The University of Vermont

Valuation of soil health ecosystem services

Vermont Payment for Ecosystem Services Technical Research Report #5

Version 2

Prepared for the Vermont Soil Health and Payment for Ecosystem Services Working Group

July 2022

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GUND INSTITUTE FOR ENVIRONMENT

THE UNIVERSITY OF VERMONT EXTENSION

Valuation of Ecosystem Services from Improved Soil Health in Vermont

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Executive Summary:

- Soil health, and the practices meant to support it, can contribute to human well-being far beyond direct impacts on agricultural productivity.
- Ambitious improvements in soil health on Vermont farms could yield more than $31/acre/year in ecosystem services, providing a total value of $25 million/year across all Vermont agricultural land.
- Soil health improvements could increase carbon storage, nearly $19/acre/year in climate mitigation benefits.
- Soil health improvements would reduce phosphorus losses, yielding nearly $8/acre/year in water quality benefits.
- Soil health improvements would reduce erosion, yielding $2/acre/year in reduced damages to waterways.
- Soil health improvements would increase water retention and infiltration, yielding an average of over $2/acre/year in reduced flooding damages to downstream communities, with values over $10/acre in some locations.
- These estimates demonstrate substantial benefits which could justify serious policy efforts to support, measure and pay for soil health improvements on Vermont farms. The estimates are preliminary, and subject to many uncertainties.
- Ecosystem services generated from large improvements in soil health are similar to ecosystem services generated by adopting best management practices on annual cropland.
- This report focuses on in-field improvements in soil health, and thus does not include edge-of-field and whole-farm practices. The impacts of these other practices on ecosystem services are often better studied than those of soil health. We refer to this research below, but estimating their economic values is beyond the scope of this report.

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Introduction

For millennia, farmers have recognized the importance of soil health for crop productivity and resilience. Recently, scientists, policy-makers, and farmers have become interested in the non-agricultural benefits of healthy farmland soils. Healthy soils can support climate mitigation through carbon sequestration, protect the health of waterways by retaining nutrients and sediments, protect downstream communities by absorbing water and protect the air by regulating gaseous emissions. These and other ecosystem services provided by healthy soils may meaningfully contribute to the health and vitality of communities and ecosystems.

In recent years, farms have struggled financially and awareness of environmental problems have grown. Policy-makers worldwide have sought ways to compensate family farms for their environmental stewardship as a means to tackle both these problems. Farmers have organized under the banner of "regenerative agriculture" to experiment with new practices and promote values provided by healthy soils far beyond the farm.

Vermont is well-positioned to become a leader in this movement; family farming and environmental stewardship are central to our collective identity and economy. There have been several efforts to develop a policy framework for soil stewardship, but none have succeeded. In 2019, Act 83 of the Vermont Legislature created a working group to explore payments for ecosystem services as a framework for linking farm supports and environmental stewardship. This report was commissioned as part of this effort.

To design a program to promote soil ecosystem services, it is necessary to generate an estimate of the magnitude of each of the benefits. If we understand the scale and value of benefits, we can then judge the cost-effectiveness of such a program compared with alternatives, such as investments in other natural systems like forests and wetlands, or investments in hard infrastructure. Because improvements in natural systems can affect many different things we care about, putting total benefits in dollar terms helps us to combine different types of benefits and to assess which benefits are largest.

In this report, we present estimates for ecosystem services from soil health using two approaches for four different services. One approach generates estimates based on soil-health practices, and the other approach is based on improvements in soil-health indicators. For soil-health practices, such as adopting best-management practices on annual corn, we utilize a set of off-the shelf empirical models widely used to estimate ecological functions on farm landscapes. For soil-health indicators, we make estimates by linking these tools with soil data and statistical models describing how soil-health parameters influence the interaction of soils with water and their environment. We provide rough monetary estimates of the value of these services, using several different standard ecological economics methods. These results are necessarily rough but can help to elucidate the relative magnitudes of different types of benefits.

Scope

This report provides a preliminary valuation estimates for four important ecosystem services in the state of Vermont from soil health improvements, including carbon storage, phosphorus (P) loading reduction, erosion control, and flood mitigation. The report also briefly addresses impacts of soil health on nitrogen cycling and pollution, but complexity and uncertainty prevents us from estimating values. While soil health has numerous benefits to yield, crop quality and climatic resilience for the individual farmers and landowners, these benefits are outside of the scope of this report. Instead, we focus on public goods provided to society at large, to inform a potential PES scheme for soil health in Vermont.

In keeping with the mandate of this project to focus on soil-health, we have excluded other management and land use changes that could have large impacts on the same ecosystem services. These include wetland restoration/construction, forested riparian buffers, conversion of agricultural land to forest, artificial ponds and stream de-channelization. While these “edge-of-field” or “whole-farm” strategies may have large impacts on the ecosystem services of interest, they are not directly “soil-health” related. The impact of these interventions on ecosystem services is also better-studied than the impact of soil health. A full assessment of the potential of farms to provide ecosystem services should consider impacts of all potential management options, but these are beyond the scope of this report.

Overall methods

This report estimates ecosystem service provision using two distinct perspectives (Figures 1,2). First, we estimate the increase in ecosystem services from ***soil health practices***, using the scenarios developed for Task 2 of our technical services contract to the PES Working Group as examples. See Table 1 for more details of these practices. For this, we use an array of existing empirical models, including the Universal Soil Loss Equation, the Curve Number Method and the Vermont Phosphorus Index to estimate the change in ecosystem services. All these scenarios take row crops with conventional tillage as their baseline for comparison. These methods assume a “normal” soil-health condition.

Table 1: Descriptions of Soil Health Practice Scenarios used in this Report. Row crops with conventional tillage was used as the baseline for comparison.

Soil Health Practice Scenario	*Description*
Corn BMPs	No-till / zone-tillage, winter rye cover crop & manure injection. These represent heavily-promoted BMPs by the state of VT for water quality.
Corn-Hay Rotation	Replacing Continuous Corn with a rotation that is half-corn, half-hay without implementing the BMPS mentioned above
Permanent Hay	Long-term perennial hay crops.
Pasture	Long-term perennial pasture[1].
Vegetable BMPs	Annual vegetable production with greatly reduced tillage with both winter and summer cover crops. This scenario uses vegetables grown conventional-tillage and no cover-crop as its baseline.

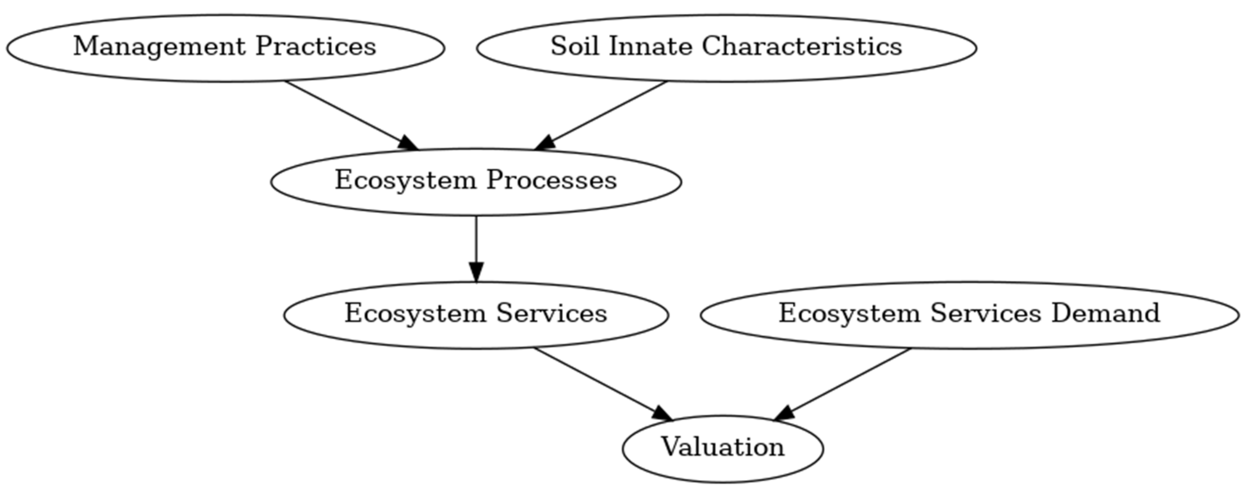

Figure 1: Conceptual Model for Estimating Impacts of Soil Health Practices on Ecosystem Services.

[1] We do not attempt to model or define different pasture management styles, which may have very different impacts. If careful pasture management has large impacts on ecosystem services, it will be due to improve soil health, and the benefits would best be reflected through estimating the direct impacts of soil-health.

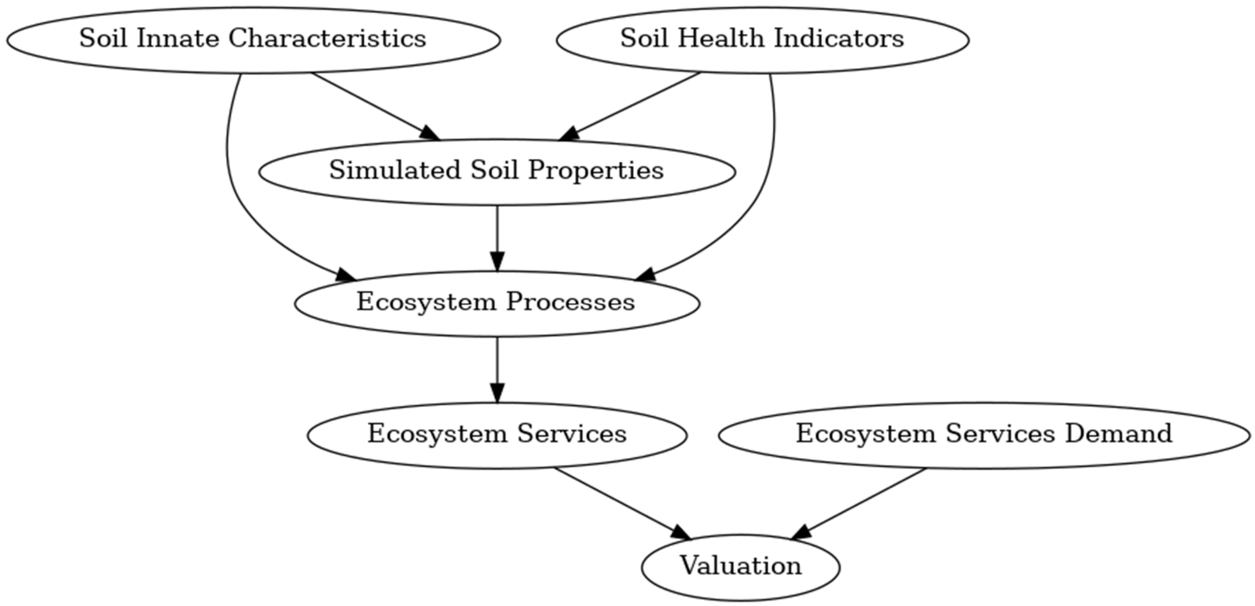

Figure 2: Conceptual Model for Ecosystem Services Assessment of Soil Health *Indicators*

Second, we estimate impacts of changes in **soil-health *indicators*** on ecosystem services. We use data from the NRCS Soil Characterization Database (Reinsch & West, 2010) to define innate characteristics and reference conditions for Vermont soil series. Innate characteristics are those that don't change with management, such as soil particle-size distribution. Reference conditions are used as typical baselines for conditions that are potentially impacted by management, such as Soil Organic Matter, Bulk Density and depth of each soil horizon. Soil innate characteristics and soil health indicators are used to simulate other soil properties, such as soil erodibility, plant available water capacity and saturated hydraulic conductivity. These parameters are then used to simulate changes to the ecosystem services of interest, using similar tools to those used for soil indicators.

We present two scenarios for moderate and large changes in soil-health and estimate their impacts relative to the reference state of the soil.

These soil health scenarios are:

"Best": Soil Organic Matter in the A horizon is 50% higher than the reference condition and bulk density 20% lower.

"Good" : Soil Organic Matter in the A horizon is 25% higher than the reference condition and bulk density 10% lower.

For each scenario, we simulate these changes on 10 different common agricultural soil-series: Tunbridge, Winooski, Agawam, Windsor, Covington, Vergennes, Cabot, Hadley, Hamlin and Georgia, and present average results, sometimes grouped by soil characteristics.

Bulk Density and Soil Organic Matter are important indicators of soil health, but their impacts on many important ecosystem processes, and therefore ecosystem services are mediated through their impacts on *other soil characteristics*. Many of these other soil properties can, in principle, be measured, but would not be feasible to include in a PES program. Instead, these characteristics, including plant available water capacity, porosity, saturated hydraulic conductivity and soil erodibility are simulated through a series of pedo-transfer functions[2]. These equations are used to estimate unknown soil properties based on known soil properties.

In this report we estimate the impacts of two different improvement scenarios for several different common Vermont Agricultural Soils and present averages of these results. The two improvement scenarios are the “high” scenario: Soil Organic Matter increases by 50% and bulk density declines by 20% and the “medium” scenario: SOM increases by 25% and bulk density declines by 10%. In both scenarios, these improvements are confined to the upper layer (A horizon) of the soil, and the decrease in bulk density is compensated for by increasing the depth of the A horizon to keep the mass of soil in the A horizon constant. For reference, agricultural soils in Vermont have average SOM contents of roughly 4.3% and bulk density of about 1.35, with substantial heterogeneity across soil types. This average soil would see SOM increase to 5.4% or 6.5% and its bulk density decrease to 1.22 g/cm^3 or 1.08 g/cm^3 in the good and best scenarios, respectively.

Additional information about the scenarios can be found in Appendix 1.

Importantly, we do not attempt to merge these two approaches and estimate the impact of soil health practices on soil characteristics themselves, and then the impacts of these soil characteristics on ecosystem services. We hesitate to do this because most tools used to assess the impact of practices on soil ecosystem functions and services do not allow us to partition between their *direct* impact on soil ecosystem services and their impact which is mediated through soil health. For instance, the NRCS Curve Number method predicts lower runoff from land that is in permanent grassland than land that is growing corn. This is due to improved soil health, greater vegetative cover and other differences, but the method gives us no way to disentangle the portion of the impact that is due to soil health itself. Hence the two distinct approaches described above.

[2] A pedo-transfer function is an equation that predicts an unknown soil property based on several known soil properties. For instance, if we know the texture of the soil, (as % sand, % silt and % clay), its bulk density and its soil organic matter content, what is the expected plant-available water capacity?

Results Summary

Overall, improvements in soil health and adoption of soil health practices have the potential to produce substantial benefits for Vermonters and people around the world. Below we summarize the results of our valuation estimates for each service.

Carbon storage benefits are substantial, valued at $18.84/acre/year in the “best” scenario, and $9.42/acre/year in the “good” scenario. We calculate these based on the reduction in warming each year due to reduced atmospheric carbon.

Flood mitigation benefits have the lowest valuations, but also the most spatially variable. Average values are roughly $2.73/acre/year for the “best” scenario and $1.10/acre/year for the “good” scenario. These values are relatively low largely because farmland in Vermont is commonly situated low in watersheds, and therefore has protects relatively fewer downstream areas compared to other runoff-generating land cover types. A small minority of farm fields have many downstream communities at risk, and those fields have potential flood-mitigation values that are 5x or 10x higher.

Erosion reduction benefits are also relatively small for most farm fields- $2.47/acre for the “good” scenario and $1.21 for the “best” scenario. These benefits are proportional to the scale of current erosion losses; fields that are flat and already have extensive soil-cover will have much smaller reductions than steeper fields or those currently in row-crops.

Phosphorus retention benefits are large in dollar terms but come with much uncertainty. Average values for the “good” scenario are $4.12 /acre/year, while average values for the “best” scenario are $7.87. The relationship between reduce soil health and P-loading loading from soils with pattern tile drainage or other direct sub-surface connections to surface-water is more complex, and this report does not draw conclusions about this. Like erosion, P-mitigation benefits from improvements in soil health are highest where potential for P loss is highest, and in watersheds where P loading is a larger problem.

Beyond the four ecosystem services we were able to value, two more deserve mention:

Nitrogen retention benefits are difficult to characterize because nitrogen can leave farm fields and damage the environment through many pathways, and practices and soil conditions that reduce one pathway may increase another. We present general estimates of the magnitude of harms from N losses from Vermont farms and demonstrate that these harms are large enough that moderate mitigation would generate substantial benefits.

Soil biodiversity benefits could be valued in several ways, but producing a monetary valuation was beyond the scope of this report.

Under the "best" scenario of soil health improvement, we estimate that farms could be credited with providing an average of $31/acre/year worth of combined ecosystem services (Figure 3). Under the "good" improvement scenario, farms could be credited for $16/acre/year. In our analysis using soil health practices (Figure 4) estimates, all management improvements from a baseline of continuous corn with normal practices create total values of at least $25/acre/year.

Table 2: Summary of Ecosystem Services Valuation of Soil-Health Improvements for two Scenarios and 4 Services.

	Valuations ($/ac/yr)			*Physical Quantities*		
Service	*Good*	*Best*	*Valuation Rate ($/unit)*	*Good*	*Best*	*Units*
Carbon Storage	**$9.42**	**$18.84**	**$1.44**	**13.1**	**6.5**	**Tons (US) of carbon /acre.**
Flood-Runoff Mitigation	**$1.10**	**$2.37**	**$8.40**	**0.28**	**0.13**	**Inches / large storm**
Erosion Reduction	**$2.29**	**$4.56**	**$11.20**	**0.20**	**0.41**	**Tons (US) /acre/year**
Phosphorus Retention	**$4.12**	**$7.87**	**$56.82**	**0.07**	**0.14**	**Lbs / acre /year**

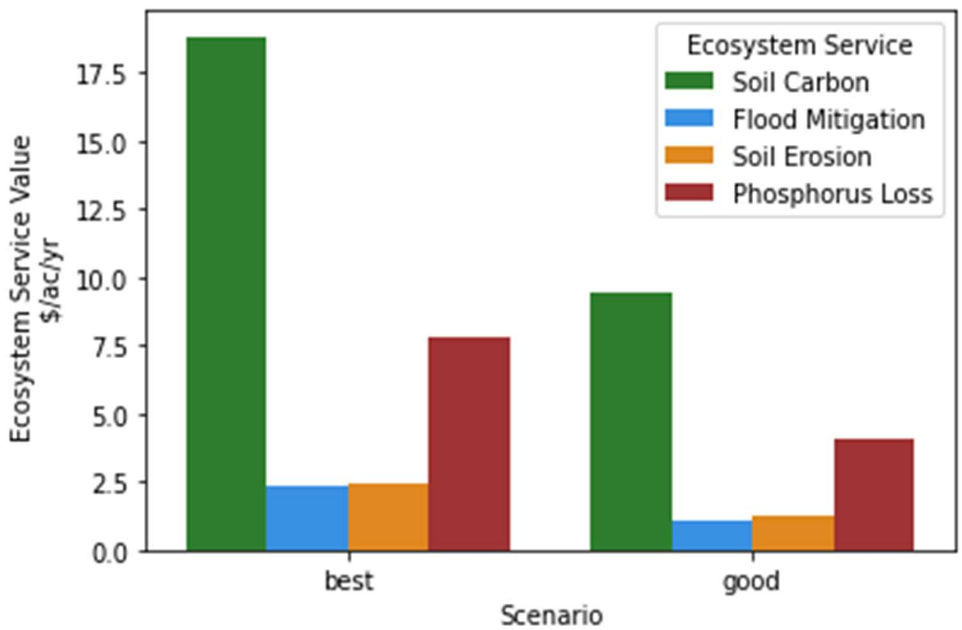

Figure 3: Predicted values of Improved Ecosystem Services resulting from Two Soil-Health Improvement Scenarios. Best: 50% increase in SOM and 20% decrease in bulk density. Good: 25% increase in SOM and 10% decrease in bulk density.

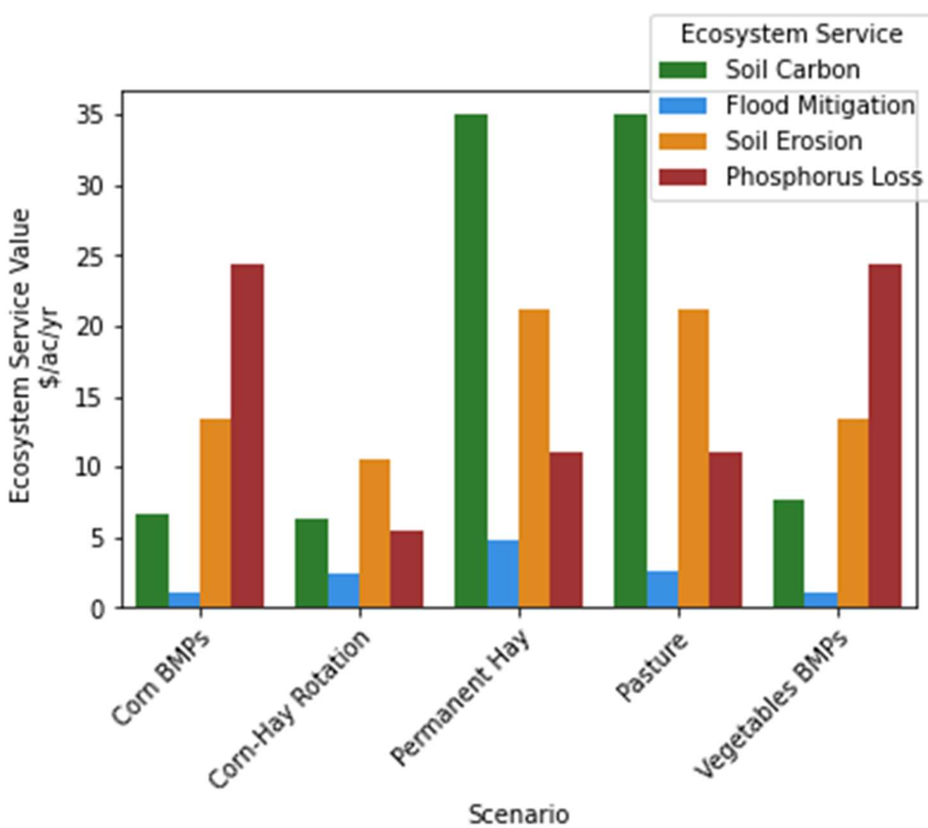

Figure 4: Values of Improved Ecosystem Services resulting from Changes in Soil-Health Practices. Practices match those developed in for Task 2. See Table 1 for descriptions.

Detailed methods and results for each ecosystem service

CLIMATE REGULATION

Healthy soils can mitigate climate change by storing carbon that would otherwise be in the atmosphere. Additionally, soil health and soil health practices can influence the production of other greenhouse gases from soils, especially methane and nitrous oxide.

Globally, soils hold an enormous amount of carbon; 3-4 times as much carbon as is currently in the atmosphere (Lal, 2003). Increasing the carbon content of soils may be an efficient way to mitigate climate change. Voluntary and regulatory markets for carbon storage make carbon storage by far the most commonly marketed ecosystem service from agriculture and other land-uses. Payments for land use-based carbon offsets now reach $1 billion / year (Dunn, 2021). Because soil carbon is directly measured as a soil-health indicator, there are fewer elements of uncertainty in the relationship between the soil health metrics and the ecosystem services of interest.

Valuing Carbon Storage and Carbon Accumulation:

There are two general approaches to valuing carbon sequestration. First, we may multiply the carbon sequestered by an estimate of the Social Cost of Carbon, as calculated by the EPA, other government agencies or academic researchers. The EPA's social cost of carbon for the year 2021 is $51/ton of CO_2 (Interagency Working Group & others, 2021). This would be equivalent to $186/ton of soil organic carbon. Alternately, we may compare them to the prices paid by voluntary or compliance-based offsets markets or other corporate programs. The Boston-based Carbon-Offset start-up Indigo Ag (Indigo Ag, 2022) currently guarantees prices in range of $10-$15/ton of CO_2, while the company Nori allows farmers to sell offsets for $15/ton (*Nori Carbon Removal Marketplace*, 2022). $15 per ton of CO_2 is equivalent to $53 for each ton of organic carbon added to farm fields. *We link values to the price of offsets ($15/ton) rather than the social cost of carbon because there is little way for Vermont government to capture the benefits of the globally avoided climate damages accounted for by the social cost of carbon*. To account for these global benefits, the values can be multiplied by 3.4.

A major area of concern for carbon sequestration payments is permanence. If a company pays for a carbon offset, or a government pays to reduce damages from carbon, that payment assumes that this carbon is permanently removed from the atmosphere, or at least removed for many decades. If this soil carbon is instead released back into the atmosphere, only a small proportion of these damages would be averted from the short-term storage of carbon, and the value of the carbon storage is greatly reduced.

Most carbon-offset programs deal with this difficulty by enforcing contracts on farmers, obligating them to continue their climate-friendly farming practices. This option seems unlikely for a state-run PES program. Some offset-generating carbon sequestration programs assume that not all carbon will be permanently stored and may reduce payments accordingly[3]. This approach could be taken by a soil PES program. Another approach could be to subtract the value of carbon losses from payments to the farmer generated by other ecosystem services. For the valuation of carbon storage from *practices*, we use a 50% withholding rate, such that farmers are only paid for 50% of the carbon they are expected to accumulate in their fields.

For soil-health indicators, and soil-health practices we must estimate slightly different values for carbon benefits. For practices, the benefits are usually measured in *carbon accumulation, in tons / year* with a change of practices. These rates of accumulation are expected to be maintained for a certain period of time (e.g. about 10 years) after transition in practices, before soil organic carbon contents stabilize at a new, higher level. For soil health indicators, soil organic matter is measured in *tons of carbon*, as a quantity. Because of this, if we measure the value of higher soil carbon using the social cost of carbon, or the sales price of offsets, we get a single lump-sum value. Not only is the number not comparable to the other values generated in this report, but impermanence and small measurement errors on farms with stable soil carbon could frequently generate substantial negative values[4].

To deal with these issues, we annualize the social cost of carbon and estimate the benefits generated by storing a ton of carbon for one year. To do this we utilize two different methods and average the results. In one method, we do this using calculations for the social cost of additional heat or "radiative forcing." In the other, we calculate a perpetual ongoing payment that is equivalent to the social cost of carbon.

The average of these two methods is $1.44/T SOC. This is valuation can be thought of as a "temporary rental" carbon offset, as opposed to a "permanent sequestration" carbon offset. Because these values are for climate mitigation benefits realized *each year,* no reduction is made to the valuation due to impermanence.

More information on this method, and its justification, can be found in Appendix 2.

Estimating Physical Quantities:

For Carbon Storage based on practices, we use estimates from the research literature compiled during Task 2. For Carbon Storage based on soil health indicators, we simply use the additional carbon in the simulated soil layers.

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[3] The California carbon market has about ¼ of forest-based credits withheld in a "buffer pool", which may not be sufficient. (Badgley et al., 2022; Herbert et al., 2020). In this instance, landowners have signed binding contracts to continue land management, which is unlikely in a PES program.

[4] See Appendix 2 for an explanation of these concerns.

Results:

Figure 5 estimates annualized increases in soil organic carbon, per acre, per year, for the soil health practices scenarios. These results are presented grouped by soil-texture class, which is the largest influence on how much carbon a soil can hold.

Figure 6 shows the estimated total soil carbon storage increase for the soil-health indicator scenarios. Because the soil-health indicator scenarios include carbon as a state variable, we cannot use them to estimate annual rates of accumulation.

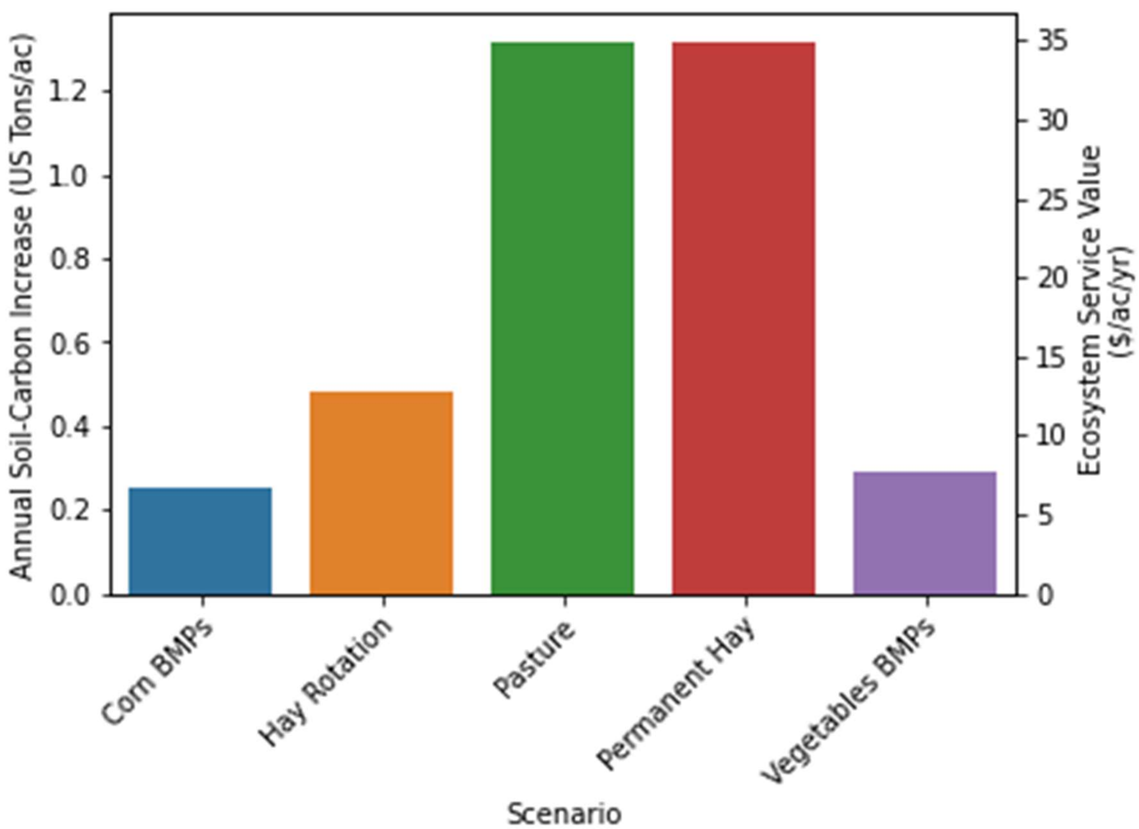

Figure 5: Total Increase in Soil Carbon and Ecosystem Service Value by Soil Health Practice Scenario[5]. Left axis reports predicted annual accrual of soil carbon, and right axis reports the economic value of these changes.

[5] Note that the Corn to Corn-Hay Rotation Numbers demonstrate the lack of durability in Soil Carbon increases: 5 years in Hay increases Soil Organic Matter dramatically, but almost half of that increase disappears when the field is rotated back into Corn for 5 years.

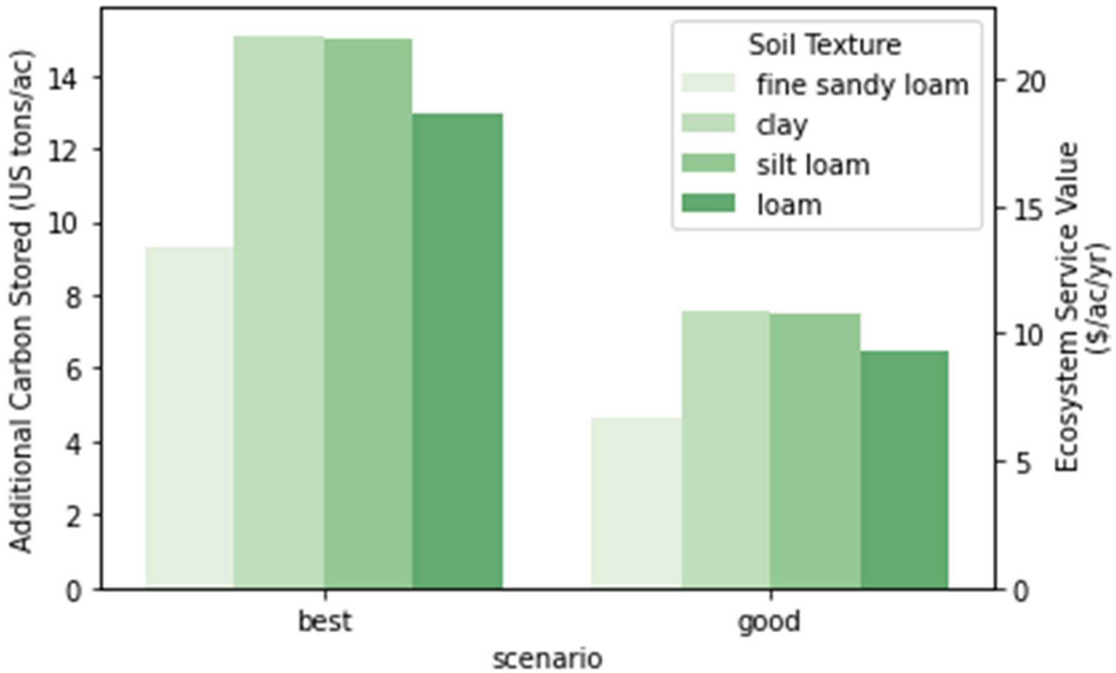

Figure 6: Total Increase in Soil Carbon by Soil Health Indicator Scenario, and Ecosystem Service Value. Left axis reports additional soil carbon stored, and right axis reports the economic value of these changes.

Variation of Service Provision and Values:

Because climate change is a global problem, the value of carbon storage is the same no matter where it is stored. For the quantity of carbon stored, farm fields with finer textures, such as clays, have more carbon storage capacity than coarse-texture soils such as sandy loams.

Caveats and Areas for Future Work:

While we have not completed more detailed simulations, in general, increased SOM results in moderate reductions in CH_4 emissions, while decreases in bulk density can moderately reduce emissions of N_2O. In temperate cropping systems, N_2O emissions are often quite substantial, especially in systems with substantial N inputs from fertilizer, legumes, or livestock manure. Methane emissions from soils, however, are relatively small, highly variable, and even sometimes negative. We discuss the general magnitude of N_2O emissions in more detail in the section on nitrogen losses.

For soil-health practices, the saturation of soil carbon-holding capacity is an important issue. The valuations provided for practice / land-use changes are only applicable for the first 10-15 years after converting from conventional corn and may not be applicable where farmland was recently converted into conventional corn.

Edge-of-Field and Whole-Farm Interventions:

Though beyond the scope of this report, a PES program compensating for carbon sequestration on agricultural land could also incorporate payments for carbon stored in woody biomass. Eligible land-uses might include silvopasture, riparian buffers, farm woodlands and other agroforestry.

FLOOD RUNOFF MITIGATION

Since the devastating flooding during Tropical Storm Irene in 2011, Vermonters have been working to make our communities safer and more resilient to flooding. Climate change is expected to increase the frequency of severe storms in Vermont, making this work even more important. Soils and vegetation upstream can play an important role in buffering peak stream-flows during storm events, protecting people, homes, and infrastructure in the valleys below. Farm fields also play an important role in protecting communities by providing space for rivers to spread out and slow down during flooding events. Flood-control services provided by coastal wetlands, riparian wetlands and upland forests are well-studied, but comparatively little research has been done on the impact of soil health in agricultural fields on flood risk[6] .

Our estimates attempt to be inclusive of all damages done by flooding, but estimates of damages, especially indirect damages, are highly imprecise. The estimates of flood-mitigation services attempt to fully account for increases in the ability of soils to infiltrate and hold both rainfall and floodwaters which inundate them but may not comprehensively account for the later.

Valuing Flood Risk:

To value reductions in flood risk from soil health practices and indicators, we must ask several questions:

- First, *what is the total, annual value of Vermont's flood risk (in $)?*
- Second, *what proportion of this risk can be attributed to runoff from agricultural land use (in %)?*
- Third, *how much of a difference does reducing runoff by a given amount reduce that runoff (in acre-inches for a reference storm)?*

We separately estimate these values for generational floods (>50 year recurrence interval) and more-frequent large floods, (10-25 year recurrence intervals). A summary of the steps that we took can be seen in Table 2.

[6] For a review of research on soil compaction and flooding, see Alaoui et al (2018), for one of soil health practices and flooding, see Basche (2017).

Table 3: Steps Taken to Estimate Flood Protection Values of Abating Agricultural Runoff

"Generational Storms"	**Number**	**Derivation**
Damages	$1 billion	TS Irene was about $1 billion in USD 2020
Frequency	50-year	TS Irene is a roughly a 100-year return time. We account for other large storms (e.g. 1973, 1938) by halving this.
Value of Risk	$20 million / year.	$1 billion / 50
Agriculture's Contribution	5%	Agricultural Land contributed 4.6% of damage-weighted runoff and was 5.6% of the landcover upstream from damaged communities (weighted by federal assistance).
Value of Agriculture's contribution	$1 million /year	5% of $20 million
Climate Change Adjustment (next 30 years)	50%	Estimates include: Wobus et al (2014) (+30% in $ damages, US) Gourevitch et al (2022) (+148% $ damages, VT, next 100 years) Swain et al (2020) (+30-127% people at risk US). These increases are driven by both larger and more-frequent storms.
Estimated value of runoff abatement (50 year flood or greater)	$0.88/acre-inch/year	$1 million * 1.5 / 1.7 million acre-inches of runoff from agriculture during Irene.
10-25 Year Floods	**Number**	**Derivation/Notes**
Damages to buildings in Champlain Basin:	$25.5 million /yr	Annualized Damages of 10 & 25 year floods from Gourevitch et al (2021)
All damages in VT	$72.9 million /yr	25.5 / .7 / .5 70% of VT structures are in Champlain Basin, about 50% of flood damages are to structures.
Damages when soil is not frozen	$56.9 million	78% of 56.9 22% of Flood Insurance claims are for damages from between December 1st and March 20th.
Agriculture's Contribution	9%	Agriculture is 9.5% of the landcover above communities damaged by non-Irene large floods (weighted by payments to towns by FEMA). It makes up a smaller proportion of runoff, though the exact proportion is not clear.
Value of Agriculture's Contribution	$5.1 million / year	9% of $56.9 million
Adjustment for Climate Change	$7.68 million / year	5.1 times 7.68. Increase 50%, as above
Estimated value of Agricultural Runoff Abatement: (10 or 25 year flood)	$7.68 / acre-inch	Assume average agricultural runoff from more-frequent storms is 1 ¼ inch per acre, yielding 1 million inches of runoff from 800 thousand acres of crops, hay and pasture.

Documentation of the flooding damages to Vermont communities from Tropical Storm Irene are useful in determining the risks posed by other extreme flooding events. Tropical Storm Irene resulted in an estimated $733 million in total damages[7], $860 million in 2020 dollars. This estimate appears to include nearly $400 million in damage to transportation infrastructure, >$10 million in damages to agriculture and $130 million to rebuild the state government complex Waterbury (VT Emergency Management, 2018). Damages to private real estate likely exceeded $150 million, and include nearly $29 million in damages assessed by FEMA and nearly $43 million in claims to the national flood insurance program (Federal Emergency Management Agency., 2021), though these are likely only a fraction of total damages to private property[8]. We account for non-financial losses from flooding (loss of life, disruption of work and school, etc) by rounding this number up to $1 billion, though a higher number may be justified. Vermont sustained one other storm of this scale in the last 100 years, in 1927, and two other, somewhat smaller major flood disasters, in 1938 and 1973.

How much does Agriculture Contribute to Flood Damages from Runoff?

Based on the National Land-Cover Dataset, 14% of Vermont land is in agriculture: cropland, hay, pasture and orchards. This land is larger located in places with lower value for flood run-off mitigation, due to lower elevation. This lower-elevation land has lower flood mitigation value due to:

1- Lower rainfall at lower elevations.

2- Fewer people and structures downstream. A large proportion of farmland is very close to Lake Champlain or the Connecticut River. Figure 7 shows that the highest concentration of farmland is in areas that flow directly into Lake Champlain, and within each sub-watershed, the largest concentration of agricultural land tends to be below the most heavily-populated areas.

An estimate using the Curve Number Method[9] yields about 10% of total run-off from agricultural lands during Hurricane Irene (Figure 8). This runoff largely occurred in areas below the most-impacted communities. Weighted by total Federal Assistance money from Irene (Vermont Public Radio, 2013), the average Irene-damaged community in Vermont had 5.6% agricultural landcover in its upstream watershed, and 4.6% agricultural runoff. Based on a 50-year return time, $1 billion damages and a 5% contribution of agriculture to damages, the annual value of agricultural runoff from generational storms is roughly $1 million/year. Adjusting 50%

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[7] The Irene Recovery Report (Rose & Ash, 2013) estimates $850 million in total assistance paid out.

[8] The NFIP claims database holds 1009 claims made on Irene in VT, while the Irene Recovery Report estimates 3500 homes and businesses damaged/destroyed and the State Hazard Mitigation Plan estimates ~5000. Assuming that 24% of damages were covered by the NFIP yields ~$180 million in damages to real estate.

[9]The NRCS curve number method is an empirical model which uses land management and soil hydrologic group to predict the rainfall-runoff relationship for a location. We additionally use an adjustment factor for slope developed by Arnold et al (2012). The CN Method is still state-of-the-art for runoff estimation, it is one of two options used for estimating runoff in the Soil and Water Assessment Tool (SWAT) and the Agricultural Policy Environmental Extender (APEX). For more information, see: https://acwi.gov/hydrology/minutes/nrcs_cn_method.pdf

upwards for climate-change risks and allocating among 1.7 million acre-inches of agricultural runoff during Irene yields $.88/acre-inch/year in large-storm runoff.

Agriculture plays a larger role in more-frequent floods. The methods for calculating its impact can be seen in the second part of Table 3. For medium-sized flood-events, we use estimates from Gourevitch et al (2022) for impacts of 10-25 year floods. This study utilized probabilistic simulation modelling of flood events in the Champlain Basin at different recurrence intervals. They estimate annualized damages of $25.5 million from storms of this scale. This number is increased to account for buildings outside the Champlain Basin and non-building damages, then decreased by 22% to account for winter flooding. Among smaller storms that still received federal disaster declarations, the average flood-damaged municipality (again, weighted by disaster assistance) in Vermont had 9.5% agricultural landcover upstream. Adjusting slightly down to 9% accounts for lower runoff from agricultural land yields $5.1 million/year in agriculture-related flood damages. Multiplying by 1.5 for climate change, and assuming an average of 1.25 inches average agricultural runoff yields $7.68/acre-inch in flood mitigation services.

More details on the methods used for valuation, their justification and uncertainty, can be found in Appendix 3.

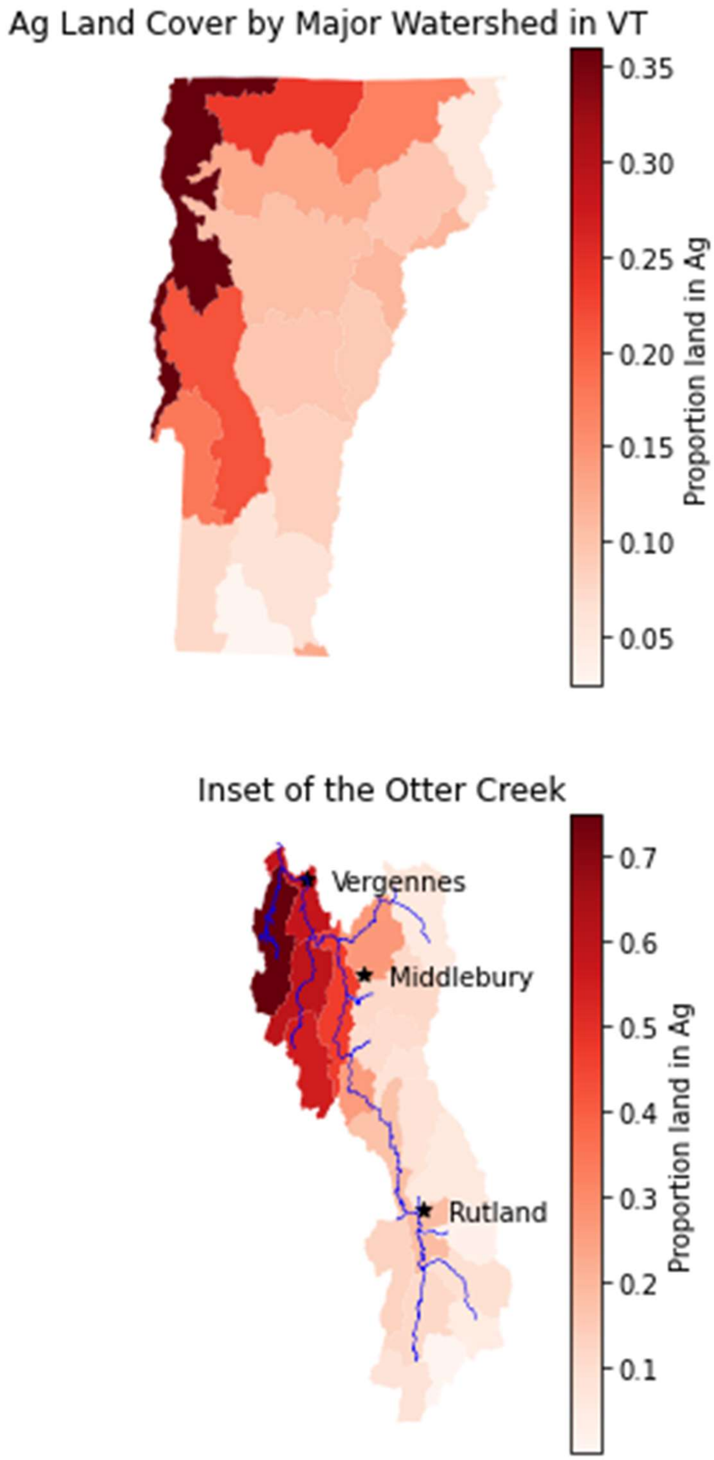

Figure 7: Percentage of Land in Agricultural Land Cover in Vermont Sub-watersheds. Data from 2014 NCLD. Agricultural land-use in Vermont is primarily close to Lake Champlain. 20% of agricultural land in VT is in sub-watersheds that flow directly into Lake Champlain.

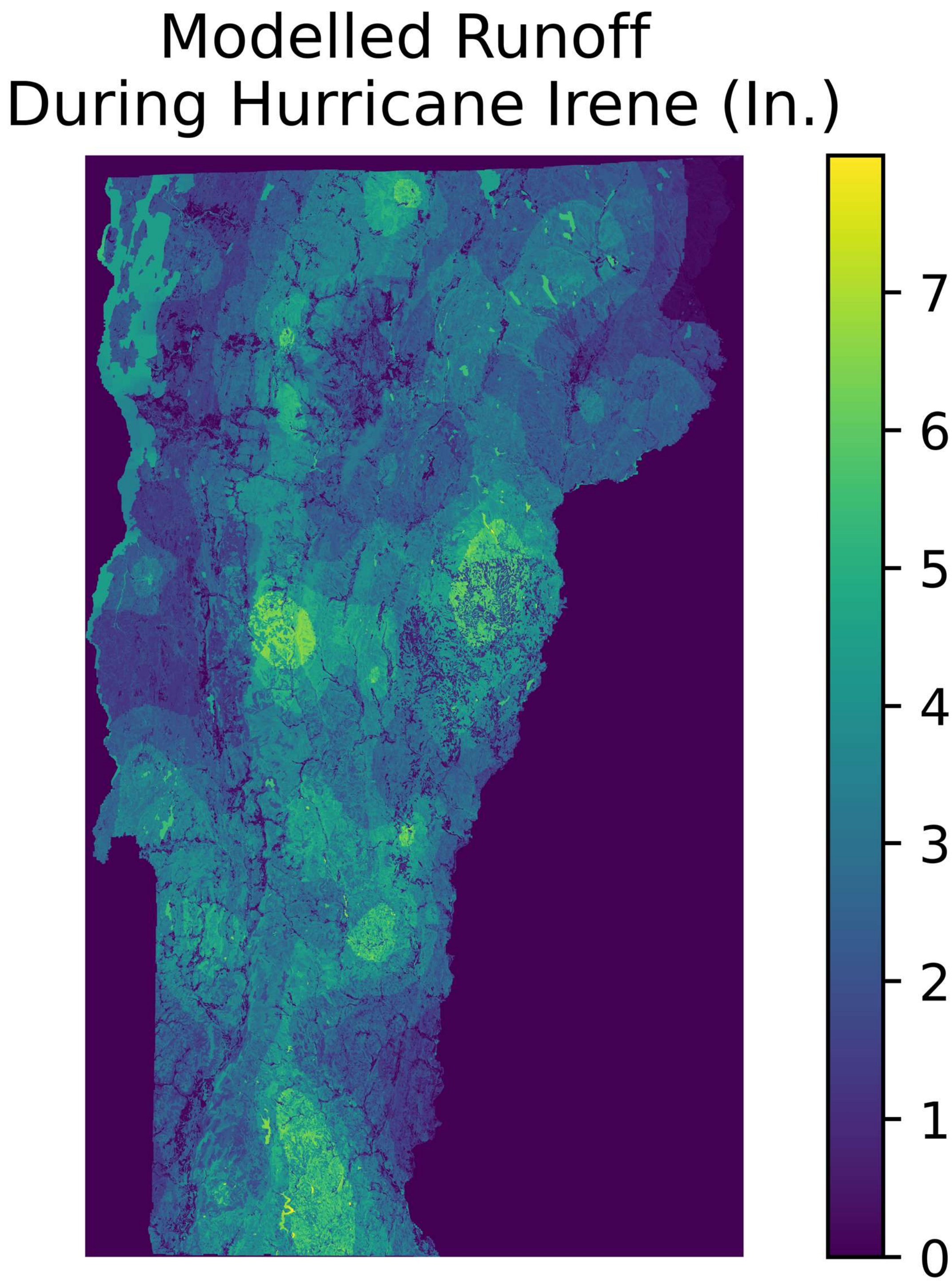

Figure 8: Runoff During Hurricane Irene, Modelled Using the NRCS Curve Number Method. Most runoff was generated from areas high in watersheds, with less agricultural landcover.

Estimating Physical Quantities:

To estimate runoff volumes for our analysis we simulate two different storms; a generational storm with 4 inches of rain falling over the course of 8 hours, and a large storm with 1.5 inches of rain falling over the course of 3 hours.

For reductions in runoff from practice changes, we use the Curve Number Method to estimate runoff volume. For very large storm events, this method is known to under-estimate runoff volumes, and thus likely exaggerates the impacts of practices.

For reductions in runoff from soil health, we use different methods for calculating flood runoff mitigation, based on soil hydrologic group. For soils in hydrologic groups C and D, we use a three-layer implementation of the Green-Ampt equation[10], while for soils in hydrologic groups A and B, we use an excess water-holding capacity method. Runoff from soils in hydrologic groups C and D is dominated by infiltration-excess runoff (runoff is generated when rainfall exceeds the soil's infiltration rate), which is well-simulated by the Green-Ampt equation. Runoff from soils in hydrologic groups A and B is dominated by saturation-excess, where runoff occurs when soils are filled to capacity. This is better simulated by available water-holding capacity in the soil at the onset of precipitation.

For both methods, we estimate soil water-parameters using a series of pedo-transfer functions and assume that the soils have 30% of their plant-available water-holding capacity available at the onset of the storm.

More details on these methods can be found in Appendix 3.

Results:

Current evidence supports only minor or moderate flood mitigation ecosystem services from soil health improvements on agricultural land in Vermont. The Figures 9 & 10 summarize the average runoff reductions for the two simulated storms. Except for conversion of row crops to hay, impacts are generally between $^1/_6$ inch and ½ inch. Monetary valuations are unlikely to reach levels relevant to farmers, at least on average. Corresponding monetary valuations are at or below $6.00/acre/year (Figure 11).

[10] The Green-Ampt equation is a simulation model describing how rainfall infiltrates into a soil, based on several soil physical parameters, including available water capacity and saturated hydraulic conductivity. For a detailed explanation, see: http://www.alanasmith.com/theory-Calculating-Effective-Rainfall-The-Green-Ampt-Method.htm. The Green-Ampt method is over 100 years old, but still widely used; along with the curve number method, it is one of two options for simulating runoff in SWAT and EPIC/APEX. We implement a Green-Ampt model with 3 distinct soil layers.

For the best soil-health scenario, runoff reductions range from ¼ to ¾ an inch. Corresponding valuations are from $1.50 - $4.00 /acre

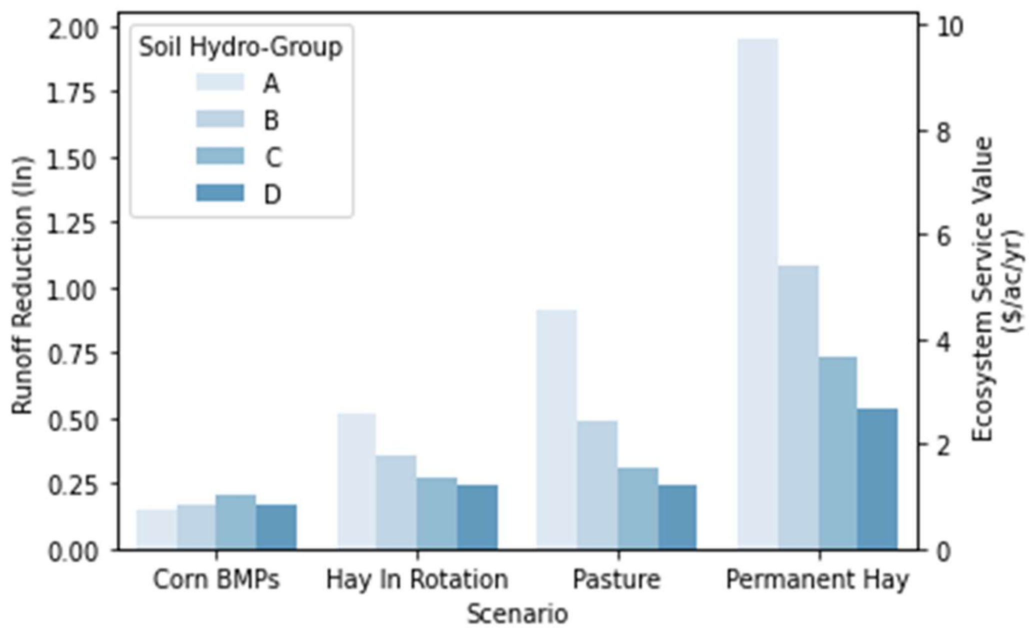

Figure 9: Runoff Reductions (4-inch storm) and Ecosystem Service Valuation for changes in Soil-Health Practices (Reference Case: Row Crops, Conventional Tillage) Left axis reports predicted changes runoff, and right axis reports the economic value of these changes.

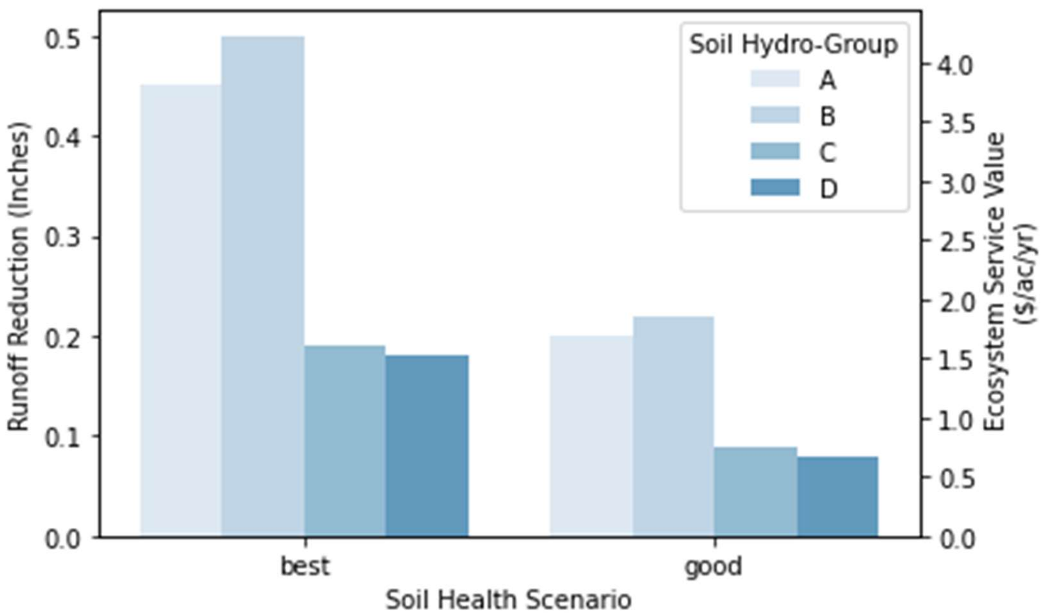

Figure 10: Runoff Reductions (4-inch storm) and valuation in Good and Best Soil-Health Improvement Scenarios. Left axis reports predicted changes runoff, and right axis reports the economic value of these changes.

Variation in Service Provisioning and Value:

There is some of variation in potential increases in runoff mitigation from farm fields; the same changes may mitigate twice as much runoff in some locations as in others. But the economic value of mitigating an inch of runoff is much more variable, spanning several orders of magnitude. As noted before, a large proportion of Vermont farmland is at low elevations, and communities with the largest historical river-flood damages are relatively high in their watersheds. To examine variability of potential flood-control services, we use the method described by Watson and colleagues (2019)[11] to quantify spatial variability in the "demand" for flood-control services. This method attempts to quantify the relative value of mitigating the same amount of runoff from different locations[12]. By normalizing the resulting scores for agricultural

[11] We assign a score to each pixel in Vermont based on the number of downstream structures at risk of flooding. It is calculated for each pixel as:

$$F = \sum_{a=1}^{n} B_A / W_A$$

Where a is each flood-prone area downstream of the pixel, B is the number of buildings in the flood-prone area and W is the area of the upstream watershed of that flood-prone area.

[12] Intuitively, a gallon of runoff to a creek which flows directly into Lake Champlain contributes far less to flood damages than a gallon of runoff from the town of Orange into the Jail Branch of the Winooski River, which will pass by thousands of structures and dozens of miles of road before reaching the Lake. This method attempts to quantify this difference.

land, we keep the *average* value of flood mitigation services on agricultural land but weight the ES value by this flood control demand score.

Our results show that the relative values of flood-protection services from farm fields follow a fat-tailed distribution[13]: the "typical" farm field has a much lower value than the "average" one. While runoff from some farm fields endangers no structures at all[14], some fields sit high in the watershed, protecting many large settlements. If payments were apportioned based on downstream flood risk, these fields could be eligible for much larger payments for their reduction in potential runoff during large storms. These farm fields are largely located in the upper reaches of the Winooski River watershed, one of the few places in the state where a high concentration of farms is upstream from substantial infrastructure and people (Figure 11). Table 4 presents the range of Ecosystem Service presents potential ecosystem services valuations for farm fields under the "best" soil health scenario. These values conserve the *average* Ecosystem Service valuation of flood runoff mitigation.

Table 4: Distribution of Ecosystem Service Valuations for Flood Reduction from Soils with "Best" Improvement Levels.

ES Value ($/acre)	*% of Agricultural Area in Range*
< $0.25	36.6%
$0.25 - $1	27.3%
$1 - $2	12.2%
$2 - $5	13.4%
$5 - $10	4.5%
>$10	5.8%

[13] Our results roughly follow the 80-20 rule: about 80% of the protection values come from 20% of farm fields.

[14] On the other hand, mitigating runoff from some these fields is likely to be very important for protecting water quality, as discussed in the section on phosphorus.

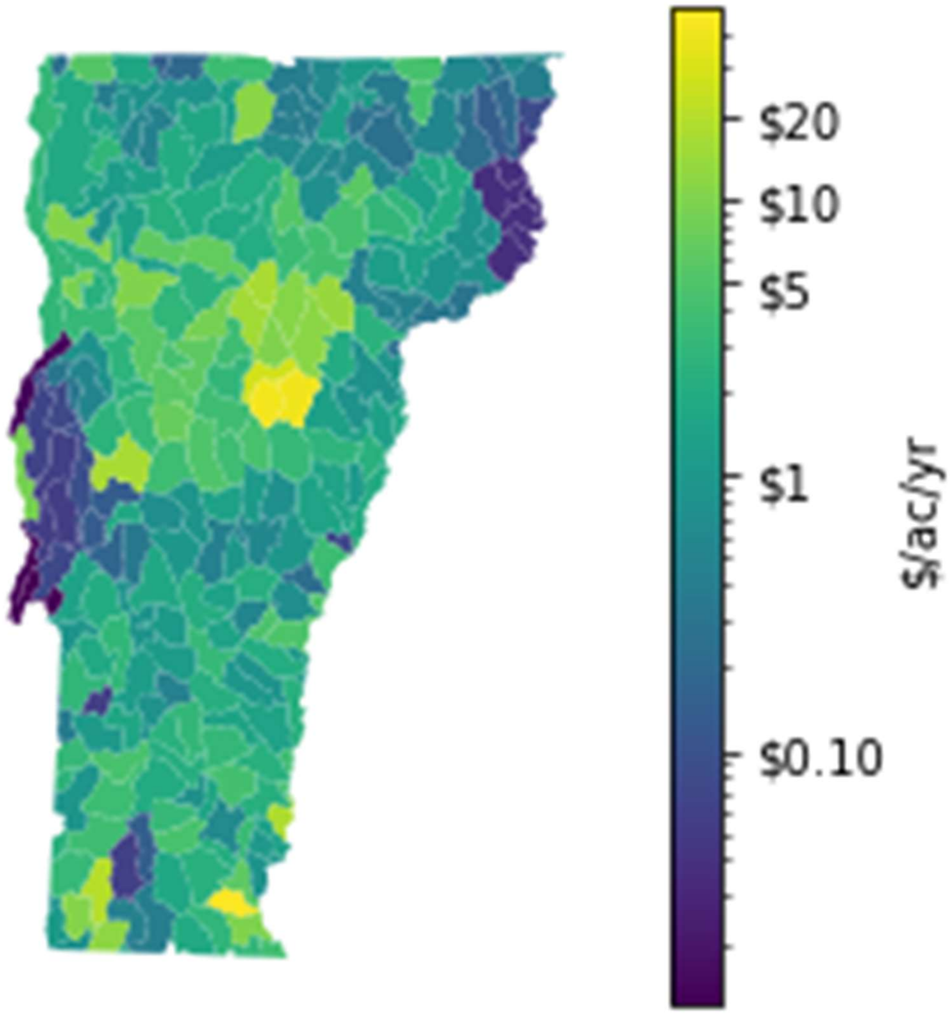

Figure 11: Ecosystem Service Value of Reducing Large-Storm Runoff from Agricultural Land by .3 inches (average for the "best" soil-health scenario.) Reducing runoff is much less valuable in areas near Lake Champlain, and much more valuable in the headwaters of the Winooski River.

Caveats and Areas for Future Work:

There are several weaknesses in our analysis, some of which may bias our estimates towards underestimating actual benefits, others which may bias them towards overestimating values. These are summarized in Table 5. Most important is the assumption of linear damages- some runoff-generating events do no damage at all, while many floods are subject to threshold effects, where a small increase in flow may cause dramatically greater damages. In addition, several types of damages may not be well-accounted for, including damages to natural capital and the economic and social costs of disruption while damaged infrastructure is un-usable.

A few factors may cause our estimates to be too high. First, some flood events occur when the soil is already complete saturated. This gives little opportunity for increased water-infiltration or holding capacity to mitigate runoff. Some of our methodological simplifications may also tilt the estimate upwards. For instance, our estimates of agricultural land-use in damaged towns' contributing watersheds are sometimes much higher than they should be to reflect the areas contributing the most to flood risk.

Edge-of-Field and Whole-Farm Interventions:

Several interventions that are not focused on soil health and are therefore outside the scope of this report are very important for flood mitigation. Overall land-use in agricultural basins is known to strongly influence stream channel flooding dynamics. Agricultural areas on well-connected floodplains provide critical opportunities to slow the movement of water and reduce storm peaks. These services may reduce downstream flood risk substantially. Other practices such as riparian buffers, constructed wetlands, artificial ponds and swales could increase infiltration, slowing and storage of floodwaters as well, and a PES program might pay for these services. Additionally, where agricultural lands are threatened by development pressures, agricultural land-cover provides substantial flood-control ecosystem services relative to developed land with substantial impervious surfaces.

Table 4: Major Sources of Uncertainty in Our Estimates of Flood Control Ecosystem Services

Factors That May Lead to Under-estimates	**Explanations/Examples**
Assumption of Linear Damages	Reducing floodwaters by 90% in many cases could eliminate 100% of damages. Given the small role of agriculture in the most disastrous floods, this is minor for "Generational Floods," but may be a larger issue for more minor flooding.
Social Costs of Infrastructure Disruptions	The costs of re-building a roadway are easy to quantify. The costs of that roadway being less usable while being rebuilt are less-so. Similar for power outages, etc. Irene was noted to cause disruptions to the crucial foliage tourism season.
Repair Costs of very minor floods.	Damages from frequent smaller floods cause damages to public infrastructure (e.g. dirt roads) that may be difficult to quantify.
Damages to Natural Capital	Flooding and fluvial erosion contribute substantially to many hard-to monetize damages from pollution. These include damages from erosion and nutrient deposition, as well as hazardous waste contamination.
Factors that May Lead to Over-estimates	
Many of the most damaging storms occur when soils are saturated.	Greater infiltration capacity gives little runoff-mitigation benefit when the soil is already saturated. Our estimates for increases in infiltration are based on soil available water capacity being 70% filled.
Town watersheds incorporate all areas upstream, sometimes overestimating the importance of agricultural landcover.	Often, small waterways with very low agricultural landcover cause a large proportion of damages. For instance, the Cold River (<2% ag landcover), accounted for a large proportion of Irene damages to Clarendon and Rutland. The total upstream agricultural landcover for both towns, which is what is used in the analysis, is >7.5%[15].
Simulating Runoff only for Large Storms	For smaller storms, the % of runoff averted by soil health is greater, but the absolute quantity will be smaller. For soil-health practices, the curve-number method is known to underestimate runoff in severe storms, leading to higher estimates of mitigation values.

[15] Similarly, most damage in the town of Hartford (~8.9% agriculture in its watershed) occurred in the Village of Quechee on the Ottauquechee River, which has less than half the upstream agricultural landcover (~3.7%). In a non-Irene example, severe flooding in Bellows Falls (Rockingham VT, 6.5% Agriculture in its watershed) in 2021 was due to the Hyde Hill Brook, which appears to have no agriculture in its watershed.

EROSION

While soil erosion is often thought of a direct threat to agricultural sustainability and productivity[16], it is also associated with many off-site environmental harms. One of the largest of these harms is the contribution of nutrients in eroded soil to freshwater eutrophication, which is covered in the Phosphorus section of this report. These costs include stream and reservoir sedimentation, which can reduce recreational value, harm wildlife and fish, increase flood risks and reduce the working life of dams.

Valuing Soil Erosion:

For soil-erosion impacts, we use a simple "value-transfer" method- we use other researchers' estimates of damage costs. Hansen and Ribaudo (2008) estimated off-site harms from erosion for every county in the United States. We exclude freshwater water-quality impacts, which should mostly be reflected in the next section on phosphorus. The number includes increases in water-treatment costs and damages to flood-control structures, farm ditches and marine fisheries. Their estimates for the 14 counties of Vermont range from $7.26 - $7.69 /ton of eroded sediment for an average of $7.38/ton in year 2000 dollars or $11.20/ton in 2020 dollars.

Hansen and Ribaudo's estimates are more geographically precise, but their estimate of average social costs of erosion for the whole United States are similar to several other estimates. Their estimate is $5.63 (USD2020) / ton while at least 3 other researchers found values between $5 and $6 per ton (Campbell, 2018; Pimentel et al., 1995; Uri, 2001). Social costs of erosion are substantially higher on a per-ton basis in Vermont and the rest of the northeast than in most other parts of the United States.

Estimating Physical Quantities:

The Universal Soil Loss Equation (USLE) is a family of simple models used to estimate soil erosion losses from farm fields. One of the parameters of USLE relates directly to soil properties, the soil erodibility or "K" factor. Wischmeier and colleagues developed an equation linking soil texture, organic matter and saturated hydraulic conductivity to the K factor (Wischmeier et al., 1971)[17]. We use this equation to estimate the impacts of soil health changes on soil erosion, using a family of reference scenarios for the other USLE parameters. Likewise, for soil-health practices,

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[16] For on-farm values of erosion control, we can consider the cost of replacing organic matter lost in eroded soil. There are roughly 400 lbs of organic matter in a cubic yard of compost. If the eroded topsoil contains about 4% organic matter, then replacing organic matter requires roughly 1 ton of compost for each 5 tons of topsoil lost.

[17] The Wischmeier equation is the default option for calculating the K factor in SWAT. Another popular option is the equation developed for EPIC/APEX by Williams (1995). The Wischmeier equation is chosen because it incorporates two soil-health parameters (Organic Matter and Saturated Hydraulic Conductivity), while the Williams equation incorporates only Organic Matter. The Wischmeier method also covers greater range of soil organic matter concentrations than the Williams method.

we alter the "C" or crop-cover factor of USLE to develop estimates of changes in erosion losses with practice changes.

Further details on these methods, including limitations, can be found in Appendix 4.

Results:

Figure 12 summarizes the reduction in soil erosion from changing practices from the reference case of conventional corn. The "hay" scenario covers all perennial forages, including rotational hay, permanent hay and permanent pasture. Figure 13 summarizes reductions in erosion from improved soil health.

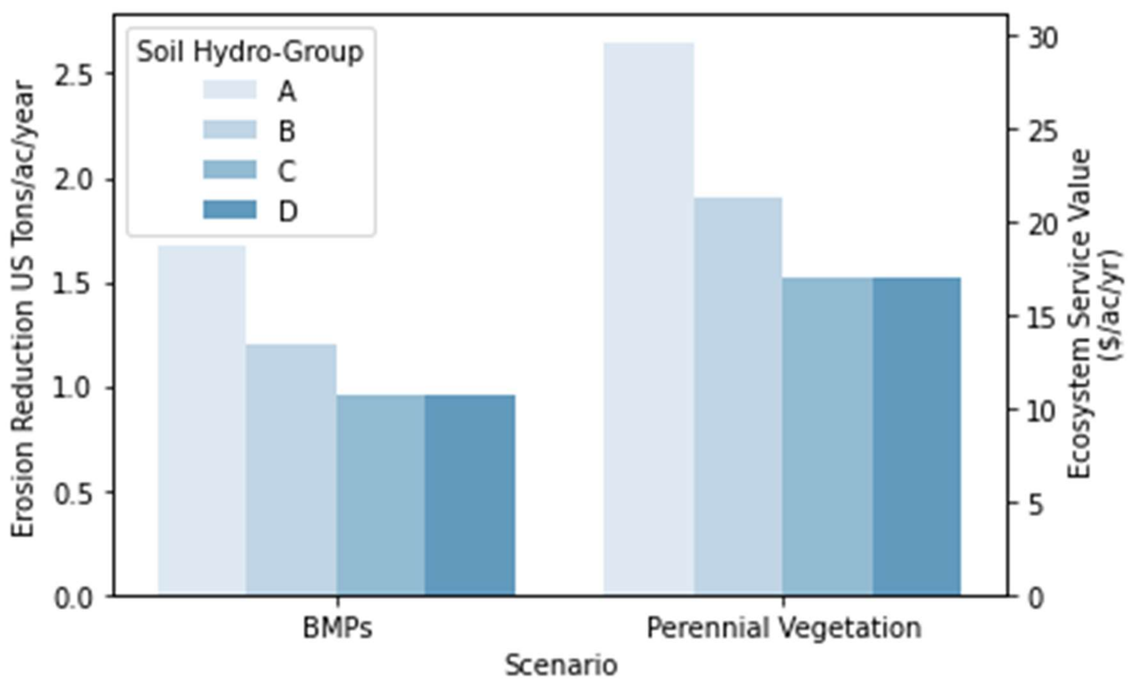

Figure 12: Predicted reductions in Erosion for Soil Health Practices and Ecosystem Service Value. Left axis reports predicted changes in erosion, and right axis reports the economic value of these changes.

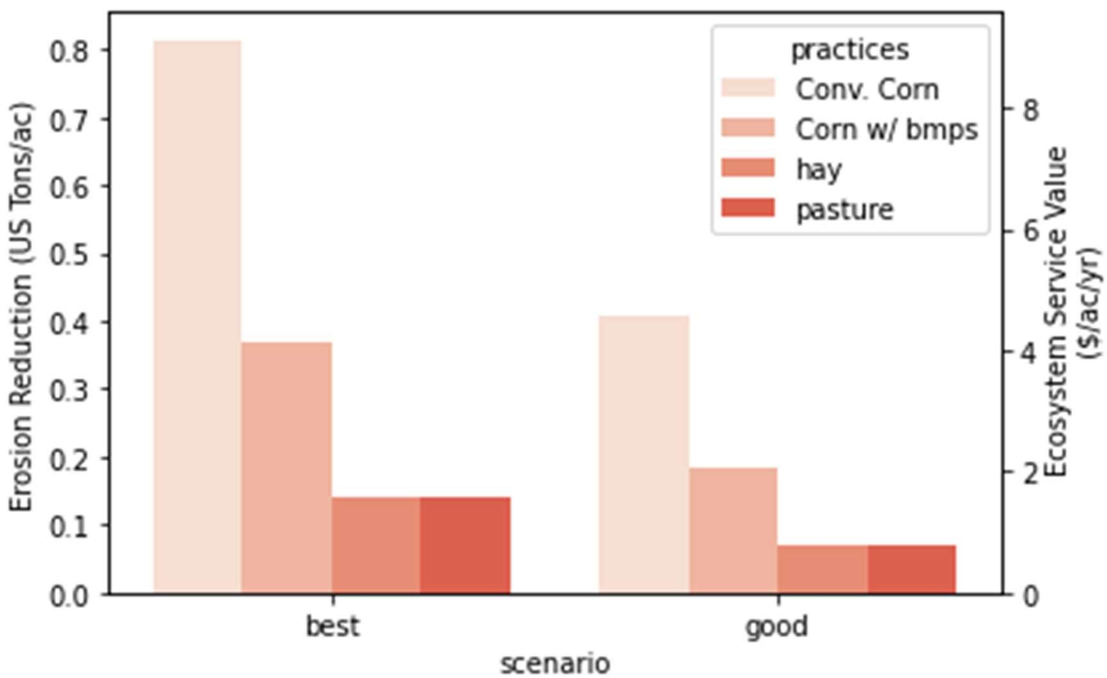

Figure 13: Predicted Reductions in Erosion for Soil-Health Indicator Scenarios and Ecosystem Service Value. Left axis reports predicted changes in erosion, and right axis reports the economic value of these changes.

Sources of Variation:

The value of erosion reduction services from healthy soil is higher on fields with steeper slopes, and higher on fields growing annual crops than those with perennial vegetation. We expect the same soil-health improvements to have similar percentage impacts on soil erosion, making the economic value much larger on fields that have high potential for erosion losses. The spatial variability in the value of damages done by a ton of eroded sediment is likely important, but not explored in this study.

Edge-of-Field and Whole-Farm Interventions:

Riparian buffer zones and other practices which can intercept eroded sediment before it enters waterways can greatly reduce the downstream damages of erosion. Likewise, substantial quantities of sediment can be generated by streambank erosion, which can be mitigated by bank stabilization practices, as well as any practices that reduce flooding as discussed elsewhere in this report. A PES program might consider paying for these services as well.

NUTRIENT RETENTION: PHOSPHORUS

Phosphorus enrichment is the largest source of freshwater eutrophication globally, and agriculture is the largest contributor. In Vermont Lake Champlain and Lake Memphremagog and several smaller waterbodies have impaired water quality due to phosphorus from agriculture. In Lake Champlain, numerous cyanobacteria blooms have degraded water quality, causing major economic, quality-of-life, and health impacts on the people living near the lake. Healthy soils and some soil-health related practices may be helpful for retaining phosphorus on farm fields and keeping it out of freshwater bodies.

Valuing Phosphorus Damages:

We estimate the damage from Phosphorus loading to Lake Champlain by roughly scaling up the work of Gourevitch et al (2021) on costs and benefits of P reductions in the Missisquoi Bay watershed. Their work combines an integrated assessment model (IAM) which links P loading to phosphorus and chlorophyll-a levels in the bay, and econometric and epidemiological models linking Chl-a levels to home sales, tourism expenditures and cases of Amyotrophic Lateral Sclerosis (ALS)[18]. While this paper does not report a "social cost of phosphorus," the annual benefits of meeting the TMDL are calculated at $2 million / year by 2050. This gives an average benefit of $10.35/lb of P mitigated. We scale this number up in two different ways and take the average of the two methods.

In the first method, we assume that economic damages from poor water-quality are linearly proportional to the economic activity in nearby areas, approximate by the number of people living within 20 km of a waterbody multiplied by the average income[19]. Further, we assume it is related to the percentage exceedance of the TMDL target. We estimate the marginal benefit curve of P reductions relative to exceedance of the TMDL using the 6 different scenarios examined by Gourevitch and colleagues and find a log-log relationship. We use data on the scale of required P mitigations under the TMDL (US EPA, 2016) and population data from the US Census. Using this, we estimate total benefits annual from meeting the TMDL, and divide these by required reductions.

This method yields an average damages of $30.42/lb of P from agriculture across Lake segments. Damages range from $6.35/lb for Otter Creek, $10.35/lb for Missisquoi Bay, to $678.83 / lb for Burlington Bay. Missisquoi Bay has large overshoot of its TMDL and the area around it is economically depressed and sparsely populated, while about 100,000 people live near Burlington Bay, including some the state's wealthiest communities.

[18] The causual linkage between ALS and cyanotoxins is still controversial.

[19] In Gourevitch et al, ALS cases are a relatively small proportion (10-15%) of monetized damages. Scaling these by income is obviously inappropriate, but damages to home prices and tourism, should scale to levels of economic activity moreso than population. On the other hand, lower-income communities may have a harder time adapting to poor water quality, and the public may have a higher willingness to pay to mitigate harms on them.

In the second method, we use estimates from Voigt et al (2015) for impacts on tourism revenues for the whole lake. Voigt et al (2015) used a series of regression models to estimate the impact of P load on water clarity (Secchi depth), and water clarity on property valuation, tourism expenditures, and regional economic activity. In their model, a 34% reduction in lake-wide mean total phosphorus concentration (corresponding to a 34% reduction in phosphorus load to meet the Vermont TMDL) would increase Secchi depth by 1.67 meters, and a 1-meter increase in Secchi depth across the lake is worth $12.6 million/year in tourism expenditures. Given the phosphorus TMDL for Lake Champlain Vermont reduction target of 234.7 US tons of phosphorus per year, this implies an average benefit of ~$45 (USD 2020) / lb of P-load reduction in tourism expenditures alone. Increasing this to reflect that Gourevitch et al estimate tourism losses as 54% of total economic damages, this yields a social cost of phosphorus of $82.72 / lb.

The average of these two values: $87.72 and $30.42, is $56.60 /lb.

Additionally, we use data from Beaulieu et al (2019) to estimate the impact of meeting the TMDL on methane emissions in Lake Champlain. They estimate that reducing P levels by 25%[20] in all global lakes with similar size and TP levels to Lake Champlain would reduce global annual methane emissions by 129000 metric tons/year. Lake Champlain's proportion of this is 3714 metric tons of methane. At a carbon-offset adjusted price of the social cost of methane, this yields over $1.8 million in annual benefits, or $3.97 / lb of P loading reduced.

Adding in this value gives $60.56 / lb of P for the Champlain Basin. We value P in the Lake Memphremagog Basin, which is also severely impaired, at the average level for Lake Champlain. About 25% of VT farmland is outside the Lake Champlain and Memphremagog Basins. We assign these areas the value for the Missisquoi Bay basin, $14.37 /lb.

This yields an average valuation of $56.82/lb. This estimate is highly imprecise, and is not exhaustive of harms done by eutrophication of freshwater bodies in VT. Not included in this analysis are the "consumer surplus" from tourism/recreational activities, above the increased spending at local businesses, other health benefits from clean water, reduced costs for treatment of drinking water and reductions in risks of catastrophic changes in the ecology of Lake Champlain. We are not able to estimate how movement of Phosphorus between different Lake segments, rather than treating segments as distinct waterbodies, might impact the valuations given.

More details on this valuation can be found in Appendix 5.

[20] The TMDL requires that P loading be reduced by 33% across the entire lake.

Estimating Physical Quantities:

To estimate reductions in P losses, we use the VT P Index (Jokela, 1999), a spreadsheet-based model used by farmers for nutrient management planning. The VT P Index includes the soil-health practice scenarios we investigate here, so these are directly simulated. The results presented average over a family of reference scenarios for innate site characteristics (slope, distance to water, soil type).

We were able to incorporate changes in soil health indicators in two ways. First, the P Index requires an erosion rate, for this we utilize the impacts on erosion losses developed previously. Second, we simulate the impacts on runoff across a wide variety of storms using the same methods as described in the section on flooding, to estimate how soil health reduces growing-season runoff, and therefore P losses in that runoff. The results presented average over reference scenarios for management parameters; which are conventional corn and the other soil-health practice scenarios.

Further details for these methods can be found in Appendix 5.

Results:

Figure 14 shows the estimated reductions in P losses for practice changes, relative to conventional corn. The corn best-management practices are simulated to have large impacts on reducing phosphorus levels. These BMPs were designed for P mitigation, so this result is unsurprising. Converting to perennial vegetation, such as hay, is modelled to have smaller benefits, and benefits that decrease with soil drainage, likely due to manure being spread on the soil surface.

Figure 15 shows our results for the soil improvement scenarios. Soil health improvements can have substantial impacts on P losses, especially from conventional corn. Soil health improvements have a smaller benefit for perennial vegetation, where P losses are lower to begin with.

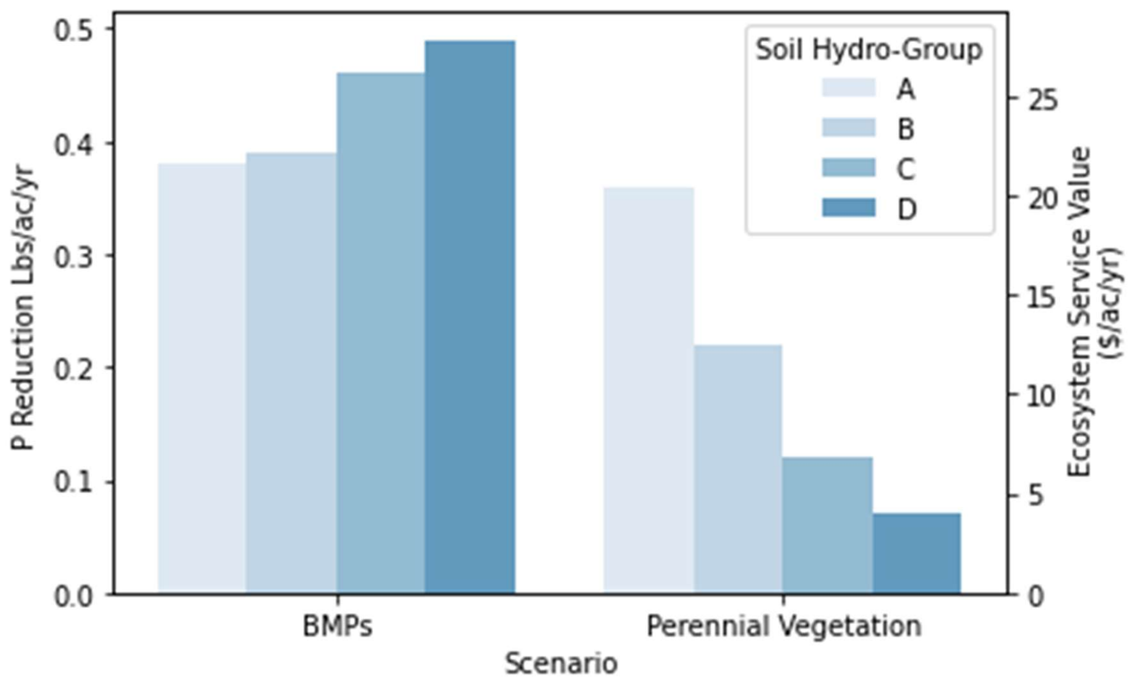

Figure 14: Reductions in P Losses for Soil Health Practices Scenarios and Ecosystem Service Value. Left axis reports predicted changes in phosphorus loading, and right axis reports the economic value of these changes.

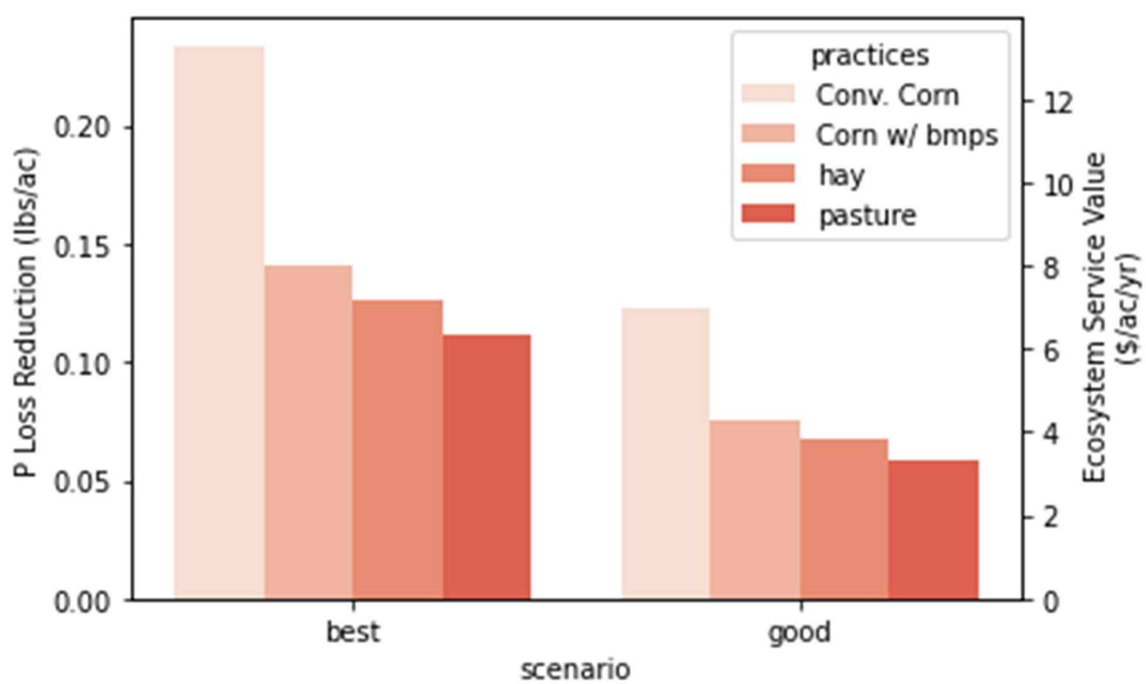

Figure 15: Reductions in P Losses for Soil Health Indicators Scenarios and Ecosystem Service Value. Left axis reports predicted changes in phosphorus loading, and right axis reports the economic value of these changes.

Sources of Variation in Service Value:

Improved soil health can reduce erosion and can reduce runoff, which are two important pathways for Phosphorus losses from farm fields. All else equal, we should expect reductions erosion and runoff to be proportional to P losses from erosion and runoff. As noted above, these reductions in P loss may be largely or fully offset by increased subsurface losses of P, on fields with substantial connections to waterways via subsurface drainage. Similar to erosion-control, the quantity of P-retention services provided by healthy soils is proportional to the field's potential to lose Phosphorus. Healthy soils provide a greater benefit in P reduction on fields growing annual crops, on steeper slopes, closer to waterways. Therefore, a large increase in soil health has a smaller value if other P-conserving practices are already implemented.

Beyond this analysis, most important soil-health indicator for P loss is **soil test phosphorus**. High soil-test phosphorus levels make it extremely difficult to keep P losses from farm fields to acceptable levels.

The largest source of variation in the value of P retention services is location in a sub-watershed. P retention is much more valuable in some sub-watersheds of the Lake Champlain Basin than others, and is generally more valuable in the Lake Champlain basin than outside of it, though this may be variable based on smaller impaired waterbodies throughout the state[21]. The variation in values is driven by the potential of downstream waterbodies to become eutrophic from Phosphorus loading, and the scale of human uses of those waterbodies. It may be even more valuable in specific sub-watersheds flowing into highly impaired lakes and ponds.

Caveats and Areas for Future Work:

Soil health metrics, and soil health practices can be effectively linked to expected reductions in erosion and runoff, nutrient losses through these pathways are proportional to these quantities, holding all else equal. Greater water infiltration may, however, increase nutrient losses downward through the soil profile, which may be especially harmful in soils with pattern tile drainage, or other direct connections to waterways via subsurface flow (Duncan et al., 2019).

Our results treat all phosphorus equally, and this assumption is untrue. Generally, soil health improvements and practices are more effective a reducing sediment-bound Phosphorus, than dissolved Phosphorus. Given that sediment-bound phosphorus is less bio-available to algae than dissolved Phosphorus, the monetary valuation for sediment bound phosphorus should be lower, and that for dissolve phosphorus should be higher.

[21]Other than Lake Champlain, there are 8 waterbodies that are either declared impaired by P and/or have had a TMDL drawn up for P since 2001. Two of these waterbodies: Ticklenacked Pond in Ryegate, and the Black River, are outside of the Champlain or Memphremagog Basins.

Our valuation of phosphorus loading is substantially lower than the State's demonstrated willingness-to-pay for phosphorus reductions from agriculture. Currently, the Pay-for-Phosphorus program pays farmers $100/lb of phosphorus load reduction and pays even more for reductions in loading from other sources. The State of Vermont is legally required to meet the TMDL, even if doing so creates more monetary costs than benefits. Using costs or payment rates for other ways of reducing phosphorus loading would result in a higher, and possibly more realistic number.

These results should be interpreted with caution. The estimates for soil-health practices are made purely using the Vermont P Index, a tool that, despite uncertainties, is widely used for communicating with and regulating farmers around water quality issues.

Edge-of-Field and Whole-Farm Interventions:

As with other services, there are several practices that contribute greatly to reducing P loads and could be incorporated into a broader PES program.

OTHER ECOSYSTEM SERVICES

Nitrogen:

There are several types of N losses from agriculture which harm ecosystems and human health through a variety of pathways. Gaseous losses, including ammonia, nitric oxides and nitrogen dioxide contribute to acidification of water and soil, and can damage air quality both directly and through their impacts on particulate formation. Water-borne losses of nitrate, including leaching and runoff, can damage drinking water resources and contribute to eutrophication of marine ecosystems. Nitrogen lost from the soil can also change form after leaving the soil - nitrate in runoff will eventually be denitrified and turn into N_2O, NO or NO_2, while some gaseous emissions will be deposited in soils that they may subsequently leach from.

Valuing N Losses:

The spatial complexity of N emissions and their harms calls for a full study of its own, but Table 6 summarizes best-estimates of the average economic harms done by different pathways of reactive nitrogen emissions in the United States. Note that some of these, such as respiratory disease, may have much smaller impacts in VT, which has low population density and few population centers downwind.

Table 5: Average US Values for Damage costs from Different types of Nitrogen Emissions, based on Sobota et al (2015).

N Loss Pathway	*Damage Valuation per Lb of N*	*Largest component*	*Notes*
NO_x	\$15.88	Respiratory Disease (79%)	Beneficial for climate
NH_3	\$6.07	Ecosystem Change (69%)	Beneficial for climate
N_2O	\$6.87	Climate Change (79%)	Climate number from (Marten & Newbold, 2012), adjusted down for offset price.
Surface freshwater	\$10.33	Eutrophication (85%)	
Groundwater	\$1.33	Colon Cancer (72%)	
Costal Water	\$12.12	Fisheries (71%)	

In estimating damages from nitrogen leaching, we take the weighted average of groundwater, costal water and surface freshwater, weighting groundwater at 80%, and the others at 10%. This yields \$3.31/lb of nitrogen.

Impacts of Soil Health on Nitrogen Losses:

To estimate impacts of soil health changes on losses of nitrogen, we parameterize the DayCent model (Parton et al., 1998) on the "current-practices" management activities for Hay and Corn used in recent agri-environmental field trials in Vermont (White et al., 2021) and the simulated soils that we generated. We use weather data from Burlington International Airport for the years 2012 - 2021   Simulated impacts of soil health on gaseous nitrogen (and methane) losses are relatively small, and highly variable. Figure 16 presents a summary of impacts of different scenarios on nitrogen and methane flows, presented in dollar terms.

In the large majority of crop-soil pairs, greater soil health resulted in larger losses of NO-N and methane, and somewhat smaller losses of N2O-N. Median values were at or below .2 lbs/acre/year, and on net correspond to roughly $3-4/acre/year in *damages* from increased soil health.

Impacts on nitrogen leaching were the largest, and also the most variable. The 'best' soil health scenario generally results in higher losses of nitrogen through leaching, this was the case for 71% of the soil-year pairs for corn, and 92% for hay. Median increases in nitrogen loss through leaching were 1.47 lbs/acre/year for corn, and 3.53 lbs/acre/year for hay, differences valued at $5 - $12/acre/year in *damages* from improved soil health.

Much of these additional N losses through leaching may be due to additional nitrogen being mineralized from soil organic matter. If farmers account for additional N from OM mineralization in their nutrient management planning, this impact may disappear. If the farmers in the 'best' soil-health scenario respond by reducing manure applications by 10%, then NOx losses are unchanged from the baseline scenario, while modelled nitrate leaching losses decrease from baseline, yielding small net benefits, rather than damages.

Daycent lacks the capacity to model surface runoff losses of dissolved nitrogen. In recent experiments, surface runoff of dissolved nitrogen from Vermont crop fields was on the order of 1 lb per acre (White et al., 2021). Given the valuations presented in Table 6, this places an upper bound of a few dollars per acre on the value of soil-health improvements for reducing nitrogen losses through this pathway.

Impacts of Practices on Nitrogen Losses:

Cover-crops show substantial reductions in loses of nitrogen through leaching; a median of nearly 5 lbs per acre, valued as a monetary benefit of over $30/acre. Impacts of cover crops on N loss through other pathways were beneficial, but very small. Manure injection, another aspect of the corn BMPs scenario may result in substantial increases in N losses, especially from leaching and from nitrous oxide (Barbieri, 2021). As currently configured, DayCent is unable to simulate manure injection.

Given these relatively small and variable values and the high uncertainty, we do not include Nitrogen and trace gases in our valuation. It is possible that this method over-estimates nitrogen losses from higher soil-health scenarios, as actual farmers often account for the increased N mineralization from organic matter in their nutrient planning and apply less N to their fields in manure and fertilizer.

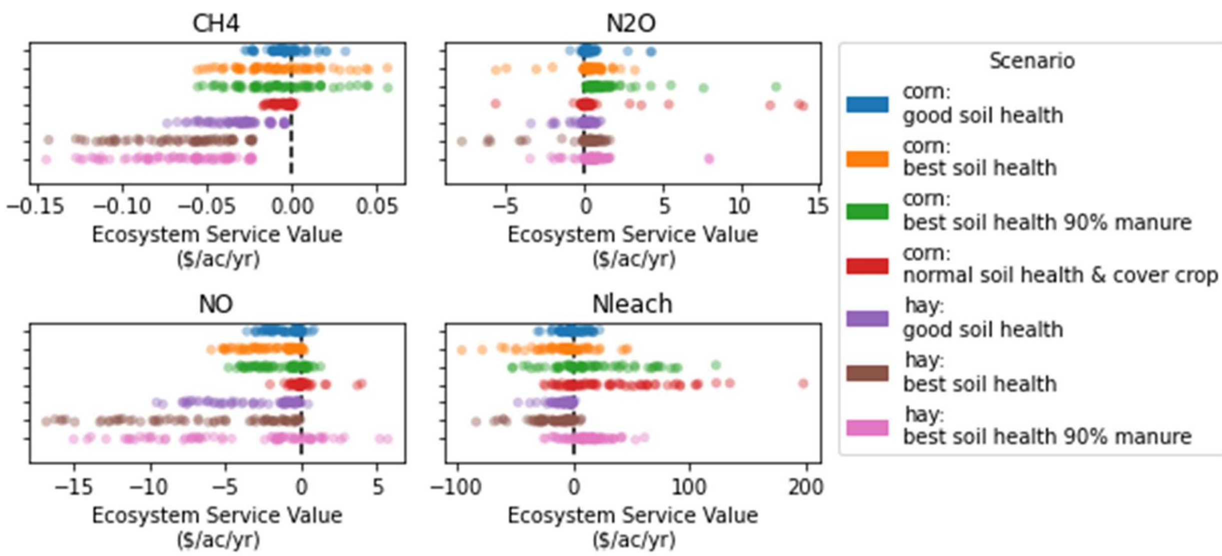

Figure 16: Value of Net Ecosystem Service Benefits from Changes Trace Gases and Nitrogen Leaching (Base Cases: Corn, normal Soil Health; Hay, normal soil health). Negative values indicate environmental damages from soil-health improvements, each dot represents a simulation of one soil series for one year. For all losses other than leaching, impacts are small, and highly variable.

Soil Biodiversity:

Several options exist for valuing soil biodiversity, though none of these are feasible within the scope of this study. There are 3 general types of values contributed by soil biodiversity. First, soil biodiversity is linked to supporting ecosystem services including nutrient cycling, predation, and soil aggregation, which may enhance other ecosystem services, including crop production and the services discussed in this paper. Second, soil biodiversity may have insurance value: soil biodiversity may enhance the resilience and stability of important soil ecosystem services. Lastly, soil biodiversity may have existence value, the people in Vermont may derive economic value from knowing that their soils are biodiverse, regardless of any direct impacts on human-wellbeing.

The first two types of value are important questions, but too little research exists to conduct a meaningful valuation of changes in soil biodiversity; no available models can link a unit-change in soil biodiversity with a unit-change in soil resilience. For existence value, stated-preference methods, such as contingent valuation surveys could be used to understand Vermonter's willingness-to-pay to improve soil biodiversity, but these methods may be unreliable for something so abstract. It would be hard, for instance, to ensure that respondents do not include any impacts on the other services examined in this report in their willingness to pay; if they did, this would result in double-counting.

Conclusion and Next Steps

In this report, we estimate the levels and values of 4 ecosystem services promoted by healthy soils and by soil-health practices. We show that the public values of these services are of reasonable size and may justify a program for payments for Ecosystem Services. While these estimates are necessarily rough, they also can provide general guidance to understanding the sources of variability in these values and their relative magnitudes.

Several areas require further work to better understand. First, better estimates of Nitrogen may be quite valuable - the relative magnitudes of benefits from reducing N losses look to be substantial. Second, estimates of the benefits from edge-of-field practices and other non-soil-health practices may also be useful. For example, it is likely that re-establishing riparian forest would have similar or greater per-acre benefits for all four of these ecosystem services than any soil-health practice or improvement[22]. Third, further research could refine the estimates of the dollar values of other Ecosystem Services. For all of the services included, the estimates that we provide for their dollar values are preliminary and would benefit from refinement.

Two areas could use deeper examination in particular. First, our valuation of phosphorus is both crude and leaves out several important harms of impaired water-quality. Better understanding these economic harms could help identify clean water beneficiaries and identify revenue sources and win-win solutions. Second, more work should be done to understand the impacts of upstream landscapes on flood resilience further downstream. Beyond the role played by agriculture and soil-health, identifying the highest-value locations and practices for flood mitigation will become increasingly important as Vermont becomes warmer and wetter.

The science on the ecosystem services from healthy soil is still in its infancy. The science linking sustainable and regenerative agriculture practices to soil health increases and ecosystems services is also new and sparse. While new research will continue to refine our understanding, the estimates provided here can guide the creation of policy with the information we have today.

[22] For instance, two recent studies (Gourevitch et al., 2020, 2022) find very large impacts from floodplain forest restoration on flood risks downstream, aboveground forest carbon storage in the Northeast exceeds 30/T acre (Heath et al., 2002) and buffer zones along agricultural fields are highly effective at reducing sediment and nutrient loading (Yuan et al., 2009).

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Technical Appendices: Valuation of Ecosystem Services from Soil Health.

Contents

Appendix 1: Scenario Development:

Practices:

The practices scenarios are derived from the list used to inform the Vermont Payment for Ecosystem Services Technical Research Report #2. These include 1) no till and cover cropped corn, 2) corn in rotation with hay, 3) transition to perennial pasture, 4) cover cropping in vegetable production, and 5) hay.

Indicators:

The % increases for soil health scenarios were partly chosen from a desire for clean, round numbers, and as such, are somewhat arbitrary. The "good" scenario represents levels of soil health differences that are often seen in long-term field experiments comparing conventional and best-management practices. In a review of long-term experiments, Crystal-Ornelas et al (2021) found that using best-management practices on organic farms increases SOC levels by an average of 14-24% compared to organic farming without these practices.

For the best scenario, we wanted to display a high bar that ambitious regenerative farmers believe that they can meet. According to data from the UVM soil testing laboratory, about 20% of commercial farm samples have SOC levels at least 50% higher than the median level for the state. This level of increase in soil organic carbon is also aligned with ambitious targets and claims by researchers and farmers in the regenerative agriculture community. For instance, the an analysis by Drawdown (Toensmeier et al., 2020) estimates that regenerative agriculture strategies on annual cropland in humid-temperate climates could sequester 7-13 tons C/acre[1] before soil carbon stops accumulating, and that managed grazing could sequester even more.

For Bulk Density, the regression model developed by Ruehlmann & Körschens (2009) predicts that the increases in soil organic carbon simulated for the scenarios would result in a 5% and a 10% reduction in bulk density respectively, due to the favorable impacts of organic matter on soil structure. This level is doubled to account for favorable impacts on soil structure from other changes.

[1]These numbers are reported as .6 Mt / ha /year for 25 to 50 years. Our "best" scenario is approximately 13 tons C / acre.

Appendix 2: Soil Carbon

Biophysical Quantities.

For soil-health indicators, the changes in carbon are assumed.

For soil health scenarios, the following data sources were used:

Scenario	Data Source	Notes	Link
Corn BMPS	Integrating Cover Crops and Manure into Corn Silage Cropping Systems		Link
Corn-Hay	Corn Cropping Systems to Improve Economic and Environmental Health		Link
Hay	Corn Cropping Systems to Improve Economic and Environmental Health		Link
Pasture	Corn Cropping Systems to Improve Economic and Environmental Health	Used Value for converting to Hay	Link
Vegetable BMPs	Evaluation of commercial soil health tests using a medium-term cover crop experiment in a humid, temperate climate (Chahal & Van Eerd, 2018)	Average of cover-crop scenarios.	Link

Table S 1: Data sources for Soil Carbon Accumulation in Practices Scenarios

Valuation:

For soil indicators, we value the climate regulation services of storing 1 ton of carbon in soil for 1 year. This approach makes the valuation comparable to the valuations of other ecosystem services, which are valued as yearly flows of benefits. The Social Cost of Carbon methodology gives a present *all* future costs and benefits of carbon stock changes It also avoids the possibility of large negative payments to farmers who are doing a reasonable job stewarding soil health.

We use two methods to estimate annual benefits of carbon storage.

In the first method, we use the "social cost of radiative forcing" as described by Rautiainen and Lintunen (2017), which describes the social cost of an additional unit of global warming in a given year. In their appendix A, Rautiainen and Lintunen estimate the social cost of radiative forcing as $358/nW/m^2. A ton of CO_2 increases radiative forcing by an average of .001476 nW/m^2 during the first 5 years after emission (Levasseur et al., 2010), which is a plausible re-sampling interval for a soil-carbon program. This gives $0.53/Metric Ton CO_2/year. Converting to imperial tons of carbon yields $1.76/ton/year, which we adjust downwards by 25% to account for the difference between the Social Cost of Carbon calculated by that study ($20/ton) and the $15/ton offset price used in this study. This yields $1.32/Ton SOC/year.

In the 2nd method, we use the social cost of carbon calculated by the EPA, and calculate the annual payment (in perpetuity) that has an equivalent net-present value at discount rate used. A lump-sum payment of the social cost of carbon of $51 is worth the same, at a 3% discount rate, as an infinite series of payments of $1.58/year. We adjust this downwards by the ratio of the social cost of carbon to the reference offset price ($15) and upwards to convert from metric tons CO2 to imperial tons carbon, yielding $1.55/T SOC.

Additional Issues:

Difficulties in Using the "raw" social cost of carbon:

A .1% change in soil organic carbon content corresponds with about .75 tons of carbon per acre. At the $53/ton of SOC offset price, this is valued at $40/acre, at the current US Social Cost of Carbon, it is valued at nearly $140/acre.

Even on relatively small crop-fields, soil samples properly taken from multiple cores will have some variability. Data from the Cornell soil lab shows that the standard deviation of soil organic matter for a small field with homogenous management can range from .13 to .39 percentage points (R. Schindelbeck, personal communication, April 15, 2022). A standard deviation of .2 percentage points in organic matter corresponds to a 0.164 standard deviation in *differences in organic carbon* between two samples from identical fields. This would mean that 27% of the time, a 2nd successive sample of the same field, on the same day, would show at least a .1% decrease in SOC, and 10% of the time would show a decrease of at least .2% SOC.

If an annualized payment/valuation strategy is used, a decrease in soil carbon content of .2% would mean a reduction in payments of about $2.88/acre/year. If a lump-sum style offset/social cost of carbon valuation were used, it would result in a *negative valuation* of roughly $75/acre, spread across the number of years until the next sample.

Bulk Density and Measurement Error:

Despite the one-to-one linkage between Soil Organic Matter as a soil health indicator, and carbon storage as an ecosystem service, there are important complications in measuring soil carbon storage. These relate to the depth of measurement, and its relationship to soil bulk density. Soil organic carbon is usually measured to a reference depth, often 30 cm. If management of a soil results in substantial soil compaction, then more soil material ends up within 30 cm of the surface, increasing measured soil carbon storage, without increasing actual carbon storage (Figure S1). Lee and colleagues (2009) demonstrate these complications and recommend that changes in bulk density not be used to assess changes in carbon storage.

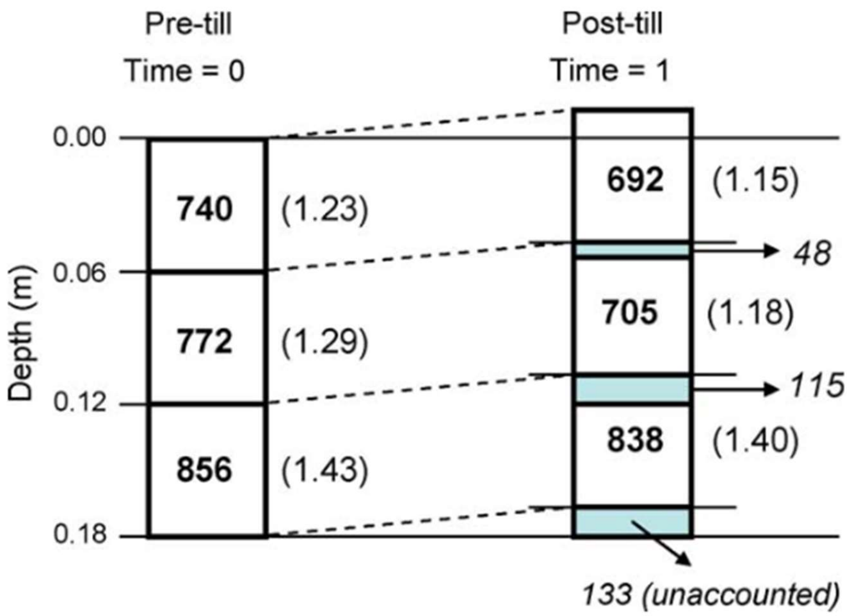

Figure S 1: Tillage decreases bulk density, expanding the volume that the soil layer takes up. Because of this expansion, some carbon is now below the depth of measurement. Figure from Lee et al (2009).

Appendix 3 Flood Mitigation:

Biophysical Quantities:

Practices:

Runoff for a 4-inch storm and a 1.5-inch storm is calculated for each land-use and the four soil drainage classes, using the NRCS curve number method. The curve number method is widely used, including as a component of basin-scale models such as SWAT and APEX. The NRCS curve number is a simple empirical model of rainfall infiltration curves.

Each combination of land-use, soil hydrologic class and practices was assigned a "Curve Number" from 30 to 100, based on decades of empirical research. The curve number is converted into a retention parameter, S through the following equation:

$$S = 25.4 \cdot \frac{1000}{CN} - 10$$

The rainfall-runoff curve is then calculated as:

$$Q = \begin{cases} \dfrac{(R - I_A)^2}{R - I_A + S} & if\ R > I_A \\ 0\ otherwise \end{cases}$$

Where I_A is normally set to $S \cdot .2$ and R is the rainfall for the day.

Indicators:

Runoff reductions for soil-health indicators are calculated as the average of two methods; the Green-Ampt equation, and additional water-holding capacity until saturation. For both, we assume that the soils start at 70% of their plant-available water-holding capacity.

For both methods, some additional parameters must be estimated first.

These include the saturated hydraulic conductivity and soil moisture contents at the permanent wilting point (ϕ_{pwp},) field capacity (ϕ_{fc},) and saturation (ϕ_s).

For these parameters, we utilize two different tools. First, we utilize the ROSETTA pedo-transfer function model from the USDA ARS (Zhang & Schaap, 2017), which calculates residual water content (ϕ_r), ϕ_s and Ksat, based on soil particle distribution (percents sand, silt, clay and organic matter) and bulk density and (optionally) $\phi_{fc}\ and\ \phi_{pwp}$. We also use the equations by Balland (2008) for calculating ϕ_{fc}:

$$\phi_{fc} = (.565 + .426 \cdot clay^{.5}) \cdot exp(-1 \cdot (.103 \cdot sand - .785 \cdot \frac{OM}{\phi_s/d_b}))$$

And ϕ_{pwp}:

$$\phi_{pwp} = \phi_{fc} \cdot (.17 + (.662 \cdot clay^{.5}) \cdot exp(\frac{1.4 \cdot OM}{\phi_{fc} \cdot -1})$$

For both equations, soil composition factors are fractions, not percentages, (0-1 rather than 0-100) and water contents are calculated on a weight, rather than volumetric basis.

Our estimates for the soil water parameters are made by first calculating ϕ_s using the Rosetta model and soil particle distribution and bulk density. The result for ϕ_s is used for the Balland field-capacity equation, whose result is used for the Balland permanent wiling point equation. The process is then repeated several times, with the values for ϕ_{fc} and ϕ_{pwp} generated by the Balland equations used to parameterize the Rosetta model. After five cycles, the final values are used.

Soil pore space is derived in 2 steps:

Particle density is calculated as (Schjønning et al., 2017):

$$d_p = 2.652 + .216 \cdot clay - 2.237 \cdot OM$$

Where OM and clay are reported as fractions.

And porosity is calculated as:

$$p = 1 - \frac{d_b}{d_p}$$

Where d_b is bulk density. This simply means that all space not taken up by particles is pore space and pore space has 0 dry-weight.

An additional parameter needed for the green-ampt equation is soil matric potential, the strength with which a soil holds the water. This is calculated using the equation developed by (Rawls & Brakensiek (1985) as described in the technical documentation of SWAT, page 109.

These results are used to parameterize the Green-Ampt Equation. We use a version with 3 unique soil layers. A general description of the Green-Ampt method can be found here. We simulate runoff for a 4-inch storm over the course of 6 hours for the "generational storm" and a 1.5 inch storm over 3 hours for 10-25 year return intervals.

Valuation:

Our estimate of the relative impacts of smaller floods vs “generational storms” in Vermont’s flood risks are intermediate between the story told by available data on past damages and simulation modelling conducted by Gourevitch and colleagues (2022). The available data shows a “fat-tailed” distribution of flooding events: a majority of flood damages are attributed to a small number of extreme storms. Gourevitch and colleagues show the opposite: over 2/3 of modelled damages come from floods with a modelled return period of 10 years or less, and about half of modelled damages are from floods with a return period of 2 years.

The historical data show that rare, extreme flooding events account for the majority of flooding damages to buildings and property (Figure S*). Tropical Storm Irene accounts for 70% of all National Flood Insurance Program payouts for non-winter flooding in VT since 1976[2] (Federal Emergency Management Agency., 2021a). Given that Irene caused severe damages outside of mapped flood zones and through landslides not covered by the NFIP, this proportion may be an underestimate of its contribution to historical flood-damages. Similarly, 71% of all flood-related payments from the USDA Crop Insurance Program since 1988 were made for damages caused by Irene (Risk Management Agency, 2021). 89% of all FEMA-assessed damage to VT homes since 2002 was associated with Irene (Federal Emergency Management Agency., 2021b). Between 65% and 91% of FEMA grants associated with flooding made to Vermont communities since 1998 were associated with Tropical Storm Irene (Federal Emergency Management Agency., 2021c)[3]. Additionally, most smaller flood events have been due to storms that featured extreme rains (>3 inches) on a more localized basis (VT Emergency Management, 2018).

[2] We would expect soil-health to have very little impact on winter flood damages from ice-dams and snowmelt, though other agricultural management practices might have an impact.

[3] This very wide range is due to the “Severe Storm” categorization – a significant proportion of damages from “severe storms” can be due to wind and ice, but much is due to flooding.

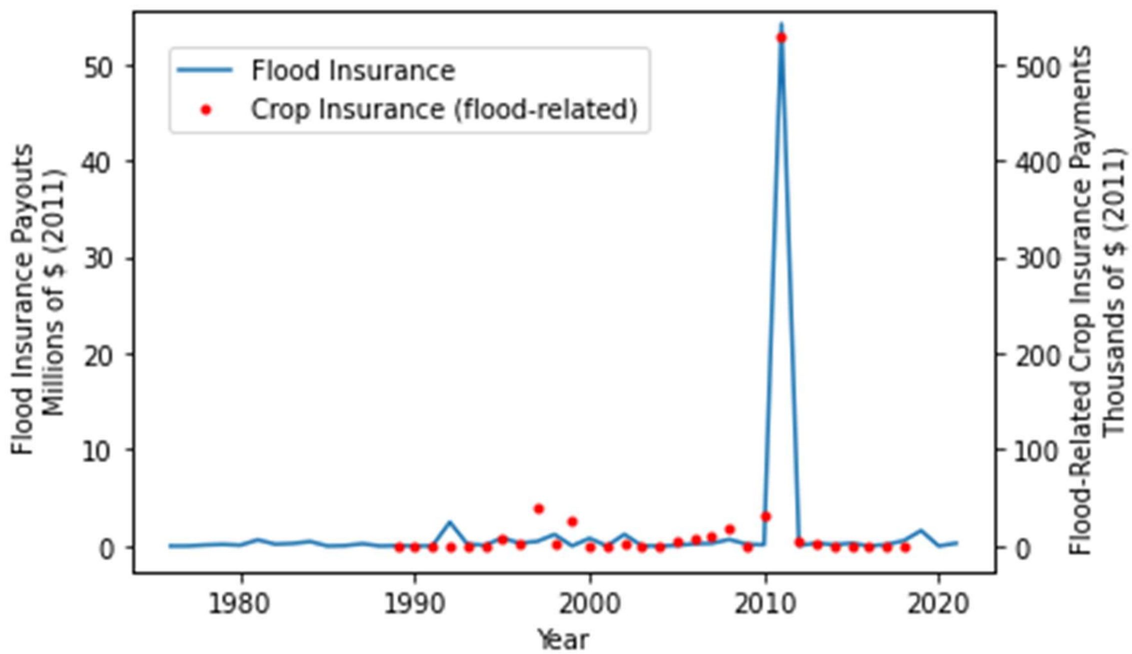

Figure S 2 Annual Payouts in Vermont for Federal Flood Insurance, and Crop Insurance Payouts for Flood-Related Damages. (Note that Crop Insurance payments are plotted at exactly 1/100th scale compared to Flood Insurance).

We reconcile the differences between these different methods by excluding modelled damages from the most frequent floods. We choose to exclude the estimated damages for these floods for several reasons.

First, it is hard to reconcile with existing data and other analyses. Other researchers consistently find the vast majority of flooding damages associated with low-recurrence floods. For instance, Wobus et al (2014) estimate that 98% of flood damages come from 25% of events.

Their estimates indicate that a year where all Vermont rivers flowing into Champlain experience a 2-year flood would yield $79 million in damages to buildings. In the last 11 years, the 75th percentile for total annual flood insurance claims for counties in the Champlain Basin is $375 *thousand*. Given that property owners whose properties are vulnerable to high-frequency flooding are substantially more likely to carry flood insurance[4], it seems extremely unlikely that flood insurance payouts would represent <.5% of *total damages to buildings* from frequent floods. For comparison, flood insurance claims accounted for about 4.8% of *total damages* from Irene, despite this storm impacting many areas that were not believed to be flood-vulnerable.

Second, a combination of intuition and the description of model uncertainty for probHAND given by the developers makes us cautious in interpreting their very large damage estimates for high recurrence floods. As Diehl and colleagues state:

[4] Indeed, a common criticism of the NFIP is that many people only live in highly flood-prone areas because subsidized insurance is available to them.

> "Inaccuracies in mapped flood extents from low-complexity models may be particularly large in urban settings or at confluences where simple process representations do not capture local hydraulic conditions [20,24] and *greatest for floods with smaller peak magnitudes*, which are more heavily influenced by local topographic and hydraulic conditions than large floods… We found that probabilistic maps capture the distribution of uncertainty within a dataset of field observations of flood extents, and from calibrated hydraulic model output."

(Diehl et al., 2021, p. 14)

While this model may capture this uncertainty well in a technical sense, it is unlikely that it accurately captures the probabilities of property flooding in high-frequency events. The choices to build, repair or abandon structures are made by people who have at least some knowledge of local flood history; people are less likely to build homes in places that are known to experience regular flooding. The more frequent the flooding, the stronger this divergence is likely to be. For 2-year return flood events, model uncertainty is high, and the location of homes provides a strong signal about the ways in which the model is likely to be wrong.

Comparative References:

Antolinin et al (2020) used SWAT to estimate impacts of agricultural best-management practices on flood damages in agriculturally-dominated (~90% of land) sub-watersheds in Iowa. In their most aggressive scenario, where about half of cropland (over 40% of watershed area) moves to no-tillage and/or cover-cropping, reduces expected annual flood damages by 5.8%.

For our soil-health "best" scenario, applied across all farmland, yields about 2-3% reduction in overall flood damages; in Vermont, agriculture is roughly 14% of landcover.

With regards to the overall scale of flood damages, Wobus and colleagues (2014) estimate $44 million in annualized (non-costal) flood damages in New England, approximately .0045% of regional GDP, from flood events large enough to be included in the National Climate Data Center storms database[5]. Our estimates give the scale of annualized flood damages in Vermont as ~$98 million, about .38% of Gross State Product, two orders of magnitude higher. The Wobus estimate only includes the largest floods, and Vermont is likely much more vulnerable to river flooding than the highly populated areas of southern New England states.

[5] This corresponds to roughly the 208 largest flood events each year in the United States.

Appendix 4 Erosion:

Biophysical Quantities.

The Soil K Factor:

The Erodibility Factor (K) is one of the five[6] parameters of the universal soil loss equation (USLE), and it is the only factor that is directly influenced by soil-health indicators. The K factor describes the susceptibility of soil particles to detachment. Because the USLE is a multiplicative model, a 10% decline in the K factor corresponds to a 10% decline in erosion, for the same field with the same management.

We use the Wischmeier equation to estimate changes in the K factor. This equation takes the form:

$$k \ = \ \frac{.00021 \cdot ((silt + vfs) \cdot (0 - cla\ \ ))^{1.14} \cdot (12 - OM) + 3.25 \cdot str + 2.5\ \cdot (perm - \ \ )}{100}$$

Where OM, clay and silt are their percentage representation in the soil, vfs is percent very fine sand in the soil. Str represents a soil structure code (integer between 1 and 4) and perm is a soil permeability class code (1-6).

We modify this equation slightly by making the soil permeability code continuous, rather than discrete, calculating it as:

$$perm \ = \ 6 - \ log(Ksat)$$

with a minimum value of .9 and a maximum of 6.5 .

The other commonly used option is the Williams equation, developed for APEX/EPIC.

Its calculation several more steps than the Wischmeier equation, but only incorporates values for clay, silt, sand, and organic carbon as percentages of soil weight.

[6] Or 6, depending on whether the length and slope factors are calculated jointly or separately.

The scenarios for the USLE simulations are parameterized as described in table S2.

Parameter	Meaning	Value Used	Notes/Source
R	Rainfall Erosivity	97	EPA R factor calculator, average of points in NW Vermont.
K	Soil Erodibility	Calculated using Wischmeier Eq	
LS	Length-Slope	.6 for Corn, .75 for Hay/Pasture	Averages Calculated for Franklin County. For practice changes, .6 is used.
C	Cover Factor	Differs by Crop	Table from here
P	Erosion Control Practice	.9	Intermediate between cross-slope and up-and-down tillage

Table S 2: Parameters used for USLE for Calculating Erosion Losses

Caveats:

Our methods gives one major source of error, which may cause an underestimate of benefits. Sediment yield, the amount of eroded sediment which actually reaches waterways is heavily influenced by runoff volumes, and improvements in soil health reduce runoff. This is reflected in the Modified Universal Soil Loss Equation (MUSLE) which calculates sediment yield (rather than erosion) for each individual runoff event. It does this by calculating the USLE R-factor as:

$$R = (Q_{peak} \cdot Q_{surf} \cdot ha)^{0.56}$$

Where ha is area of the field in hectares, and Q_{peak} is maximum 15-minute runoff rate in m^3/s. Because of its exponential scaling, the need to calculate peak runoff for each event and scaling with size of field, this method is substantially harder to implement.

Valuation:

Described fully in the primary document.

Appendix 5: Phosphorus

Biophysical Quantities.

Our simulations of P loss utilize a slightly modified version of the Vermont Phosphorus Index. For simulating impacts of soil health, there are two sets of impacts. First, the reduced erosion rates reflect themselves in less soil lost through erosion. This is estimated simply by inputting the erosion rates for different soil-health scenarios into the P-Index model.

Second, improved soil health results in reduced growing-season runoff. To estimate these reductions in runoff, we first estimate curve-number adjustments from improved soil health based on runoff simulations used in the Flood Mitigation section and equations described by Baiamonte (2019) to translate the results of a Green-Ampt simulation into an approximate Curve number. These were combined with average rainfall data for Burlington, VT to estimate changes in total seasonal runoff. These results were translated into custom runoff adjustment factors for the P Index[7].

Otherwise, the P Index was parameterized as shown in Table S3.

Parameter	Value	Notes
Elevation	600	Most Vermont farm fields are at relatively low elevations
Soil Test Phosphorus ppm (Modified Morgans)	6	Considered a medium-high level. Crop fields in VT have a mean MM P of ~6.5 and a median of ~3 .
Soil Test Aluminum	40	Crop fields in VT have a mean Al level of 49 & a median of 31.
Tile Drain	Not Present	
Distance to water	25 feet	
Buffer Width	15 feet	

Table S 3: Parameters Used For the VT-P Index to calculate phosphorus losses

Valuation:

For the first method, we transfer the estimates of economic damages calculated for the Missisiquoi Bay by Gourevitch et al to other Lake Segments. To do this, we assume that the total economic damages of exceeding the TMDL for each Lake segment are determined by 3 quantities:

1: How much, in % terms, the segment's P Load exceeds the TMDL.

[7] Modifying the runoff adjustment factors found on page 6 of the VT P Index technical documentation.

2: The number of people living within 20 km of the Lake Segment (people living within 20 km of more than 1 Lake Segment are divided evenly between them.)

3: The average household income of the people living near that Lake Segment.

The second two quantities approximate the overall level of economic activity in the area; economic damages/benefits relating to ecosystem services are usually proportional to economic activity. A more exact calculation would use home-price and tourism revenue data.

For the first parameter, we find that a log-log model best fits the outputs of the base scenario and 6 different reduction scenarios from Gourevitch et al. We calculate the damage scaling factor M as (Figure S3):

$$M = (.721 + \ln(P_{base}/ P_{tmdl}) \cdot .705)^{e}$$

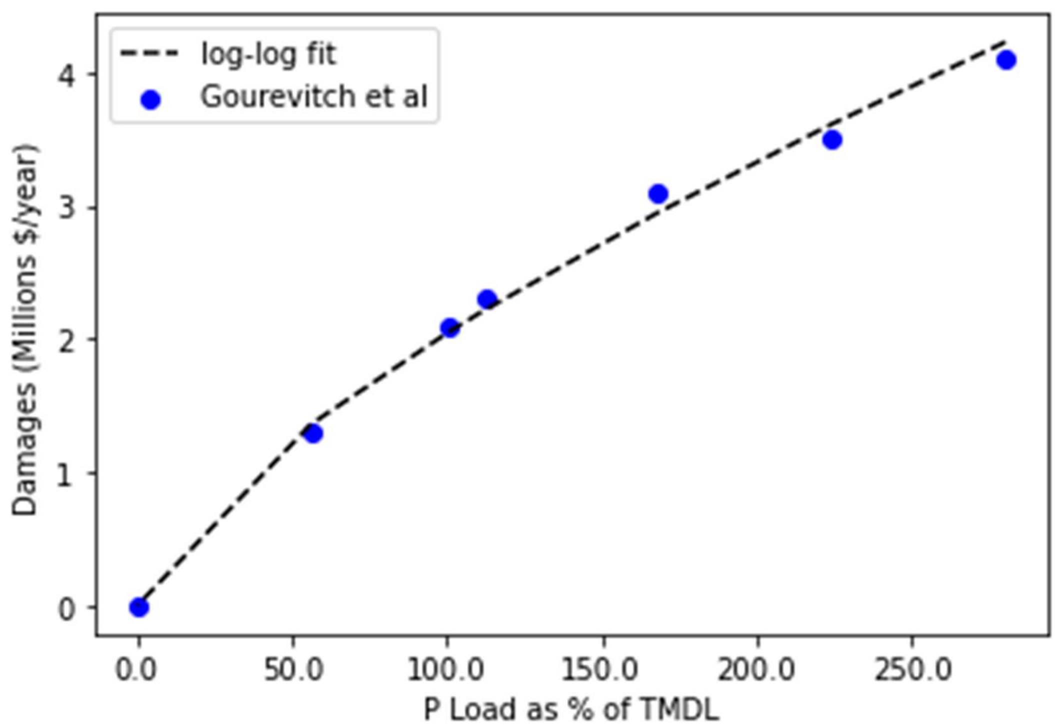

Figure S 3: Relationship between P Loading as a % of TMDL target and Total Damages, for the Missisquoi Bay in simulations run by Gourevitch et al.

For the 2nd and 3rd parameters, we use census block-group population data, and average household incomes for each county.

For each Lake Segment, we estimate total damages from Phosphorus as:

$$D_S = D_{MB} \cdot \frac{Pop_S}{Pop_{MB}} \cdot \frac{HHI_S}{HHI_{MB}} \cdot \frac{M_S}{M_{MB}}$$

And damages at the TMDL as

$$DTM_S = DTM_{MB} \cdot \frac{Pop_S}{Pop_{MB}} \cdot \frac{HHI_S}{HHI_{MB}}$$

Total Benefits from meeting the TMDL are calculated as:

$$B_S = D_S - DTM_S$$

Where D, DTM, P, HHI and are total damages from P, total damages from P if the TMDL is met, population within 20 km, household income, and damage scaler and the subscript S represents a given segment and the subscript MB represents the Missisquoi Bay.

The final valuation of Phosphorus is calculated as the benefit of meeting the TMDL divided by the required reduction.

This method yields the values shown in table S4 below. We calculate the average value of reducing a lb of Phosphorus from agriculture as the average weighted by the agricultural phosphorus load to each segment.

Segment	Phosphorus Valuation ($/lb)
Burlington Bay	678.83
Isle La Motte	83.86
Main Lake	16.39
Malletts Bay	38.21
Missisquoi Bay	10.35
Northeast Arm	71.92
Otter Creek	6.35
Port Henry	160.12
Shelburne Bay	111.09
South Lake A	76.99
South Lake B	18.43
St. Albans Bay	65.16

Table S 4: Valuation of Benefits from Reducing Phosphorus Losses, $/lb

Both methods used to scale up from the estimates made by Gourevitch et al are quite imprecise. Note as well that their paper was not exhaustive in its treatment of economic damages from water quality. Not included in their analysis are the "consumer surplus" from tourism/recreational activities, above the increased spending at local businesses, other health benefits from clean water, reduced costs for treatment of drinking water and reductions in risks of catastrophic changes in the ecology of Lake Champlain. We are not able to estimate how movement of Phosphorus between different Lake segments, rather than treating segments as distinct waterbodies, might impact the valuations given.

Counter-intuitively, Gourevitch et al show increasing marginal benefits from reducing phosphorus loads. If we calculate the price of phosphorus based on the total modelled benefit of reducing P loading to 0, then the valuation of phosphorus roughly doubles.

Comparisons:

Another way to conceive of the benefits reducing P loads is to consider VT's obligation to meet the TMDL as a fixed commitment, and therefore, benefits of reducing P loads by 1 pound are the costs of the next cheapest alternative method. Using this approach would give a higher value than our damage-cost methods. The Vermont Pay-for-Phosphorus program currently pays $100/lb of P reductions, with substantial overhead costs. Costs of other opportunities to reduce P loads may be an order of magnitude higher. For instance, costs of reducing P from some VT wastewater treatment facilities are fairly low, but increasing these reductions see sharply increasing marginal costs (Figure S4). Additional reductions from urban areas or rural roadways may cost hundreds of dollars per lb of P.

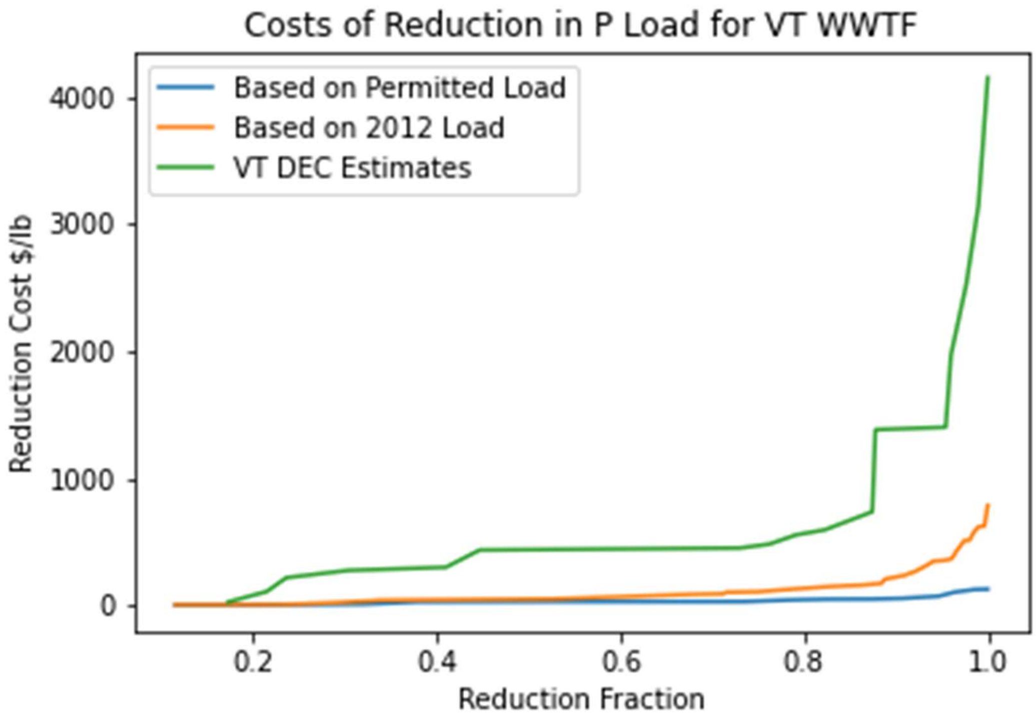

Figure S 4: Abatement Curves for Reducing Phosphorus Loads for Vermont Wastewater Treatment Plants.

Given that Vermont would not give up on the TMDL based on findings that the costs exceed the benefits, this approach might give a more realistic sense of the monetary benefits of reducing Phosphorus but understanding the exact costs of alternative measures may be difficult.

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