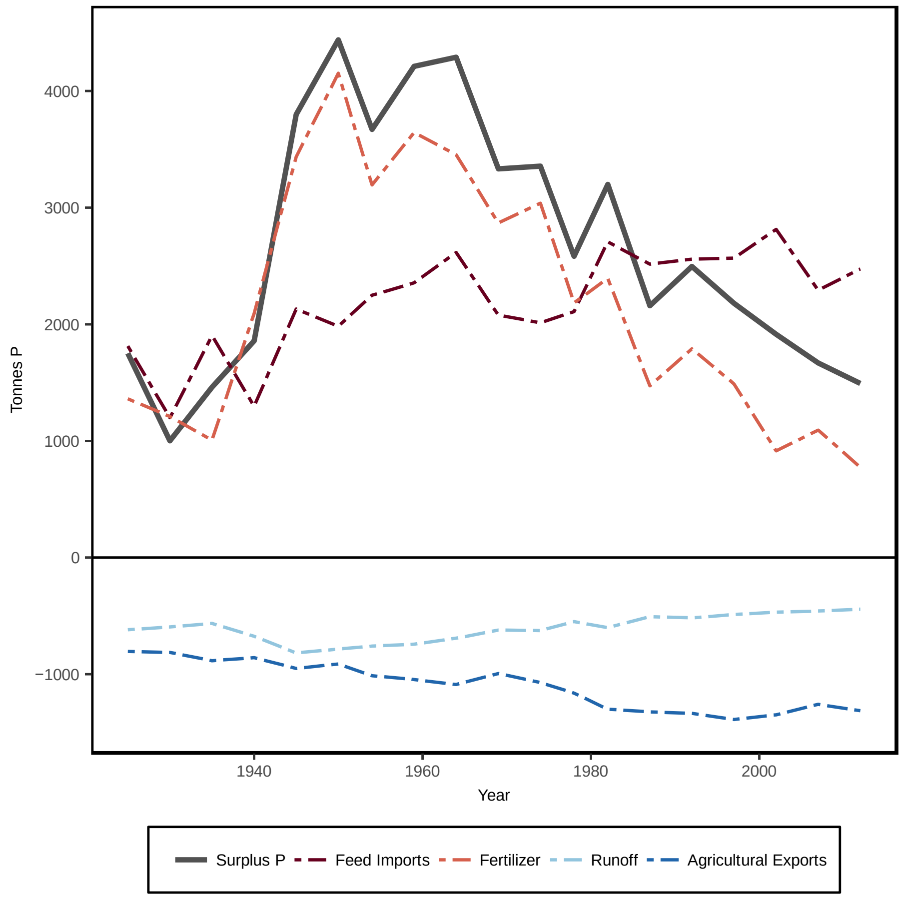

Figure 5. Phosphorus flows and net balance in Vermont from 1925-2012. Negative flows (values below zero) indicate outflows from Vermont's agricultural system.

Increasing animal unit density means there is less land available to assimilate manure generated by the animal herd. Manure was the largest source of P inputs into Vermont's agricultural soils for the entire period analyzed (Appendix A2, Figure A2-2). The effects of increased AU density are partly offset by increased crop uptake rates, which rose nearly four-fold since 1925 (Figure 6b), reflecting productivity improvements at the level of the crop and shifting crop cover.

Most cropland in Vermont is dedicated to growing animal feed and forage and has been for the entire period analyzed (Figure 1). Land dedicated to agriculture has declined more than four-fold since 1935, when more than 60% of the state was cultivated; less than 15% of the state was cultivated in 2012. Notably, the area devoted to pasture, which has a relatively low P uptake rate, declined by an order of magnitude; the area of corn, which has a high P uptake rate, scarcely changed. The result is that despite productivity improvements, 30% less P was taken up from Vermont farmland in 2012 compared to 1925.

Overall, the decline in cultivated land outpaced the decline in animal units, leading to an increase in animal unit density of more than 250% (Figure 6c). Animal production itself changed, with P exports per AU more than doubling (Figure 6d), largely driven by an almost five-fold increase in average milk production per cow and a doubling of total milk production (Census Bureau, 1925-1992; USDA-NASS, 1997-2012). The drive for increased P output per animal (as milk and, secondarily, meat) has led to increases in feed intake per animal.

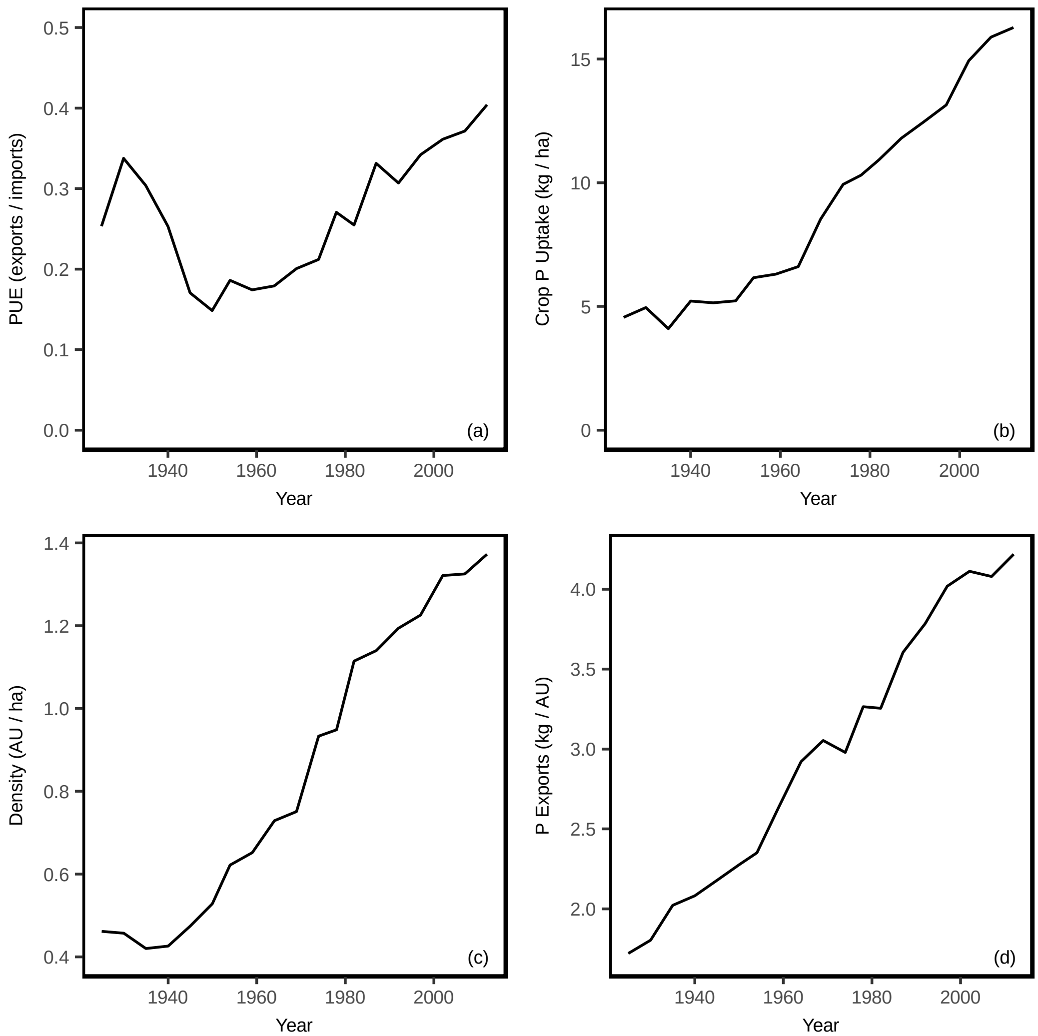

Figure 6. Indicators of agricultural intensification and phosphorus use efficiency. Phosphorus use efficiency (PUE) (a), crop P uptake rate (b), animal unit (AU) density (c), and P exports per AU (d) in Vermont from 1925-2012.

The net effect is best captured in the P surplus per unit area (kg ha^{-1} y^{-1}), an important indicator of farm- and regional-level nutrient management (Soberon et al., 2015). The surplus represents the total amount of P added to agricultural soils beyond the amount removed via crop harvest, grazing, or runoff. The surplus per unit area is reported using three different denominators – see Section 3.4 for an explanation. Vermont farmland has had a surplus of > 5 kg ha^{-1} since 1960 (Figure 7).

Available data suggest that less than 10% of pasture receives fertilizer or manure beyond what is deposited while animals graze (USDA-NASS, 2017), implying that the effective surplus on many fields may be closer to the "cropland" rate, which exceeded 16 kg ha^{-1} from 1964-1974, than the "cropland + all pasture" rate. It should be noted that we report mean values, which inevitably mask heterogeneity at the level of the county, farm, and field.

The state-level trends mask considerable county-level variation (see Appendix A2). While total P surplus has declined in all counties (Appendix A2, Figures A2-3 and A2-4), Addison, Franklin, and Orleans counties have seen their share of Vermont's surplus increase (Appendix A2, Figure A2-5). All other counties have seen their share of the surplus remain stable or decline. Addison, Franklin, and Orleans counties have accounted for more than half of Vermont's surplus since the 1980s, with each county accumulating > 200 tonnes of legacy P in 2012 (Appendix A2, Figure A2-4 and Table A2-1). Franklin County had the largest total surplus (453.9 tonnes); if Franklin County completely stopped applying fertilizer, it would still have a surplus of 230.9 tonnes.

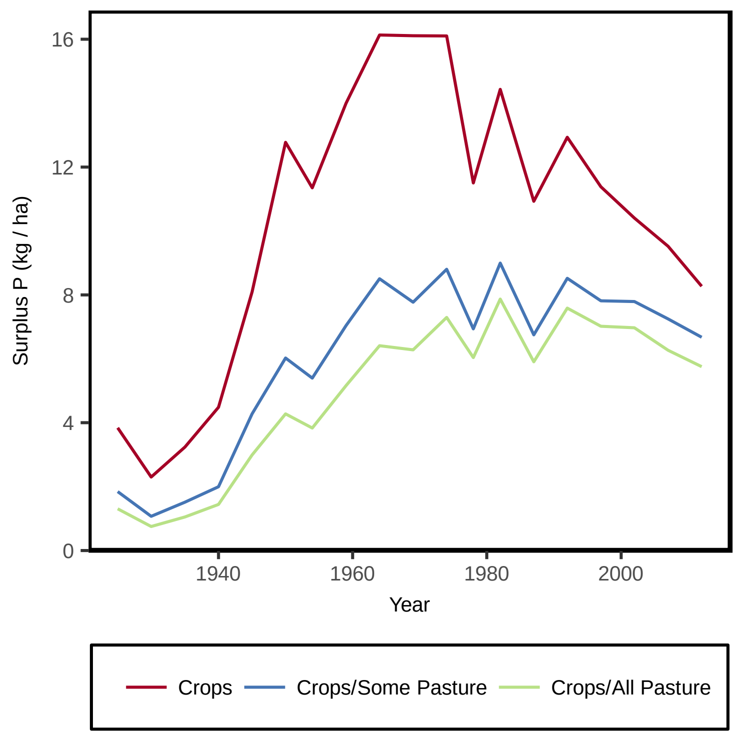

Figure 7. The rate of legacy P accumulation. Surplus P accumulation is reported with three denominators – see Section 3.4 – to facilitate comparison with other research.

Franklin and Orleans counties also appear to be accumulating legacy P at an increasing rate each year (Appendix 2, Figure A2-6), at least when measured as surplus P divided by "cropland + some pasture" and "cropland + all pasture." These two counties had the highest area-weighted surpluses in 2012 (10 to 14 kg ha^{-1} y^{-1}, depending on calculation method), which is important since they collectively represent 27.6% of Vermont's agricultural land and have the highest animal unit densities in the state, as well as below-average PUE (Appendix A2, Table A2-1). Conversely, Essex and Rutland counties achieved area-weighted surpluses <2 kg ha^{-1} y^{-1} in 2012. However, these counties account for less than 10% of Vermont's agricultural land. While available data prohibit a more detailed analysis of the factors contributing to high P accumulation rates, there is a strong correlation ($r^2 = 0.50$, $P < 0.001$) between AU density and P surplus per unit area (Figure 8).

The increasing importance of Addison, Franklin, and Orleans counties reflects their dominance of Vermont's milk production, as well as the decline of agriculture in most of the southern and eastern parts of the state. In the period we analyzed, agriculture steadily concentrated in the Lake Champlain and Lake Memphremagog basins, which are the two major Vermont watersheds subject to TMDL limits on P loading.

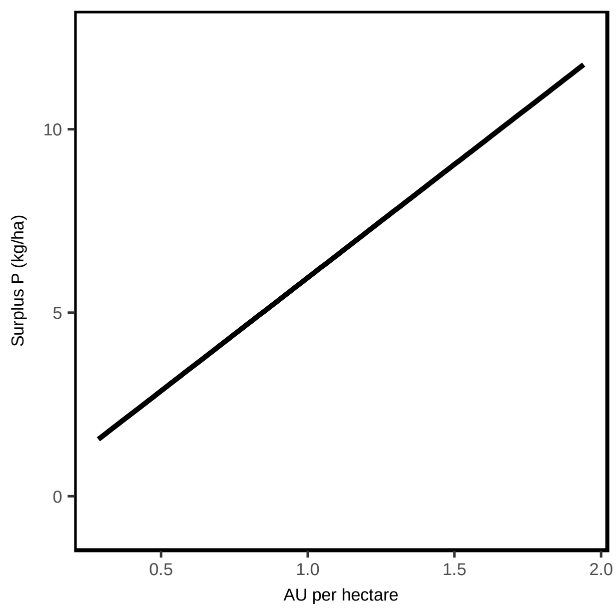

Figure 8. Surplus P (kg ha^{-1}) vs. animal unit density (AU ha^{-1}). The shaded area represents the 95% confidence interval for the linear fixed-effects model. Surplus P/ha was calculated using the "cropland + all pasture" method.

4.5 Discussion

Vermont continues to accumulate significant quantities of legacy P in its agricultural soils, although the rate has declined significantly since its peak following World War II. This reflects declining P fertilizer use and increased efficiency in converting inputs into commodities, trends observed in other long-term P MFAs in Europe and North America (Antikainen et al., 2008; Bouwman et al., 2009; Hale et al., 2013; MacDonald and Bennett, 2009; Withers et al., 2001).

4.5.1 Changing P Inputs

Declining P fertilizer use has been observed in Europe and North America as farmers have built up soil P stocks, increased efficiency, and taken advantage of specialty fertilizers (Scholz et al., 2013). In the U.S., P fertilizer use has declined nearly 25% from its peak in the 1970s (Nehring, 2017). In Vermont, where the peak occurred two decades earlier, P fertilizer use in 2012 was 80% less than peak use. This discrepancy is partly explained by the decrease in cultivated land area. When adjusted for the amount of harvested cropland, fertilizer use in 2012 was 64% below peak. Farmers may also be responding to public concern about eutrophication that has persisted since the 1970s, as well as state policy to reduce P use (Osherenko, 2013).

In an animal-dominated agricultural region, there are limits to how far P surpluses can be reduced via efficient fertilizer use. First, it can be expensive and difficult to move manure to some fields (Kellogg et al., 2000; Sims et al., 2005). Continued fertilizer use may be required to maintain soil P in these cases. Second, farmers may undervalue the potential of manure as a substitute for fertilizer (Nesme et al., 2015). Third, farmers have an incentive to reduce fertilizer use because it is a direct expense, whereas manure is a byproduct that must be disposed of even when there is more than can be easily recycled (Gollehon et al., 2001; Haygarth et al., 2009). Our results underscore this challenge. In 2012, if Vermont's farmers completely eliminated P fertilizer use, they would still face a surplus of more than 700 tonnes, or 2.8 to 4.0 kg ha^{-1}.

While fertilizer use and the net P surplus have declined, feed imports have risen to where, in 2012, they were a source of three times more P than fertilizer. This is noteworthy because the use and sale of P-containing animal feed has not been tracked in Vermont. The recent Vermont Clean Water Act (64) of 2015 requires commercial feed dealers to report tonnage of feed sold in Vermont for two years beginning in 2016 (Vermont, 2015); it is unclear if this will be converted into P equivalents and if and when these data will be made publicly available. Notably, Act 64 imposes a modest water quality tax on fertilizer but fails to tax feed. Our results suggest this is an important oversight in Vermont's efforts to reduce P pollution.

4.5.2 Managing Legacy P

The nearly 240,000 tonnes of accumulated legacy P will remain in Vermont's soils until it is lost via runoff or removed via grazing and crop harvest. To stop accumulating legacy P, the P applied to agricultural soil must equal the P removed by crops and runoff. The persistent water quality problems in Vermont, combined with the history of P accumulation we document, suggest some farmers may need to actively mitigate legacy P, in addition to taking measures to reduce runoff losses (e.g., through buffer strips, cover cropping). Legacy P can be mitigated by reducing fertilizer use, increasing or maintaining crop uptake, exporting manure P to regions facing a P deficit, and/or reducing feed imports. Each strategy has limitations and secondary effects.

Managing fertilizer use has already played an important part in reducing P surpluses in Vermont, yet is insufficient on its own, especially in counties such as Franklin and Orleans. A second strategy is to use crops to remove legacy P. Crop uptake rates (the quantity removed per hectare) climbed steadily from 1925-2012, reflecting changes in technology, cropping patterns, and management. If inputs are lower than uptake rates, crops can draw down legacy P. For farmers with soils that have P levels above desired agronomic levels, legacy P could constitute a “free” stock of fertilizer. Syers et al. (2008) suggest that up to 90% of P applied as fertilizer and manure can eventually be accessed by crops, as stable forms of P break down into labile forms. Where soil P levels are high, farmers may be able to cease fertilizer and manure application for a period without detrimental effects on yields, providing an alternative pathway is available for manure disposal (Rowe et al., 2016).

The ability to draw down P levels via crop harvest is complicated by land cover change observed since 1925 (Figure 1). The period prior to 1970 saw a rapid decline in land in agricultural production; this period accounted for half the legacy P accumulation documented in our study (Appendix A2, Figure A2-1). It is likely that highly productive agricultural soils have been continuously cultivated throughout our study period, as reflected by the shift in agriculture away from the Green Mountains toward the Champlain Valley, the most fertile region in Vermont. Yet, a portion of the legacy accumulation likely occurred on land that shifted to another land cover. For land that transitioned into forest, the net effect is likely a reduction in total P losses due to the lower runoff losses associated with forest (Smil, 2000; Vuorenmaa et al., 2002). Additionally, some legacy P may be assimilated into biomass during succession (Vitousek and Reiners, 1975). For land that transitioned from agriculture to other uses, legacy P may be mobilized by construction, forestry, and other disturbances. However, there are no data available to estimate how much legacy P is stored in non-agricultural soils.

Another option is to haul manure to farms that need it, although widespread P surpluses suggests supply of manure exceeds demand. Additionally, there is potential for converting manure into a product that can be exported (e.g., compost, fertilizer, etc.). Several constraints limit this pathway: first, phosphate fertilizer is relatively cheap, which makes it hard for alternatives to compete (Keplinger and Hauck, 2006); second, most dairy manure in Vermont is managed via a liquid system (USDA, 2016), which simplifies management but creates a heavy, diluted product (Jokela et al., 2010) that is expensive to haul long distances (Keplinger and Hauck, 2006); third, the nearest major agricultural region in the US facing a P deficit is in northern Maine, hundreds of miles from Vermont by road (Jarvie et al., 2015). Off-farm manure disposal schemes could be facilitated via dewatering technology, packed bed compost systems, and/or anaerobic digestion (Petersen et al., 2007). Anaerobic digestion has been promoted in Vermont to create value from excess manure, e.g. through the Cow Power program (Wang et al., 2011). The limitation is that digestion does not remove P, so an end-use is still needed for the residuals. In Vermont, residuals are often used as a bedding replacement, which has financial benefits for farmers but is not a net flux of P from the agricultural system (ibid.). While there is promise for off-farm manure diversion, it is unlikely to handle significant volumes of P under current economic conditions and technology.

The most likely method to lower the P surplus in the short-term is to reduce feed imports. Reducing feed imports could be achieved through precision feeding (carefully managing animals' dietary nutrient intake to meet but not exceed requirements), placing land into production to grow more feed locally, and/or reducing herd size. Efficiency gains from precision feeding could reduce surplus without impacting herd size or output (Ghebremichael et al., 2008; Soberon et al., 2015). The potential reduction in P excretion due to precision feeding for dairy cows could be as high as 10 to 25% (Maguire et al., 2007; Ghebremichael and Watzin 2011). Evidence suggests some reduction in the P content of Vermont manure during the 1990s, possibly reflecting precision feeding (Jokela et al., 2010). In addition, idle land could be put into production, increasing local feed and forage production but also increasing runoff losses, undermining any benefits (Vuorenmaa et al., 2002). However, cultivated land consistently declined throughout our study period for myriad factors, including the low productivity of much of Vermont's farmland (Albers, 2002).

A final strategy is to reduce herd size to match the capacity of agricultural land to absorb manure and fertilizer P. If farms have more manure than can be responsibly managed on- or off-farm, then a herd reduction may be necessary. While P accumulation rates were greatest during the period from 1964 to 1992, with the highest per unit of cropland recorded in 1964 in Lamoille County (28.53 kg ha^{-1}), some counties (Franklin, Orleans) continue to have surpluses > 10 kg ha^{-1}. These counties may have insufficient land to absorb the manure produced locally.

Farmers frequently point to the decline in the size of Vermont's milking herd as an indicator of reduced environmental impact; however, the decline in animal units is more than offset by intensification, with milk production per cow increasing nearly five-fold over our study period. Additionally, as large-stature Holsteins came to dominate smaller-stature breeds, each "milking cow" became effectively larger in both size and nutrient requirements. Our results suggest animal unit density is a good predictor of nutrient surplus per unit area, echoing other studies (Nesme et al., 2015; Soberon et al., 2015).

4.5.3 Uncertainty and Information Gaps

We have attempted to draw on the best available data, although some uncertainty remains in our estimates (see the Supplemental Information, including Appendix A2,

Figure A2-7). Notably, we adjusted pasture production to account for changing yields over time by using hay as a proxy, an improvement upon previous P MFAs (e.g., Hale et al., 2013; Kellogg et al., 2000; MacDonald and Bennett, 2009). Yet, pasture production (P uptake) remains uncertain, with reported uptake rates varying substantially from source-to-source (Appendix A1). Additional research regarding prevailing pasture management and grazing practices could help improve estimates of pasture uptake and manure deposition.

Furthermore, manure P excretion factors vary widely (Appendix A1, Table A1-4), especially for dairy. While we improve upon previous P MFAs by accounting for historical change in dairy manure P excretion, the regression model used is based on contemporary research and feeding practices and likely overestimates manure production in earlier decades (ASAE, 2005). Because this drives feed imports, our estimates of feed imports in earlier years may be an overestimate, although less than it would have been without adjustment.

Finally, our simplified approach to estimating runoff likely provides a reasonable approximation yet fails to capture some spatial and temporal variance. Several recent studies have sought to better understand the connection between P inputs, legacy P accumulation, and losses at the regional-scale. For example, Motew et al. (2017) found a strong positive relationship between legacy P storage (assessed via scenario due to lack of data on soil P levels) and total P transport from agricultural watersheds in Wisconsin. Hale et al. (2015) used a statistical model to demonstrate that P inputs (agricultural and non-agricultural) were a predictor of variation in loading over space, whereas runoff was the strongest predictor of variation over time. Metson et al. (2017) report results for the US for the year 2012; yet, their study does not factor soil P levels into the analysis (actual legacy P), which they highlight as a potential reason for the weak relationship they find between P inputs and riverine P transport. These studies highlight the positive relationship between P inputs, legacy P, and P losses via runoff, but they also point to the need to improve data collection.

Spatially-explicit data on soil P levels could facilitate legacy P management and increase understanding of the connection between legacy P and runoff losses. Soil testing is already part of Vermont's strategy to reduce P losses from agriculture (USDA-NRCS, 2014). Nutrient management requirements prohibit the application of P beyond crop- or soil-specific rates in soils with a high risk of P loss (as estimated using the Vermont P-Index), and ban application in soils with a very high P-Index (USDA-NRCS, 2014). However, data on soil P levels are not collected in a manner that allows researchers or regulators to understand where critical source areas lie, or what fraction of soils require legacy P management. Evidence from New York suggests nearly half of soils have P levels at or above the critical agronomic value (Ketterings et al., 2005). Given the importance of legacy P as a factor driving runoff losses, as well as widespread evidence of highly-variable soil P levels in agricultural regions (Page et al., 2005; Tóth et al., 2014), the development of spatial datasets documenting soil P levels would fill an important knowledge gap.

4.6 Conclusion

The increasing importance of animal feed as a source of P reflects a broader global trend of decoupling animal agriculture from the land base (Gollehon et al., 2001; Naylor et al., 2005). Importing feed allows farmers to increase livestock density beyond what can be sustained locally, presenting important challenges for policymakers. While milk and dairy products are exported, manure remains, driving the accumulation of legacy P. In 1925, 77% percent of the P fed to animals was from feed and forage grown in Vermont, while in 2012 it was 63%. As pasture has been abandoned and forests have regrown, animals have been confined to fewer hectares of land. Simultaneously, Vermont farmers effectively use land elsewhere to grow feed that is timported, a case of "virtual land trade" (Würtenberger et al., 2006). In 2012, we estimate that Vermont imported more than 2,000 tonnes of P as animal feed. If this was produced in Vermont at the state's average P uptake rate for crops, forage, and pasture, it would require > 150,000 ha of land be placed into production, an increase of more than 50%. This is greater than the total land area of Chittenden County, home to the state's largest city, Burlington.

Reducing legacy P will require careful management of the P balance, including imports and exports. In Vermont, tracking animal feed sales is an important step, although it should become an ongoing effort with better accounting at the county, watershed, and ultimately farm scale. Currently there is no formal accountability to ensure suppliers are selling feed rations that meet – but do not exceed – a herd's needs. By accounting for feed and fertilizer use at the farm-scale (e.g., via the point-of-sale), Vermont could enhance existing nutrient management planning, encourage precision feeding, and facilitate regulatory enforcement. Coupling nutrient management plans – which only indirectly capture feed imports – with whole-farm nutrient balances is a promising approach (Soberon et al., 2015, 2013).

Efficiency gains encouraged by accountability can help reduce legacy P accumulation; however, the rate and duration of legacy P accumulation in some Vermont counties suggest that, absent a market for manure, the livestock herd may need to shrink. Herd buyouts have been used for nutrient management in the Netherlands (Sims et al., 2005). The challenge for Vermont, which exports the majority of its agricultural products, is that reducing the herd size will have little impact on commodity market prices, so reducing production will have direct economic costs. Similarly, efforts to internalize the costs of pollution (e.g. through tradeable permits, a P tax, etc.) may be difficult to pass on to aggregators and (eventually) consumers. Raising production costs would undermine Vermont farmers' competitiveness in a global market. Vermont already has one of the highest costs of dairy production in the US, but benefits from proximity to large markets (USDA-ERS, 2017).

Ultimately, the mismatch between the scale of the market (global) and the scale of the pollution problem (regional) constrains the options available to regional policymakers. For widely-traded agricultural commodities, policy intervention may be needed at a higher level. Beyond efforts to encourage efficiency and improve accountability, potential strategies at the regional-level could include facilitating the transition to specialty markets that reward low-input, agroecological production methods, reducing reliance on commodity markets through value-added production, and governance efforts to coordinate regulatory requirements at a level that more closely approximates the market. As animal agriculture continues to intensify and concentrate in specific regions, policymakers must seek to devise new solutions to resolve the challenge of mismatch in agricultural water quality governance.

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CHAPTER 5: INCREASING ACCOUNTABILITY IN AGRICULTURAL PHOSPHORUS MANAGEMENT: A CASE STUDY AND POLICY PROPOSAL

5.1 Introduction

Phosphorus (P) runoff is a driver of eutrophication, especially in freshwater ecosystems where P availability often limits phytoplankton growth (Smil, 2000; Sterner, 2008). For decades, public policy in the United States (US) has sought to control P runoff by targeting the major sources of P loading to water bodies, for example through Section 319 and Section 303(d) of the US Clean Water Act (Dowd et al., 2008; Shortle et al., 2012; Sims et al., 2005) . Despite these efforts, P runoff continues to degrade water quality in the US, Europe, and beyond, with some researchers suggesting that we have now crossed a "planetary boundary" for P pollution (Carpenter and Bennett, 2011).

Researchers have pointed to the need for improved P stewardship, emphasizing the fact that P is a non-renewable, essential input for food production as well as an important source of water pollution (Cordell and White, 2014; Jarvie et al., 2015). By far the majority of the world's P consumption is for agricultural purposes, where P is used as a fertilizer and as a dietary supplement for livestock (Cordell and White, 2014). Unlike other historically important sources of P pollution (e.g. detergents), there are no substitutes for P in agriculture (Smil, 2000). Farmers must ensure adequate P availability if they want to maximize yields (Sharpley et al., 1996), hence in many rural regions agriculture is the largest source of P runoff (Hale et al., 2015; Metson et al., 2017; Powers et al., 2016).

Much of the P applied to soils as fertilizer and manure binds to the soil, at least until P saturation occurs (Sharpley et al., 1996). Up to 90% of this stored P may eventually become available for plant uptake (Syers et al., 2008). The accumulation of P in soils is common in intensively cultivated agricultural regions, especially those where high livestock density creates an abundance of manure to be managed (Metson et al., 2016; Reijneveld et al., 2010). This legacy P can be a long-term source of P runoff, confounding efforts to mitigate P pollution (Carpenter, 2005; Haygarth et al., 2014; Motew et al., 2017). In general, higher levels of soil P yield greater loss rates as runoff (McDowell et al., 2001; Pote et al., 1996). Increasing soil P levels provides an agronomic benefit up to a critical threshold, above which further application provides no benefit to crops (Sharpley et al., 1996). For fields above the critical threshold, a critical part of sustainable P management is avoiding additional accumulation (Ketterings et al., 2005; Reijneveld et al., 2010; Rowe et al., 2016).

Mitigating P runoff in agricultural systems is challenging because many P fluxes are diffuse, spatially and temporally heterogeneous nonpoint sources of pollution that do not lend themselves to traditional "end-of-pipe" pollution control solutions (McDowell et al., 2009; Schoumans et al., 2014). The sheer number of farms, their often modest individual contributions, the diversity of farming practices, variable environmental conditions, and varying watershed-level susceptibility to eutrophication make it complicated and costly to regulate each farm on its own terms (Larson et al., 1996; McCann and Easter, 1999; Ribaudo et al., 1999; Rørstad et al., 2007). This is the root of the P challenge in agriculture: how to manage and regulate a substance that is an essential input, yet becomes a pollutant only under very particular, spatially- and temporally-heterogeneous conditions?

Legacy P complicates the picture further, because avoiding legacy accumulation requires managing the P flowing in *and* out of the farm to minimize P surpluses. This can be difficult in animal-dominated systems, where imports of feed can represent a major P influx into the system, eventually ending up as manure (Sims et al., 2005). Despite high rates of on-farm manure recycling, evidence suggests that many animal-dominated, intensively cultivated regions face significant P surpluses (MacDonald et al., 2011; Metson et al., 2016; Schipanski and Bennett, 2012). This may be the result of intensification and spatial concentration of livestock production, leading to increased stocking rates and the spatial decoupling of animal production from crop agriculture (Sims et al., 2005; Wironen et al., 2018). Legacy P also contributes to long lag times between interventions to reduce P runoff and measurable water quality improvements, challenging policymakers (Meals et al., 2009; Rowe et al., 2016).

In the US, efforts to mitigate P pollution in agriculture typically combine structural and cultural best management practices (BMPs) designed to reduce P transport to water bodies (e.g. cover crops, buffer strips, no-till, manure injection, etc.) with field-specific nutrient management plans (NMPs). A NMP aims to match nutrient application rates with crop and soil needs, taking full advantage of available manure, while minimizing runoff losses (Beegle et al., 2000; Dowd et al., 2008; Shortle et al., 2012).

While NMPs are promising, recent research indicates that farmers frequently fail to adopt some or all of the practices specified in their NMP, undermining its effectiveness as a management and pollution control tool (Osmond et al., 2015; Perez, 2015). In some jurisdictions where NMPs have been required, evidence indicates that limited enforcement effectively made implementation voluntary (Perez, 2015). In the Netherlands, despite strong manure spreading and nutrient management policies in place since the 1980s, P levels in non-pasture soils continue to increase, with the rate of change positively correlated with livestock density (Reijneveld et al., 2010). This suggests there may be value in considering alternative or complementary policy approaches, increasing accountability for nutrient flows and potentially streamlining enforcement (Shortle et al., 2012). One possible approach is the use of nutrient budgets or mass balances, also known as whole-farm nutrient balances (WFNBs) (Lampert and Brunner, 1999; Lanyon and Beegle, 1989; Öborn et al., 2003). A WFNB can provide a more comprehensive accounting of the P fluxes that cross the farm-gate, which govern legacy P accumulation.

In this paper, we draw on a case study from Vermont (VT) in the northeastern US to investigate how current and alternative approaches to managing P flows in agricultural systems can help improve agricultural P management, including legacy P. We emphasize the need for alternatives that can scale from the farm- to the regional-level, providing accountability that can facilitate improved water quality governance. We begin by describing the VT context, highlighting the role of agricultural P runoff in contributing to VT's water quality challenges. We review the current regulatory context, emphasizing the central role of NMPs in VT's strategy to reduce P pollution. We critically assess the strengths and weaknesses of NMPs as a tool for mitigating P pollution and legacy accumulation. Next, we evaluate WFNBs as an alternative approach. We conclude by demonstrating how NMPs and WFNBs could be combined to create a more comprehensive adaptive management system, arguing that this could increase accountability for P flows at the farm- and regional-levels, supporting improved P governance and water quality.

5.2 Case Study: Phosphorus Pollution in Vermont's Agricultural System

Animal agriculture has been an important part of VT's landscape and economy for two centuries (Albers, 2002). For much of this period, more than half of VT's land was dedicated to agriculture, although the past seventy years has seen the large-scale abandonment of farmland, with less than 20% of the state's land in agriculture today (USDA-NASS, 2012), Dairy production dominates, accounting for more than 80% of farmland and 60-70% of sales (Vermont Dairy Promotion Council, 2014b). More than 85% of dairy production is exported to other parts of the US and abroad (Parsons, 2010). Despite a more than three-fold decline in agricultural land since 1945, dairy production has doubled (Wironen et al., 2018). This reflects a broad trend of intensification in VT's agricultural sector; since 1925, VT's stocking rate (animal unit density) has increased four-fold, partly enabled by increased imports of feed (ibid.).

Since at least the 1970s, policymakers have been concerned about the effects of P runoff on water quality, particularly in Lake Champlain (Osherenko, 2013). Lake Champlain is the sixth largest freshwater body in the US, with a watershed shared by New York, VT, and Quebec. The majority is located in VT. Concerns about water quality degradation led the US Environmental Protection Agency (EPA) to impose a Total Maximum Daily Load (TMDL) limit on P runoff in 2002. The VT TMDL was updated in 2015, mandating a 53% reduction in P loading from agriculture, the largest source of P into the Lake (U.S. EPA, 2015). A P TMDL has also been established for Lake Memphremagog in the northeast of the state (VT-DEC, 2017). The eight counties falling within these two watersheds account for more than 70% of VT's agricultural land (USDA-NASS, 2012), meaning the majority is subject to a P constraint (Figure 1).

Since the 1970s, considerable effort and tens of millions of dollars have been expended to reduce P loading (Osherenko, 2013). The contribution from wastewater treatment plants – the main point source of P in VT – has declined at least 86% since the 1970s due to improvements in technology and the elimination of P in detergents, among other factors (Smeltzer et al., 2012). In agriculture, programs such as the USDA's Environmental Quality Incentive Program (EQIP) along with state-funded efforts have paid for infrastructure, equipment, and the application of BMPs to reduce P losses (Osherenko, 2013). These efforts typify the "pay the polluter" approach common in US agricultural nonpoint source pollution programs (Shortle et al., 2012). Vermont also mandated the development of NMPs for Large Farm Operations in 1999 and Medium Farm Operations in 2007 (Osherenko, 2013).

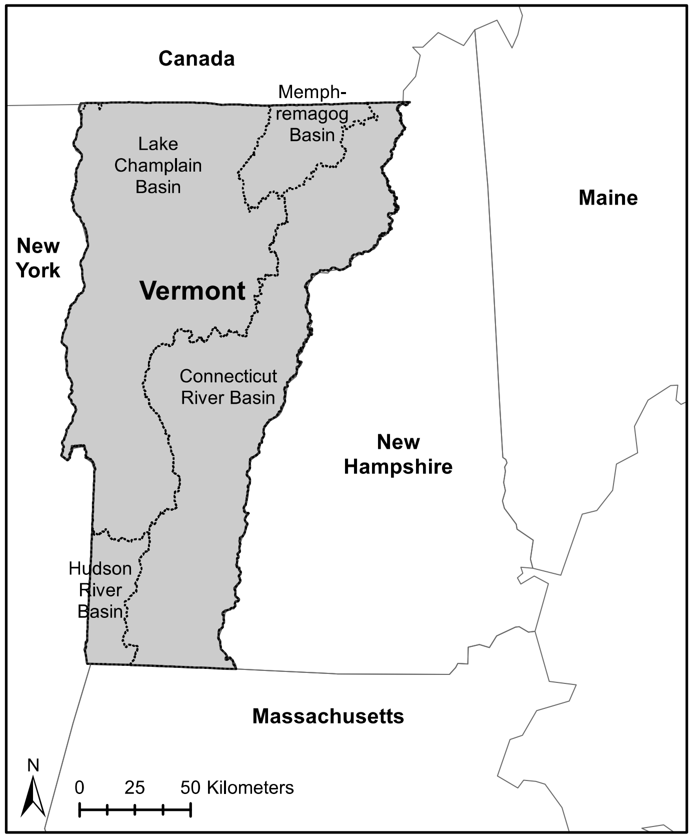

Figure 1. Vermont and its major (HUC-8) watersheds. The Lake Champlain and Memphremagog Basins are sub-watersheds of the St. Lawrence, and hence drain northward.

The results have been mixed, at least for the Lake Champlain Basin. Long-term monitoring data (1979-2009) indicate increased P concentrations in four segments of the lake, decreased concentrations in three segments, and no significant change in three others (Smeltzer et al., 2012). Monitoring of TP loading from tributaries draining into Lake Champlain gives similarly mixed results (Medalie, 2014). This P loading persists despite evidence that application of P fertilizer has declined 80% since its peak in 1950 (Wironen et al., 2018). Some of the decline was offset by P in animal feed, which now accounts for the majority of P entering the watershed (Wironen et al., 2018). Vermont's agricultural system continues to import more P than is exported, with the greatest rate of legacy P accumulation occurring in the counties with the most agriculture. The impact of agricultural BMPs and NMPs may be partly by the long lag time of legacy P in agricultural watersheds (Meals et al., 2009). No data are available to determine what proportion of VT's agricultural soils have soil test P (STP) levels above the critical level, i.e. have accumulated more P than is agronomically beneficial. Research from neighboring New York suggests that nearly half of agricultural soils have STP levels at or above the critical level, with STP levels rising fastest in dairy-dominated agricultural regions (Ketterings et al., 2005).

The recent update of the Lake Champlain TMDL spurred VT to take further action to reduce P fluxes from agriculture and other sources, resulting in the passage of the VT Clean Water Act in 2015 (Vermont, 2015). Central to the bill is the establishment of Required Agricultural Practices (RAPs), mandatory BMPs that must be implemented by all farmers (VT-AAFM, 2016). Additionally, NMPs are now required of all Certified Small Farm Operations, significantly increasing the proportion of VT's farms covered under a NMP.[6] This represents a strengthening of earlier efforts to mitigate runoff through BMPs and NMPs but does not constitute a significant shift in approach.

The persistent P surpluses in VT agriculture, as well as evidence from other regions in the US of weak implementation of NMPs at the farm-level, raises questions about how effective NMPs are as a policy and management tool in P-constrained, animal-dominated agricultural regions. In VT, much of the state's agriculture is subject to an explicit regional-level limit on annual P runoff. Can NMPs, which are central to VT's efforts to reduce P pollution in agriculture, be scaled up to the regional-level to provide accountability for P flows, including runoff? Given the long timeframe and large investment needed to achieve the TMDL and meet VT's water quality standards, metrics for farm- and regional-level accountability may help bridge scales and levels in support of P governance and stewardship.

We set aside questions about whether, how, and why farmers adopt and use NMPs, focusing instead on evaluating the NMP framework through a systems and multi-level governance lens. We review NMP requirements in VT to assess which farm P flows are captured, where there is uncertainty in NMP development and implementation, and whether and how NMPs can be scaled from the farm- to regional-level. We repeat this process for WFNBs, concluding with a synthesis.

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[6] The definition of Large, Medium, and Certified Small Farm Operation can be found here: http://agriculture.vermont.gov/sites/ag/files/RAPsummaryPDF.pdf

5.3 A Critical Assessment of Nutrient Management Planning in Vermont

Nutrient Management Plans are presently required for Large, Medium, and Certified Small Farm Operations, accounting for the majority – at least 90% – of VT's dairy herd, based on 2012 Agricultural Census data (USDA-NASS, 2012). The dairy herd accounts for 85% of VT's cattle, with cattle comprising more than 90% of total animal units (ibid.). The NMPs must be completed according to NRCS Conservation Practice Standard Code 590 (a "590 NMP"), adapted for Vermont (USDA-NRCS, 2014). A 590 NMP is built around:

- **Farm Setting**: A spatially-explicit accounting of a farm's infrastructure, livestock, fields, soils, and other natural resources;
- **Soil Conditions**: An erosion and a P loss risk assessment (VT P-Index), based on STP levels (assessed via sampling) and other soil and site characteristics, to determine the likelihood of sediment and P losses;
- **Crop Conditions**: Cropping plans, yield goals, and P uptake for each field;
- **Available Nutrients**: An accounting of P nutrients available for mechanical application to crop soils (Figure 2).

This information is used to determine allowable nutrient application rates for each field in an effort to match nutrient demand with available supply. For example, a field highly susceptible to P runoff losses (High or Very High P-Index) will be assigned a lower P application rate than a field with lower runoff risk (Low or Medium P-Index). If a plan demonstrates that a farm has a surplus of available nutrients, measures must be taken to either reduce the surplus (e.g. reducing P fertilizer use) or find an alternative end use (e.g. hauling manure to a neighboring farm with a deficit). There is some flexibility in what is included, as well as regional differences to account for varying conditions and practices; hence, we focus on the explicit minimum requirements for a 590 NMP in Vermont.

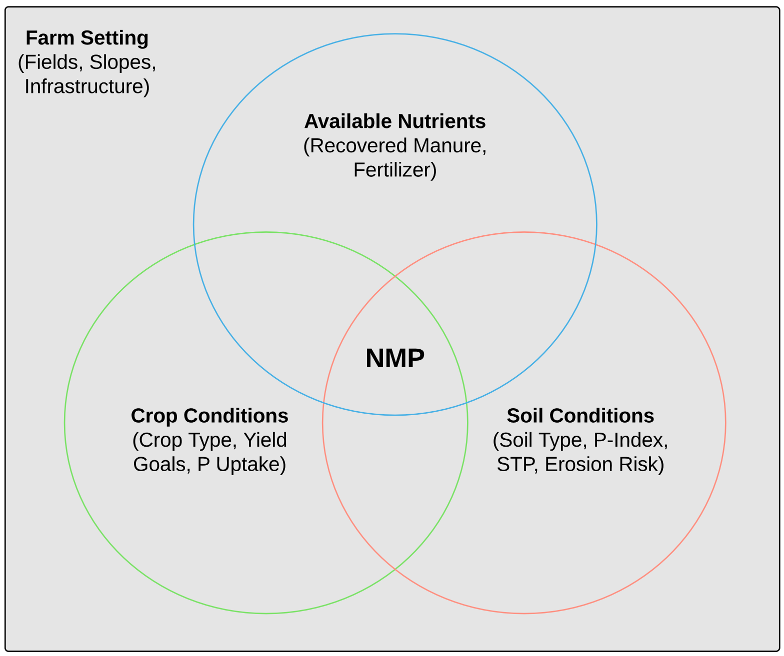

Figure 2. The main components of a Nutrient Management Plan. A 590 NMP combines information on the farm setting, available nutrients, soil conditions, and crop conditions to balance environmental and agronomic objectives. Acronyms are defined in the text.

It should be noted that a 590 NMP is distinct from a Comprehensive Nutrient Management Plan (CNMP). A CNMP builds on a 590 NMP, adding a Manure and Wastewater Handling and Storage Plan that addresses the condition and functioning of structural systems for handling manure and managing wastewater from production areas (USDA-NRCS, 2018). In most of the US, a CNMP also includes a Feed Management Plan, but this has been excluded from the Vermont requirements due to the absence of qualified, local technical service providers (Sandra Primard, USDA-NRCS, personal communication, 16 February 2018). A CNMP is required by NRCS if a farmer seeks funding for a structural improvement (e.g. a manure storage facility) (ibid.).

We define a “typical” farm to serve as the basis for our analysis. The typical farm is a dairy with some land devoted to pasture and to crops. In 2012, more than 80% of VT dairy farms had some land in pasture, with pasture accounting for approximately a third of cultivated and grazed land on dairy farms (USDA-NASS, 2012). Cropland (hay, corn, etc.) accounted for the rest. Manure management infrastructure data are unavailable, but discussion with local experts suggests most larger farms use a liquid/slurry collection system, with smaller farms more likely to use dry or bedded pack systems.

5.3.1 Accounting for Farm P Stocks and Flows in a NMP

A typical 590 NMP directly accounts for the crop/soil system as well as livestock nutrient flows from the manure storage facility “downstream” to the farm field (Figure 3 and Table 1). The volume of manure available for land application is estimated based on farm characteristics and infrastructure (USDA-NRCS, 2014). The P concentration is measured via lab testing, which introduces uncertainty due to the variable P content of manure across time and within the manure facility (Dou et al., 1997; Jokela et al., 2010).

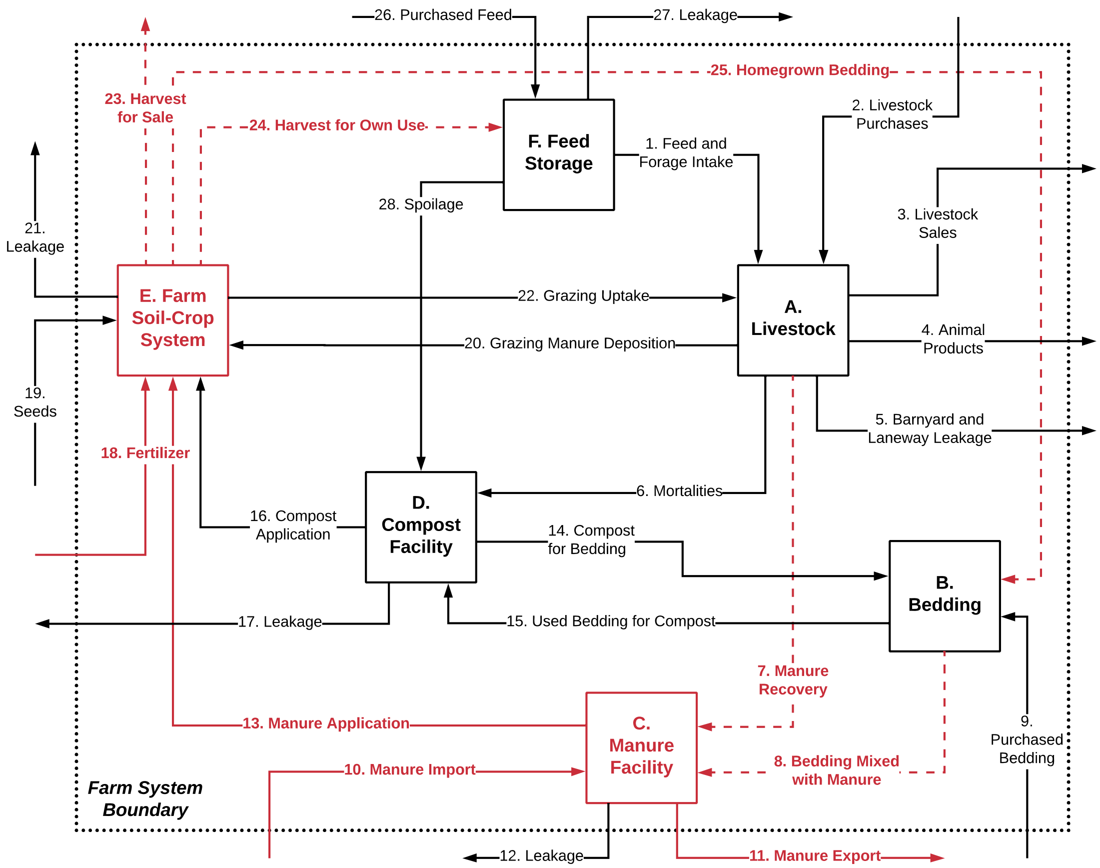

Figure 3. Systems diagram of farm-level P stocks and flows. The stocks (boxes) and flows (lines with arrows) are typical of a semi-confinement dairy system. The area in grey represents the farm system, with flows that cross the dashed border entering or leaving the farm system. The flows in solid red are directly captured in a 590 NMP. The flows in dashed red are grouped with other flows and so not directly quantified. Flows in black are not required in a 590 NMP. Leakage refers to fluxes of P into the environment, which includes runoff losses and fluxes into soil pools that are not cultivated (e.g. buffer strips, barnyard soils, adjacent forest, streambanks). Not all leakage ends up as discharge into a water body. Atmospheric transport and weathering are ignored since they are largely out of the control of the farmer. We developed the figure based on previous studies (Ghebremichael and Watzin, 2011; Soberon et al., 2013), our own knowledge of VT farm systems, and the *Integrated Farm System Model* (Rotz et al., 2016).

Modern, well-sized, and well-maintained manure storage systems are designed to minimize runoff and leaching losses, e.g. by lining lagoons and covering stacks. Not all systems meet these criteria, so a manure facility can be an important source of P leakage from the farm system, which is not captured in a NMP. Manure imports and exports are captured in a NMP, as is the manure spread on farm fields.

Manure (including urine) that is not recovered in the manure facility is ignored in the NMP: this includes manure deposited while grazing and in laneways and streams. Depending on the farm system, some portion of manure deposited in the barnyard may not be recovered, e.g. due to runoff losses and accumulation in barnyard and adjacent. Bedding, which is typically straw or sawdust (USDA, 2016), is only captured in a NMP if it is mixed with manure. Compost is only captured if it is part of the manure management system and is mechanically spread on fields (e.g. a bedded pack system).

The soil-crop system is included in a NMP, although the quantity of P stored in the soil rooting zone and in biomass is not estimated. The 590 standard requires farmers to measure STP levels every three years, with each sample representing an area of 20 acres (8.1 ha) or less (USDA-NRCS, 2014). The STP level is a proxy for the quantity of P stored in soil. Fertilizer applied to fields is captured, runoff and leaching losses are excluded, and seed inputs are excluded. Plant uptake is captured insofar as crops are harvested. The amount of P removed via crop harvest is estimated based on yield goals and historic records, although the breakdown of crops harvested for sale, for own use, and for bedding is not captured. Estimates can be optimistic, with farmers planning for ideal rather than typical yields (Osmond et al., 2015; Perez, 2015). Grazing P uptake is not captured; pastures are typically excluded from a NMP unless nutrients are mechanically applied to pasture or the farm is grazing-based.

Table 1. Farm-level P stocks and flows. The rightmost columns indicate whether these stocks and flows are captured in a 590 NMP or a farm-gate whole-farm nutrient balance (WFNB).

ID	Name	Description	Part of NMP	Part of WFNB
A	Livestock	P in the biomass of livestock on-farm	No	No
B	Bedding	P in bedding material (straw, sawdust, etc.)	No	No
C	Manure Facility	P in manure lagoon, stacks, bedded pack or other facility	Yes	No
D	Compost Facility	P in a compost facility, e.g. for mortalities or spoiled feed	No	No
E	Farm Soil-Crop System	P in the rooting zone of farm soils and contained in biomass	Yes	No
F	Feed Storage	P in stored grain, silage, hay, supplements, and other feedstuffs	No	No
1	Feed and Forage Intake	P fed to livestock	No	Yes
2	Livestock Purchases	P in the biomass of purchased livestock	No	Yes
3	Livestock Sales	P in the biomass of sold livestock, incl. for slaughter	No	Yes
4	Animal Products	P in milk, eggs, meat processed on farm, etc.	No	Yes
5	Barnyard and Laneway Leakage	P in manure and urine deposited in the barnyard, laneway, streams, or other non-productive areas that is not recovered	No	No
6	Mortalities	P in the biomass of livestock mortalities	No	No
7	Manure Recovery	P in manure, urine, barnyard runoff, and process water that is collected and recovered	Yes*	No
8	Bedding Mixed with Manure	P in bedding that is mixed with manure	Yes*	No
9	Purchased Bedding	P in bedding that is purchased and brought on-farm	No	Yes
10	Manure Import	P in manure that is imported on-farm	Yes	Yes
11	Manure Export	P in manure that is exported off-farm	Yes	Yes
12	Leakage	P in manure that is lost from the manure facility	No	No
13	Manure Application	P in manure that is mechanically applied to a farm field	Yes	No
14	Compost for Bedding	P in compost used for bedding	No	No
15	Used Bedding for Compost	P in bedding that is sent to a compost facility	No	No
16	Compost Application	P in compost that is mechanically applied to a farm field	No	No
17	Leakage	P lost from a compost facility	No	No
18	Fertilizer	P fertilizer applied to fields	Yes	Yes
19	Seeds	P in seeds and non-fertilizer chemical treatments (e.g. gyphosate) applied to fields	No	Yes
20	Grazing Manure Deposition	P in urine and manure deposited on pasture while grazing	No	No
21	Leakage	P lost from the soil root zone in fields	No	No
22	Grazing Uptake	P in crop and forage biomass consumed by grazing livestock	No	No
23	Harvest for Sale	P in crop and forage biomass harvested and sold	Yes*	Yes
24	Harvest for Own Use	P in crop and forage biomass harvested for use on-farm	Yes*	No
25	Homegrown Bedding	P in bedding material (e.g., straw) grown on-farm	Yes*	No
26	Purchased Feed	P in feed and supplements purchased for livestock	No	Yes
27	Leakage	P lost from a feed storage facility	No	No
28	Spoilage	P in animal feed stored on-farm that spoils	No	No

* = Indirectly captured, see text for explanation.

Most dairy farms have a high rate of internal nutrient recycling, although this is not fully captured in a NMP. Crops harvested for own-use are typically stored on-farm and supplemented with imported feed; the storage system is not captured in a NMP, nor are feed imports, which in VT are the single biggest operating expense and source of P imports for dairy farms. Spoiled feed, may be 5-10% of feed stored on-farm (Antikainen et al., 2005; Hale et al., 2013; MacDonald et al., 2012), is included in a NMP if it is mixed with manure, but not if composted. Leakage from feed storage systems (e.g. silage leachate) is not captured in a NMP.

The livestock herd is a standing pool of P that is not captured in a NMP. Purchase and sale of livestock is excluded, with the herd assumed to be at steady-state (Lauren Gibson, Vermont Association of Conservation Districts, personal communication, 16 January 2018). Animal mortalities are typically composted on-farm and excluded from the NMP. Animal products, which in VT are the largest P flux out of the farm system, are not captured in a NMP. Milkhouse wastewater and other process water may be mixed in with manure or drained into a treatment system and discharged. If it is mixed in with manure, then it will be captured indirectly in the manure component of the NMP. If not, it represents a source of leakage that is not captured.

5.3.2 Nutrient Management Plans: Strengths and Weaknesses

Nutrient management planning is a well-established approach to managing P flows to balance agronomic and environmental objectives. The NMP framework provides tools needed to help farmers adaptively manage their farm, making efficient use of recovered manure nutrients and minimizing fertilizer demand. It also promotes farm practices that aim to reduce the risk of P losses and erosion. Yet, a NMP does not provide a comprehensive account of all major P fluxes in the farm system. In particular, NMPs do not provide accountability for P flows crossing the farm-gate, nor do they account for runoff losses (Table 1).

Strengths of the NMP Approach

A NMP is intended to match available, recovered P stocks with a beneficial end-use, factoring in field-level details such as soil type, runoff and erosion risk, and STP levels. By focusing on recoverable manure, a NMP guides farmers in maximizing the value of nutrients available in a farm's manure storage system. This can point to opportunities to reduce P application rates, cutting fertilizer expenditures and conserving a nonrenewable resource. Soils with High and Very High STP levels are unlikely to receive an agronomic benefit from P application (Ketterings et al., 2005); a NMP provides this information to farmers, helping maximize nutrient use efficiency. However, allowable P application rates are governed by the VT P-Index, rather than STP level, so it is conceivable that a NMP may allow P application even when there is no agronomic benefit, for example because the risk of P loss and hence environmental impact is low.

A NMP includes multiple measures to reduce P runoff losses. An erosion estimation must be completed for each field, with farmers obligated to manage fields to keep erosion below a defined threshold (USDA-NRCS, 2014). There are also prescriptive practice requirements which aim to reduce P losses. The P-Index combines soil test results (STP and reactive aluminum), soil type, erosion rate, slope, crop cover, crop type, location/elevation (to estimate precipitation), P application timing/source/method, distance to water conveyance, presence of tile drain, and details on some BMPs (buffers strips, sediment traps) (Faulkner and Tilley, 2018). The P-Index estimates the risk of P loss via three pathways – sediment-bound P, dissolved P, and tile drainage P – to give an overall risk score, which can help identify fields that are priorities for BMP application. Depending on the P-Index, farmers may be allowed to apply P at rates that match or exceed crop removal rates; for soils with a Very High P-Index, P application is prohibited to allow crop removal to draw down soil P (USDA-NRCS, 2014).

Some limitations of NMPs as planning documents (e.g. overestimation of P uptake by crops, weather conditions affecting manure and fertilizer application, uncertainty in manure estimation, spreader calibration issues) can be overcome through adaptive nutrient management (USDA-NRCS, 2011). In adaptive nutrient management, careful record-keeping (soil and manure test results, manure application rates, actual yields, crop rotations, new practices) are used to refine a NMP over time. The use of a NMP in adaptive management demands not only good practice and record-keeping, but faith in the science underlying NMPs, something that is not always present (Osmond et al., 2015).

Weaknesses of the NMP Approach

A 590 NMP developed to meet VT requirements is only a partial picture of the P flows of importance on the farm system (Figure 3). In 2012, pasture made up nearly 30% of VT's active agricultural land (i.e. land grazed or cultivated), yet less than 10% received mechanically-applied fertilizer and manure, meriting exclusion from a typical NMP (USDA-NASS, 2017, 2012). However, pasture benefits from careful nutrient management (Gibon, 2005; Kristensen et al., 2005). Pasture soils deficient in P can be a potential sink for excess manure, provided spreading equipment can access them; at the same time, productive pastures can contribute to a herd's dietary needs, offsetting the import of feed.

Flows of P on- and off-farm are only partly captured in a NMP, yet the balance determines whether legacy P accumulates. Fertilizer and manure imports are captured; feed, livestock, and seed imports are not. Exports of harvested crops and manure are captured; exports of milk, eggs, and livestock are not. While the risk of runoff loss is captured through the P-Index and erosion calculations, actual loss rates/volumes are not estimated.

Critically, NMPs fail to account for at least part of the manure that is not collected in a manure storage facility and reused. Manure recoverability rates vary depending on practices and environmental conditions; in Vermont, up to 90% of dairy manure may be recoverable (Kellogg et al., 2000). Unrecovered manure may include some of the P that is deposited in the barnyard, in laneways *en route* to pasture, directly into streams, or that leaches or spills from manure, composting, and feed storage facilities and spreading equipment. These losses diminish the amount of P that can be recovered for value-added reuse and represent a direct loss from the farm into the broader ecosystem. Even if unrecovered P accumulates in soil (e.g. in a buffer strip, etc.) rather than runoff into a water body, once it leaves the rooting zone of a farm's arable land or pasture it is lost as an agronomic resource.

The partial P flow data makes it impossible to compute a farm-level mass-balance or calculate P use efficiency (PUE). Ultimately, a NMP does not provide ready accountability for P inflows, outflows, the P balance, PUE, P runoff losses, or P accumulation, all of which may be relevant to policymakers, regulators, and the public.

As conceived and executed in VT, NMPs are difficult to scale to the regional- and watershed-levels, which are important from a governance and policymaking standpoint. For example, VT's TMDL load allocations are set at the watershed-level. The absence of any streamlined means to aggregate, for instance, STP/P-Index data and NMP-prescribed P application rates makes it difficult to know where P is accumulating, where and at what rate it is being applied, and where potential hotspots are for intervention. This weakens science and impedes both voluntary and regulatory efforts to mitigate P pollution and legacy accumulation.

5.4 Whole-Farm Nutrient Balances as an Alternative Approach

Whole-farm nutrient balances (WFNBs) provide a different perspective on nutrient flows and accumulation. In VT, WFNBs have been developed at the farm-level and the regional-level as part of research initiatives (Ghebremichael and Watzin, 2011; Wironen et al., 2018). There are no regulatory requirements pertaining to WFNBs; all previous efforts have been voluntary.

> **Key Acronyms**
>
> NMP = Nutrient Management Plan
>
> WFNB = Whole-Farm Nutrient Balance
>
> NMPB = Enhanced Nutrient Management Plan and Balance

A WFNB can be developed with several different system boundaries, depending on purpose and available data (Öborn et al., 2003; Oenema et al., 2003). The *farm-gate* balance approach focuses on the flows of materials on- and off-farm, ignoring internal cycling (the focus of a traditional NMP) and runoff losses. A *soil-surface* balance more closely approximates an NMP, focusing on nutrients entering (as manure, fertilizer) or leaving (via crop uptake) the farm soils stock, although as in a NMP runoff losses are typically ignored. A *soil-system* balance extends the soil-surface model to include the soil system, therefore including losses via leaching and runoff. Hybrid approaches are also possible.

The *farm-gate* balance has been used in the northeastern US for farm-level studies (e.g. Soberon et al., 2013, 2015; and Cela et al., 2014). It involves a comprehensive accounting of P flows that pass via the farm-gate, including livestock, fertilizer, animal feed, bedding, and seeds (Figure 4). It excludes losses via runoff. The farm-gate balance is therefore the sum of inputs minus the sum of outputs, which provides an indication of nutrient use efficiency.

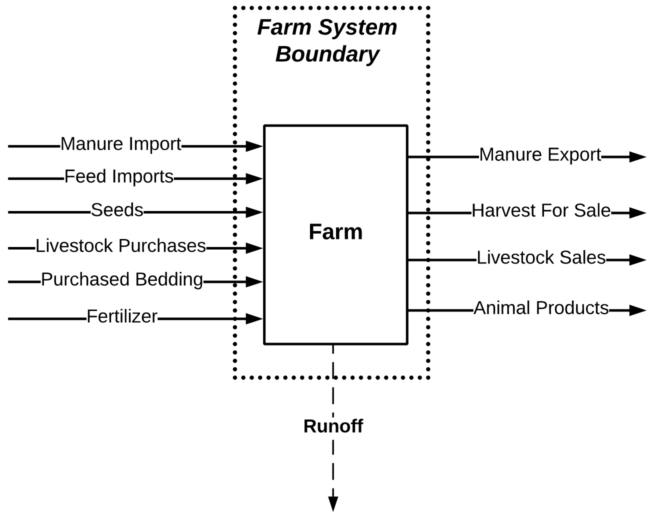

Figure 4. Farm-gate Whole-farm Nutrient Balance (WFNB) system diagram. The flows entering or leaving the farm system are denoted by solid black lines with arrows, with the exception of runoff (dashed black line with arrow). Unlike a NMP, the internal flows are not detailed or quantified; the farm is treated as a single stock. Atmospheric transport and chemical weathering are also excluded.

The P inputs can be quantified by using a specific material's disclosed nutrient content, lab testing results, or by using a "book value" for that category of material. Fertilizer, feed supplements, and animal feed purchased from a grain dealer typically has a stated chemical composition that can be used. For other inputs (e.g. bedding, livestock, feed and forage purchased from a nearby farm, seeds) a book value may be used, presenting some uncertainty (Öborn et al., 2003). For row crops, pasture, hay, and haylage, the mix of species and other factors affect nutritional value and P content, so farmers may send forage samples for testing. For imported manure, the P content may be estimated from lab sampling results (as in a NMP) or using a book value.

The P outputs can be quantified similarly. For crop sales, milk, and eggs, book values or lab testing may be used. For livestock, book values will typically be used. This is necessary because much of the P stored in livestock is in the bones, so estimates based on the P contained in meat harvested at a slaughterhouse will underestimate the amount of P that walks off-farm (Karn, 2001). For exported manure, the P content may be estimated from lab sampling or using a book value.

Runoff is not typically included, although some tools that report farm-gate WFNB results, e.g. the IFSM model, will estimate runoff losses to calculate a more complete balance (Rotz et al., 2016). Runoff is typically excluded due to the extra level of effort required to estimate losses with sufficient confidence (Öborn et al., 2003).

The WFNB approach has the advantage of scalability. A regional WFNB can be compiled from farm-level WFNBs or developed as an independent effort. Aggregating farm-level WFNBs will only give a reasonable estimate of regional conditions if the farm-level data are comprehensive and consistent, covering most or all farms in a defined region (Öborn et al., 2003). Absent a regulatory requirement pared with improved data collection efforts, this approach is likely infeasible. Hence, most regional WFNBs are compiled using secondary data and various estimation procedures, both of which introduce uncertainty.

In the case of VT, no data are available regarding the physical quantity and composition of feed imports, exports, or sales (Wironen et al., 2018). Fertilizer sales data are available in recent decades at the county-level, although a sale in a given county may not be applied in the same county, so there is some uncertainty (IPNI, 2012). Livestock sales, but not purchases, are reported at the county-level, making it difficult to determine the net balance of livestock in a given county (USDA-NASS, 2012). Crop harvests for major crops are reported at the county-level, but there is no indication whether harvested crops are consumed on-farm (i.e. fed to livestock), sold within the county, or exported (ibid.). Pasture P uptake rates are unknown and must be estimated, e.g. using a proxy such as hay yield (Conrad et al., 2016). The P content of hay and pasture varies widely (Dairy One Forage Lab, 2017). The major gaps in available data mean that analysts must use simplifying assumptions and/or different system boundaries.

Previous efforts to estimate partial or full WFNBs in VT for agricultural P flows at a regional-level (county, watershed, and/or state) have used an emissions factor approach, where each livestock class is assigned a manure excretion factor (Wironen et al., 2018; MacDonald and Bennett, 2009). This is used to estimate the amount of manure produced by the livestock, which is needed for estimating a soil-surface balance or, alternatively, the quantity of feed imported. For a soil-surface balance, the manure volume may be multiplied by a “recoverability factor” to account for the portion of manure that is actually available for land application (Kellogg et al., 2000). This approximates the NMP approach.

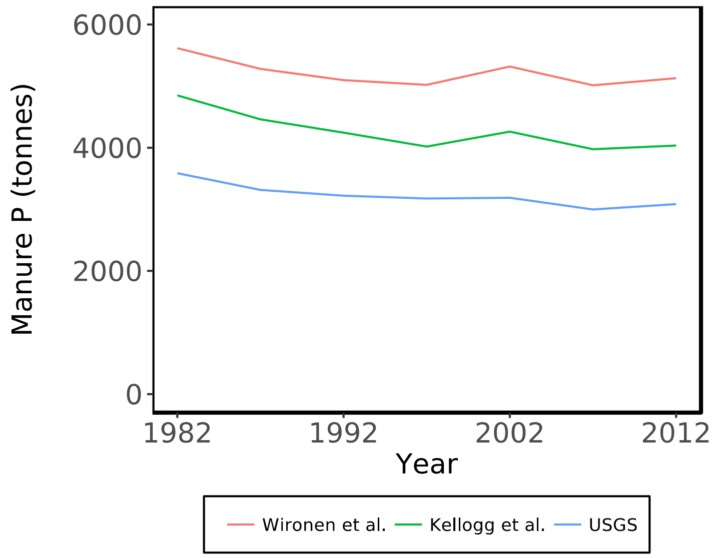

Figure 5. Three estimates of Vermont manure P production, 1982-2012. Note that Kellogg et al. (2000) only report a value for 1997; the other years are estimated using an adaptation of their method.

The emissions factor method is subject to some significant uncertainty due to the fact that manure P volume and concentration varies a lot based on diet, breeding, farm infrastructure, and other factors that change over time (ASAE, 2005; Jokela et al., 2010). The choice of emission factor can have a significant impact on the results (Velthof et al., 2015); estimates of total manure production in VT from three studies give wide-ranging results (Figure 5) due to the use of different emission factors, different approaches to quantifying the livestock herd, and variations in which livestock are included in the study. This discrepancy points to the need for improving data collection at the regional-level and/or increasing the coverage of farm-level WFNBs.

5.4.1 Whole-farm Nutrient Balances: Strengths and Weaknesses

Whole-farm nutrient balances are a well-established tool for understanding and managing P flows on- and off-farm, since they allow for farm-level benchmarking and can scale to the regional level. Yet they fail to capture runoff, lack detail on internal P cycling relevant to farm management, and require additional data and record-keeping.

In a farm-gate WFNB, the flows that are principally responsible for legacy P accumulation are captured, with the exception of runoff losses. A WFNB can help farmers benchmark performance over time and against similar farms, while also helping identify opportunities for efficiency (Soberon et al., 2013). The inclusion of animal feed highlights precision feed management as a means of reducing P inputs. A WFNB can allow farmers to directly measure the effect of precision feeding on their P balance, something only indirectly captured in a NMP via manure testing.

For benchmarking to be useful, the methods must be standardized and efforts taken to reduce uncertainty in measuring major flows (Öborn et al., 2003). If WFNBs are completed for most or all farms within a region, they can be a useful governance tool. A WFNB can be used to estimate PUE, the percent of imported P that is exported in the form of livestock, crops, and animal products. Farms with a low PUE may present opportunities for management improvements. A WFNB can be combined with information about a farm's productive land to estimate an area-weighted P balance, which can indicate whether a farm is operating with a significant P surplus or deficit.

Like a NMP, a typical farm-gate WFNB excludes runoff losses. This means the resulting balance is not directly representative of legacy P accumulating in the farm-system, and hence complicates interpretation of results. As Öborn et al. (2003: 216) note, "at best, nutrient balances and surpluses provide an integrated measure of the total nutrient loss potential," rather than actual runoff losses or legacy accumulation. Exactly what balance is considered acceptable will vary depending on farm conditions, policy objectives, and other factors. Research in New York indicates that most dairy farms can achieve farm-gate P surpluses below 13 kg ha^{-1} (tillable) simply through efficiency and cost-effective management improvements, with many farms achieving much lower surpluses (Cela et al., 2014). In 2012, two VT counties (Franklin and Orleans) had a mean surplus > 13 kg ha^{-1}, suggesting ample room for improvement, especially since they also contain non-dairy agriculture (Wironen et al., 2018).

A WFNB is of limited use as a stand-alone management tool; it does not provide the detail or agronomic utility that a NMP provides. Some invaluable aspects of a NMP – matching nutrient application rates to soil and crop needs, factoring in P runoff and erosion risk – are ignored. While a farm-gate WFNB is built around data typically recorded as part of farm management (Öborn et al., 2003), because WFNBs are not mandated, the additional record-keeping may be a burden. To facilitate this, Soberon et al. (2013) developed a WFNB calculator tool in New York that is being adapted for VT as part of the Go-Crop software package. For organic dairies, much of the required documentation (e.g. tracking feed purchases) is already required.

5.5 Discussion: Increasing Accountability for P Flows in Agricultural Systems

Phosphorus management in agricultural watersheds is often characterized by long lag-times between interventions (e.g. reducing P inputs and runoff losses) and visible or measurable results (e.g. better water quality) (Meals et al., 2009). Legacy P accumulation is an important contributing factor to this phenomenon (Rowe et al., 2016). The result is an example of governance mismatch, which is a "problem of fit involving human institutions that do not map coherently on to the biogeophysical scale of the resource, either in space or time" (Cash et al., 2006: 8). This is a common characteristic of complex, multi-level and multi-scale environmental challenges (Cash et al., 2006; Young, 2002). The mismatch between the pace of the political process and that of the P cycle challenges policymakers, who are tasked with making significant investments that may not bear fruit for years.

To bridge the temporal mismatch, there is a need for better metrics to measure and report on progress in P management. Better metrics can improve accountability for P flows, aiding regional-level scientific understanding of the P cycle, farm- and regional-level regulatory efforts, and farm-level management. One means of achieving these objectives could be combining NMPs with WFNBs to create an enhanced Nutrient Management Plan and Balance (NMPB). Linking NMPs and WFNBs has been proposed by others, for example Öborn et al. (2003) and Koelsch (2005). To explore the implications, we first discuss some metrics that could be calculated; second, we investigate how NMPBs could improve accountability for P flows at different levels;

third, we highlight challenges to the implementation of NMPBs, raising questions for further research.

5.5.1 Combining NMPs and WFNBs: Opportunities for Improved Performance Metrics

Both a 590 NMP and a farm-gate WFNB are partial pictures of P flows in the farm system (Figures 3 and 4, Table 1). A NMPB could strengthen a 590 NMP by providing full accounting of missing internal flows (pasture P uptake and deposition; compost; animal mortalities), farm-gate inflows of P (e.g. animal feed, seeds, bedding, and livestock), and farm-gate outflows of P (e.g. livestock, animal products, crops sold). A NMPB would enable calculation and reporting of a series of performance metrics that provide broad insight into P management at the farm-level, with potential to scale to the regional-level (Table 2).

A 590 NMP includes sufficient information to report on the number of farms with a completed NMP, although since pasture is frequently excluded it is inappropriate to treat all of a farm's land as "covered" by a NMP. A NMP also provides sufficient information to estimate total agronomic P inputs, potentially excluding pasture deposition, seeds, and compost application. This can be used to calculate area-weighted inputs (rate of application) at the field-, farm-, and regional-level. When paired with estimated crop uptake, this information can be used to calculate a partial soil-surface nutrient balance at the field-level. The soil-surface balance excludes runoff losses, so it cannot be directly interpreted as the rate of legacy P accumulation or drawdown. In theory, the soil-surface balance can be scaled up to the farm- and regional-level, but this would only reflect the aggregate of the farm fields contained within the farm or region (i.e. excluding laneways, barnyards, etc.), with potential for misinterpretation.

A NMP also contains important information about a farm's fields, including the STP levels and P-Index. This information could be used to identify the number and area of farm fields with high STP and/or high P-Index at the farm- and regional-level. Because these factors affect the allowable rate of P application, they could be useful in determining the total amount of P that can be land applied in a given region (e.g. a watershed). This could be combined with information about animal unit density (stocking rate) to identify hotspots for P management.

A WFNB provides a valuable complement to a NMP, because it can be used to calculate net farm imports and exports of P. This can be used to calculate a more complete farm-balance, although runoff losses are still excluded. These same metrics can be used to estimate PUE, a measure of how efficiently a farm converts P imports into P exports. This can support benchmarking, although it could be misinterpreted since a farm can conceivably be more than 100% efficient, for example if it minimizes P imports and draws upon existing legacy P stocks in crop production.

Table 2. Comparison of P Management Performance Metrics. This table compares metrics that can be calculated using information available in a 590 NMP, a farm-gate WFNB, and the proposed NMPB.

Indicator	Calculation	Definition	Relevance	NMP	NMB	Enhanced NMPB	Level
NMP Coverage	NMPs farms / all farms	Number or % of farms subject to a NMP	Indicator of proportion of farmland subject to restrictions on P use and erosion rates	Yes&	No	Yes	Region
Total Soil P Inputs	SUM inputs (kg P)	Total P applied to farm soils	Indicator of the amount of P applied to farm soils	Yes*	No	Yes	Field, Farm, Region
Area-Weighted Soil P Inputs	kg P ha-1	Total P applied per unit area	Indicator of the rate of purposeful P application to farm soils	Yes*	No	Yes	Field, Farm, Region
Field (Surface) P Balance	P inputs - crop uptake (kg P)	Surplus or deficit per field	Indicator of the quantity of P added to or removed from a field's soils, excluding runoff	Yes*	No	Yes	Field
Area-Weighted Field Balance	Field balance / field area (kg P ha-1)	Rate of P accumulation/depletion for field^	Indicator of the rate of P accumulation/depletion in a field, excluding runoff	Yes*	No	Yes	Field
High STP Farmland	SUM high STP fields (ha)	Land area with high STP level	Indicator of absolute or relative (%) area of soils with P levels above critical value	Yes	No	Yes	Field, Farm, Region
High P Index Farmland	SUM high P index fields (ha)	Land area with high P index	Indicator of absolute or relative (%) of farm soils at high risk of P runoff	Yes	No	Yes	Field, Farm, Region
Net Farm-Gate Imports	SUM imports (kg P)	Total P imported on-farm	Indicator of the amount of P brought on-farm	No	Yes	Yes	Farm, Region
Net Farm-Gate Exports	SUM exports (kg P)	Total P exported off-farm	Indicator of the amount of P exported off-farm	No	Yes	Yes	Farm, Region
Net Farm-Gate Balance	Imports - exports (kg P)	Imports minus exports, total P surplus or deficit	Indicator of the total amount of P accumulating in or removed from the farm system	No	Yes	Yes	Farm, Region
Area-Weighted Balance	kg P ha-1	Rate of P accumulation/depletion for farm^	Indicator of the magnitude of surplus or deficit given the land area under production.	No	Yes	Yes	Farm, Region
P Use Efficiency	Exports / imports	Proportion of imported P that is exported~	Indicator of efficiency of farm in converting P imports to exports	No	Yes	Yes	Farm, Region
% P Inputs Grown On-Farm	*See text*	Proportion of P inputs produced on-farm	Indicator of farm reliance on P inputs purchased/imported from off-farm	No	No	Yes	Farm, Region
% P Recovery	*See text*	P recovery rate	Efficiency of a farm in recovering and reusing P	No	No	Yes	Farm, Region
Total Runoff	kg P	Total P lost via runoff	Indicator of the quantity of P lost via runoff; a proxy for net pollution contribution	No	No	No	Field, Farm, Region
&	Pasture may be excluded, so using total cultivated area (to give hectares under NMP) may be misleading.						
*	Seeds and compost inputs may be excluded. Pasture may be excluded.						
^	This metric is reported using different denominators in the literature. Some report total productive land area (culivated cropland + pasture); some include only cultivated cropland; others in between.						
~	This is a proxy; a portion of the P exported is derived from soil stocks, rather than from the P applied during a particular time period.						

The NMPB approach would combine the two approaches, making several additional changes described earlier. First, by including pasture deposition and uptake, it would integrate a major component of the farm system frequently excluded from a NMP. By including compost application and seeds, it would integrate potentially important sources of P entering the soil pool. This would improve the P input and soil-surface balance metrics.

The NMPB approach would also enable the calculation of several new metrics. First, by including the P exported as crops as a distinct flow, it would allow one to apportion the estimated crop uptake from the NMP into two fluxes – one for exports and one for on-farm use. This could be used to estimate the percent of livestock P inputs produced on-farm, i.e. the percent of a herd's diet satisfied by homegrown feed and forage. This metric would be an indication of how dependent a farm is on imported sources of P. Farms that import more of their P inputs are more likely to have surplus P (Cela et al., 2015).

Additionally, the NMPB approach would allow for an estimation of the percent of manure P recovered on-farm. This is important because a higher recovery rate means more manure P is being captured and factored into nutrient management planning; i.e. the farm's manure management system is less "leaky." The P recovery rate can be captured as:

- Recovery Rate = [($Manured_{Recovered}$ / $Manure_{Total}$) * 100]
- $Manure_{Recovered}$ = ($Manure_{Pit}$ + $Manure_{Grazing\text{-}Deposition}$)

- $\text{Manure}_{\text{Total}} = [(\text{Feed-Forage} + \text{Livestock}_{\text{Imports}} + \text{Grazing}_{\text{Uptake}}) - (\text{Livestock}_{\text{Exports}} + \text{Animal-Products} + \text{Livestock}_{\text{Mortalities}})]$

Total manure production is calculated as a remainder, which assumes steady-state conditions (i.e. no net assimilation of P as biomass over a year). Alternatively, total manure production could be estimated using a diet-based equation (ASAE, 2005). Ideally, estimates of the amount of manure available in a facility (part of a 590 NMP) could be compared with estimates of the amount spread to help validate the estimated quantity.

5.5.2 Increasing Accountability to Facilitate P Governance

The NMPB approach provides the ability to account for most important P flows entering, exiting, and internal to the farm system at the field-, farm-, and potentially the regional-level. It also provides information needed to calculate performance metrics that can inform farm management, policy development and implementation, watershed science and modeling, regulatory enforcement, and governance more broadly.

At the field- and farm-level, the NMPB would support farm management, providing a more comprehensive accounting of P flows that could enable farmers to more closely match their P application rates with crop and soil needs. By including pasture, farmers may be able to manage pasture STP levels to enhance productivity and absorb surplus manure P. Additionally, by explicitly tracking the P contained in animal feed and forage, farmers would have access to information to more precisely match dietary P intake with an animal's needs (i.e. to implement precision feed management). Evidence from VT suggests that many farmers feed their livestock more P than is needed (Anderson and Magdoff, 2000; Ghebremichael and Watzin, 2011); precision feeding would increase PUE and reduce manure P content, potentially reducing P surpluses at the field- and farm-level. Precision feeding can also help reduce feed expenditures by reducing or eliminating the need for P supplements or P-rich concentrates. The net economic effect will depend on feed prices and other factors.

The data contained in an NMPB could inform policy development and implementation, provided it can be compiled into a central database. First, the performance metrics could be used to identify farms that could benefit from participation in state- and federal-level programs to implement BMPs, allowing for targeted outreach. Second, if NMPBs are extended to all farms currently required to complete a 590 NMP, the level of coverage would allow for more precise analysis of the P "carrying capacity" at the farm- and regional-level. Data on STP levels and the P-Index could be used to determine land eligible for P application at rates greater than, less than, or equal to crop uptake. Cropping information could be used to determine the total assimilation capacity of a field, farm, or region's soils, which could be compared with available manure and fertilizer P to see if there is a surplus or deficit. This information would be critical in designing manure transfer or P trading programs. It could also inform the deployment of technology (digesters, manure dewatering equipment), targeting of farm buyouts, and other strategies to mitigate surplus P.

Because data contained in 590 NMPs is not presently captured in a centralized database or repository, scientists working in Vermont must make general assumptions about critical parameters such as STP levels, P application rates, P recovery rates, etc. This introduces unnecessary uncertainty into modelling efforts, for example the attempt to identify critical source areas for P runoff in the Missisquoi Basin (Winchell et al., 2011) and the Soil and Water Assessment Tool analysis completed to set the TMDL for Lake Champlain (U.S. EPA, 2015), both of which assumed nearly uniform STP levels and P application rates. Similarly, efforts to study regional flows of nutrients are hampered by the lack of data on feed imports and other flows entering and exiting the farm system. A NMPB program could serve as the basis for collecting data of general use to government and the scientific community.

There is some potential for NMPBs to facilitate regulatory enforcement efforts, although this would likely engender political conflict with the farm community. By allowing for estimation of P recovery rates and PUE, farms that are especially “leaky” or have low PUE can be targeted for inspection. Similarly, the broader set of metrics can be used for facilitating NMP review and developing “presumptions,” where farms are assumed to be in compliance based on the metrics in their NMPB. This can facilitate the efficient deployment of limited enforcement resources. At present, only Large Farm Operations are required to submit NMPs and undergo inspections annually, with Medium and Certified Small Farm Operations subject to reduced reporting and inspection requirements.

In terms of general governance, the systematic adoption of NMPBs could help elected officials and agency staff report on progress to the general public and key constituencies such as farmers, farm and environmental advocates, and federal regulators. By tying NMPBs to a central database, state agencies could report on key metrics (e.g. total surplus, area-weighted surplus, PUE, etc.) that would serve as progress indicators; given the long lags between making change at the farm-level and seeing results at the watershed-level, these performance metrics periodically could help improve government and farmer accountability to the public.

5.5.3 Considerations for Implementation and Future Research

At a minimum, a NMPB would correct for several important weaknesses in the 590 NMP framework: incomplete accounting of internal P stocks and flows, especially pasture; incomplete accounting of P flows that enter and exit the farm system; and, limited utility for performance reporting and benchmarking. Because a NMPB is inherently an adaptive management tool, these improvements on the 590 NMP requirements could lead to better performance outcomes at the field- and farm-level.

To maximize the benefit of an NMPB approach and improve accountability for nutrient management, it is necessary to collect, assemble, and/or extract key data to create a central database. Even with the 590 NMPs, there is sufficient data contained in the plans – which are typically submitted in hard copy or maintained in a binder on-farm – to support better regional governance and accountability. A NMPB program that includes centralized data collection would make better value of existing data contained in the 590 NMPs (e.g. STP results, P-Index calculations) while also enhancing that data with new, valuable information on P flows (Table 1). Some data could potentially be collected directly at the point-of-sale, for example feed and fertilizer purchases. Tying these to a farm registration ID or similar system would allow for centralized tracking of nutrient sales, facilitating data collection, enabling auditing of NMPBs, and laying the groundwork for a taxation or deposit-refund scheme. Use of a standardized animal registration system (e.g. the National Uniform Eartagging System) could facilitate tracking of livestock.

Implementing an NMPB program would present significant challenges, some of which point to future research needs. A NMP is a forward-looking planning tool that can (but infrequently does) serve as an adaptive management tool. A WFNB is typically conducted as a post-facto research and management tool, but can also be used as an adaptive management tool (Soberon et al., 2013). Integrating the two as envisioned in a NMPB would require farmers to adopt and embrace adaptive management. For some, it may entail considerable additional data collection and record-keeping, which comes at the very least with an opportunity cost. In Vermont, tools such as Go Crop – a web- and mobile-based program for developing and maintaining a 590 NMP – are being adapted to include WFNB functionality. This could facilitate adoption and, if mandated, compliance with a NMPB program. Lessons can be drawn from New Zealand, which has linked NMPs and WFNBs in the Overseer program (Wheeler et al., 2003).

Some of the additional components of a NMPB present challenges for scientists. Estimating grazing P uptake and deposition is difficult because it varies so much based on management and site conditions. The Overseer program uses a metabolic model to estimate grazing uptake and deposition, which could be adopted. This does not address the issue of uneven manure deposition in pastures, which can create "hotspots" for P (Öborn et al., 2003).

As proposed here, a NMPB would not include runoff loss estimation. This could be added; for example, VT has adapted the Agricultural Policy EXtender (APEX) model, which could serve this purpose. Absent the inclusion of runoff losses, it is difficult to set an appropriate sustainability target. Combining NMPBs with APEX could help test whether complying with the NMP and RAP requirements of the VT Clean Water Act is sufficient to meet the TMDL load reduction targets in a given region.

The data collected as part of a NMPB could be very useful for regional-level planning, analysis, policy implementation, and regulatory enforcement if aggregated in a comprehensive, spatially-explicit manner. This could create privacy concerns (e.g. related to personally-identifiable data) and potential conflict with federal agencies and laws governing information use, such as the USDA (e.g. FSA and NRCS). Data could be aggregated for use by agency staff and researchers, with limited, anonymized, spatially-inexplicit data available for public release. Or, in the spirit of transparency and accountability, data could be made publicly available in its raw or aggregated form. As it stands, very useful information in 590 NMPs (e.g. STP results, nutrient application rates, manure volumes, etc.) is not being used for regional-level purposes, which is a missed opportunity.

Finally, given the evidence of limited farmer adoption of nutrient management planning in both voluntary and mandatory policy contexts in the US, significant effort is undoubtedly needed to create the social and political conditions necessary for farmers to embrace the adaptive management approach espoused in a NMPB. Environmental concerns are only one factor in farm decision-making.

5.6 Conclusion

Nonpoint source nutrient pollution from agricultural systems is a wicked problem, presenting scientific, managerial, economic, and governance challenges that are confounded by the growing integration of farms into complex, global supply chains. For P management, the challenge is reducing water quality impacts and conserving an essential, nonrenewable resource without undermining farm viability. Legacy P presents an additional challenge, introducing a troublesome temporal mismatch between intervention and outcome.

Our case study shows that a key component of VT's strategy to reduce P pollution from agriculture – nutrient management planning – can be effective as a farm-level tool but fails to capture all P flows of management and policy significance. Yet, important information contained within NMPs – STP levels, allowable P application rates, etc. – is not collected or made available in a manner that enables regional-level planning and analysis. The emerging recognition of the need to manage legacy P demands greater accountability for P flows that enter and exit the farm system, information mostly missing from the NMP framework.

A WFNB can complement a NMP, providing additional accountability for P flows that can be scaled to the regional-level. Combining both approaches to create a NMPB program could improve farm- and regional-level accountability for P flows, facilitate research and governance, and provide metrics that help bridge the temporal mismatch between action and outcome. A NMPB would formalize the shift from a planning to an adaptive management paradigm, something already proposed as a best practice by NRCS (USDA-NRCS, 2011). This approach would also provide information needed for establishing a nutrient cap or budget at the farm- and regional-level, as well as a tax or deposit-refund system. While this would be groundbreaking in the US, examples from other countries such as New Zealand could provide lessons for design and implementation. The shift toward a paradigm of full P accountability presents an opportunity to extend responsibility for responsible nutrient use from the farmer upstream to feed and fertilizer suppliers, facilitating efficient regulatory enforcement and incentivizing efficiency. Increasing accountability for P flows in agriculture is a necessary step in the effort to mitigate legacy P accumulation and improve water quality in agricultural regions.

CHAPTER 6: DELIBERATION, ECOLOGICAL ECONOMICS, AND A SUSTAINABLE FOOD SYSTEM TRANSITION: LESSONS FOR VERMONT

Achieving Vermont's water quality standards in Lake Champlain and Lake Memphremagog will require considerable effort to mitigate phosphorus (P) runoff from farms. Yet the challenge is not simply about water quality: the debate about how to achieve the requirements of Vermont's total maximum daily loads (TMDLs) for P is interwoven with concerns about public health, economic development, justice, and more. Ultimately, water quality is part of a broader discussion about what Vermonters want the state to look and feel like, which is sometimes described using the label a "working landscape." Is the future one of bucolic landscapes dotted with red barns and black and white Holsteins grazing on verdant pasture? Or one of thriving, dense communities built around 21st century industries (technology, services, knowledge), with rural areas serving as a pristine place for a weekend getaway? Is Vermont going to drive the transition toward a new, sustainable food system, built around concern for farmers livelihoods, health, nutrition, and the environment? Can all these visions be true? Are they achievable? If not, which do we pursue? Seeking to answer these questions raises fundamental questions about tradeoffs, democracy, and social choice.

Ecological economics provides a framework for tackling the central challenge of mitigating P pollution in agriculture that may also help address the interlinked questions about the future of the Vermont landscape (Daly, 1992; Daly and Farley, 2010). The traditional ecological economics framing can be adapted to address the specifics of

Vermont's agricultural water quality problématique (Figure 1). Vermont must establish the **sustainable scale** for each resource issue of concern, such as has been (partly) done with the P TMDL that establishes a cap on P loading. Where possible, specific interventions can be designed to increase resource **efficiency** helping achieve the sustainable scale target. Finally, Vermont must consider **justice and distribution** in devising additional policy to achieve sustainable scale targets, addressing contemporary resource distribution, historical legacies, and future generations in a rich, multi-criteria context.

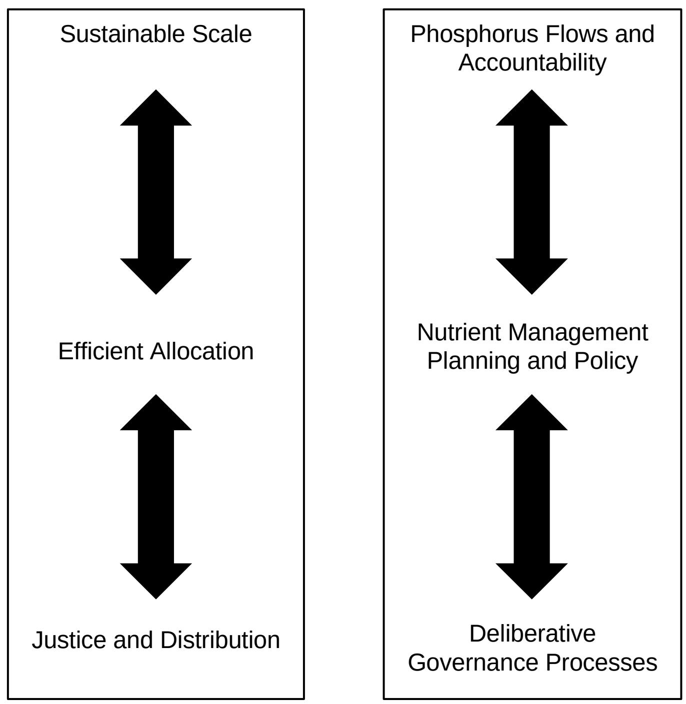

Figure 1. Ecological economics goals and mitigating agricultural P in Vermont.

6.1 Sustainable Scale: Phosphorus Accountability from Field to Watershed

More than half of Vermont's land is subject to a P TMDL, which sets an absolute numerical cap on the total P discharge into specific water bodies (in Lake Champlain, each lake segment has its own cap) (i.e. point A in Figure 2). The cap is then allocated to different sectors (agriculture, wastewater treatment, etc.) (U.S. EPA, 2015). While the TMDL sets the sustainable scale limit for agricultural loading, governance challenges remain: first, if and how to allocate the sectoral load amongst individual farms in a watershed; and second, how to connect, monitor, and manage P flows at the field- and farm-levels (i.e. points B-F in Figure 2) to achieve P loading limits. These two challenges are intimately connected.

The current approach to managing agricultural P in Vermont does not explicitly allocate the sectoral P loading limit to individual farms. Instead, a prescriptive approach that mandates Required Agricultural Practices (RAPs) and Nutrient Management Plans (NMPs), among other interventions, has been pursued to reduce P loading (State of Vermont, 2016). It remains to be seen whether this approach will be sufficient in all lake segments. An alternative performance-based approach would be to establish a specific P cap for each farm, allowing farmers to select strategies that will enable them to meet their target. This has been considered in places such as New Zealand, generating considerable debate about the means of allocating pollution permits (the cap) (Greenhalgh et al., 2015). Different allocation approaches draw on competing theories of justice (e.g. egalitarian, consequentialist, libertarian, etc.). Legacy effects due to historic land use, farm management, and other factors could underlie claims for special consideration. Should Vermont eventually consider this approach, a deliberative process may be needed to develop agreement on a fair procedure for allocating pollution permits.

A = Discharge (Loading) into Water Body (*Google*)
B = Runoff at Edge-of-Field (*Google*)
C = Nutrients Applied to Field (*UVM*)
D = Runoff from Production Areas (*VPR*)
E = Nutrients Brought On-Farm (*Seven Days*)
F = Nutrients Shipped Off-Farm (*Agri-Mark*)

Figure 2. Sustainable scale monitoring points.

Any system for regulating nutrient flows at the sub-basin level – including farm-level permit allocation – needs to acknowledge that the agricultural loading targets for watersheds in the Lake Champlain basin are only loosely connected to field- and farm-level management and regulation, due to both scientific uncertainty and information gaps. Much research is seeking to improve understanding of the connection between specific farm practices and P mobilization, fate, and transport. Not all P that runs off or leaches from a farm field ends up in the lake, so this research could inform efforts to regulate P at the edge-of-field (point B in Figure 2). This raises questions about how field-level nutrient application rates (point C), combined with agronomic practices, influence edge-of-field runoff rates. To-date, much attention in VT has focused on the connection between points C, B, and A in Figure 2.

The research described in this dissertation (esp. Chapters 4 and 5) draws attention to points D, E, and F in Figure 2, suggesting that understanding and managing the entirety of P flows on- and off-farm is important to meet VT's sustainable scale constraints. In Chapter 3, we estimated the quantity of P entering and exiting the farm system at the county- and state-level, quantifying nutrients brought on-farm (point E in Figure 2), nutrients shipped off-farm (point F), and total farm runoff (points B and D). This analysis demonstrated a persistent imbalance, leading to legacy P accumulation in agricultural soils. In Chapter 4, we critically assessed how flows of P in the farm system are tracked and regulated as part of VT's NMP requirements, emphasizing the need for greater accountability for P flows that cross the farm-gate (points E and F). The farm-gate balance is an important driver of legacy accumulation and P loading (Öborn et al., 2003; Soberon et al., 2015). This paper also highlighted the importance of accounting for and managing runoff from production areas, laneways, and other areas upstream of the manure recovery facility (point D).

Our research highlights the value of a systems perspective in seeking to understand and manage P flows in Vermont's agricultural sector. A systems perspective reveals that P can be managed for scale at multiple points: loading into water bodies, which is the ultimate concern (point A in Figure 2); edge-of-field losses (point B); nutrient application rates (point C); production area losses (point D), and nutrient imports (point E). Nutrient imports are the main source of P entering the farm system; research described in Chapter 4 demonstrates how farmers have used feed imports to increase livestock production levels while taking land out of production, leading to intensification that has driven large P surpluses. Ultimately, imports must be constrained by the ability of a farmer to convert these imports into exports (point F) and to legally and responsibly dispose of the remainder (point C). While some loss from production areas (barnyards, etc.) is inevitable, this represents a loss of no agronomic benefit and should be minimized (point D). The allowable rate of nutrient application (point C) will depend on field conditions that govern the rate at which nutrients are lost from the edge-of-field (point B). Soil test P (STP) levels, which reflect legacy P accumulation and are integrated into the VT P-Index, are an important factor in determining the allowable rate of nutrient application. Data on STP levels and the P-Index – already recorded as part of mandatory NMPs – should be centralized to facilitate sustainable P management.

Vermont's current regulatory infrastructure includes many of the elements needed to manage agricultural P flows to achieve sustainable scale. In Chapter 5, we outlined ways in which the NMP framework could be expanded to facilitate collection of information on nutrient imports and exports (points D and E) at the farm- and regional-levels. This would involve tracking sales of feed, fertilizer, and agricultural exports, ideally at the level of the individual farm. We also outlined ways to capture missing nutrient flows and estimate nutrient recoverability rates, which are important in optimizing and accounting for nutrient application rates (point C). Estimating manure recoverability is important to identify the amount of manure P that is never recovered for application to farm soils. In linking nutrient application rates with edge-of-field losses (point B), there are open science questions (e.g. regarding the effects of tile drainage on P fluxes) that must be addressed. Once acceptable edge-of-field loss rates have been established, it may be necessary to test tools like the P-Index to see if they are calibrated to deliver these loss rates under typical conditions. In addressing these gaps to create a multi-point system for monitoring and managing P flows in agriculture, Vermont could improve accountability and facilitate P governance in support of clean water.

6.2 Efficiency: Resource Stewardship, Technology, and Policy

The research described in this dissertation points to several opportunities for increased efficiency of P use in agriculture, while raising some important questions about efficient policy and enforcement. Given that P is an essential, non-renewable resource, efficient stewardship is critical.

The results described in Chapter 4 indicate that, while overall P use efficiency is rising, there remains a surplus at the county-level, with considerable county-to-county variation. This surplus is in spite of the fact that the analysis assumes that all P removed via crop harvest and runoff is replaced via manure and fertilizer P application, an assumption that evidence suggests is false since not all fields receive manure and fertilizer application (USDA-NASS, 2012). There is therefore potential to better match nutrient supply and demand at the field-, farm-, and regional-levels to maximize the beneficial reuse of recoverable manure P.

At the field- and farm-level, nutrient management planning is designed to support farmers in maximizing the beneficial reuse of manure P. However, this implies farmers follow their NMP and that both real and perceived barriers do not inhibit compliance, such as difficulty in getting manure spreading equipment to a given field or infrastructure constraints. Greater accountability for nutrient flows – as proposed in Chapter 5 – could help improve use of and compliance with NMPs, although this will not in itself address issues of farmer motivation. Yield insurance programs could give farmers confidence to bear the risk of yield declines as they reduce nutrient inputs. Dewatering technology and other systems to recover or concentrate nutrients in manure could help make it easier to store and transport P longer distances. Manure hauling could also be incentivized to support intra- and inter-basin transfers to match supply with demand. Research is needed to see if this is best accomplished with a trading scheme or otherwise.

Measures to promote more efficient use of existing manure resources can help reduce demand for fertilizer, but results presented in Chapter 4 suggest that this alone will not be enough to reduce legacy P accumulation in counties such as Franklin and Orleans. Better distribution of manure resources in the landscape may also boost yields, reducing demand for imported feed. Additional efficiencies may be possible through precision feed management, where farmers aim to minimize over-feeding of P and other nutrients. This could be encouraged through informational (such as the use of whole-farm nutrient balances described in Chapter 5), economic (via feed taxes or a deposit-refund system), or regulatory means (such as a cap on permissible P imports). Precision feeding would reduce the amount of P entering VT agriculture and subsequently the amount of manure P requiring land disposal or hauling. It can also save farmers money, becoming a win-win solution.

There is a real opportunity to collect new data and make better use of existing information to target the application of BMPs and other investments in clean water, as described in Chapter 5. Collecting and assembling readily-available STP and P-Index information from NMPs could help identify hotspots for P loss and therefore targets for intervention. This same information could facilitate rollout of manure hauling and other programs (e.g. digester/nutrient recovery facility siting).

6.3 Justice: Democratizing Water Quality Governance in Vermont

In setting a sustainable scale and seeking to maximize efficiency within the current political-economic paradigm, Vermont may be able to make great strides toward reducing P pollution, legacy P accumulation, and improving water quality. Yet this may not be enough, requiring additional policy and action that could entail tradeoffs that point toward different visions for the future of Vermont and its landscape.

Vermonters pride themselves on the state's natural beauty, the quality of the environment, their tradition of independence, and the state's legacy of environmental leadership (Moser et al., 2008). Yet more than anything else, Vermonters value "the state's working landscape and heritage" (ibid.: 4). The idea of a working landscape is a relatively recent one, but tangled up in the concept is a sense that a landscape can be multifunctional, providing economic, social, and environmental benefits (Hall et al., 2004). This clashes with traditional conservationist notions that afford pride of place to landscapes bereft of people.

Reducing P loading to improve water quality and achieve regulatory compliance will require action by farmers, forest owners, developers, local government, and others, all of which could impact both the look and function of the working landscape Vermonters value so deeply. In changing the economic, social, and environmental systems that "co-create" the working landscape to mitigate P pollution, tradeoffs will arise. If the goal were simply to improve water quality subject to a constraint of cost effectiveness, technical and economic analysis may provide sufficient insight to support decision-making. This could be done with a neoclassical or environmental economics approach to policy.

Managing a landscape for a multiplicity of benefits is more complicated, demanding that stakeholders and/or decision-makers weigh and evaluate tradeoffs across multiple criteria, not all of which will fall into their areas of previous experience or substantive expertise (Gregory et al., 2012; Hockley, 2014). This is where a deliberative, ecological economics perspective distinguishes itself. Each party to the decision will likely value different dimensions of the decision differently: this is the challenge of normative evaluation, as described in Chapters 2 and 3.

For instance, a farmer, conscious of market competition, may privilege interventions that impose few costs on farmers or even pay for mitigation measures (a "pay the polluter approach"); yet, they may advocate a "polluter pays" approach for mitigating P runoff from new urban development. A conservationist may argue that no economic activity is worth sacrificing clean water, demanding strict and immediate action. A locavore may be concerned with preserving small family farms, especially organic farms. Many may have no clearly defined preferences about how to mitigate P pollution, and whether to weigh economic, social, or environmental criteria more strongly (Hall et al., 2004). The maintenance of a working landscape – the ostensible shared goal – remains underspecified, making it difficult to chart a path that integrates individual and social preferences into clean water policy.

The current system, whereby citizens voice their preferences – however partial or poorly formed – via elections and interest groups, has not helped to clarify the values and competing normative claims at play in the debate about how to achieve clean water (Dryzek, 2002). Public meetings and comment/review periods, as well as public processes led by various civil society organizations, provide for a modicum of citizen input. The legislature is the direct representative of the public, yet as a part-time body with few staff and resources, there are limits to what it can accomplish. Substantive expertise lies within the executive branch agencies, yet as nominally apolitical bureaucracies, they are not designed as value-articulating institutions. In devising public policy, both knowledge and the understanding of the target populations are socially constructed, demanding public interrogation and deliberation to avoid replicating existing inequalities (Schneider and Ingram, 1993, 1997). In Vermont, there remains a democratic deficit around agricultural water quality policy and the future Vermont landscape. Preferences should be formed in light of extant knowledge, public reasons given, and claims scrutinized, all with the aim of forming some partial consensus or ranking of claims about the future rural landscape. Otherwise, the notion of a working landscape will remain an underspecified, vague concept, one that can be wielded as a rhetorical instrument in service of any number of particular interests.

Deliberative processes and institutions may offer some potential for improving Vermont's ability to devise widely-supported, legitimate policy and programs to mitigate P pollution. We identify three opportunities: citizen representation, perhaps in the form of a jury, that can support agricultural regulatory enforcement; deliberative polling to support revision/development of the TMDL Phase 2 Implementation Plans (Tactical Basin Plans); and, a deliberative forum to clarify and form consensus around the definition of a working landscape in Vermont. Deliberation can breathe legitimacy into a body of policy and practice that remains contested on multiple fronts.

A central source of water quality governance "illegitimacy" in the eyes of some critics is that the Agency of Agriculture, Food, and Markets (AAFM) is responsible for both promoting and regulating agriculture, creating a potential conflict of interest. A recent government accountability report to the Vermont General Assembly states that "the same State entity should not both promote and regulate a program" (Vermont-GAC, 2018: 13). Hence, there have been numerous proposals, including a current Senate Bill (S.220), to transfer authority for regulating water quality on farms from the AAFM to the Agency of Natural Resources (ANR). The authority to regulate Vermont's water quality resides with ANR, which is also the entity responsible for reporting to the U.S. EPA on progress in implementing Vermont's various TMDLs. Yet a Memorandum of Understanding between ANR and AAFM assigns responsibility for agricultural nonpoint source pollution regulation to AAFM. Point source (Concentrated Animal Feeding

Operation, or CAFO) permits are administered by ANR, although no farms in VT currently have a CAFO permit (Vermont, 2018).

De-delegating regulatory authority from AAFM to ANR would improve government accountability but is only a partial solution. The enforcement action process for agricultural water quality violations remains opaque and considerable discretion is left to agency staff, under oversight from the legislature. Conceivably, a citizen panel or jury could be used to evaluate facts and evidence gathered by agency personnel in response to a potential violation; the citizen jury would be able to deliberate to determine whether an enforcement action is necessary and if so, the type and level of action. Integrating the public into the process of regulatory enforcement would further enhance accountability and provide for independent deliberation that weighs not only the technical and legal issues, but also the values and tradeoffs that may factor into the level of enforcement action applied to the case.

Better enforcement of existing rules and regulations is an important part of achieving compliance with the Lake Champlain P TMDL, but in itself may not be enough. To define the larger strategy for achieving compliance, the State elaborated a Phase 1 Draft Implementation Plan that sets out the means by which the state will make progress; public and U.S. EPA comments informed the Plan's development (State of Vermont, 2016). The second phase of implementation builds on the existing tactical basin planning process to develop Phase 2 plans for each sub-watershed subject to the TMDL (ibid.).

While the TMDL and Phase 1 Plan processes featured public meetings, comment periods, and other means of public participation, they did not explicitly include deliberative democratic processes. For example, sectoral load allocations (i.e. the permissible loading for agriculture, urban areas, wastewater treatment plants, etc.) were determined as a technocratic exercise, under- or un-informed by citizen preferences and public deliberation. A deliberative process may have been better-suited to balancing the claims of those who bear the costs and those who benefit from water pollution, helping devise politically legitimate solutions that reflect reasoned public values in addition to expert opinion about technical constraints and feasibility.

In revising and updating the Phase 1 Implementation Plan over its twenty-year implementation timeframe, as well as in developing the more detailed Phase 2 plans, deliberative polling could help integrate deliberatively-formed citizen preferences. Deliberative polling involves selecting a random group from the public to participate in a multi-day process that combines information sharing (e.g., presentations by AAFM, ANR, U.S. EPA, and civil society organizations) with opportunities for discussion and deliberation about the issues at hand (Fishkin, 1991; Fishkin and Luskin, 2005). Polling is used to assess public opinion, with the presumption that the preceding information-sharing and deliberative process will lead to better-informed and more developed preferences than simple opinion polling (ibid). While cost may prohibit deliberative polling for each tactical basin, it could be used to inform the general scoping of the basin planning process or on an experimental basis in a few priority basins. Deliberative polling can help legitimate the tactical basin plans, insofar as deliberatively-expressed preferences end up reflected in the resulting plan (Dryzek, 2010).

Public deliberation can be most effective in targeted settings – a jury, a consensus conference, or for polling on a specific set of issues – yet deliberative processes can also help advance larger-scale visioning processes. In trying to grasp what the future Vermont “working” landscape looks like, deliberative forums could play a role. Ongoing efforts to map, model, and envision the future of the New England landscape, such as the Wildlands & Woodlands initiative and the New England Food Vision, provide some resources and content for organizing a deliberative forum (Donahue et al., 2014; Foster et al., 2017). Ongoing processes organized by VT Farm-to-Plate could also play a role. The central value of a deliberative forum on the future VT landscape is that it provides a platform for learning about the processes that shape the VT landscape and, in deliberation, airing and evaluating different normative claims about what a collectively desirable future looks like. The intention is not to leave with specific policy recommendations, but rather to foster a public dialogue on the values that matter in crafting policy and shaping the State’s future.

Deliberation can play a valuable problem-solving role as Vermont seeks to reduce agricultural water quality pollution. It can also help deepen democracy, sowing seeds for a more public-minded, better-informed, deliberative future (Dryzek, 2005; Schneider and Ingram, 1997). The modest proposals outlined in this dissertation can help sow a few seeds; it remains to be seen if they will germinate and thrive.

6.4 Trade and Regional Governance

There is real potential to make significant progress in improving water quality and reducing agricultural P pollution in Vermont; in fact, the research in this dissertation demonstrates that progress has already occurred, albeit not enough to meet the TMDL requirements. But, this research highlights that Vermont is not an island – it is embedded in complex trade and governance networks that powerfully shape the working landscape and the possibilities for water quality governance.

Vermont's farmers export most of their products out-of-state, representing a tiny fraction of the agricultural commodities – mostly milk – produced in the US each year (Parsons, 2010). They are therefore subject to price fluctuations they have little ability to affect. This is compounded by a regionalized federal Milk Market Order that constrains prices but not supply, leading to a persistent glut of conventional milk. As a high cost producer, Vermont is constantly threatened by lower-cost regions such as California, Idaho, and Wisconsin, benefiting largely from its close proximity to major markets in Boston and New York. Anything done in VT to protect local water quality that raises the cost of production puts Vermont farmers at a further disadvantage.

Simultaneously, Vermont's dairy farms are increasingly dependent on imported feed, making them vulnerable to swings in the price of inputs. Changing feed prices can have real landscape consequences, as when northeastern corn production increased to hedge against the corn price spike of 2008-2009 (Ghebremichael et al., 2009). Price volatility is compounded by the fact that hundreds of Vermont dairy farmers buy from only a handful of feed and fertilizer dealers and sell to a handful of co-ops and processors, typical of highly-concentrated agricultural markets (Carolan, 2012). Concentration stifles competition and can be economically damaging for farmers, but it also presents opportunities. For example, if feed and fertilizer are to be tracked or taxed, extending responsibility for responsible use upstream of the farm, then there are relatively few entities to regulate.

In trying to govern a region's environmental quality in a time of deep economic interdependence driven by trade, Vermont will be forced to make economic and environmental tradeoffs. This is why a deliberative approach to policymaking is imperative. One path forward is to "de-commodify" agriculture, catering to specialty markets for which Vermont is already renowned. The potential is great, albeit constrained by market demand. For conventional dairy, which accounts for the bulk of the State's agricultural output, there is room for efficiency (as described earlier), but the market remains a powerful force. An underexplored avenue for change is for the State to engage with some of the concentrated market powers that sell to, lend to, and buy from its farmers to help shape the political economic context that drives poor water quality and economic outcomes.

Vermont has the opportunity to set important national precedents on nutrient management and accountability in agriculture, developing and promulgating new regulations and programs to address the "wicked problem" of nonpoint source agricultural pollution (Patterson et al., 2013). The precedent that Vermont sets could impact other jurisdictions and potentially help change the discourse around food, farming, water quality, and working landscapes. Given the challenge of governing environmental and economic flows that span multiple scales and levels, it is important that local experimentation and leadership are recognized, shared widely, and emulated, helping form collective norms that can guide change at a higher level. In this sense, Vermont can help drive a systemic transition toward a sustainable food system.

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APPENDICES

The appendices comprise supplementary information included with papers developed as part of this dissertation.

Appendix 1: Supplementary Methods and Materials for Chapter 4

We developed a material flow analysis (MFA) that estimates the quantity of elemental phosphorus (P) flowing in and out of Vermont's farm system. The MFA was completed for all years in which the United States (US) *Census of Agriculture* was released, starting in 1925 and ending in 2012. Our account is comprehensive: we include all livestock flows (manure, meat, etc.) accounting for > 0.01% of VT's animal units; we include all crop flows accounting for > 0.01% of planted area.

In preparing the MFA, we compiled flow data in their native units (bushels, hundredweight (cwt), pounds (lb.), acres, short tons, etc.). These data were then converted into flows of elemental P via conversion factors or equations. In this Supplementary Information (SI) section, we describe the process by which material flow data were compiled (including sources, estimations methods, and assumptions) and the P conversion factors applied. We also discuss our methods for estimating area-weighted surplus P and our uncertainty analysis.

The main P flows in VT's agricultural system are listed in Table A1-1. We describe each flow, explaining why it was included or excluded, the data sources used, any estimation procedures employed, and important assumptions, uncertainties, and/or limitations.

Table A1-1: P Flows in VT's Agricultural System

Inflows	Internal Flows	Outflows
Imported Animal Feed~	Manure: Cattle, Horse, Sheep, Lamb, Goat, Chicken, and Turkey	Agricultural Outputs: Milk~, Meat~, Eggs~, Fruit, Vegetables
Fertilizer	Feed and Forage: Hay and Non-Corn Silage; Corn (Grain and Silage); Other Feed Crops	Runoff (Dissolved and Particulate Losses)
Bedding*	Pasture	Animal Exports*
Seeds*	Spoilage	Manure Exports*
Animal Imports*	Mortalities*	Aeolian Erosion*
Atmospheric Deposition*		
Chemical Weathering*		
* These flows have been excluded from the MFA – see text for details. ~ These flows are typically excluded from soil surface and soil system P balances (Oenema et al. 2003).		

The material flow data were compiled from statistics reported in the Census of Agriculture, referred to here as the *Census* (Census Bureau, 1925-1992; USDA-NASS, 1997-2012), the Annual Agricultural Statistics, referred to as the *Annual Ag. Stats.* (USDA-NASS, 1936-2014), and the Annual Statistical Bulletin – New England, referred to as *NE Ag. Stats.* (USDA-NASS, 2001-2013).[7] Additional resources are cited accordingly.

[7] For each of these sources, we drew upon multiple years' reports to compile flow data. The year listed in the citation is the "nominal" year; for example, the Census of Agriculture for the nominal year 1925 was released in 1927.

Estimating Undisclosed Values

The Census occasionally reports a "(D)," signifying undisclosed, in lieu of an actual value for a given field at the county or, rarely, state level. The disclosure restrictions have been in place since 1974 and are intended to protect the privacy of the small number of farmers engaging in the measured activity (e.g., planting alfalfa hay) (USDA-NASS, 2014b). For counties, we used the reported state value and a proxy variable to allocate the unattributed quantities to the counties with undisclosed values. For example:

- State Value = A
- County Values = B_1:B_{14} with subset (e.g., B_3, B_6, and B_9) containing (D) as the value
- Residual to Allocate = C = A – $\sum B1: B14$, skipping all (D)
- The residual, C, needs to be allocated to the subset for which (D) was recorded as the value, in this example B_3, B_6, and B_9.

To do this, we produce a vector of county weights based on observed values for a variable that can be considered a proxy for the variable with undisclosed values.

- County Weights = D = D_1:D_{14} based on proxy variable.

With this information, the residual is allocated according to the following formula:

- Allocated Value for B_3 = C * [D_3 / $\sum D_{3,\ 6,\ 9}$]

This procedure was followed in every case where a state total was reported but a subset of county values was undisclosed. In most instances, the residual to allocate was less than 5% of the state total.

Inflows

Imported Animal Feed

We estimated animal feed imports based on the law of conservation of mass. The basic principle is that all P flowing out of the animal stock in steady-state conditions (no stock change) must be compensated by inflows of animal feed and forage. Outflows must equal inflows of local feed and forage, pasture, and imported feed. Although animals can draw on their skeletal P stock in times of dietary deficiency (Karn, 2001), farmers and animal nutritionists typically aim to provide a minimum P content in the diet to avoid deficiency (A. Kitsos, UVM Extension, personal communication, 2 June 2017).

To enable estimation of imports, we assume that all animal feed crops grown in Vermont are consumed in Vermont. Vermont is a net feed importer. The main agricultural regions of Vermont are isolated from neighboring states' agricultural regions; the exception is the Montérégie of Quebec in Canada. While the sale of feed crops grown in Vermont is common, in most cases these sales are from one farm in Vermont to another (R. Parsons, UVM Extension, personal communication, 31 January 2017). Some minor crops (e.g., soybeans) may be sold as cash crops to enable purchase of animal inputs, but this practice is modest in scale (ibid.). Any export sales would require a compensating import of P, making our estimate of imported animal feed a likely underestimate. The basic equation for estimating imported animal feed is:

- $\text{Feed}_{\text{Imports}}$ = [Manure + Animal-Products] – [$\text{Feed\_Forage}_{\text{Local}}$ + Pasture]

Animal products are milk, meat, and eggs. Animal feed and forage includes, silage, grain, and other VT-grown feed crops.

Fertilizer

We used three approaches (Table A1-2) to estimate fertilizer consumption in Vermont (VT) at the state and county levels for all Census years from 1925-2012:

1. When available, using annual reported P2O5 consumption data at the state and county level; else,
2. Using annual reported P2O5 consumption data at the state level and downscaling it to the county using proxy variables; else,
3. Estimating state consumption by using reported fertilizer consumption (in short tons) and average nutrient composition, and then allocating to counties using proxy variables.

Primary data sources were:

1. The Annual Ag. Stats., which report tons of phosphate (P2O5) applied at the state level for all Census years from 1945-1982.8 Additionally, five-

[8] The USDA collected state-level data on commercial fertilizer consumption from 1942 to 1985, reporting the values in the Annual Ag. Stats. Starting in 1985, the AAPFCO took responsibility for collecting this information.

year averages are reported for 1935-1939 and 1940-1944. They also report the average nutrient composition of commercial fertilizer used in the US (in terms of N, P2O5, and K2O) for the years 1925-1985 and provides the data needed to calculate state-level composition from 1935-1982.

2. The American Association of Plant Food Control Officers (AAPFCO), which reports county and state level P_2O_5 use for all Census years from 1987-2012 (IPNI, 2012).
3. The Census, which reports data on proxy variables (e.g., fertilizer expenditures or tonnage applied) used to downscale national and state fertilizer consumption data.

For 1925 and 1930, we multiplied the total tonnage of fertilizer used in VT (per the Census) by the average fertilizer nutrient content for VT (for 1935-1939) to estimate P consumption. During this period, VT used disproportionately more P_2O_5 than K_2O and N, compared to the national average (Annual Ag. Stats.), so we used the 1935-1939 VT average (the last reported) rather than national data for 1925 and 1930 (Annual Ag. Stats.).

For 1935, we multiplied the reported average annual tonnage of P_2O_5 consumed in VT for 1935-1939 by 1935's proportion of the period's fertilizer tonnage:

- $P2O5_{1935} = [Avg\text{-}P2O5_{1935\text{-}1939} * 5] * (Tonnage_{1935} / \sum Tonnage_{1935\text{-}1939})$

For 1940, the process was similar with a slight modification:

- $P2O5_{1940} = [(Avg\text{-}P2O5_{1940\text{-}1944} * 5) - (Reported\text{-}P2O5_{1943} + Reported\text{-}P2O5_{1944})] * (Tonnage_{1940} / \sum Tonnage_{1940\text{-}1942})$

Because fertilizer use was increasing so much during this period, our estimates are likely more representative of the temporal trend than if the 5-year average were applied, although the estimates are still subject to error due to inter-annual variability.[9] A general weakness of calculating a balance for Census years only is that short-term spikes are lost; if they are normally distributed this weakness is insignificant.

Table A1-2: Fertilizer Estimates

Year	Approach	Source and Direction of Error
1925, 1930	VT tonnage * average VT fertilizer composition (1935-1939); allocated to counties using proxy variables	Values are reported as “applied on farm” but unclear if non-farm uses removed[10]
1935	Average reported value for 1935-1939; allocated to counties using proxy variables	Values are reported as “applied on farm” but unclear if non-farm uses removed
1940	Average for 1940-1944 adjusted for reported values in 1943 and 1944; adjusted average for 1940-1942 applied; allocated to counties using proxy variables	Values are reported as “applied on farm” but unclear if non-farm uses removed
1945 - 1982	Reported state values; allocated to counties using proxy variables	Values are reported as “applied on farm” but unclear if non-farm uses removed
1987 - 2012	Reported state and county values from AAPFCO	Non-farm uses removed

[9] For example, in 1942 reported fertilizer tonnage spiked; if the same method is used to estimate a value for 1942, the applied P_2O_5 in 1942 is 254% of the applied P_2O_5 in 1940.
[10] Ruddy et al. (2006) report that non-farm phosphorus consumption was less than 2% of total phosphorus consumption in 1997; the impact of non-farm uses is therefore likely negligible.

In selecting reported values to use for 1945-1982, we compared the state-level values reported in the Annual Ag. Stats. and in Alexander and Smith (1990), who derive their values from data collected for Crop Reporting Board (CRB) reports (USDA-CRB, 1982, 1976, 1971). From 1970 onward, the state-level values are consistent (<5% difference). The values diverge significantly prior to 1970. In looking at the CRB reports, P_2O_5 consumption is reported for New England, but not for VT (USDA-CRB, 1971). It appears that Alexander and Smith (1990) estimated P_2O_5 consumption for VT based on VT's share of fertilized acreage, leading to much greater estimated P consumption than reported in the Annual Ag. Stats. Hence, we used values reported in the Annual Ag. Stats., which are derived directly from USDA data. This means that fertilizer inputs in our MFA are significantly lower from 1945-1970 than those documented by Hale et al. (2013), who used the values from Alexander and Smith (1990).

Since fertilizer is reported as P_2O_5 applied, we converted the values into elemental P using the following conversion factor:

- 1 unit P_2O_5 = 0.4364 units of elemental P

For all Census years prior to 1987, it was necessary to downscale state P consumption to the county level. To reflect data availability and our desire to compare estimation methods, we downscaled state data using multiple proxy variables: area of harvested cropland, fertilizer expenditures, and tonnage applied (at national and regional scales). Not all proxies were available over the time series. The approach for allocating state sums was:

- $\text{County-Sum}_A = \text{State-Sum} * (\text{County-Proxy}_a / \sum \text{County} - \text{Proxy}_{a\ldots N})$

In the MFA, we used fertilizer expenditure data for downscaling for every Census year except 1935, 1945, 1950, and 1959, when expenditure data were unavailable. For 1959, we used reported tonnage for downscaling. For all other years, we used harvested cropland acreage as a proxy.

In general, expenditures or tonnage of fertilizer were the preferred proxies as they capture differences in agricultural intensity in input use at the county level. However, they remain subject to the assumption that the type of fertilizer being purchased does not vary from county-to-county.

Bedding Material

Bedding material for cattle, horses, and other large livestock typically consists of wood shavings/sawdust, sand, straw/hay, or composted manure, with wood shavings/sawdust and straw/hay the most common in dairy systems in the US (USDA, 2016). Used bedding is usually mixed with manure and applied to farmland (ibid.).

Each bedding material has a different P content, source and lifecycle; for example, composted manure may be produced on site, whereas sawdust is usually purchased and imported onto the farm. No comprehensive data are available regarding prevailing practice and the amount of bedding consumed in Vermont, although sawdust and straw/hay are widely used (R. Parsons, UVM Extension, personal communication, 31 January 2017).

A study for a major VT watershed reported bedding material consumption of 1.4 kg animal unit^{-1} day^{-1} (Winchell et al., 2011), based on survey data collected in the 1990s in VT by Cassell et al. (1998). However, the composition of this material is unspecified in the study and it is unclear whether it represents a flux into the system or simply a transfer from farm-to-farm.

Cela et al. (2014) surveyed 102 dairy farms in New York, estimating that bedding and other miscellaneous sources of P accounted for less than 2% of total P imports. Most of this flow comprised purchased straw/hay bought from a neighboring farm (Dr. Quirine Ketterings, Cornell University, personal communication, 31 July 2017).

Similarly, in VT most straw/hay is likely produced on-farm (or within county) and thus is not a net flow of P into the farm system. For sawdust, the P content is low: if we assume sawdust is 40% moisture by weight, with a C:P ratio of 1800:1 (Smil, 2000), then the P content is 0.00022%. Using the consumption rate from Winchell et al. (2011), this amounts to an import of 0.14 kg P animal^{-1} year^{-1}, so much less than 1% of total feed intake. Hence, we ignore bedding as a significant flow of P into the farm system, excluding it from the model.

Seeds

Seeds are a source of P input to agricultural soils (Senthilkumar et al., 2012b). The magnitude of this flow is small, likely less than 1% of total inputs to soil (ibid.). It is difficult to estimate the flow of P in seeds for VT because there is no available information on seed application rates. Additionally, hay and pasture (the dominant crops by land area) may receive periodic over-seeding as well as re-seeding after plowing. Calculations for corn in 2012 – VT's most important row crop – suggest that the influx of P is < 0.5 tonnes P y^{-1}. Given these facts, we have excluded seeds from the MFA as *de minimis*.

Animal Imports

The MFA is run as a steady-state model; the inventory figures are taken as a snapshot in time; no change in animal stocks is included, as would be needed for a dynamic model (Peters et al., 2014). Therefore, animal imports and exports are ignored (alternatively, assumed to balance), except as flows of meat.

Our approach is reasonable on an annual basis: while some animal production sectors may potentially have important sub-annual stock changes (e.g., beef feed-lot/fattening operations; poultry operations) these sectors are small in Vermont (USDA-NASS, 2012). Additionally, given the slow rate of change in the animal herd in Vermont (typically less than 1% of animal units), the net import/export flow likely balances. There are no sources reporting inter-state sales of animals (either to other US states or from VT to Canada) distinct from intra-state sales, making it impossible to know how much the stock change is due to sales of live animals versus herd management practices. Since the import/export sales volume is unknown and the net balance is suspected to be nearly zero, it is omitted from the model.

Atmospheric Deposition

Atmospheric deposition of P occurs worldwide, and Tipping et al. (2014) report a geometric mean deposition rate of 0.027 g P m^{-2} y^{-1}, or 0.27 kg ha^{-1} y^{-1}.[11] They note that transport of larger particles is likely due to entrainment and re-deposition of material within a short-range (e.g., entrainment in one agricultural field and deposition in another). Thus, much of the reported flow is likely local and does not represent an input at the county or state scale. The exception is in dusty regions where aerosol entrainment and deposition can be important at sub-continental and continental scales, such as in the Sahara and Arabian Peninsula, and in areas with large fossil fuel combustion facilities (Mahowald et al., 2008). Since neither exception applies to VT, we assume atmospheric deposition is negligible and exclude it from the model. This approach aligns with other large-scale nutrient balance studies; e.g., Ruddy et al. (2006) include atmospheric N deposition in their balance while excluding atmospheric P.

Chemical Weathering

Rates of chemical weathering of phosphorus (release of phosphorus from bedrock to soil) vary widely (Gardner, 1990), with global estimates estimating a net flow from bedrock to soil of 15-20 TG of P y^{-1}, or 0.3-0.4 kg ha^{-1} y^{-1}(Bennett et al., 2001). No published flow estimates are available for VT. Weathering rates and P availability depend

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[11] This nearly matches the rate used by Sattari et al. (2012) in a global study of P flows; their figure derived from calculations made by Liu et al. (2008).

on factors including parent material (e.g., presence and quantity of apatite) and microbiota (including mycorrhizae) (Nezat et al., 2008).

Given the low magnitude of weathering flows compared to anthropogenic inputs, the inability of farmers or policymakers to meaningfully affect the rates, and the spatial and temporal heterogeneity, chemical weathering has been excluded as a flow in the model.

Internal Flows

Manure

Manure production was estimated using state and county-level animal inventory data compiled from the Census. The animal classes included were: cattle (all, broken down by type), chickens, turkeys, horses, goats, sheep and lambs, and swine. These classes account for more than 99% of animal units in each year. Livestock inventory data were typically multiplied by a manure excretion factor to yield manure P estimates.

In this section, we discuss: first, the process of compiling livestock inventory data; second, the P excretion factors used; third, the estimation of manure recoverability; fourth, animal unit calculations.

Livestock Inventory

Cattle dominate Vermont's livestock sector in all years analyzed. Most cattle classes are complete (fully-disclosed) for all years, although the way in which cattle classes are defined and reported varies. Inventory reports for beef cows and "cattle on feed" contain some undisclosed values: we used the county-level cattle inventory as a proxy variable to estimate these values.

Horse inventory data were complete and available for all Census years except 1964, when the horse inventory was unreported at both state and county levels. Horse inventory data are not reported in the Annual Ag. Stats. Because the inventory changes in a consistent direction year-to-year (downward for most of the time series, and then steadily upward), we assigned the median of the 1959 and 1969 values to 1964.

Sheep and lamb inventory data were complete and available for all Census years except 1978, 1982, 1992, and 1997. For these four years, there were several undisclosed values to fill. For 1978 and 1982, the 1974 inventory served as a proxy. For 1992 and 1997, 1987 served as a proxy. These proxy years were the nearest complete cases in time, hence their use.

Goat inventory data were reported for some but not all Census years, likely due to the absence of a significant goat herd in VT for much of the mid-20th century. For years when goats were reported at the state level but some counties were undisclosed, that year's harvested cropland acreage was used as a proxy. No better proxy variable was identified. In most cases, the undisclosed quantity to allocate was less than 5% of the goat population.

Chicken and turkey inventory data are included in the model. Other fowl (e.g., ducks) are excluded because their population is small (<0.01% of animal units) or nonexistent in all Census years.

Turkey inventory data are derived from the Census for 1935 – 1940 and 1974 – 2012. For 1930 and 1945 – 1969, state totals are taken from the Annual Ag. Stats. No data are available for 1925 and no clear trend exists to use for estimation purposes, so we excluded turkeys for that year. The impact on the MFA is small given that turkeys are less than 0.1% of animal units in every Census year. For undisclosed values (including the years where only state level values were reported), harvested cropland acreage was used as a proxy.

For chickens, inventory values were reported in the Census for all years. For undisclosed values, harvested cropland acreage was used as a proxy. Because VT has only a few large chicken producers, in some cases 80% or more of the state's inventory was undisclosed. This was only a problem starting in 1974, when disclosure restrictions were initiated. Typically, only a few counties have undisclosed values, meaning most counties are directly represented in the reporting, reducing the spatial distortion.

The way in which chickens are classified and inventoried in the Census varies considerably over time, with data reported for all chickens, chickens > 3 months of age, chickens > 4 months of age, hens and pullets of laying age (>13 or >20 weeks), and so on. For the purposes of the MFA, an aggregate category called "all chickens" was created, which includes:

- 1925: chickens of all ages
- 1930 – 1940: chickens 3 months and older
- 1940 – 1964: chickens 4 months and older

- 1969 – 1982: chickens 3 months and older
- 1987 – 2012: layers 20 weeks and older

Broilers and other meat-type chickens are reported from 1974 onward; these are included in the analysis.

For all years prior to 1978, the inventory of chickens less than 3 or 4 months of age is unreported, except for 1925. Pullets of varying age classes are reported starting in 1978; however, there are so many undisclosed values (including state totals) that we excluded pullets. This means we likely underestimate chicken manure production.

Phosphorus Excretion Factors

The manure P excretion factors are of critical importance because they are used to estimate one of the largest P flows in our MFA. Contemporary manure P excretion factors vary considerably from source to source (Table A1-3), because the amount of manure P excreted by an animal depends on animal diet, feed stocks, genetics (breeding), age, life stage, and other factors (ASAE, 2005). Most previous MFAs for P have estimated a balance for a single year, typically in recent decades (Antikainen et al., 2005; Chen et al., 2008; Cooper and Carliell-Marquet, 2013; Lander et al., 1998; Liu et al., 2008; MacDonald et al., 2012; Metson et al., 2016; Suh and Yee, 2011; Van Dyne and Gilbertson, 1978). Authors have generally employed fixed "as excreted" conversion factors, estimated based on contemporary data and available in standards such as ASAE D384.2 or other recent literature. The choice of P excretion factor has a major impact on the balance, and hence merits scrutiny.

Table A1-3: Comparison of Manure P Excretion Factors for Multiple Livestock Classes

<table>
<tr><th rowspan="2">Livestock group</th><th colspan="7">kg P animal^{-1} y^{-1}</th></tr>
<tr><th>Goolsby et al. (1999)</th><th>Lander et al. (1998)</th><th>ASAE (2005)</th><th>Chen et al. (2008)</th><th>Withers et al. (2001)</th><th>Sheldrick et al. (2003)</th><th>Van Dyne and Gilbertson (1978)</th></tr>
<tr><td>Cattle on feed</td><td>NA</td><td>14.20</td><td>7.87&</td><td rowspan="6">17.00 (all cattle)</td><td rowspan="6">9.30 (all cattle)</td><td rowspan="6">10.0 (all cattle)</td><td>8.28</td></tr>
<tr><td>Beef cattle</td><td>19.35</td><td>19.77 (beef breeding herds)</td><td>16.06 (confinement beef)</td><td>8.12</td></tr>
<tr><td>Milk cows</td><td>11.68</td><td>17.93</td><td>25.97^ or eq. 22</td><td>11.75</td></tr>
<tr><td>Heifers</td><td>6.57</td><td>6.23</td><td>7.30 (dairy heifer)</td><td>Allowance made in beef and dairy cows</td></tr>
<tr><td>Steers</td><td>17.52</td><td>14.20</td><td>NA</td><td>Allowance made in beef and dairy cows</td></tr>
<tr><td>Slaughter cattle</td><td>5.78 for 170 days (12.41 for 365 days)</td><td>10.21></td><td>NA</td><td>Allowance made in beef and dairy cows</td></tr>
<tr><td rowspan="3">Hogs and pigs</td><td rowspan="3">4.38</td><td>4.44 (breeders)</td><td>1.88 (slaughter) ~</td><td rowspan="3">3.00</td><td rowspan="3">2.76</td><td rowspan="3">4.00</td><td rowspan="3">5.24</td></tr>
<tr><td>2.41 (slaughter)</td><td>4.14 (breeders) "</td></tr>
<tr><td>2.84 (weighted) '</td><td>2.35 (weighted) '</td></tr>
<tr><td>Chickens and hens</td><td>0.22</td><td>0.21</td><td>0.18</td><td rowspan="2">0.20 (all fowl)</td><td rowspan="2">0.27 (all fowl)</td><td rowspan="2">0.19 (all poultry)</td><td>0.23</td></tr>
<tr><td>Pullets and broilers</td><td>0.11</td><td>0.12</td><td>0.12</td><td>0.09</td></tr>
</table>

Livestock group	kg P animal^{-1} y^{-1}						
	Goolsby et al. (1999)	**Lander et al. (1998)**	**ASAE (2005)**	**Chen et al. (2008)**	**Withers et al. (2001)**	**Sheldrick et al. (2003)**	**Van Dyne and Gilbertson (1978)**
Tom turkeys	0.27 for 130 days (0.73 for 365 days)	0.66 (turkeys for slaughter)	0.44				0.47
Hen turkeys	0.15 for 112 days (0.47 for 365 days)		0.26				
Sheep and lambs	1.46	1.46 (from Brosch (2010))	NA	2.00 (sheep and goats)	0.99	2.00	1.04
Horses and ponies	8.03	NA	4.75 (sedentary) 12.05 (intense)	NA	NA	8.00	NA
Goats	NA	1.19 (from Brosch (2010))	NA	2.00 (sheep and goats)	NA	2.00	NA
Also used by:	(Ruddy et al., 2006)	Brosch (2010); Kellogg et al. (2000); MacDonald et al. (2012); Maguire et al. (2007)*			Cooper and Carliell-Marquet (2013)		

& = 3.3 kg P animal^{-1} over 153 days, annualized
^ = calculated by weighting: 12/14 months lactating; 2/14 months dry
> = Average of steers and heifers rate; per Ruddy et al., half of steers and half of heifers are classed as "slaughter cattle"
~ = sum of nursery + grow/finish stages, annualized
" = calculated by weighting: 122/143 days gestating (115 gestating + 7 between weaning and insemination); 21/143 days lactating
' = weighted, assume 21% of inventory are breeders (VT avg. over period in which data reported)
* = Maguire et al. (2007) use a value of 3.9 kg P animal^{-1} day^{-1} for heifers, which is less than the value reported in Lander et al. (1998) and Kellogg et al. (2000), the purported sources for their values

The challenge is that, as noted by the ASAE (2005, p. 1), "the reported typical values may become obsolete with time due to changes in animal genetics, feeding programs, alternative feeding technologies, and available feeds." Because our analysis extends to 1925, excretion values based on recent research, and therefore contemporary genetics and feeding practices, are liable to misrepresent earlier eras. Despite this, previous studies that have estimated P MFAs for long timeframes, such as Hale et al. (2013), MacDonald and Bennett (2009), Sheldrick et al. (2003) and Withers et al. (2001), have used contemporary excretion factors without adjustment.

For livestock cultivated for meat production, the impact of changing diets, feed stocks, and breeding on total P excretion per unit output is likely to be less significant than for animals such as dairy cows (Dr. D. Anderson, Iowa State University, personal communication, 28 March 2017). Conversely, dairy cows are fed to maintain body mass and condition while maximizing milk, with milk production being the largest source of P demand (National Research Council, 2001). Because dairy dominates the VT animal herd, we investigated the likely changes to P excretion due to breeding, diet, and other factors.

First, the dairy herd composition in VT has changed since 1925. While detailed data are unavailable at state and national levels, partial national level data indicate a shift toward Holsteins (Covington, 2013). In 1935, Holsteins accounted for 40% of the US dairy herd. Jerseys were the dominant breed, accounting for 42% of the herd. By 1985, Holsteins had risen to nearly 93% of the dairy herd, remaining above 90% through 2007

(USDA, 2008). Vermont's herd has followed national trends, although it tends to have more representation from minor breeds (Jersey, Ayrshire, Brown Swiss, etc.) than the national average (Dr. Sabrina Greenwood, UVM, personal communication, 7 February 2017). The annual milk production of Holsteins exceeds other breeds, although the fat content of their milk is lower than breeds such as Jerseys. On an animal unit basis, the USDA-NRCS (2009) estimates that Jersey cows excrete more P than Holsteins per unit milk (for cows producing 75 lb. milk d^{-1}, Holsteins excrete 0.12 lb. P d^{-1} and Jerseys 0.15 lb. P d^{-1}). However, Jersey cows are smaller than Holsteins (1.0 animal units per mature Jersey milker, compared to 1.4 for Holsteins). Thus, VT's dairy herd in the 1930s would likely have more animals per animal unit than its milking herd in the 1980s. This has been factored into our animal unit estimation.

Milk production per cow has increased more than fourfold since 1925. In 1925, VT's dairy cows averaged 4,055 lb. of milk cow^{-1} y^{-1} at 4.05% milkfat. In 2012, they averaged 19,316 lb. cow^{-1} y^{-1} at 3.92% milkfat. This rose to 20,964 lb. cow^{-1} $year^{-1}$ in 2016 (USDA-NASS, 2017). This reflects the shift toward Holsteins, improved breeding, changes to feeding practices, and other management improvements. As one indicator, the quantity of grain and concentrates fed to dairy cows in VT rose from 1,770 lb. cow^{-1} y^{-1} in 1950 to 5,810 lb. cow^{-1} y^{-1} in 1997, the last year in which these data were reported (Annual Ag. Stats.).

Ration composition and quantity are major predictors of P excretion rates (Alvarez-Fuentes et al., 2016; ASAE, 2005). Feed rations are typically designed to ensure that minimal nutrient requirements are met, as specified by the National Research Council (NRC) (A. Kitsos, UVM Extension, personal communication, 2 June 2017). As animal science improves and awareness of water quality issues tied to livestock production and P excretion expands, NRC guidelines regarding P composition of rations have changed. For example, in the most recent revision, the absorption coefficient for some feed stuffs was increased, meaning less feed is now required to meet a cow's P intake requirements (National Research Council, 2001). Data from VT indicate that P concentration in manure has declined since the early 1990s, perhaps reflecting more precise feeding practices (Jokela et al., 2010). Yet, it remains common for rations to exceed the nutrient levels specified by the NRC (Cela et al., 2015; Ghebremichael et al., 2008; Maguire et al., 2007; Van Horn et al., 1996).

There are no data available regarding typical rations fed to dairy cattle in VT, either at present or historically. Yet, given the major changes that took place during the timeframe analyzed, it would be unwise to apply a contemporary excretion factor in all years. Hence, we used ASAE D384.2 equation 22 (ASAE, 2005) to adjust the P excretion factor for lactating dairy cows:

- $P_{excreted} = (Milk * 0.773) + 46.015$
- $P_{excreted}$ = g/animal/day
- Milk = kg/animal/day

This equation captures some of the variation in P excretion over time; however, it was derived using data from contemporary, Holstein-dominated herds so it is unlikely to capture the full variation (Dr. D. Anderson, Iowa State University, personal communication, 28 March 2017). No better alternative exists.

In applying the P excretion factors, it is necessary to group animals by type and life stage (Table A1-4). This adds a further complication because many animals have life stages that are less than one year in length; some animals (e.g., broilers, tom turkeys, slaughter pigs, veal) typically have an entire lifespan that is less than a year. Because the Census provides a snapshot of each county's animal inventory at a particular time (in recent Censuses, inventory on December 31st), it does not capture sub-annual dynamics in the livestock population (Kellogg et al., 2000; Peters et al., 2014).

Table A1-4: Proposed Livestock Groups and P Excretion Factors

Livestock category	Livestock group from Census	P excretion factor (kg P animal^{-1} y^{-1})	Source
Cattle on feed	Cattle on feed, inventory (2002- 2012)	7.8725	ASAE (2005)
Beef cattle	Beef cows, inventory (1925 - 2012)	16.0600	ASAE (2005)
Milk cows	Milk cows, inventory (1925 - 2012)	EQ 22	ASAE (2005)
Heifers	Heifers and heifer calves, inventory (1925 - 1935; 1950 - 1964; 1974 - 1997)	6.5700	Goolsby et al. (1999)
	0.25 * Other cattle, inventory (1940-1945;1969; 2002 - 2012)		
Steers	Steers and bulls (incl. calves), inventory (1925 - 1935; 1950 - 1964; 1974 - 1997)	17.5200	Goolsby et al. (1999)
	0.25 * Other cattle, inventory (1940-1945; 1969; 2002 - 2012)		
Slaughter cattle	0.5 * Other cattle, inventory (1940-1945;1969; 2002 - 2012)	12.4100	Goolsby et al. (1999)
Hogs and pigs	Hogs and pigs, inventory (1925 - 2012)	2.8363	Lander et al. (1998)
Chickens and hens	All chickens, inventory (1925 - 2012) - see SI for details	0.2073	Lander et al. (1998)
Broilers	Broilers (1974 - 2012)	0.1164	Lander et al. (1998)

Livestock category	Livestock group from Census	P excretion factor (kg P animal^{-1} y^{-1})	Source
Tom turkeys	0.5 * Turkeys, inventory (1930 - 2012)	0.7300	Goolsby et al. (1999)
Hen turkeys	0.5 * Turkeys, inventory (1930 - 2012)	0.4745	Goolsby et al. (1999)
Sheep and lambs	Sheep and lambs, inventory (1925 - 2012)	1.4600	Goolsby et al. (1999)
Horses and ponies	Horses and mules, inventory (1925 - 2012)	8.0300	Goolsby et al. (1999)
Goats	Goats, inventory (1925 - 2012)	1.1914	Brosch (2010)

Table A1-5: Animal Unit Conversion Factors

Livestock category	AU Conversion Factors
Cattle on feed	1
Beef cattle	1
Milk cows	1.0 - 1.4*
Heifers	0.7
Steers	0.7
Slaughter cattle	0.7
Hogs and pigs	0.17
Chickens and hens	0.004
Pullets and broilers	0.0026
Tom turkeys	0.017
Hen turkeys	0.00757
Sheep and lambs	0.1
Goats	0.07
Horses and ponies	1
*See SI for explanation.	

In VT, most livestock production is dairy or dairy-beef, which operates at a near constant level year-round and is not subject to major seasonal fluctuations. The annual change in the size of the livestock herd is small. Thus, the Census inventory figures were assumed to be representative of the population over the course of the year. This steady-state assumption mirrors that of Senthilkumar et al. (2012) and Withers et al. (2001). Similarly, the US Geological Survey (USGS) estimates of manure excretion use a steady-state assumption for most animal types, with the exception of slaughter cattle and turkeys (Gronberg and Arnold, 2017; Mueller and Gronberg, 2013; Ruddy et al., 2006).

The final challenge is the way in which animals are grouped in the Census. Van Dyne and Gilbertson (1978), Lander et al. (1998), and Ruddy et al. (2006) explicitly match their excretion factors with Census-reported categories. Other sources, for example ASAE (2005), break livestock down into more or different categories (life stages and sizes) than are reported in the Census. Some authors include all cattle and all poultry (or fowl) in their own aggregate grouping; they then apply an excretion factor to each aggregate (Chen et al., 2008; Sheldrick et al., 2003; Withers et al., 2001).

Our approach to grouping animals generally follows Ruddy et al. (2006); Table A1-4 presents our animal groupings, mapping them to Census categories (which change over time), and our proposed P excretion factors.

Animal Units

For the entire study period, VT's livestock herd has been dominated by dairy cattle, with some change in composition. We converted animal inventory counts into animal units (AUs) to understand how livestock composition has changed over time (Table A1-5).

In dairy, the average size and breed has changed substantially since 1925. Holsteins – a large breed – rose from 40% to more than 90% of the dairy stock nationwide between 1935 and 1985, crowding out smaller breeds such as Jerseys. Data from the Annual Ag. Stats. suggest that small breeds accounted for more than 50% of new dairy cattle registration through 1950; we assumed a linear rate of change in the AU conversion factor for dairy cattle from 1.0 AU animal^{-1} in 1954 to 1.4 AU animal^{-1} in 1982.

Feed and Forage

We estimate feed and forage P uptake by multiplying the quantity harvested (yield) by a P conversion factor (Table A1-6). Most P conversion factors were derived from the Crop Nutrient Tool (USDA-NRCS, 2017) or Jokela et al. (2004).

Table A1-6: P Conversion Factors for Crops, Forage, and Pasture

MFA Flow	Class	P Uptake	Units	Source
Corn grain, bushels (1925 - 2012)	Feed	0.151	lb./bu	Crop Nutrient Tool: USDA-NRCS 2017
Wheat, bushels (1925 - 2012)	Feed	0.224	lb./bu	Crop Nutrient Tool
Oats, bushels (1925 - 2012)	Feed	0.109	lb./bu	Crop Nutrient Tool
Barley, bushels (1925 - 2012)	Feed	0.177	lb./bu	Crop Nutrient Tool
Soybeans, bushels (1969 - 2012)	Feed	0.363	lb./bu	Crop Nutrient Tool
Buckwheat, bushels (1925 - 1950)	Feed	0.152	lb./bu	Crop Nutrient Tool
Mixed grains, bushels (1930 - 1974)	Feed	0.170	lb./bu	Own calcs - avg. of oats, wheat, and barley, per Crop Nutrient Tool
Soybean hay, tons (1930 - 1945)	Forage	4.377	lb./T	Crop Nutrient Tool

MFA Flow	Class	P Uptake	Units	Source
Corn silage, tons (1925 - 2012) Corn hogged or grazed, tons (1925 - 1964)	Forage	1.702	lb./T	Jokela et al., 2004
Sorghum hay, wet tons (1925 - 2007)	Forage	1.193	lb./T	Crop Nutrient Tool
Oats harvested unthreshed, tons (1925 - 1950)	Forage	4.255	lb./T	Crop Nutrient Tool
Alfalfa hay, tons (1925 - 2012)	Forage	4.931	lb./T	Jokela et al., 2004
Small grain hay, tons (1925 - 2012)	Forage	4.146	lb./T	Jokela et al., 2004
Other tame hay, tons (1925 - 2012)	Forage	4.582	lb./T	Jokela et al., 2004
Wild hay, tons (1925 - 2012)	Forage	4.582	lb./T	Jokela et al., 2004
Haylage, silage, greenchop, all green tons (1950 - 2012)	Forage	2.226	lb./T	Jokela et al., 2004

Hay and Non-Corn Haylage, Silage, and Greenchop

Hay and non-corn haylage, silage, and greenchop yields were drawn from the Census. In most cases, both acreage and yield (in dry or wet tons) were reported. Prior to 2002, three green tons were assumed to equal one dry ton; from 2002 onward, a conversion factor of 0.4943 was used to convert all grass silage, haylage, and greenchop into dry tons (USDA-NASS, 2004). For all years prior to 1940, non-corn haylage, silage, and greenchop were not reported and, it is assumed, not cultivated.

For 1925, the hay yield for reported hay sub-types were unavailable. To estimate yields, we used the 1930 yield per acre at the state level to estimate 1925 county yields based on reported acreage. So:

- $\text{County-Hay-Sub-Type\_Yield}_{1925} = \text{State-Hay-Sub-Type\_Yield}\cdot\text{Acre}^{-1}{}_{1930} * \text{County-Hay-Sub-Type\_Acres}_{1925}$

Then, we summed the estimated sub-type yields for the counties and subtracted the sum from the reported state summary yield. This was used to estimate a residual for the state and each county.

- Residual to Allocate = $\text{All-Hay\_Yield}_{\text{Reported}} - \sum \text{Hay-Sub-Type\_Yield}_{\text{Estimated}}$

This residual was then allocated to each geographic area based on the relative weight of each hay sub-type (in acreage) multiplied by the residual to allocate.

- $\text{Final-Hay-Sub-Type\_Yield}_{1925} = \text{Hay-Sub-Type\_Yield}_{1925} + [\text{Residual} * (\text{Sub-Type\_Acres}/\text{All-Hay\_Acres})]$

This means that the estimated yield at the county level (across all sub-types) matches the reported yield, but that the county yields for an individual sub-type do not sum to the estimated state value.

For 1969 and 1974, total hay (including haylage, silage, etc.) acreage and yields were reported for the state and each county. However, the acreage and yield for hay sub-types was only reported for farms with annual sales greater than $2,500 (which represents >90% of farm acreage and yields). To estimate the full value for each sub-type's acreage and yieldwe converted silage tonnage (reported as wet tons) into dry tons. Then, the residual acreage and tonnage (state total minus sum of sub-types) was allocated to each sub-type. This was then allocated to each county based on each county's proportion of that sub-type's value.

For hay sub-types with undisclosed values (e.g., alfalfa hay), total hay acreage served as a proxy variable for weighting.

For the entire timeframe sorghum was a minor crop, accounting for less than 1% of acreage. Where values were reported, an attempt was made to include the acreage and yield in the MFA. For 1940, the yield was not reported and therefore the 1945 state yield acre^{-1} was used. For 1969 and 1974, the state yield acre^{-1} from 1978 was used. For 2007 and 2012, state yields were unreported and acreage was insignificant (<50 acres) and so these years were excluded. Undisclosed county-level values were estimated using harvested cropland as a proxy.

Yield and acreage of hay sub-types was reported across most of the period analyzed; sub-types of haylage, silage, and greenchop were not. Hence, haylage, silage, and greenchop is grouped into a single category with its own P conversion factor.

Corn Grain and Silage

For 1935, 1945, and 1969, corn silage acreage but not yields are reported. The average yield for the state from the Annual Ag. Stats. (1935 = 10.5 tons acre^{-1}, 1945 = 9 tons acre^{-1}, 1969 = 15 tons acre^{-1}) was applied uniformly to the county-level acreage.

For 1974, corn silage acreage and yield were reported for farms with sales >$2,500. Total corn acreage and corn grain acreage were reported for all farms. Corn silage acreage was thus calculated as all corn acreage minus corn grain acreage. We applied county-level yields per acre for farms with sales >$2,500 to the all farm silage acreage values to estimate all farm yields.

For all years reporting corn acreage hogged or grazed, the county-level silage yield was applied to the acreage (i.e., treating hogged/grazed corn as equivalent to silage). No other procedure for estimating yield was identified in the literature.

Other Feed Crops

For other feed crops (e.g., barley, oats, wheat, soybeans, etc.), acreage and yield were reported in the Census. Undisclosed values were estimated using harvested cropland as a proxy. Some reported crops were omitted from the analysis, including rye, emmer and spelt, sugar beets, popcorn, flax, triticale, and canola. These crops were almost always grown on less than 100 acres, amounting to much less than 0.01% of total acreage.

In 1969 and 1974, acres of mixed grains harvested were reported, but yield (in bushels) was unreported. The nearest complete case was 1959. Acreage and yield is reported from 1930 to 1959 (continuously); there is no evident time trend in yields (e.g., as would be expected due to increased use of fertilizers). Hence, the state-level yield per acre was estimated for each year (1930-1959) and then averaged, giving 28.1 bushels per acre as an average yield. This was applied to the acreage in 1969 and 1974.

For the years prior to 1950, most of the soy grown in VT was harvested as hay or green chop rather than for beans, which was typical for the US during this timeframe (Blount et al., 2002). However, no yield data are available in either the Census or the Annual Ag. Stats. for that period. A yield of 1.5 dry tons of soybean hay per acre (~10% moisture) was used for the non-bean harvest for years prior to 1950. This yield is within the range reported for the 1980s and earlier (Hintz et al., 1992); it is on the low end for studies conducted recently using improved varieties (Blount et al., 2002). Soy does not reappear in the Census until 1969, when it is reported as a bean crop. From 1969-2012, all soy is reported as harvested for beans.

For 2002 and 2007, the state-level barley acreage and yields were undisclosed, which suggests that few people planted barley and that the acreage was likely quite small. The two closest values to estimate from are 1997 and 2012. Therefore, the values estimated for 2002 and 2007 were the median values (over three time steps) between 1997 and 2012. This likely gives a close approximation of the values.

For the years prior to 1954, oats were reported as harvested for grain and as harvested for feeding unthreshed. Yield in bushels was reported for oats harvested for grain; only acreage was reported for oats harvested for feeding unthreshed. From 1954 onward, oats harvested for feeding unthreshed were included within the small grain hay category for all states. The yield for small grain hay across all Census years is typically between 1.5 and 2 dry tons acre^{-1}; this yield aligns with those reported in multiple academic and extension publications (Aydın et al., 2010; Caballero et al., 1995; George et al., 1982; Larson et al., 1952; Lithourgidis et al., 2006; Long et al., 2005). Therefore, a yield of 1.75 dry tons acre^{-1} was used to estimate yield for oats harvested and fed unthreshed for all years prior to 1954.

Pasture

Pasture was the main agricultural land cover in Vermont prior to the 1970s; in more recent decades it has remained an important land cover and source of forage. Over this period, grazing practices have evolved, partly in response to economic pressure on farmers to use their land more efficiently and increase productivity (Winsten et al., 2000). Unfortunately, pasture yields are unreported in the Census.

We estimated pasture yields using the method reported in Conrad et al. (2016). The yield of "other tame hay" at the county-level was adjusted for curing losses and the cut portion to estimate production in units of dry matter/acre. This biomass production was then grazed at a defined harvest efficiency. The procedure is:

1. Calculate "other tame hay" dry tons per acre yield at county and state level
2. Adjust for curing losses (divide by 0.8)
3. Adjust for cut portion (divide by 0.85)
4. Adjust for percent of standing forage consumed (harvest efficiency – multiply by 0.3 to 0.4 depending on year, see below).
5. Calculate short-tons of dry matter by multiplying the earlier result (yield) by total acreage of pasture
6. Convert from short-tons/acre to metric units

An important variable in this approach is harvest efficiency, i.e. the amount of standing forage consumed during grazing. Conrad et al. (2016) apply a 30% efficiency to permanent pasture and 35% to cropland used as pasture for continuous grazing, resulting in mean dry matter yields for the northeastern US of approximately 2,000 kg ha^{-1}; Tichenor et al. (2016) apply a harvest efficiency of 50% and obtain mean dry matter yields of nearly 3,000 kg ha^{-1} for the same region. The higher harvest efficiency used by Tichenor et al. (2016) is intended to reflect yields from management-intensive grazing.

Harvest efficiency can be interpreted as a proxy for grazing practices. In management-intensive grazing, pastures are grazed intensively for short periods of time (often less than a day), and then allowed to regrow undisturbed before the next round of grazing (Murphy, 1987). The amount of standing biomass harvested during a grazing period may exceed 65% (ibid.). Continuous grazing systems will be grazed less-intensively, but without a recovery period. This can impact the amount and quality of forage yield, resulting in a lower effective harvest efficiency (Murphy, 1987). The dry matter yield $acre^{-1}$ may decline by half, although protein yields may be equal or higher (Morrison, 1949).

Survey results suggest that more than 10% of dairies in the northeast practice management-intensive grazing (Winsten et al., 2010) and that more than 30% of VT farmers practice at least moderately intensive grazing (Winsten et al., 2000). Similarly, recent farm-scale studies in the northeast report or estimate dry matter yields from pasture that range from 3.5 – 6.9 tonnes ha^{-1} (Ghebremichael et al., 2008; Rotz et al., 2002), which suggest significantly greater harvest efficiency than proposed by Conrad et al.

(2016). We capture the shift in recent decades toward better grazing practices on more productive land in two ways:

1. We couple our estimated pasture production to hay production, so yield improvements in hay automatically result in pasture yield improvements; and
2. We adjust harvest efficiency over time.

Yields for "other tame hay" have nearly doubled since 1925. One possible reason for the improved hay yield is the buildup of P in the soil due to fertilizer and manure application. Hay fields are more likely to be fertilized than pasture (USDA-NASS, 2012), and hay is typically rotated with corn in VT, with corn almost always receiving some fertilizer (R. Parsons, UVM Extension, personal communication, 31 January 2017). As P builds up in soils, it will (to a point) increase yields. While hay fields are more likely to receive inputs of fertilizer and recovered manure, pasture receives manure inputs directly during grazing (Kellogg et al., 2000).

Another reason is that much pasture land has been abandoned since 1925, particularly land classified in the Census as "woodland, pastured." Rational economic behavior would be to abandon the least productive land (e.g., fields at higher elevations, on hillslopes, etc.) or the most difficult to harvest (e.g., rocky fields). As pasture acreage has declined, it is likely that the remaining acreage is more productive and actively managed, e.g., under management intensive grazing (Winsten et al., 2010). In a recent national survey, Sanderson et al. (2012: 30) note that "pastureland management is relatively intensive and technology based, commonly with inputs of seeds, fertilizers, and pesticides."

Additionally, we adjust harvest efficiency over time to account, in part, for the shift from continuous toward intensive grazing. We apply a harvest efficiency of 30% from 1925-1969, 35% from 1969-1987, and 40% from 1987-2012. There are no historical data with which to corroborate these values, although estimated average dry matter yields in recent decades fall within the range reported in the literature, including data from dozens of dairies in neighboring New York State (Dr. Quirine Ketterings, Cornell University, personal communication, 31 July 2017).

For pasture, we derived our P conversion factor from data reported by the Dairy One Forage Lab (2017). The Dairy One Forage Lab analyzes thousands of feed and forage samples from across the US (primarily the northeast); they report summary values for four types of pasture: mixed mainly grass, mixed mainly legume, legume, and grass (Dairy One Forage Lab, 2017). The range of values for P uptake ton^{-1} of dry matter is from 6.08 lb. ton^{-1} to 7.52 lb. ton^{-1}, with the higher rates corresponding to a greater proportion of legumes. We assume that 20% of pasture area is leguminous (Goslee, 2014; Tichenor et al., 2016). This is lower than the 35% figure reported by Soberon et al. (2015) but is considered regionally representative. Hence, we calculated a weighted P conversion factor (80% mixed mainly grass hay; 20% mixed mainly legume hay) of 6.66 lb. ton^{-1} of dry matter. This equates to pasture P content of 0.33% on a dry matter basis.

Spoilage

Most harvested row crops, feed, and forage grown in Vermont is stored on-farm and fed to animals. All food grown locally offsets food that must be imported. If any of the reported harvest spoils, it must be replaced by purchased food. It is common for some spoilage to occur, although most literature focuses on the degradation in feed value (nutritional content) due to material transformation, rather than physical losses that would impact P content (Ruppel et al., 1995; Wilkinson, 1981). MacDonald et al. (2012) applied a 4% spoilage factor to all animal feed stocks in their study of national (US) phosphorus flows. Antikainen et al. (2005) apply a 5% spoilage factor to hay and 10% to other animal feed stocks. Hale et al. (2013) apply a 10% spoilage factor. In the absence of local data on spoilage rate, we applied a 7.5% spoilage factor to all feed and forage (excluding pasture).

Mortalities

Some animals are slaughtered for meat. These mortalities are accounted for in the slaughter (meat) flow. Otherwise, animals that die are assumed to be composted on-farm, representing no net flow of P off-farm.

Outflows

Agricultural Outputs

The main agricultural outputs were milk, meat, eggs, fruit, and vegetables. We multiplied yields by a P conversion factors to generate P flows (Table A1-7).

Table A1-7: P Conversion Factors for Agricultural Outputs

MFA flow	Class	P uptake	Units	Source
Milk (1925 - 2012)	Animal Product	0.422	g/lb.	National Nutrient Database for Standard Reference 28
Beef (1925 - 2012)	Animal Product	0.71	% of liveweight	Antikainen et al. (2005)
Pork (1925 - 2012)	Animal Product	0.55	% of liveweight	Antikainen et al. (2005)
Lamb (1925 - 2012)	Animal Product	0.56	% of liveweight	Cooper et al. (2013)
Chicken (1925 - 2012)	Animal Product	0.67	% of liveweight	Antikainen et al. (2005)
Turkey (1930 - 2012)	Animal Product	0.67	% of liveweight	Assumed identical to chicken
Eggs (1925 – 2012)	Animal Product	0.111	g/egg	National Nutrient Database for Standard Reference 28
Apples, 1000 pounds (1925 - 2012)	Fruit	0.094	lb./1000 lb.	Crop Nutrient Tool
Sweet corn, tons	Vegetables	1.488	lb./T	See text
Potatoes, hundredweight (cwt) (1925 - 2012)	Vegetables	0.056	lb./cwt	Crop Nutrient Tool
Dry edible beans, bushels (1925 - 2012)	Vegetables	0.478	lb./bu	Crop Nutrient Tool
Other vegetables, acres (1925 - 2012)	Vegetables	11.015	lb./acre	See text

Milk

Milk production is reported at the state level for all years, either in the Census or Annual Ag. Stats. For most years, milk production is not reported at the county level; therefore, county-level milk production was estimated using the milk cow inventory as a proxy.

Some milk is fed to calves or consumed on-farm by farmers. Milk fed to calves is an internal flow (within-farm) and therefore is not factored into the model. The magnitude of this internal flow has generally declined (from ~3% of milk production to less than 1%) as the animal population has declined and animal husbandry practices have changed: for example, it was 32 million lb. in 1940, 18 million lb. in 1969, and 13 million lb. in 2012 (Annual Ag. Stats.). Milk consumed by people (farmers, etc.) on-farm is included in the flow of milk "off-farm," since it is leaving the farm system and entering the human consumption (wastewater) system.

Milk fat content varied considerably over the timeframe analyzed, from a low of 3.67% in 1982 to a high of 4.10% from 1940-1950. Most P is contained in the proteins in milk (Lenstrup, 1926), so variation in milk fat content itself cannot itself be used as a proxy for milk phosphorus content (Alvarez-Fuentes et al., 2016; Klop et al., 2014). Hence, the National Research Council (NRC) no longer advises normalizing milk on a fat-content basis when estimating phosphorus balance and ration requirements (National Research Council, 2001). We did not normalize milk volumes based on fat-content.

Meat

The Annual Ag. Stats. is the primary source for data on meat production, with the NE Ag. Stats. serving as a resource for 2002-2012. Production is reported in pounds live weight, which includes bones and other non-meat products. This is important because bones are the largest repository for phosphorus (Field et al., 1974; Karn, 2001). Production is reported at the state-level and allocated to the counties based on the corresponding animal inventory, which is reasonable since there is little dedicated beef production (including fattening operations) and most chicken production is for eggs. No data are available to account for county-level discrepancies in herd management and culling rate, which impact meat production particularly in dairy-beef systems (Tichenor et al., 2016).

Meat flows were estimated for cattle (beef and veal), swine (pork), poultry (chicken and turkey), and sheep and lambs (lamb, mutton). Goats, ducks, and various other livestock were not included due to lack of data and their small population (much less than 0.1% of animal units). Total production figures were used: it is irrelevant if the animal was slaughtered on farm, at a slaughterhouse in Vermont, or at a slaughterhouse elsewhere, if they were raised in VT.

For the years 1925 and 1930, meat production data are unavailable for all meat types. Linear regression models were developed to predict slaughter weight based on animal inventory, using available data for model estimation. For turkeys, production data were unavailable for 1974 and from 1982-2012; therefore, the same regression model was used to estimate production for these years.

Eggs

Egg production is reported at the state level in the Annual Ag. Stats. and is allocated to each county based on the chicken inventory chickens. Losses are assumed to be composted on-farm and so they do not represent a flow out of the farm system.

Fruit

Apples are the main fruit crop in Vermont. Prior to 1969, acreage was not reported for different classes of fruit trees in the Census; in lieu, the number of trees was reported. Apples account for more than 90% of all orchard acreage and/or fruit trees in Vermont for all years analyzed, except for 2012, where they account for 84% of acreage. Orchard acreage includes tree fruit (apples, peaches, plums, cherries) as well as grapes. The decline in apples' standing in 2012 is due to an increase in grape production.

Berries are also cultivated, although total acreage amounts to a few hundred acres or less for all Census years analyzed. Because the P content of berry crops is low, the total harvest represents an insignificant flow of P; in 2012, the year with the most acreage devoted to berry production, the berry harvest contained less than 120 kg of elemental P. Therefore, berry crops are excluded from the model.

Apple production is reported in the Census for all years prior to 1997 (except 1969). Production data for 1997-2012 are taken from the NE Ag. Stats. The 1969 value was taken from the Annual Ag. Stats. Given the limited availability of yield data for other fruit crops, the approach taken to estimate a fruit flow was to take the apple yield (lb./acre or lb./tree, depending on the year) and extrapolate that to all reported acres or trees to generate a total "apple" or orchard fruit production for each county for all years.

Vegetables

Data on acreage and yield of potatoes are reported for all Census years and are thus included in the MFA as a distinct flow.

Dry edible beans are reported most years and are also included as an individual flow. For 1925, dry beans acreage but not yield data are available. Therefore, the average state yield for 1930 was applied. For 1992-2012, very small quantities of dry beans were grown in VT (<50 acres each year, statewide). Hence, these are ignored as *de minimis*.

For all other vegetables, total acreage is reported from 1969 onward, although it is not broken down by crop type until 1997. Prior to 1969, only acreage of the main crops is reported, rather than total vegetable acreage. For most years between 1969 and 1997, the acreage of sweet corn is reported but nothing else. In most years, sweet corn is the dominant crop, representing more than 50% of vegetable acreage. Post-1992, sweet corn's dominance declines, representing only 28% of planted acreage in 2012.

To estimate a P flow from other vegetables, the acreage was split into two pools: sweet corn and "other vegetables," with the assumption that the state distribution of acreage was mirrored at the county level. For 1969 and 1974, sweet corn acreage is unreported; therefore, the adjusted median of the 1964 and 1978 values is used. Sweet corn yield is unreported in the Census; the NE Ag. Stats reports Vermont sweet corn yields of 65 cwt $acre^{-1}$ in 1992, 75 cwt $acre^{-1}$ in 1997, 50 cwt $acre^{-1}$ in 2002, 65 cwt $acre^{-1}$ in 2007, and 50 cwt $acre^{-1}$ in 2012. For years prior to 1992, a yield of 50 cwt $acre^{-1}$ was applied. We assumed that a 70 lb. bushel yields 56 lb. of shelled corn (80% of harvested ear weight = shelled corn) (Lander et al., 1998). At 0.93 g of P kg^{-1} of shelled corn (USDA-ARS, 2017), a ton of harvested sweet corn is equivalent to 1.49 lb. P ton^{-1}. This assumes that the stover is not harvested but rather composted *in situ*, so we may underestimate actual P removal.

For “other vegetables” (all vegetables excluding potatoes, sweet corn, and dry edible beans), we created a composite P conversion factor, capturing the main vegetable crops in Vermont during the timeframe analyzed: beans (snap, pole), cabbage, tomatoes, squash, pumpkins, lettuce and peppers. Yields are unreported in the Census; the NE Ag. Stats reports yields in VT for 2012. The 2012 yields were compared to “typical” yield ranges in Laboski and Peters (2006); based on this, we used the lower-range yield values from Laboski and Peters (2006) in the composite indicator. The composite conversion factor assumes an equal mix of the seven crops (in terms of area); given the incomplete information on crop acreage for many Census years, this assumption is necessary.

Animal Exports

These are not included in the model – see “animal imports.”

Manure Exports

Manure export is one possible means of disposing of surplus phosphorus, whether at the farm or regional level (Metson et al., 2016; Ribaudo et al., 2003). It is known that some farms in VT export manure off-farm, although typically the manure is transported to a nearby farm that has a nutrient deficit (Heather Darby, UVM Extension, personal communication, 6 September 2017). Because manure is so heavy and has such a low P concentration (Jokela et al., 2010), especially liquid manure, it is uneconomical to transport it long distances (Kaplan et al., 2004; Keplinger and Hauck, 2006). Hence, we assume that any off-farm exports of manure are local (within-county) and therefore can be excluded from the MFA.

Aeolian Erosion

This is not included in the model – see "atmospheric deposition."

Runoff: Dissolved and Particulate Losses

Runoff losses are important because they drive eutrophication and are a major flow of P off-farm. If farmers seek to maintain soil test phosphorus at optimal agronomic levels, they must compensate for runoff losses (Sattari et al., 2012).

The literature on runoff losses of P from agricultural fields is vast; reported loss rates from pasture and hay are typically several times lower than from row crops such as corn, with substantial variation at the field-scale due to topography, soil type, and other edaphic factors. Estimating runoff losses accurately is difficult without a detailed, spatially-explicit field-scale model (Vadas et al., 2015); this type of analysis is outside the scope of our study, and perhaps impossible given the absence of detailed spatial data on land cover for much of the timeframe analyzed.

In a global study, Sattari et al. (2012) assumed losses were equivalent to 10% of fertilizer and manure inputs. In a study of dairies in the northeastern US, Rotz et al. (2002) propose a 5% loss rate. We used a rate of 7.5% to approximate runoff losses, the median value.

To assess model sensitivity to this important parameter, we conducted a sensitivity analysis using a 5%, 7.5%, and 10% loss factor (see Table A1-8 and Figure

A1-1). In 2012, the estimated surplus ranged from 1345.6-1640.8 tonnes of P, depending on the loss factor applied. The surplus ranges from 4-11% from the reported value, depending on year and loss factor applied. This is insufficient to change the interpretation of results or the general conclusions.

Table A1-8. Sensitivity analysis results.

Year	Surplus (7.5%)	Surplus (5%)	Surplus (10%)
1925	1752.13	1958.30	1545.96
1930	1000.91	1199.01	802.81
1935	1460.23	1648.38	1272.07
1940	1857.38	2082.44	1632.33
1945	3799.49	4072.12	3526.86
1950	4438.65	4700.10	4177.21
1954	3671.66	3924.80	3418.51
1959	4212.33	4459.93	3964.74
1964	4290.61	4521.06	4060.16
1969	3332.83	3539.93	3125.72
1974	3355.70	3564.11	3147.30
1978	2583.14	2766.01	2400.27
1982	3200.18	3400.41	2999.94
1987	2159.51	2328.34	1990.69
1992	2496.39	2668.57	2324.21
1997	2180.09	2342.84	2017.34
2002	1914.55	2070.37	1758.73
2007	1669.25	1821.94	1516.57
2012	1493.09	1640.63	1345.55

We also used results from the Soil and Water Assessment Tool (SWAT) model prepared for the Lake Champlain Basin TMDL. We used the estimated loss rates (kg of total P ha^{-1}) for each land class and Hydrological Response Unit to calculate an average loss rate for each land class. These average loss rates were then applied to all land fitting within that land class at each time step. It is notable that the estimated runoff losses using this approach fall within the values estimated using the 5 and 7.5% loss factor from 1969 onward. Prior to 1969, the estimated runoff losses climb rapidly, likely reflecting the significantly greater land area under cultivation relative to the amount of manure and fertilizer inputs. This suggests that out estimate of total P losses due to runoff may be low for the period prior to 1960.

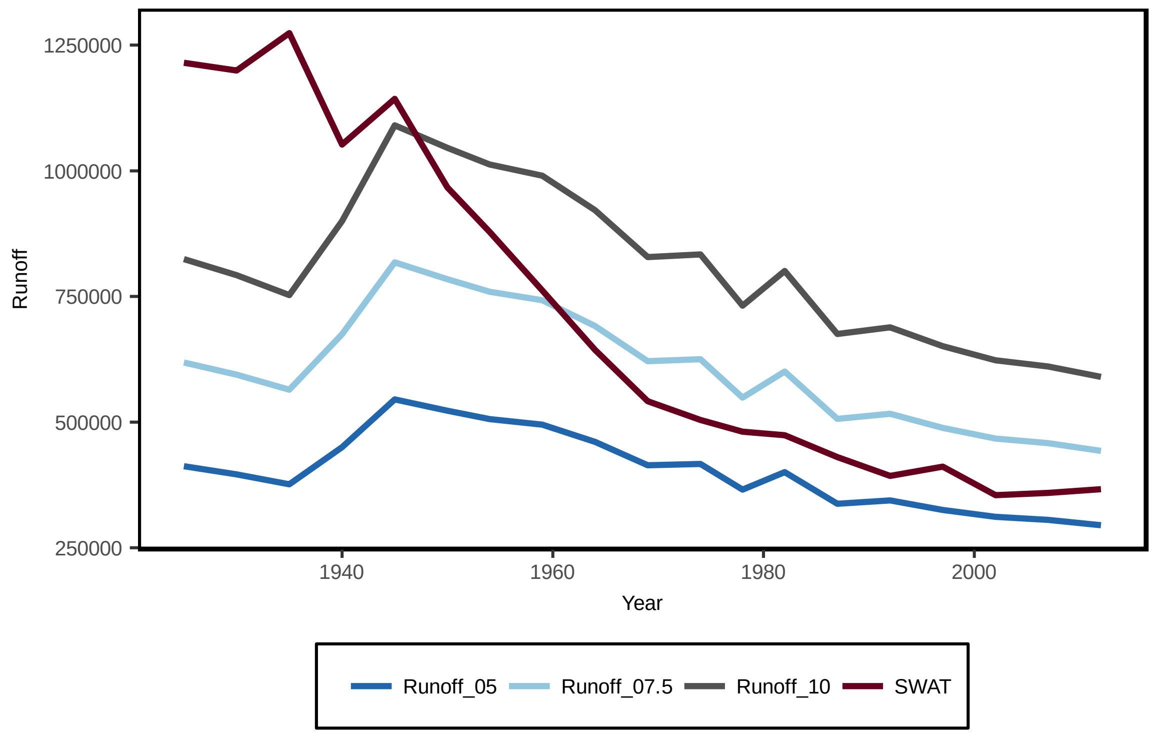

Figure A1-1. Runoff losses using different estimation approaches.

In addition to calculating a balance that factors in runoff losses, we also estimate total accumulation of legacy P in VT's agricultural soils. To do this, we follow

MacDonald and Bennett (2009) in using a linear interpolation to fill in years between MFA balance estimates.

Estimating Area-Weighted Surplus

The balance calculations assume that P is applied over all agricultural land at rates equivalent to actual P removal, with any remaining P considered "surplus." Surplus P is reported in absolute and relative (per hectare) terms to give an indication of how much excess is likely accumulating in soils in a geographic region. Actual P application rates, P surpluses, and accumulation in soils is in fact spatially heterogeneous for multiple reasons:

1. Fields vary in their nutrient needs; farmers typically manage nutrients based, in part, on soil test results. Soils that are deficient in P may require surplus applications to raise levels to the agronomic optimum; conversely, soils with excessive P may need a draw-down period to bring levels in line with agronomic recommendations (Sharpley et al., 1996).
2. Some fields are inaccessible, partially or fully, to equipment for spreading manure and fertilizer. For example, Kellogg et al. (2000) assume that 50% of permanent pasture in the US is inaccessible to mechanized equipment.

3. Actual uptake varies based on weather, disease, pests, and other factors; farmers likely apply nutrients at a rate closer to the maximum uptake rate, plus some safety factor (Sheriff, 2005).
4. P deposition of manure on pasture is spatially variable, in part because animals congregate in areas with water, shade, and other favorable properties (West et al., 1989).
5. Not all manure P is recoverable (Kellogg et al., 2000). In general, confinement operations have a higher recoverable fraction for P than grazing-based operations. However, much "unrecoverable" P in grazing-based systems is likely applied in situ, so it is not truly lost from the system.
6. Farmers may have insufficient land to assimilate the nutrients generated by their livestock. In this case, they must rely on the willingness of neighboring farms to take their excess nutrients or else apply nutrients at rates that exceed agronomic need.

Hence, in interpreting our results, it is important to understand the absolute surplus or deficit as well as the balance regarding:

1. The land that likely *did* receive nutrient inputs: all cropland
2. The land that likely *could* receive nutrient inputs: all cropland plus some pasture
3. The total land in production: all cropland plus all pasture

Cropland area: The Census reports acreage that received commercial fertilizer at the county level for all years from 1954-2012. Additionally, from 2002-2012, the acreage receiving manure is reported at the county level. Prior to 1954, no information is available regarding the acreage that received nutrient inputs. Available state data indicate that:

- The percentage of harvested cropland receiving fertilizer rose from 24% in 1954 to ~56% in 1978. Since then, the percentage has remained around 50% (+/- 5%).
- The percentage of pastureland receiving fertilizer has been 10% or less for all reported years.
- For the three Census years in which the land receiving fertilizer and manure inputs is reported, the area treated with manure is roughly equal to the area treated with fertilizer; the total area is similar to the area of harvested cropland (90% - 106% of harvested cropland). This assumes no land receives both manure and fertilizer P, which is reasonable. The N:P ratio of manure is low, so it is easier to meet P requirements with manure than N requirements; i.e., farmers may apply manure to meet P requirements and supplement with nitrogenous fertilizers. Available data suggest that farmers in VT continue to increase the amount of N applied as fertilizer, while fertilizer P consumption declines (IPNI, 2012).

Hence, we report the likely area that received fertilizer and manure as equivalent to the area of harvested cropland. While it is unlikely that all harvested cropland receives fertilizer and/or manure each year, it is reasonable to assume that over a multi-year, multi-rotation period nearly all cropland receives some nutrient inputs.

Cropland + some pasture: We follow Kellogg et al. (2000) in assuming that all harvested cropland, all pastured cropland, and half (50%) of other pasture land (this includes permanent pasture and pastured woodland) may potentially receive nutrient inputs (beyond grazing manure). Idle/fallow cropland, woodland, orchards, and land dedicated to farmsteads, roads, and other built structures are excluded.

Cropland + all pasture: We calculated the total area that could receive and assimilate nutrients as the sum of all cropland (including fallowed or idle cropland) and all pastureland. Woodland and land dedicated to farmsteads, roads, and other built structures were excluded.

Uncertainty Analysis

Uncertainty is rarely addressed in P MFAs. In MFAs more generally, uncertainty can be difficult to address using traditional methods because some parameters may have neither a known distribution nor an alternative data set to use for corroboration/validation (Hedbrant and Sörme, 2001).

To address uncertainty, we adapted the approach of Hedbrant and Sörme (2001), which has been used in a P MFA previously by Antikainen et al. (2005, 2008). We defined a set of uncertainty factors and assigned factors to each variable (flow data, conversion factors, etc.), based on reported statistical properties or expert judgment (Table A1-9). The assigned factor was often adjusted to reflect changes in data quality over time.

We used the equations defined in Hedbrant and Sörme (2001) to calculate the uncertainty factors that resulted from the sum or product of multiple variables: for example, a P conversion factor and the associated material flow, each with its own uncertainty factor, may be multiplied to calculate a P flow. An uncertainty factor will be calculated for this P flow, and both the flow value and the uncertainty factor may be used in subsequent operations.

Hedbrant and Sörme (2001) assume a normal distribution for the upper-bound of the data, i.e. the uncertainty factor is multiplied by the mean to calculate the upper bound and the mean is divided by the uncertainty factor to calculate the lower bound. The upper interval corresponds to one standard deviation from the mean. The calculated interval for surplus P was doubled to equate to the 95% confidence interval.

Table A1-9: Uncertainty Levels, Factors, and Flow Magnitudes

Flow	Type	Equation	Uncertainty Levels^	Uncertainty Factor	Flow Magnitude (Tonnes) in 1925	Flow Magnitude (Tonnes) in 2012
Imported Feed	Inflow	Direct Output	Direct Output	Varies	1813	2475
Fertilizer	Inflow	Direct input	1925-1940:3; 1945-1982: 2; 1987-2012: 1	1.05, 1.1, 1.2	1361	774
Manure	Internal Flow	Animal Inventory *	1*(2-4, depending)	1.112, 1.206, 1.334	6886	5128

Flow	Type	Equation	Uncertainty Levels^	Uncertainty Factor	Flow Magnitude (Tonnes) in 1925	Flow Magnitude (Tonnes) in 2012
		Conversion Factor				
Pasture	Internal Flow	Yield * conversion factor	3*4	1.467	3165	810
Feed and Forage	Internal Flow	Yield * conversion factor	2*1 (Feed + Corn Silage) 2*3 (Non-Corn Forage; Corn Hogged; Oats Unthreshed)	1.112 1.224	2875	3392
Spoilage	Internal Flow	Constant * Feed and Forage	4* Feed and Forage	Varies	216	254
Milk	Outflow	Production * conversion factor	1*1	1.071	478	1093
Meat	Outflow	Production * conversion factor	2*2	1.141	266	194
Eggs	Outflow	Production * conversion factor	1*1	1.071	8	7
Fruits and Vegetables	Outflow	Estimated yield per acre * acreage * conversion factor	Fruits: 3*3*2 Sweet Corn, Other Vegetables: 5*3*3 Potatoes; Dry Beans: 2*1	1.3 1.574 1.112	52	19
Runoff	Outflow	Constant * (Fertilizer + Manure)	5* (Fertilizer + Manure)	Varies	616	443
Animal Stock	Stock	Value = 0; assumed steady-state conditions	NA	NA	NA	NA
Soil Stock	Stock	Calculated as remainder of:	NA	NA	NA	NA
^Uncertainty Levels: Level 1 = */1.05, Level 2 = */1.1, Level 3 = */1.2, Level 4 = */1.33, Level 5 = */1.5						

Appendix 2: Supplementary Results for Chapter 4

Our material flow analysis (MFA) assessed county- and state-level phosphorus (P) flows and balance. In the Supplementary Information (SI) document we present additional results that we were unable to include in the main research paper.

Total Legacy P Accumulation

Between 1925 and 2012, we estimate that Vermont's agricultural soils accumulated more than 230,000 tonnes of legacy P (Figure A2-1).

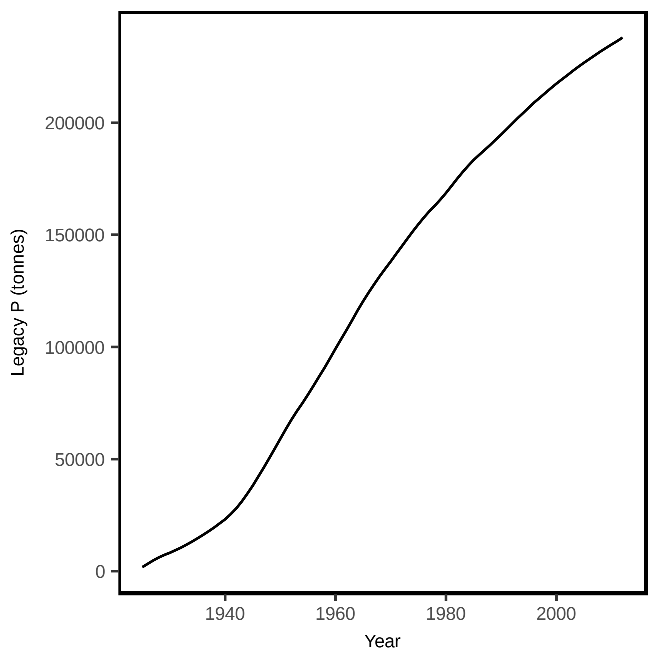

Figure A2-1. Total legacy P accumulation in VT, 1925-2012.

Changing Inputs

Manure is the main source of P applied to VT's agricultural soils since 1925 (Figure A2-2). Fertilizer P was most important as a source of P inputs from 1945-1982, declining since.

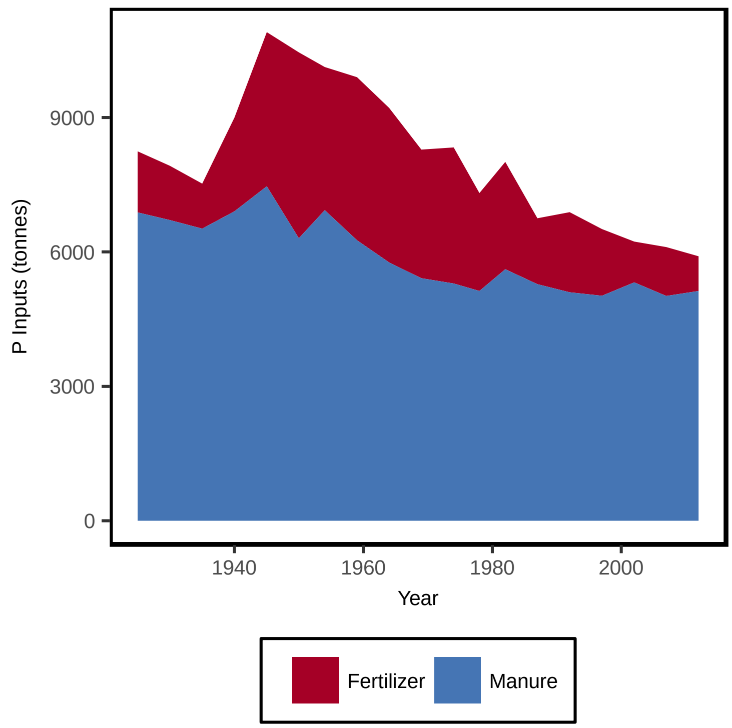

Figure A2-2. Agricultural P inputs by source in VT, 1925-2012.

County P Balance

The state-level results reported in the main research paper mask considerable spatial and temporal diversity at the county-level. While the total surplus has declined substantially (Figure A2-3), some counties continue to accrue large surpluses (Figure A2-4). Notably, Addison, Franklin, and Orleans counties – the largest agricultural producers in VT – continue to accrue > 200 tonnes of legacy P each year. These same counties have seen their share of VT's surplus increase (Figure A2-5), possibly reflecting the shift in the spatial center of gravity for agriculture to the northwest (Figure 2). This shift is important since the two major watersheds subject to a P TMDL – Lake Champlain and Lake Memphremagog – are located in the northwestern part of the state. Agricultural has spatially concentrated, intensified, potentially mitigating some of the beneficial effects of the declining P surplus. It is noteworthy that these are the only counties to see a marked increase in their share of the state's surplus; all other counties have seen their share of the state's surplus remain stable or decline (Figure A2-5).

In looking at the county-level surplus P (Figure A2-4), both Addison and Franklin counties exhibit considerable inter-annual variance in their total surplus. This may be due to these counties accounting for the majority of VT's corn land, which has high inter-annual variability in yields. In 2012, the two counties accounted for 57% of VT's corn land.

Counties varied widely in the rate at which they accumulated legacy P (Figure A2-6). When the rate is measured as surplus P divided by cropland, virtually all counties exhibit a decline in the rate of legacy P accumulation in the past two decades. When the rate is measured as surplus P divided by cropland plus some or all pasture land, the pattern is more variable. As with other trends, Addison, Franklin, and Orleans counties appear to be accumulating legacy P at an increasing rate each year. Conversely, most other counties have seen their rates of legacy P accumulation decline or remain relatively stable over time.

A detailed analysis of results for 2012 (Table A2-1) further corroborates the outsized importance of Addison, Franklin, and Orleans counties, while also highlighting significant county-level variability.

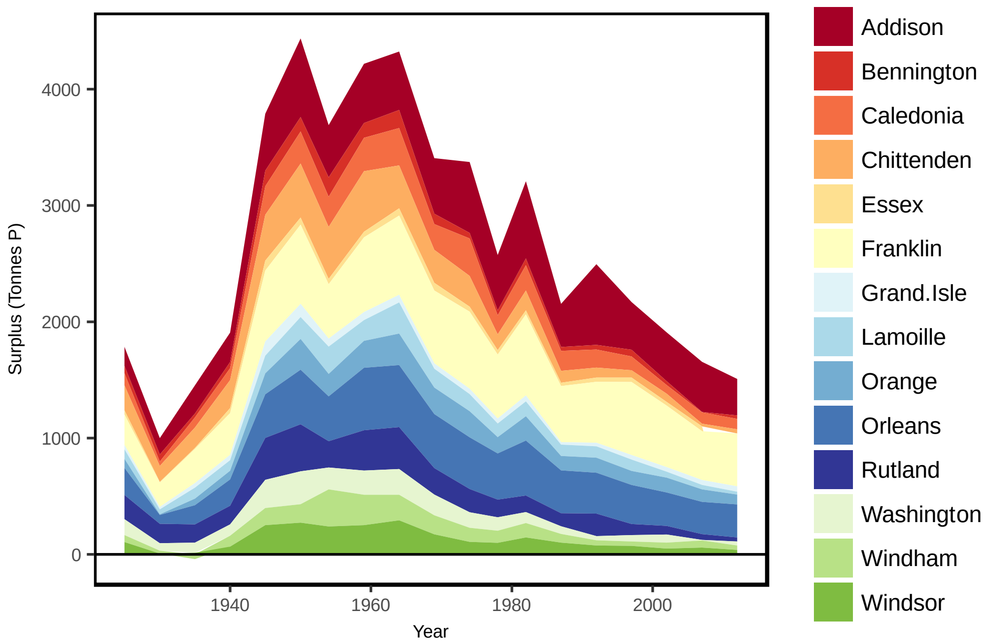

Figure A2-3. Surplus P in VT's counties, 1925-2012.

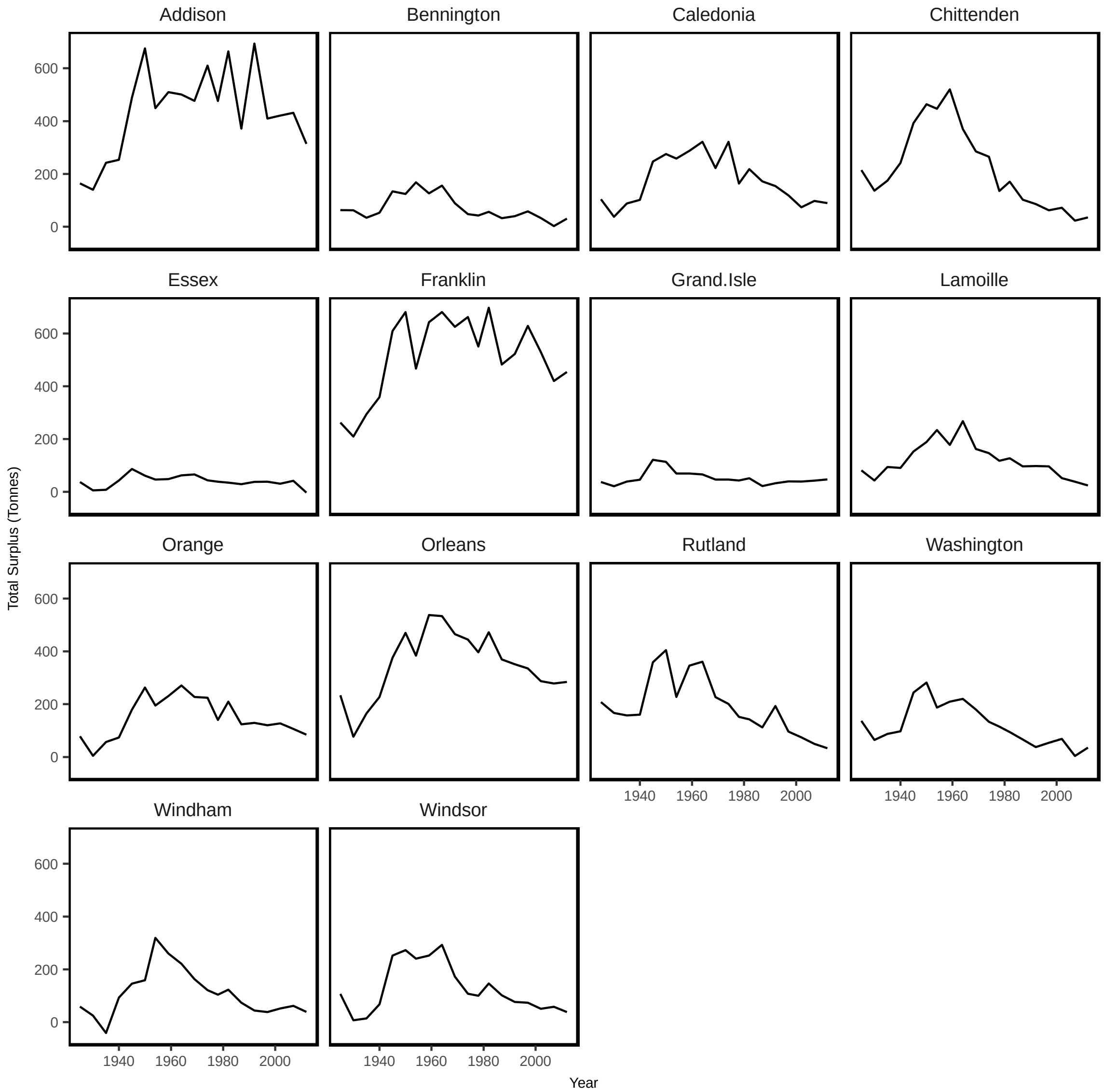

Figure A2-4. Total P Surplus for VT Counties, 1925-2012.

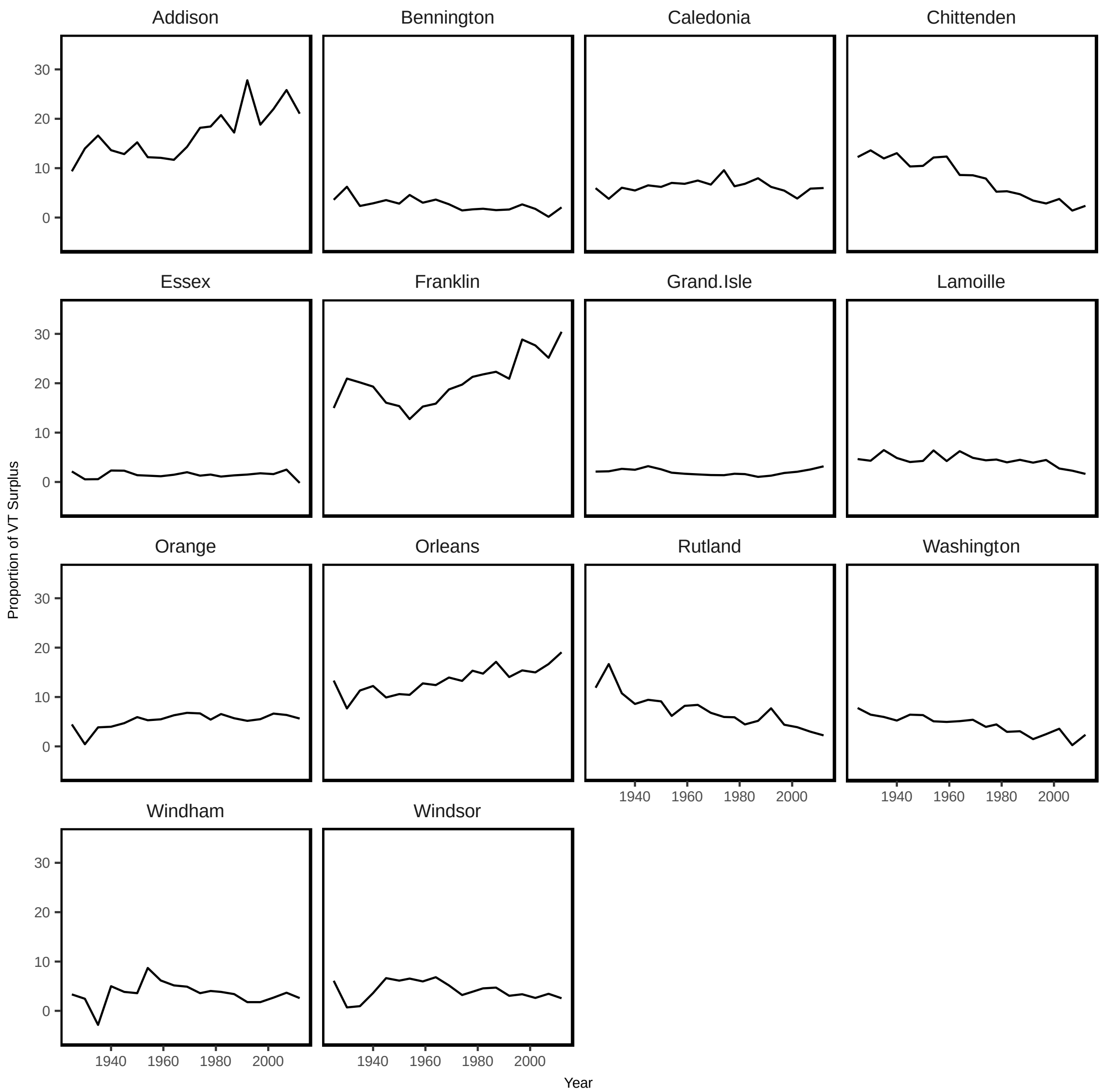

Figure A2-5. Each County's Proportion of VT's P Surplus, 1925-2012.

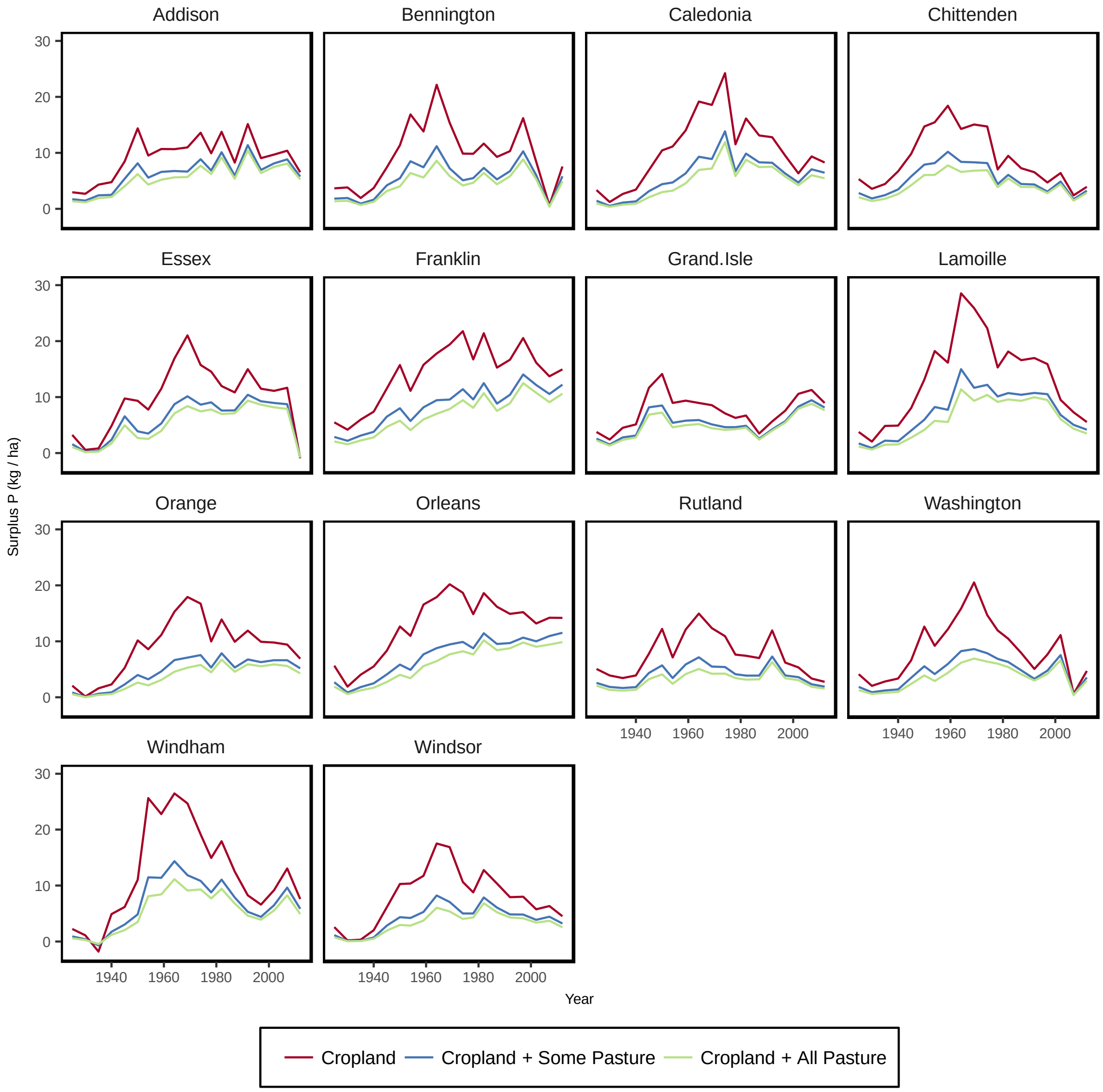

Figure A2-6. Legacy P Accumulation Rates in VT Counties

Table A2-1. County and state summary for 2012

	Cultivated Land		Animal Units		Flows (tonnes)					Performance Indicators			Surplus (kg/ha)	
Region	*Ha*	*% of VT*	*Total*	*AU/ ha*	*Fert-ilizer*	*Feed Imports*	*Manure*	*Runoff*	*Food*	*Surplus (tonnes)*	*PUE*	*Legacy P (tonnes)*	*All Ag. Land*	*Crop-land*
Addison	59,692	23.0	75,833	1.39	120.0	619.3	1,291.8	105.9	319.4	314.0	0.43	38,820	5.3	6.5
Bennington	6,214	2.4	4,805	0.92	19.8	32.1	71.6	6.9	14.5	30.5	0.28	6,337	4.9	7.6
Caledonia	16,383	6.3	16,519	1.20	53.1	126.2	274.1	24.5	65.7	89.1	0.37	15,666	5.4	8.3
Chittenden	12,521	4.8	12,361	1.12	37.4	68.6	196.9	17.6	52.9	35.5	0.50	19,669	2.8	3.9
Essex	3,639	1.4	4,389	1.30	13.6	8.9	74.8	6.6	18.9	-3.0	0.84	3,581	-0.8	-1.0
Franklin	42,783	16.5	72,095	1.94	223.0	679.4	1,229.1	108.9	339.6	453.9	0.38	45,646	10.6	15.0
Grand Isle	6,187	2.4	7,909	1.36	25.6	66.6	129.5	11.6	33.2	47.4	0.36	4,614	7.7	8.9
Lamoille	7,037	2.7	6,256	1.07	23.9	36.4	103.0	9.5	26.3	24.5	0.44	10,777	3.5	5.5
Orange	19,781	7.6	21,124	1.30	32.1	169.0	350.8	28.7	88.1	84.3	0.44	13,308	4.3	6.9
Orleans	28,827	11.1	42,902	1.74	91.0	453.5	719.3	60.8	199.2	284.5	0.37	31,132	9.9	14.2
Rutland	21,677	8.4	14,341	0.82	54.4	50.7	231.1	21.4	50.2	33.5	0.48	17,227	1.5	2.8
Washington	11,821	4.6	11,343	1.14	27.0	69.2	183.6	15.8	45.0	35.4	0.47	10,774	3.0	4.7
Windham	7,866	3.0	7,140	1.08	26.9	55.1	115.7	10.7	32.8	38.5	0.40	9,636	4.9	7.6
Windsor	14,982	5.8	9,865	0.83	26.2	56.6	156.5	13.7	30.9	38.1	0.37	11,376	2.5	4.5
VERMONT	259,412	100.0	306,883	1.37	774.0	2,474.8	5,127.8	442.6	1,313.1	1,493.1	0.40	237,934	5.76	8.27

Uncertainty Analysis

To address uncertainty, we adapted the approach of Hedbrant and Sörme (2001), which has been used in a P MFA previously by Antikainen et al. (2005, 2008). Our results indicate, unsurprisingly, that uncertainty in our estimate of surplus P was greater in earlier years (Figure A2-7).

Figure A2-7. Surplus P accumulation in Vermont. The bounds of uncertainty are indicated in grey.