


2018

Conservation of Ecosystem Services and Biodiversity in Vermont, USA

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CONSERVATION OF ECOSYSTEM SERVICES AND BIODIVERSITY IN
VERMONT, USA

A Dissertation Presented

by

Keri Bryan Watson

to

The Faculty of the Graduate College

of

The University of Vermont

In Partial Fulfillment of the Requirements
for the Degree of Doctor of Philosophy
Specializing in Natural Resources

January, 2018

Defense Date: August 17, 2017
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ABSTRACT

Supporting a growing human population while avoiding biodiversity loss is a central challenge towards a sustainable future. Ecosystem services are benefits that people derive from nature. People have drastically altered the earth's land surface in the pursuit of those ecosystem services that have been ascribed market value, while at the same time eroding biodiversity and non-market ecosystem services. The science required to inform a more balanced vision for land-cover change in the future is rapidly developing, but critical questions remain unanswered regarding how to quantify ecosystem services and ascribe value to them, and how to coordinate efforts to safeguard multiple ecosystem services and biodiversity together. This dissertation addresses several of these challenges using Vermont as a model landscape. Specifically, we begin by estimating the economic value of flood mitigation ecosystem services and show that the externalized value of ecosystem services can be quite high. Second, we assess the role of demand from human beneficiaries in shifting the spatial distribution of ecosystem services, and address the biodiversity and human wellbeing implications of that shift. Third we analyze the tradeoffs and synergies inherent in pursuing multiple ecosystem services and biodiversity through conservation, and show that overall ecosystem service conservation is more likely to boost biodiversity outcomes than to undermine them. Finally, I implement statewide scenarios of land-cover change and flood risk in order to assess our ability to quantify ecosystem service outcomes and identify spatial priorities for the future despite land-cover change dynamics that are complex and unpredictable.

CITATIONS

Material from this dissertation has been published in the following form:

Watson, K.B., Ricketts, T.R., Galford, G.L., Polasky, J., and O'Neil-Dunne, J.. (2016) Quantifying flood mitigation services: The economic value of Otter Creek wetlands and floodplains to Middlebury, VT. *Ecological Economics*, 130, 16-24.

Material from this dissertation has been submitted to Ecosystem Services on August, 16, 2017 in the following form:

Watson, K.B., Ricketts, T.R., Galford, G.L., Sonter, L.J., and Koh, I.. Beneficiary demand impacts the human wellbeing and biodiversity outcomes of ecosystem service conservation. *Ecosystem Services*.

ACKNOWLEDGEMENTS

I would like to thank my advisors, Taylor Ricketts and Gillian Galford, who have provided invaluable guidance throughout my PhD. I would also like to thank Donna Rizzo, Jon Erikson, and Kimberly Wallin who served on my committee and advised me through the process of producing this dissertation. I would also like to thank Laura Sonter for her support and thoughtful insight, and Insu Koh for his incredible patience and R wizardry. I would also like to thank Jarlath O’Niel Dunne and Steve Polasky for their insights and contributions as co-authors on research included in this dissertation. Thank you to Mike Kline for his valuable insights into the dynamics of flooding in Vermont, and Roy Schiff for his insights and datasets regarding flood risk, and Liz Thompson for her kind eye into the practice of conservation in Vermont, and to all of the conservation professionals who dedicated time and thought to the generation of the scenarios presented in this dissertation. I would like to thank Diedre Zoll, Charlie Nicholson, Steve Posner, Courtney Hammond, Lindsay Barbieri, and Alison Adams for their support, insight, and friendship. Finally, I would like to thank my husband and daughter, Will and Amelie Watson, whose unending support and love have provided encouragement, balance, and strength through my PhD.

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CHAPTER 1: INTRODUCTION AND BACKGROUND

People have drastically altered the earth's land surface. Humans are the dominant force on a full 40% of all land (Ramankutty and Foley 1999), directly or indirectly influence over 80% (Sanderson et al. 2002), and appropriate one third to one half of global productivity (Vitousek et al. 1986). These land-cover changes have occurred largely through the pursuit of those ecosystem services whose value is captured in economic markets, but have eroded the planet's biodiversity (Newbold et al. 2015) as well its provision of non-market ecosystem services (Millennium Ecosystem Assessment 2005; DeFries, Foley, and Asner 2004; Foley et al. 2005). Approximately four fifths of all threatened terrestrial birds and mammals are declining primarily due to land-cover change (Tilman et al. 2017), and annual ecosystem service losses due to land-cover since 1997 have been valued at \$20 trillion/year (Sutton et al. 2016).

Sustainable development requires a more balanced vision of how to manage the earth's land surface. This will involve balancing tradeoffs between the provision of market and non-market ecosystem services, and between ecosystem services and biodiversity. The former can be informed by estimating the economic value of non-market ecosystem services, so that both market and non-market benefits can be taken into account in land-cover planning (Polasky et al. 2008; Bateman et al. 2013). The latter will be aided by a careful understanding of the spatial distribution of ecosystem services and biodiversity (Chan et al. 2006; Naidoo et al. 2008). Long term planning for each requires

understanding how land-cover change may proceed into the future (Peterson, Cumming, and Carpenter 2003).

Each of these areas of research is rapidly developing. It is now a widely accepted best practice to measure the economic value of ecosystem services in terms of marginal value (Ricketts and Lonsdorf 2013), and to specifically identified beneficiary groups (Arkema et al. 2013). Because of this, valuations from one context are not easily transferred to another (Balmford et al. 2002), and site-specific valuation remains an important tool in incorporating ecosystem services in economic decision-making. In general, the evidence accumulated thus far indicates that the value of non-market ecosystem services can be quite large (Balmford et al. 2015; Gallai et al. 2009; Van der Ploeg and De Groot 2010). The loss of non-market services outweighs the economic benefits of land conversion in many cases (Balmford et al. 2002), and the returns on investment in protecting ecosystem services exceed the costs of conservation (Balmford et al. 2015; Kovacs et al. 2013). Those ecosystem services with technological substitutes often prove more cost effective than their alternatives (Jones, Hole, and Zavaleta 2012). Those without substitutes are unsuited to economic valuation; their contribution to human well-being is irreplaceable (Farley 2012).

There has been increasing interest in ecosystem services from the private sector (Goldman et al. 2008; Ruckelshaus et al. 2013), governments (Donovan, Goldfuss, and Holdren 2015; Pittock, Cork, and Maynard 2012; Bateman et al. 2013; Liu et al. 2008),

and international organizations (Díaz et al. 2015; Van der Ploeg and De Groot 2010; Millennium Ecosystem Assessment 2005; Lammerant et al. 2013). Furthermore, ecosystem services are now explicitly targeted by many leading conservation organizations. This has led to concerns that ecosystem services are drawing from resources traditionally allocated for the protection of wild nature, and as such are detracting from biodiversity conservation (McCauley 2006; Goldman, Daily, and Kareiva 2011). The severity of the tradeoff between ecosystem services and biodiversity depends on the extent to which priorities for each overlap in space. The principles of conservation planning for biodiversity have been applied to identify spatial priorities for ecosystem services, and to compare them with existing biodiversity priorities (Maes et al. 2012; B. Egoh et al. 2009; Nelson et al. 2009; Bhagabati et al. 2014; Naidoo et al. 2008; Chan et al. 2006). However, the evidence to date is mixed. Some studies have identified win-win opportunities (Turner et al. 2007; B. N. Egoh et al. 2010), while others have cautioned that these win-wins may be infrequent (Anderson et al. 2009; Naidoo et al. 2008), or have found correlations that are positive but weak (Chan et al. 2006).

In order to safeguard biodiversity and ecosystem services in the long term, we need to know where they are most valuable now, and where they will be most important in the future. This presents a challenge. Human driven changes to the landscape interact via complex feedbacks with each other, with ecosystem services, and with human responses to those changes (S. R. Carpenter 2002). The consequence is that we know the future will be fundamentally different from the past, but are unable to foresee what that future will

be like (Clark et al. 2001; Raskin 2005). Scenario planning has been presented as a tool for making decisions under conditions of high uncertainty, and has been applied in the context of sustainable development (Raskin 2005) conservation planning (Peterson, Cumming, and Carpenter 2003) and ecosystem services (S. Carpenter, Bennett, and Peterson 2006). Where scenarios of land-cover change have been coupled to ecosystem service outcomes, they have provided powerful insights about tradeoffs and feedbacks between ecosystem services and land-cover decisions (Thompson et al. 2016; S. Carpenter, Bennett, and Peterson 2006; Bohensky, Reyers, and Van Jaarsveld 2006; Bateman et al. 2013). However, we still lack generalizable conclusions about how best to target our actions today given the degree of uncertainty we face about the future.

Critical questions remain unanswered regarding how to quantify ecosystem services and ascribe value to them, and how to coordinate efforts to safeguard multiple ecosystem services and biodiversity. This dissertation addresses several of those challenges using Vermont as a model landscape. I begin by estimating the economic value of flood mitigation ecosystem services through a case study of the Otter Creek watershed, and show that the externalized value of ecosystem services can be quite high. Second, I assess the role of demand from human beneficiaries in shifting the spatial distribution of ecosystem services statewide, and address the biodiversity and human well-being implications of that shift. Third, I analyze the tradeoffs and synergies inherent in pursuing multiple ecosystem services and biodiversity through conservation, and show that overall ecosystem service conservation is more likely to boost biodiversity outcomes than to undermine them. Finally, I implement statewide scenarios of land-cover change and flood

risk in order to assess our ability to quantify ecosystem service outcomes and identify spatial priorities for the future despite land-cover change dynamics that are complex and unpredictable.

Ethical Framework and Theory of Change

The research presented in this dissertation is scientific, but the motivation for pursuing this line of research, and for selecting this specific set of research questions, is value-laden. Here, I describe the ethical framework and theory of change that have motivated this body of work.

Ethical Framework

Pursuing a more sustainable future is, to me, a moral imperative. Doing so involves achieving three things: providing for the needs of people today, providing for the needs of future generations, and doing so within the ecological boundaries of our planet such that all other life on earth is also able to thrive. Humanity is simultaneously a plain member (Leopold 1989), and uniquely a steward, of the ecological community of our planet. Thus we are morally compelled to consider the wellbeing of all people, and that of non-human nature, in our personal and societal decisions.

In my view, people have a moral obligation to protect the integrity of earth's ecosystems for two reasons. First, nature has utility value to people, and is critical to the wellbeing of our human communities. Thus we must treat our entire planetary ecosystem responsibly

in order to treat our fellow man ethically. Second, it is my personal ethic that living things and ecosystems other than ourselves have inherent worth. The economic system that currently drives our natural resource use does not account for nature's intrinsic value, and accounts for its utility value insufficiently. It operates under the implicit assumption that nature's contribution to utility, and by extension human well-being, are signaled by market value. However, ecosystems contribute to human wellbeing in ways that are externalized from our economy. It is critical to our own well-being that we account for the role that nature plays in human well-being. Doing so is the most basic goal of ecosystem service science. It is also critical that we seek to protect our planet's biodiversity for its own sake. Doing so is at the heart of conservation. The research questions I pursue in this dissertation are motivated by the moral imperative to pursue each of these goals, human well-being and biodiversity conservation, simultaneously.

Theory of Change

Our current economic system is incompatible with long term sustainability in several ways. For example, infinite economic growth is fundamentally unsustainable (Daly 1992). Unlike internalizing ecosystem service value, correcting these problems would require a fundamental restructuring of our economy and our relationship to nature, a paradigm shift that is unlikely to occur in the near future. Thus in thinking about how to bring about change, we face an inherent tension between idealism and urgency.

It is my hope that making the value of ecosystem services more transparent will do three things to facilitate change towards a more sustainable system. First, I hope that it will allow for the rapid incorporation of nature's value in decision-making contexts where it is currently overlooked, shifting decisions to be more sustainable in the short term. I acknowledge concerns that economic valuation may result in "crowding out" of motivations for protecting nature that are not based in self-interest (Rode, Gómez-Baggethun, and Krause 2015), and concerns about the commodification of nature (Luck et al. 2012; McCauley 2006). However, I think that there are many contexts where the motivations likely to be crowded out already play a minimal role, and where attributing ecosystem services some value will lead to better decision making than implicitly attributing them no value. Second, I hope that mainstreaming the idea that nature is critical to human well-being will lay the stage for a more fundamental shift in our relationship to nature and a restructuring our economic system. Finally, quantifying ecosystem services can enable these services to be more efficiently protected in light of scarce resources for conservation.

As the ecosystem service concept is mainstreamed, it is important that the value of nature for people is considered in addition, not instead of, nature's intrinsic value. Research efforts treat the relationship between biodiversity and ecosystem services in a variety of ways. Some consider biodiversity to be an ecosystem service or to underpin ecosystem services, thereby implicitly focusing on nature's utility value. Others consider ecosystem services as a way of rebranding conservation to bolster the conservation community's ability to protect biodiversity, implicitly focusing predominantly on nature's intrinsic

value. Instead, in my research framework I deliberately consider ecosystem services (a representation of utility value) and biodiversity conservation (a representation intrinsic value) to be two separate conservation objectives that are each important in their own right.

CHAPTER 2: QUANTIFYING FLOOD MITIGATION SERVICES: THE ECONOMIC VALUE OF OTTER CREEK WETLANDS AND FLOODPLAINS TO MIDDLEBURY, VT

Abstract

Functioning ecosystems can buffer communities from many negative impacts of a changing climate. Flooding, in particular, is one of the most damaging natural disasters globally and is projected to increase in many regions. However, estimating the value of “green infrastructure” in mitigating downstream floods remains a challenge. We estimate the economic value of flood mitigation by the Otter Creek floodplains and wetlands to Middlebury, VT for Tropical Storm Irene and nine other floods. We used first principles to simulate hydrographs for scenarios with and without flood mitigation by upstream wetlands and floodplains. We then mapped flood extents for each scenario and calculated monetary damages to inundated structures. Our analysis indicates damage reductions of 84-95% for Tropical Storm Irene and 54-78% averaged across all 10 events. We estimate that the annual value of flood mitigation services provided to Middlebury, VT exceeds \$126,000 and may be as high as \$450,000. Economic impacts of this magnitude stress the importance of floodplain and wetland conservation, warrant the consideration of ecosystem services in land use decisions, and make a compelling case for the role of green infrastructure in building resilience to climate change.

Keywords

Ecosystem services, economic valuation, flood mitigation, green infrastructure, climate resilience

Highlights

- We present a simple approach to quantifying and valuing flood mitigation services.
- Wetlands and floodplains reduce flood damages by 54-78%.
- The economic value of this service warrants consideration in land use decisions.

Introduction

Ecosystems support human well-being in myriad ways. In many places, human activities have altered ecosystems to such an extent that real consequences on well-being are apparent [1]. To respond to these changes, the focus of conservation is broadening to include not only the negative impacts that people have on nature, but also the benefits nature provides to people [2, 3]. These benefits, or ecosystem services, include the many ways in which our communities and economies rely on functioning natural landscapes [4]. Such services have real and quantifiable value, although they are largely unrecognized externalities in our economy [5]. Economic valuation of ecosystem services can be instrumental in decision making that incorporates the contributions of nature to human well-being [6].

One way that ecosystems support well-being is by providing resilience to climate change. For example, coastal ecosystems can buffer against impacts from severe storms [7-10]; diverse ecosystems provide natural checks that limit the spread of infectious diseases [11]; and freely flowing rivers can alleviate the impacts of severe storms and flooding expected as climate changes [12]. Increasingly, “green infrastructure,” the network of functioning ecosystems that confer benefits to people [13, 14], is recognized as a method of building climate resilience [15], that may be more cost effective than engineered solutions in many cases [16, 17].

In particular, floods cause more human fatalities than any other natural disaster [18, 19] and are the most frequent natural disaster in many regions [18]. The potential of wetlands and floodplains to reduce flooding is widely recognized. Wetlands are areas where water is the primary factor driving plant and animal life [20]. Floodplains are the flat lands adjacent to rivers created by their lateral migration [21]. Both can act as green infrastructure to mitigate flooding by storing and slowing floodwater so that it arrives downstream gradually rather than in a single large pulse [22, 23]. Wetlands are thought to be most effective in reducing small, frequent flood events [24], whereas floodplains can reduce downstream peak flows for more severe events as well [21, 25]. Many climate scenarios indicate an increase in severe precipitation events [26], which suggests that the importance of wetlands and floodplains for human wellbeing will increase.

Despite the importance of wetlands and floodplains for alleviating floods, both have undergone widespread loss resulting from human interference with river geomorphology, such as the construction of levees and river channelization [20, 27]. These practices promote incision and disconnection of rivers from their floodplains and associated wetlands. By rapidly channeling water downstream, these hard engineering solutions reduce flooding locally, but can increase floods downstream [28, 29]. Both wetland loss and floodplain disconnection are being targeted by conservation and restoration projects with green infrastructure goals. The non-market benefits of wetlands and floodplains are often undervalued or completely unaccounted for in local decisions [22] because these benefits are externalities that mostly accrue downstream. Quantifying the economic value

of flood mitigation services, in terms of real and avoided flood damages, can influence regional-scale planning decisions regarding the use of green and built infrastructure [30] by connecting upstream decisions to downstream impacts. In order to responsibly allocate conservation resources to protect wetlands and floodplains, we need to know when expected returns on that conservation investment will be positive.

Current techniques to quantify water-related ecosystem services generally fall within three categories. First, empirical approaches are used to measure the biophysical supply of services, such as measuring the water storage capacity of wetland soils [31] or relating the development of wetlands to flooding frequency [32]. Second, advanced hydrological models are modified to inform ecosystem service decisions; however, these models do not tend to produce results necessary to evaluate benefits to specific stakeholders [33]. Finally, models developed as support tools for ecosystem service decision-making seek to provide more direct measures of human well-being outcomes [34] [35]. There are existing hydrologic models and empirical approaches that measure the impacts of land use on flooding [31, 32, 36-38] and other models that measure the impacts of flooding on people [39], but we do not know of an existing model designed for ecosystem service decision making. Although it may not be possible to consider biophysical and socioeconomic dynamics each in depth, it is crucial that valuations of hydrologic services consider both [40].

We present a first-order approach to estimating the value of flood mitigation services provided by wetlands and floodplains built upon ecologic, hydrologic, and economic principles. Our approach is novel in linking biophysical flooding dynamics to human beneficiaries at the watershed scale. To illustrate this approach, we quantify the economic value of flood mitigation in terms of avoided damages to human beneficiaries provided by the wetland-floodplain complex of the Otter Creek (which remains highly connected to its floodplain and associated wetlands) to Middlebury, Vermont (USA). Specifically, we address two questions:

- 1) What was the value of the Otter Creek wetlands and floodplains in reducing flood damage during Tropical Storm Irene in 2011?
- 2) Beyond this single event, what is the expected annual value of the wetlands and floodplains in mitigating flood damages?

These valuations allow us to quantify the damages of a high-profile storm that has focused attention on role of wetlands and floodplains in bolstering climate resilience, and to estimate the damages avoided in an average year. The latter is more likely to be actionable information for decision makers than the damage costs of a rare event, although both are important given that storm intensity and rainfall are increasing in this region [41]. This work enables explicit consideration of flood mitigation by wetlands and floodplains in land use and resource decisions.

Methods

We estimated the value of flood mitigation services as the damage to downstream communities that was avoided as a result of wetlands and floodplains. Quantifying avoided damages is a well-established method of non-market valuation [42, 43]. Specifically, we estimated the difference in expected damages between current conditions (where the river is connected to wetlands and floodplains, hereafter referred to as the “wetlands” scenario) and two hypothetical scenarios where the river does not have these connections. One of these counterfactual scenarios represents a large effect of wetlands and floodplains (“no wetlands-high” scenario) and the second represents a more conservative effect (“no wetlands-low” scenario). These scenarios apply theoretical conditions to the Otter Creek to illustrate the potential range of benefits provided by the wetland-floodplain complex, rather than predicting the precise value of those benefits. More advanced process-based modeling would be appropriate if specific predictions were needed given expected marginal changes in access to wetlands and floodplains. The use of scenarios is a well-established method of illustrating the envelope of possible outcomes given large uncertainties [44].

To evaluate flood damages, we followed a five-step process: First, we modeled hypothetical flood peaks representing conditions where the Otter Creek lacks connection to its floodplain and wetlands (henceforth referred to as “no-wetlands” scenarios for simplicity). Next, we estimated flood extent for wetlands and no-wetlands scenarios. Third, we identified flooded structures in each scenario. Fourth, we calculated expected

damages for each structure as a function of flooding depth and house value. Finally, we estimated the value of avoided damages by pooling costs for each scenario and calculating the difference in total damage between wetlands and no-wetlands scenarios. We followed these steps for Tropical Storm Irene and for nine additional historic flooding events in order to estimate the annual value of flood mitigation.

Study System

We focused on Otter Creek in Middlebury, VT (Figure 2.1). The Otter Creek is a useful case study for several reasons. First, Vermont's land use pattern, with development concentrated along rivers in low-lying floodplain areas, is typical of many rural regions. Second, recent extensive flood damages related to very large storms have pushed flood resiliency forward as a priority in Vermont and the Northeast. Finally, climate projections estimate that precipitation will increase, and will more often occur in high energy precipitation events, a trend that has already been observed over the last half century [41, 45]. This indicates that flood resiliency will increase in importance. Finally, the Otter Creek remains well connected to its floodplain, and thus has the potential to illustrate the value of maintaining functional access to floodplains and wetlands for the purpose of mitigating floods.

Otter Creek flows north through a large wetland complex and a relatively wide, connected floodplain from Rutland, VT to the town of Middlebury (Figure 2.1). Although three-quarters of Vermont streams and rivers are incised, and thus disconnected from

their floodplains [46], stream geomorphic assessment indicates that there is virtually no stream incision on the main stem of Otter Creek [47]. The watershed is predominantly forested (60%), 5% of land-cover is developed, 24% is agricultural, and 8% is wetland. Wetlands comprise a total of 18,000 acres, most of which are forested swamplands. USGS gauging stations on the Otter Creek are positioned in the towns of Rutland (hereafter, “upstream”) and Middlebury (hereafter, “downstream”). The river meanders 36 river miles between the gauges, and elevation change is modest, dropping from 475 to 336 feet above sea level [48]. The downstream gauge has a drainage area twice as large as the upstream gauge (628 vs 307 square miles). The paired gauges record flow dynamics during rain events and enable us to value flood mitigation provided by the wetland-floodplain complex in the absence of an advanced hydrological model.

Tropical Storm Irene hit Vermont on August 28, 2011. Every town in Vermont reported flood damages [41], including Rutland and Middlebury. Rutland experienced the highest peak flow on record on August 28th and suffered serious flood damages over the five days following the storm. Roughly thirty miles downstream and a week later, Middlebury experienced a much lower peak and flooding was minor because floodwater arrived gradually over a longer time interval (several weeks instead of about five days) (Figure 2.2). Locally, the observed difference in flood damage was touted as an example of flood mitigation by wetlands and floodplains, and of green infrastructure bolstering the resiliency of local communities to extreme rain events [49]. We focus our valuation on

the town of Middlebury itself, which encompasses 14 square miles and has a population of roughly 6,600 [50].

A hydrograph is a plot of discharge as a function of time — typically in cubic feet per second (CFS). We accessed hydrographs for upstream and downstream gauges over the interval of the downstream storm water pulse (17:00, 8/27/11 to 11:00, 9/22/11) [48] (Figure 2.2). We included a long tail on the hydrograph’s falling arm to ensure a conservative estimate of the pulse duration and magnitude (The falling arm is where discharges of the two hydrographs are most similar). Flood volume is the sum of areas area under the hydrograph curve. We calculated volume as a Reimann sum:

Equation 2.1
$$V = \sum_{i=0}^{n-1} q_i * \Delta t$$

where V is total water volume in cubic feet, q is discharge (cfs) for each time interval i , and Δt is the time between discharge measurements at the gauge (15 minutes).

Modeling Peak Flows

We developed two scenarios to estimate peak flows in cases where wetlands and floodplains were eliminated completely. Although the Otter Creek is not under immediate risk of losing its wetlands or its connection its floodplain, such losses are common elsewhere and reduce the capacity of landscape to mitigate downstream flooding. Further,

“total loss” scenarios such as these are needed to determine the ecosystems’ total value for flood mitigation. Our two no wetlands scenarios differ in terms of the size of the impact that disconnection from wetlands and floodplains has on downstream flooding, By providing a high and low estimate of this effect, they illustrate the range of effects wetlands and floodplains may have on downstream flood damages.

No-Wetlands High Scenario

The no-wetlands high scenario represents a case where the difference in the shape of the upstream and downstream hydrographs (the timing of floodwater arrival) was solely attributable to the wetlands and floodplains that lie between the two gauges, but where the wetlands and floodplains had no impact on the total floodwater volume.

We normalized the upstream hydrograph by dividing the volume for each time interval by the total upstream water volume, and then multiplied these incremental volume measures by the total volume recorded at the downstream gauge:

Equation 2.2
$$v_{iNo-Wetlands\ High-Impact} = \frac{v_{iUpstream}}{V_{Upstream}} * V_{Downstream}$$

Where v is water volume for a time interval i , and V is total water volume.

By modeling the no-wetlands hydrograph using the upstream hydrograph shape and downstream floodwater volume, we simulated a case that does not allow for any dissipation of the storm peak or temporary water storage by the landscape, but that does contain all the rainfall that occurred between the upstream and downstream gauges. In doing so we also assumed that much of the water entering between the gauges would contribute to the downstream hydrograph peak. Essentially, this simulated a case in which floodwater moved downstream through an impervious channel, and where all of the water that fell between the upstream and downstream gauges entered the channel exactly in proportion to the passing flood peak. Because of these non-conservative assumptions, this scenario represents an upper bound on the value of the wetland-floodplain complex.

No-Wetlands Low Scenario

We created a more conservative scenario that differed from the no-wetlands high scenario in two ways. First, we assumed wetlands and floodplains only affected water that entered the Otter Creek above the upstream gauge. To model this, we assumed water entering the Otter Creek between the gauges did so with timing proportional to the downstream hydrograph (instead of proportional to the upstream hydrograph). We calculated the difference in observed water volumes recorded at the upstream and downstream gauges using Riemann sums, multiplied the volume of water that entered the channel between the two gauges by the normalized downstream hydrograph, and multiplied the upstream water volume by the normalized upstream hydrograph. This assumption causes us to

underestimate the impact of the wetland floodplain complex, thus this scenario represents a lower bound on their value.

Secondly, wetlands and floodplains were considered to be only partially responsible for flood mitigation. Floodwaters would have dissipated to some extent due to factors other than wetlands and floodplains. Others have shown that wetlands are the only land-cover type that impacts flood peaks in this region [51]. However, topographic effects other than floodplains such as storage and friction within the channel will also reduce flood peaks, so that larger drainage basins tend to have lower flood peaks relative to their flood water volume even when they do not have floodplain access. To account for these effects, we regressed discharge per unit area against drainage basin size for 10-year floods at Vermont USGS gauges (Figure S2.1, [51]). Using this relationship we determined that the unit discharge expected for the drainage area of the downstream gauge was 11% lower than that expected for the drainage area of the upstream gauge. We decreased the volume of the upstream hydrograph for each time interval by this dissipation factor. Because most rivers in Vermont have been disconnected from floodplains through incision, this dissipation factor provides us with an estimate of how much the flood peak would dissipate while traveling downstream from the upstream to the downstream gauge in the absence of wetland and floodplain effects. In sum, the no wetlands low hydrograph was calculated as:

Equation 2.3

$$v_{iNo-Wetlands Low} = \left(\frac{v_{iUpstream}}{V_{Upstream}} * V_{Upstream} * 0.89 \right) + \left(\frac{v_{iDownstream}}{V_{Downstream}} * V_{Between} \right)$$

Where v is water volume for a time interval i , and V is total water volume, as above.

Although this is a much more conservative estimate of the potential impact of wetlands and floodplains on peak flows, it does not represent an absolute lower bar of that affect.

Determining Flood Extent

For each scenario, we used a rating curve built from a log-log regression of the highest daily mean water level for every year from 1927 to 2012 ($r^2=0.96$, $p<2.2e-16$; Figure 2.3) to relate discharge (cfs) to stage (river height, feet). From the rating curve, we calculated the flood elevation associated with downstream peak discharge from the wetlands and no-wetlands hydrographs. While many of the annual peaks in our dataset represented cases where Otter Creek overflowed its channel and inundated the surrounding floodplain, the no-wetlands discharge exceeded all recorded annual peaks so we were forced to extrapolate beyond our data to determine flood elevation.

A 15 meter waterfall occurs in Otter Creek at Middlebury just below the downstream gauge. Thus we adjusted flood heights for areas below the falls (north) by subtracting 15 meters (Figure S2.2) but otherwise assumed that the rating relationship and flood elevation were equal throughout Middlebury (i.e., a “bathtub” model of flooding). In

reality water volume, not height is conserved as a flood pulse travels downstream because the relationship between volume and height is sensitive to floodplain geometry. The benefit of this assumption was the use of a single metric, flood height, which could be robustly estimated (Figure 2.3).

We defined the flood extent as areas in Middlebury that were hydrologically connected to the Otter Creek and that fell below the flood elevation. This flood extent was identified using a high-resolution 1-meter Digital Elevation Model (DEM) derived from LiDAR data acquired under leaf-off conditions in 2014.

Identifying Flooded Structures

We overlaid the flood extents for each scenario with a point database of Middlebury's structures that was created for emergency response efforts [52]. Structures were determined to be flooded if they fell within the flood extent, or if they fell within a 100 ft buffer of the extent and were within two feet of the flood elevation. The latter criterion accounts for structures above the flood level with basements that may have flooded. The Federal Emergency Management Agency estimates monetary damages beginning with flood depths of -2 feet for residential structures [39], and most homes in Vermont have basements. We calculated each structure's flooding depth as the structure's ground elevation, as determined by the LiDAR DEM, subtracted from the flood elevation.

All hydrograph manipulations and flood elevation calculations were performed using R statistical software [53]. Flood extent scenarios were performed in Quick Terrain Modeler [54]. All other GIS analyses were done using the ArcGIS software package [55].

Monetary Damages

We calculated expected damage for each structure as a function of flooding depth and property value (Figure 2.4). We applied a depth-damage function for residential structures with basements from FEMA's HAZUS guidelines [39]. This function is developed from national insurance claims, with adjustments for uninsured losses. We merged a publicly available database of property tax records with the spatial dataset of structures. The matching of these datasets had to be verified and cleaned by hand due to discrepancies such as spelling errors, and duplicated entries. We also verified and, in some cases, updated property estimates from Zillow [56]. Publicly owned structures with no tax record were assigned the lowest property value of the identified flooded structures.

Valuation of Avoided Damages

We calculated the value of flood mitigation services provided to Middlebury by the upstream wetlands and floodplains as the difference in total damages for all structures between the wetlands and no-wetlands scenarios.

The Mean Annual Value of Flood Mitigation

The method outlined above resulted in an estimate of avoided damages for a single event, Tropical Storm Irene. To quantify the annual expected avoided damages, we repeated the procedure for Irene and nine additional flooding events using historical data. Prior to 2007, discharge data were shown as mean daily values rather than in 15-minute intervals. We obtained these hydrograph data for the seven largest events on record at the upstream gauge (including Tropical Storm Irene), plus three floods whose peak discharge approximated those of two-year and five-year floods [57]. For each storm event, we included data for one month before and one month following the upstream flood peak.

Using wetlands and no-wetlands damage estimates for these ten events, we determined mean annual value by establishing a probability-damage function that relates expected damages to annual exceedance probability, paralleling the methodology of the U.S. Army Corps of Engineers for risk analysis [58]. Annual exceedance probability, p , is the probability that a discharge Q is equaled or exceeded in a given year, and is the reciprocal of the return interval. For example, a flood expected to occur approximately every 20 years has an exceedance probability of 0.05, i.e., a 5% chance of occurring in any given year. We fitted an exponential decay function to the peak discharge of FEMA designated 2, 5, 10, 25, 50, 100 and 500-year floods [57] and used this function to determine the annual exceedance probability of each flood we modeled, based on downstream discharge in the wetlands scenario. Finally, we created damage-probability functions by fitting negative exponential curves to the expected damage against

exceedance probability for each historic flood and for both wetlands and no-wetlands scenarios.

We estimated expected annual damages as the integral of the probability-damage function over the range of exceedance probabilities from zero to one, and determined the mean annual value of flood mitigation services as the difference in expected annual damages between the wetlands and no-wetlands scenarios.

Net Present Value Calculation

We calculated net present value based on this average annual value of flood mitigation benefits by assuming that this value will be accrued in perpetuity and that future values are discounted relative to present value. We applied a range of plausible discount rates: the standard US discount rate for water resource decisions is 3.375% [59]. This rate is lower than the standard discount factors used by FEMA (4.125%) [39] and the US Army Corps of Engineers (7%) [60]. However, it is much higher than discount rates applied to long term environmental benefits elsewhere, such as the declining discount rate suggested by the UK Treasury [61], and the 1.4% discount rate adopted by the Stern Review on the economics of climate change [62].

We compared these estimates of net present value to the costs of conservation by assuming these costs are equal to the costs of purchasing all 18,000 acres of Otter Creek

wetlands for the county average value of farmland (\$3044 and \$2718 per acre in Addison and Rutland counties, respectively) [63].

Results

Middlebury's peak discharge for Tropical Storm Irene in the wetlands scenario corresponds to a flood height of 7.4 ft above the downstream gauge (Table 2.1). In contrast, our modeled no-wetlands scenarios indicate flood heights of 13 to 18 ft above the gauge and greatly expanded flood extents. We identified 21 to 54 flooded structures in the no-wetlands scenarios, compared to just nine in the wetlands scenario (Figure 2.5). The total damages for all flooded buildings was \$100,600 in the wetlands scenario, which is similar to the \$70,000 in actual reported damages in Middlebury [64]. We estimate damages of \$626,600 to \$1,900,800 in the no-wetlands scenarios (Table 2.1). These differences correspond to an 84-95% reduction in financial cost of floodwater inundation and between \$525,900 to \$1,800,200 in avoided damages.

Expected damages across the 10 modeled floods ranged from \$45,000 to \$338,000 in the wetlands scenario, and from \$130,400 to \$1,339,000 in the no-wetlands scenarios (Table 2.2). The average damage reductions were 54% to 78% for low and high scenarios, respectively. Reductions tended to be greater for smaller, more frequent floods (Figure 2.6). For each scenario, we fit probability-damage functions to these ten events. Based on these damage functions, we calculated expected annual damages to be \$75,000 in the wetlands scenario, \$201,400 in the no-wetlands low scenario and \$534,000 in the no-

wetlands high scenario (Figure 2.7). The mean annual value of flood mitigation services provided to Middlebury is therefore \$126,000 based on our low scenario, and \$459,000 based on our no-wetlands high scenario.

By applying the US standard discount rate for water resource decisions [59] to our high estimate of annual flood mitigation value, we estimate that the net present value (NPV) of mitigation services exceeds 12 million dollars, which is over a quarter of our estimated costs of conservation (Table 2.3). Using the declining discount rate suggested by the UK Treasury, NPV rises to approximately 16 million dollars, or 30% of the costs of conservation. Using the 1.4% discount rate adopted by the Stern Review on the economics of climate change [62], NPV triples and amounts to over 60% of land acquisition costs. When we apply a discount rate back-calculated from mean agricultural land values and rents [62] (i.e., assuming rents reflect annual benefits accrued in perpetuity), this value rises to 95% of conservation costs (Table 2.3). Using our low estimate of flood mitigation values and these same discount rates and cost estimates, we find that net present values range from \$1,800,000 to \$14,000,000, which is 3-27% of our estimated costs of conservation.

Discussion

We show that wetlands and floodplains can provide valuable flood mitigation services and increase community resilience to climate change. Specifically, we find that the Otter Creek wetland-floodplain complex reduces downstream flood inundation costs by up to

92% across a range of flood intensities (Table 2.2). For Tropical Storm Irene alone, these wetlands and floodplains provided between \$627,000 and \$2,000,000 in avoided damages (Table 2.1). Beyond this one event, the expected annual value exceeds \$126,000, and may be as high as \$450,000. These values will likely increase under a changing climate, with extreme rain events already becoming more common. Our findings support the potential of wetlands and floodplains to act as green infrastructure that builds community resilience to climate change.

Our damage estimates represent only a fraction of the flood mitigation value provided. We focused on avoided damages caused by inundation of buildings in the town of Middlebury, omitting damages to infrastructure, profits lost to businesses, erosion damages (which often exceed those from inundation [65]), insurance costs, agricultural losses, and less tangible impacts on human health. All of these factors may also be mitigated by upstream wetlands and floodplains.

The estimated mean annual value of \$126,000 to \$459,000 for this wetland complex is large enough to warrant explicit consideration of flood mitigation services in land use decisions. When we compare this value to rough estimates of the costs of wetland conservation we find that flood mitigation benefits alone “pay-back” at least a quarter of the expense of conserving the Otter Creek wetland-floodplain complex (Table 2.3). This conclusion holds over a range of discount rates for our high scenario, and over all but the highest discount rates for the low scenario. High fixed discount rates are inappropriate

both to human preferences over long time spans and to precautionary environmental decision-making [61, 66]; thus, we find the lowest discount rates presented here are most applicable. Furthermore this conclusion is conservative because we are likely to have overestimated conservation costs. Most of these wetlands are already protected under state and federal legislation [67], and conservation is increasingly achieved through easements, which are more cost effective than land acquisition [68].

That flood mitigation alone could pay back over a quarter of the costs of conservation is remarkable, since conservation would also protect biodiversity and a number of other ecosystem services that provide quantifiable benefits to people, such as hunting, bird watching, recreation, and water filtration [20]. A full analysis of the return on investment (ROI) in wetland conservation is beyond the scope of our study, and would require more accurate estimates of acquisition and opportunity costs, as well as information on development risk. However, our rough comparison illustrates that ROI is likely to be generally positive, given that wetlands are under high risk globally [20].

While damage reductions were substantial in all ten historic cases, we found that the flood mitigation effects decreased for larger floods (Figure 2.6). This result reinforces existing findings that wetlands are less important for larger, less frequent flood events [24, 69]. Beyond some threshold, the capacity of wetlands to absorb flood water may be overwhelmed, in which case no additional mitigation can be provided [70]. Green infrastructure solutions may therefore be best suited to address flood events with medium

return intervals, whereas built infrastructure and careful development planning are more effective for the most extreme events.

Our findings support a growing body of literature indicating wetlands and floodplains can have large impacts on peak flows [71]. Indeed, previous findings correspond more closely with our higher estimates of peak flows. For example, studies in New England using more advanced hydrological models have shown complete removal of wetlands can increase peak flows by over 200% [72]. Elsewhere, river channelization is estimated to increase peak flows by 50-150% [21]. Additionally, the discharge we estimated in Tropical Storm Irene under the no-wetland high scenario corresponds almost exactly to the 10-year flood discharge from a regional statistical model developed by the USGS when we remove the effect of wetlands ([51],Table S2.1).

The economic value of flood mitigation services per area of wetland presented here is considerably lower than values obtained elsewhere via other methods. We estimated the value of the Otter Creek wetlands complex at less than \$100 per hectare per year (\$459,000 divided by 7280 ha). Ming and colleagues [31] have calculated the water storage capacity of wetlands in the Mogome National Reserve in China and value this storage function at \$5700 per hectare per year using a replacement cost technique. Thibodeau and Ostro [73] use an avoided damages approach to arrive at a similar value of \$5000 per hectare per year. In the Economics of Ecosystems and Biodiversity (TEEB) database [74], there is only one study related to water flows that does not transfer values

from other studies; this study uses an avoided damages approach to calculate values of over \$9000 per hectare per year [75].

The quantity of ecosystem service depends on demand from human beneficiaries as well as biophysical supply [3], and demand will vary widely depending on downstream population and infrastructure [76]. Here we value benefits to a relatively small population of downstream beneficiaries, which may explain why the biophysical impacts we find are in line with other research efforts whereas our economic valuation is substantially lower than values found elsewhere. Although more sophisticated models exist to evaluate separately the hydrologic dynamics [36-38] and economic damages [39] of flooding, this dynamic stresses the importance of accounting for both biophysical supply and beneficiary demand.

We see three limitations to our approach. First, our no-wetlands scenarios rely on simplifying assumptions (Table S2.2) that result in a wide range of possible values. Future research is needed to reduce this uncertainty, to evaluate the effects of marginal (i.e. small) changes in wetland area, and to allocate value spatially within a watershed. Second, we extrapolate beyond the observed rating curve (Figure 2.3), and assume this rating relationship applies throughout Middlebury. Many of the annual floods used to establish the rating relationship overtopped the main channel into the floodplain, which does not include a second topographic tier that we would expect to shift the rating relationship for any floods other than the most extreme cases modeled. In these most

extreme cases height may be slightly overestimated (Figure S2.2). Because all floods inundated a wide floodplain throughout the study area, very large changes in volume would be required to cause noticeable differences in flood height, making our results less sensitive to this “bathtub” assumption. Further, our modeled flood extent are similar to flood extents from FEMA flood insurance rate maps despite this assumption (Figure S2.3; [77]). Floods of historically unprecedented proportions resulting from land use and climate change will fall outside the observed rating curve, so preparation for these events necessitates extrapolation. Third, our damage functions are poorly fit to the data in the no-wetlands cases (Figure 2.7). Variation in modeled flood peaks is to be expected given differences in temporal and spatial rainfall patterns, flood sizes, etc. While we cannot estimate the shape of the no-wetlands damage function with confidence, there is a consistent and significant vertical shift in the damage function as a result of wetland and floodplain loss (Figure 2.7). This emphasizes the importance of natural landscapes for flood mitigation regardless of the functional form of the damage curve.

If the conservation of wetlands and floodplains provides large returns, why do wetland loss and river channelization continue? The value of wetlands is often considered to be negligible, even negative, in many decision-making contexts [78]. Further, the costs of conservation and the benefits of avoided damages are realized by different groups. For instance, the costs of flood inundation are often spread among many downstream property owners and insurance agencies, whereas the opportunity costs of conserving wetlands must be borne by relatively few upstream landowners and municipalities.

Economic valuation can help clarify the impacts of land use decisions on people. Our findings provide evidence that preventing rivers from flooding surrounding wetlands and floodplains may only displace, and potentially increase, the total cost of flood damage [29]. Our most basic infrastructure, the ecosystems that support us, are in worldwide decline. In Vermont and nationwide, significant efforts are reconnecting rivers to their floodplains and conserving wetlands. This study illustrates that the benefits of these efforts are potentially quite large, and that the omission of ecosystem service outcomes from land use decisions may have real and severe consequences for people.

Acknowledgements

Special thanks to Brendan Fisher, Drew Guswa, Mike Kline, Roy Schiff, and Charlie Nicholson for their thoughtful comments, as well as Ned Swanberg, Arne Bomblies, Paul Marangelo, Kim Greenwood, and Michael Coe for their input and advice. The authors also thank the Lintilhac Foundation and the USDA McIntire-Stennis award #2014-32100-06050 to the University of Vermont for funding. J. O'Neil-Dunne was partially funded by the AmericaView consortium and from the National Science Foundation under grant DEB1053566.

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Tables

Table 2.1 Comparative summary of peak flows, flood height above the gauge, flooded structures, and expected damages following Tropical Storm Irene.

| Scenario | Peak Discharge (cfs) | Flood height (feet above gauge) | Structures affected | Expected Damages |
|---------------------------------|-------------------------|--|------------------------|---------------------|
| Wetlands | 6,180 | 7.4 | 9 | \$100,600 |
| No –wetlands low estimate | 15,600 | 12.8 | 21 | \$626, 600 |
| No-wetlands high estimate | 27,100 | 17.9 | 54 | \$1,900,800 |

Table 2.2 Value of wetlands and floodplains in terms of avoided flood damages for ten flood events in Middlebury, VT. Annual exceedance probability (AEP), damages with and without wetlands, and the resultant percent reduction and reduction in damages (value) for each flooding event are shown.

| Year | AEP | Damages under each scenario | | | | Value of wetlands and floodplains | | Estimated damage reduction (%) | |
|------|------|-----------------------------|-----------------|------------------|-----------|-----------------------------------|------|--------------------------------|------|
| | | Wetlands | | No-wetlands high | | Low | High | Low | High |
| | | Wetlands | No-wetlands low | No-wetlands high | High | Low | High | Low | High |
| 1976 | 1.02 | \$45,760 | \$204,962 | \$453,198 | \$159,202 | \$407,438 | 78% | 90% | |
| 1947 | 0.44 | \$49,974 | \$130,429 | \$325,698 | \$80,455 | \$275,724 | 62% | 85% | |
| 1956 | 0.42 | \$49,975 | \$161,232 | \$449,418 | \$111,258 | \$399,444 | 69% | 89% | |
| 1984 | 0.25 | \$68,169 | \$157,101 | \$527,783 | \$88,932 | \$459,614 | 57% | 87% | |
| 1948 | 0.14 | \$100,633 | \$243,401 | \$675,893 | \$142,769 | \$575,260 | 59% | 85% | |
| 2011 | 0.14 | \$100,632 | \$498,760 | \$1,338,654 | \$398,128 | \$1,238,022 | 80% | 92%** | |
| 1938 | 0.1 | \$127,032 | \$325,713 | \$1,043,294 | \$198,681 | \$916,262 | 61% | 88% | |
| 1977 | 0.08 | \$152,857 | \$204,956 | \$439,190 | \$52,098 | \$286,333 | 25% | 65% | |
| 1987 | 0.07 | \$157,088 | \$243,404 | \$457,925 | \$86,316 | \$300,837 | 35% | 66% | |
| 1936 | 0.01 | \$338,114 | \$325,708 | \$523,519 | -\$12,405 | \$185,405 | -4% | 35% | |

** Tropical Storm Irene. The use of daily discharge data for historic flooding events underestimates flood damages; in the specific analysis of Tropical Storm Irene, we used 15-min discharge data and found a 95% reduction in flood damages.

Table 2.3 Value of flood mitigation services relative to conservation costs. Net present value (NPV) is calculated using a range of discount rates, and is compared against conservation costs as estimated by the cost of land acquisition. Ranges reflect low and high scenarios.

| Source of discount rate | Discount rate | NPV (millions US\$) | NPV/Cost of land acquisition |
|---|----------------------|----------------------------|-------------------------------------|
| Mean agricultural land values & rents [63] | 0.9% | 14 - 49.8 | 27 – 95 % |
| Stern Review [62] | 1.4% | 9 – 32.8 | 17 – 62 % |
| UK standard for cost-benefit analysis [61] | DDR* | 4.4 -16 | 8 – 30 % |
| US standard: water & related land-use policy decisions [59] | 3.375% | 3.7 - 13.6 | 7 – 26 % |
| US FEMA [39] | 4.125% | 3 – 11.1 | 6 – 21 % |
| US Army Corps of Engineers [58] | 7% | 1.8 – 6.6 | 3 – 12 % |

* Declining discount rate defined by the UK Treasury for 100 years, then a 2.5% discount rate from 100 years onward.

Table S2.4 Model Comparison. Olson et al. [51] have created a regression model to estimate discharge in ungauged basins in VT. We use this model to estimate the change in discharge when wetland area is 0. The regression does not include floodplain storage, and as a result overestimates discharge for the downstream gauge when wetlands are present relative to our model and to recorded discharge measures. Our no-wetlands cases are analogous in representing cases where both wetland and floodplain effects are absent. This comparison illustrates that our simple assumptions approximate the results of more complex modeling efforts and the importance of floodplain storage, as well as wetland effects, in flood peak mitigation.

| | Our model | Statistical Regression [51] |
|-------------|--|--|
| Wetlands | 6,180 (wetlands and floodplains) | 20,700 (no floodplains) |
| No Wetlands | 27,100 (no wetlands or floodplains) | 27,481 (no wetlands or floodplains) |

Table S2.5 Summary of Biophysical Assumptions Made in Modeling Hydrographs

| No-Wetlands High Scenario | No-Wetlands Low Scenario |
|---|--|
| All floodwater recorded at the downstream gauge was impacted by wetlands and floodplains | Only water recorded at the upstream gauge was impacted by wetlands and floodplains |
| Water entering the creek between the upstream and downstream gauges does so proportionally to the timing of water observed at the upstream gauge. | Water entering the creek between the upstream and downstream gauges does so proportionally to the timing of water observed at the downstream gauge. |
| Water storage by other land-cover types is negligible. | Water storage by other land-cover types is negligible. |
| Channel storage, friction, and routing effects and natural peak dissipation with watershed size assumed to be negligible. | Channel storage, friction, and routing effects and natural peak dissipation with watershed size assumed to be equivalent to statistically derived average effects of 10-year floods in the area. |

Figures

Fig. 2.1 Map of the Otter Creek watershed. The Otter Creek flows northward from Rutland to Middlebury.

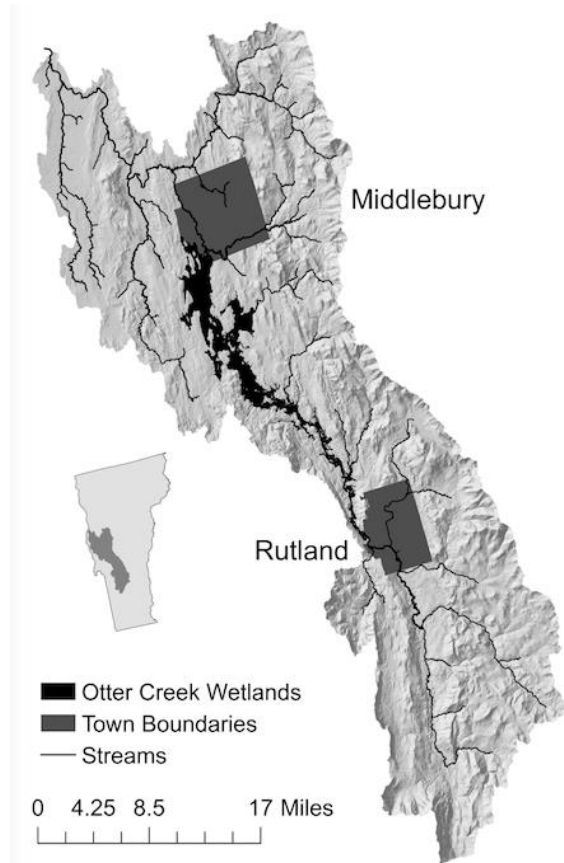


Fig. 2.2 Observed and modeled hydrographs for Otter Creek, VT.

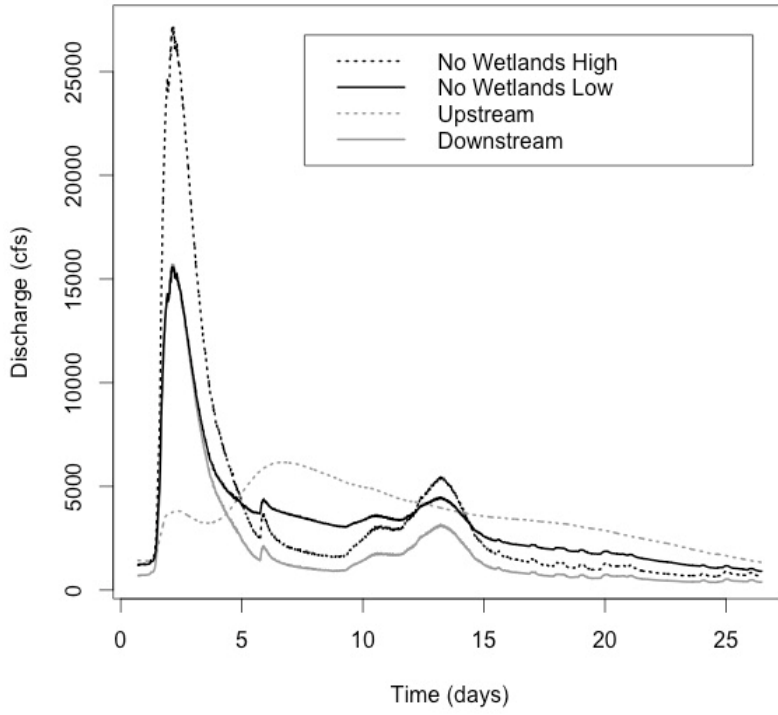


Fig. 2.3 Rating curve relating discharge and flood height at the downstream gauge
($r^2=0.96$, $p<2.2e-16$).

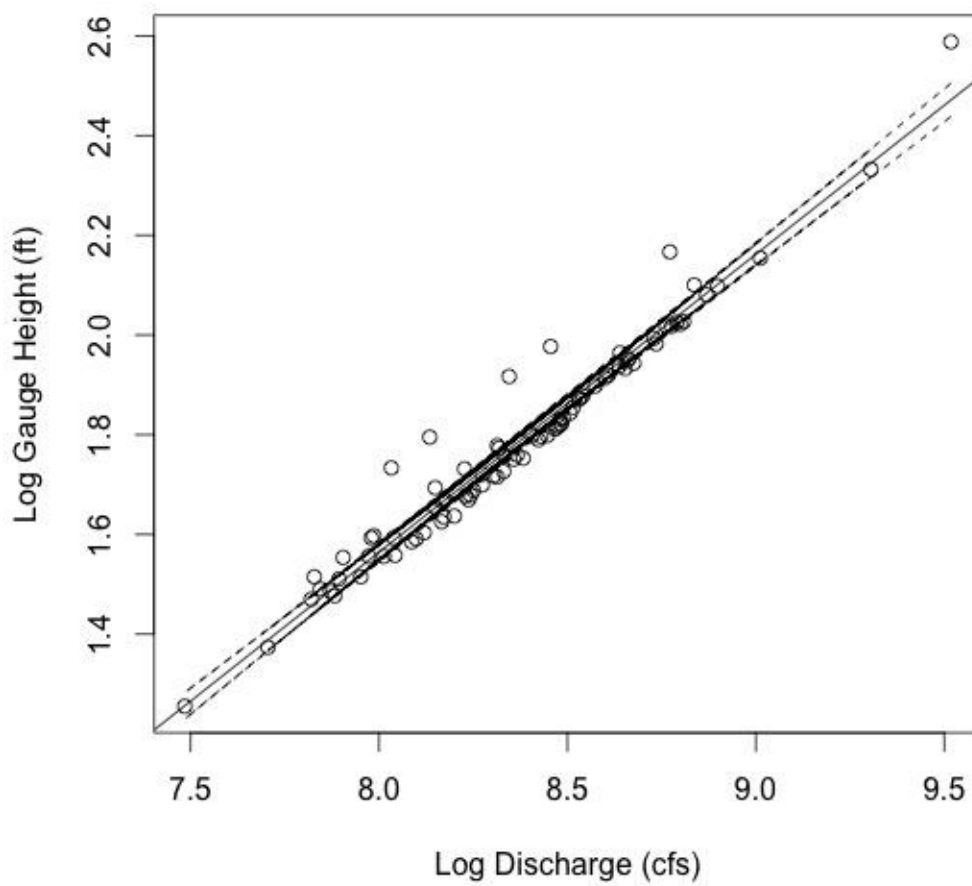


Fig. 2.4 Depth-damage curve used to relate flood depth of flooded structures to percent loss of the structure value due to flood damages [39].

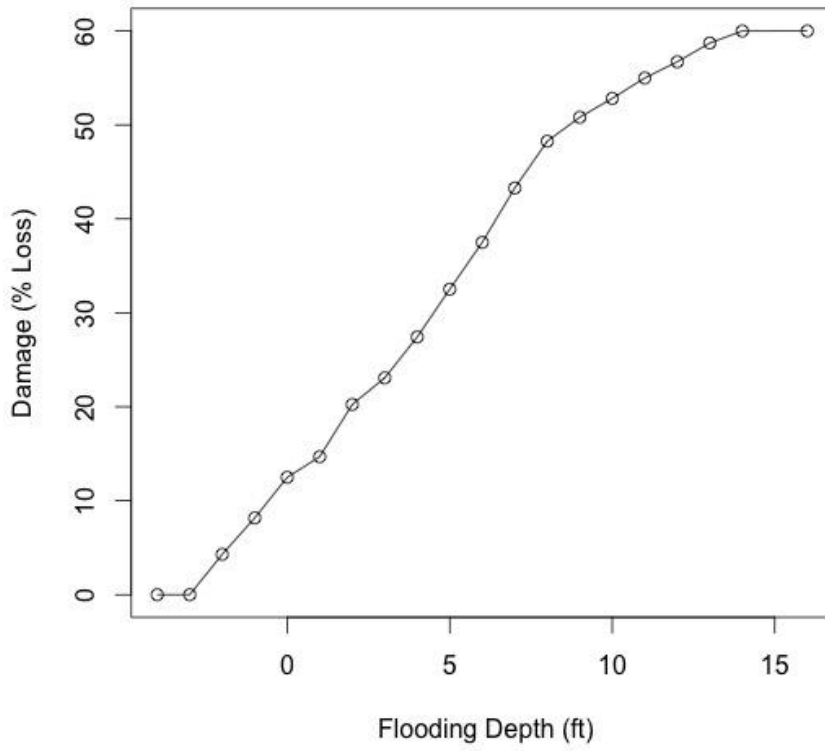


Fig. 2.5 Flood extent and damages to flooded structures in Middlebury following Tropical Storm Irene. Panel A: wetlands scenario, Panel B: no-wetlands low scenario, Panel C: no-wetlands high scenario.



Fig. 2.6 The percentage reduction in damages resulting from flood mitigation services as a function of the annual exceedance probability of ten historic floods. Hollow black: No-wetlands low, Solid black: No-wetlands high.

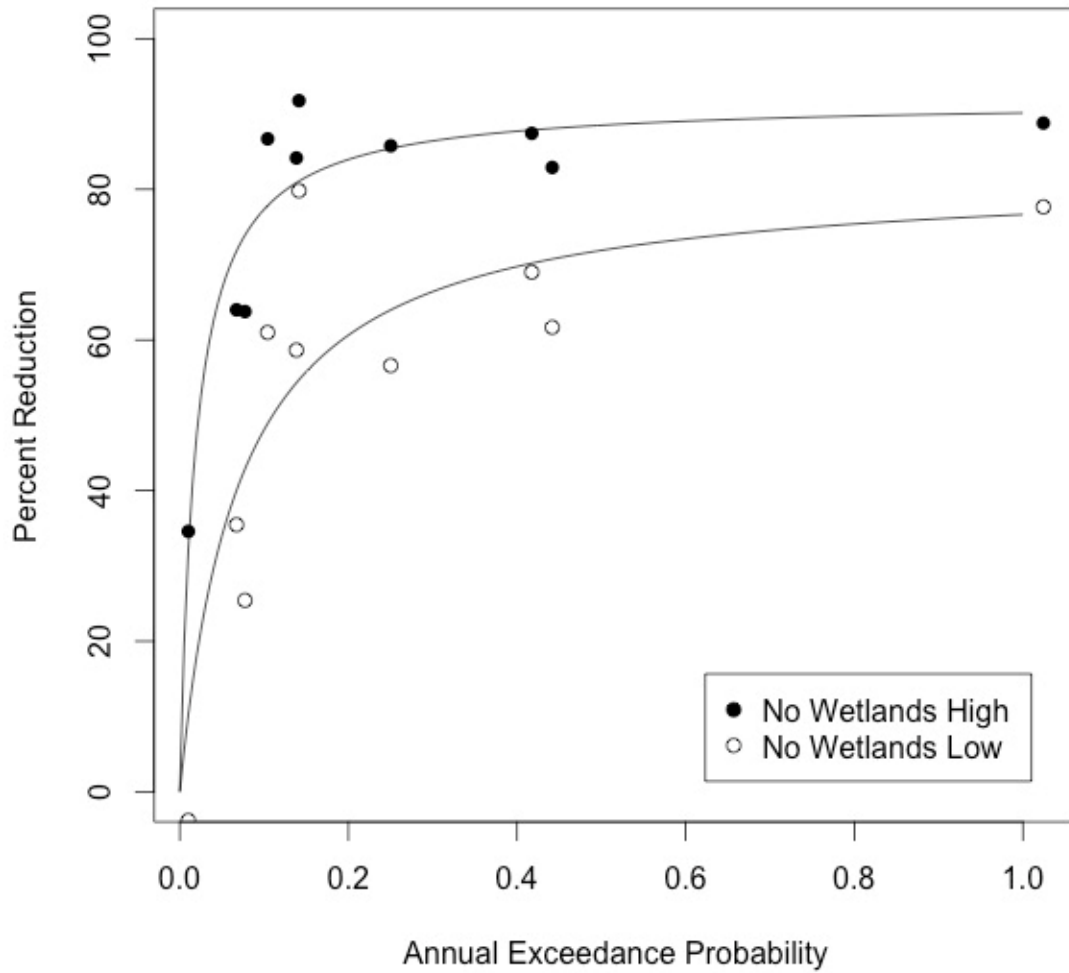


Fig. 2.7 Damage probability functions. Grey diamond: wetlands scenario

($D=e^{10.55757}p^{-0.48927}$, $p= 4.367e-07$, $r^2= 0.9646$), Open black circles: no-wetlands low scenario ($D=e^{12.02817}p^{-0.16884}$, $p= 0.1119$, $r^2= 0.2851$), Filled black circles: no-wetlands high scenario ($D=e^{13.11465}p^{-0.07055}$, $p= 0.5626$, $r^2= 0.04361$).

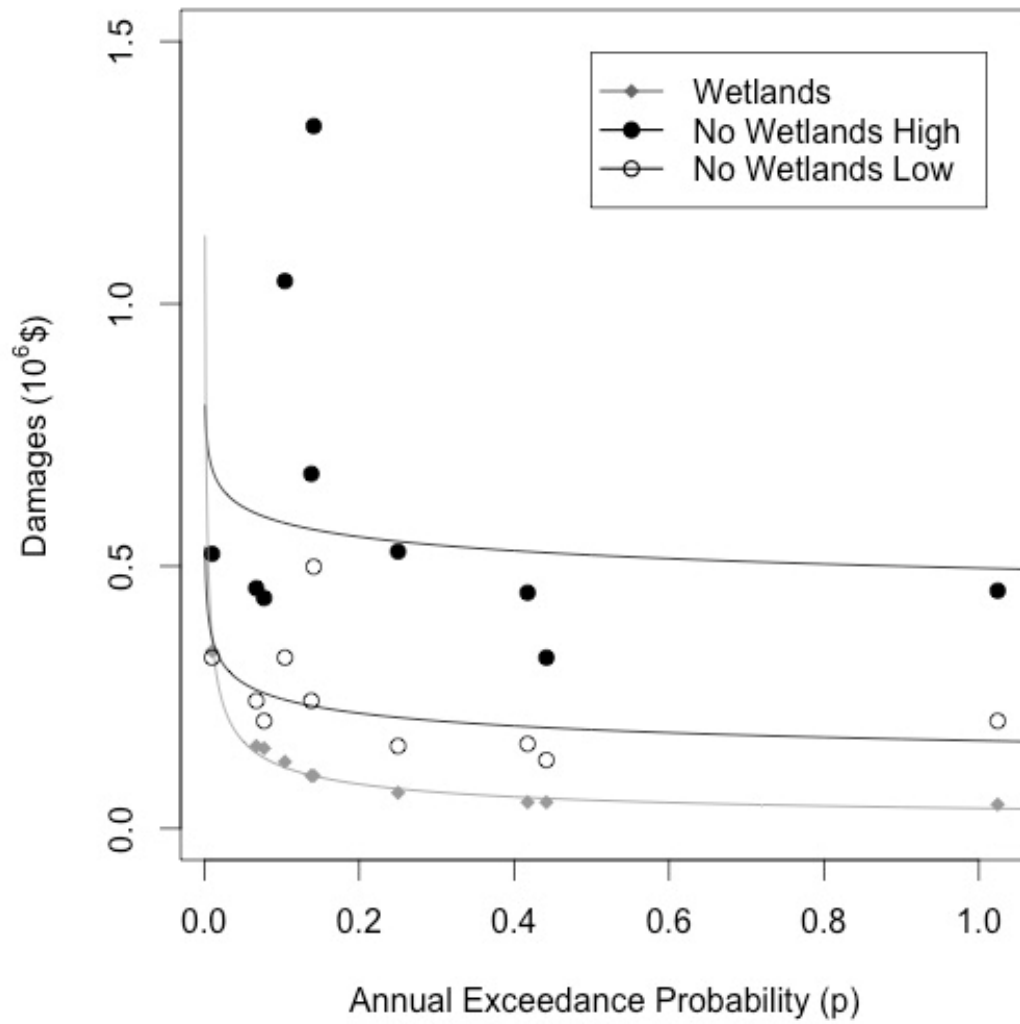


Fig. S2.8 Discharge per unit area as a function of drainage basin size for Vermont watersheds ($Q=e^{4.61406*} a^{0.16072}$, $p<0.001$, $r^2=0.2865$).

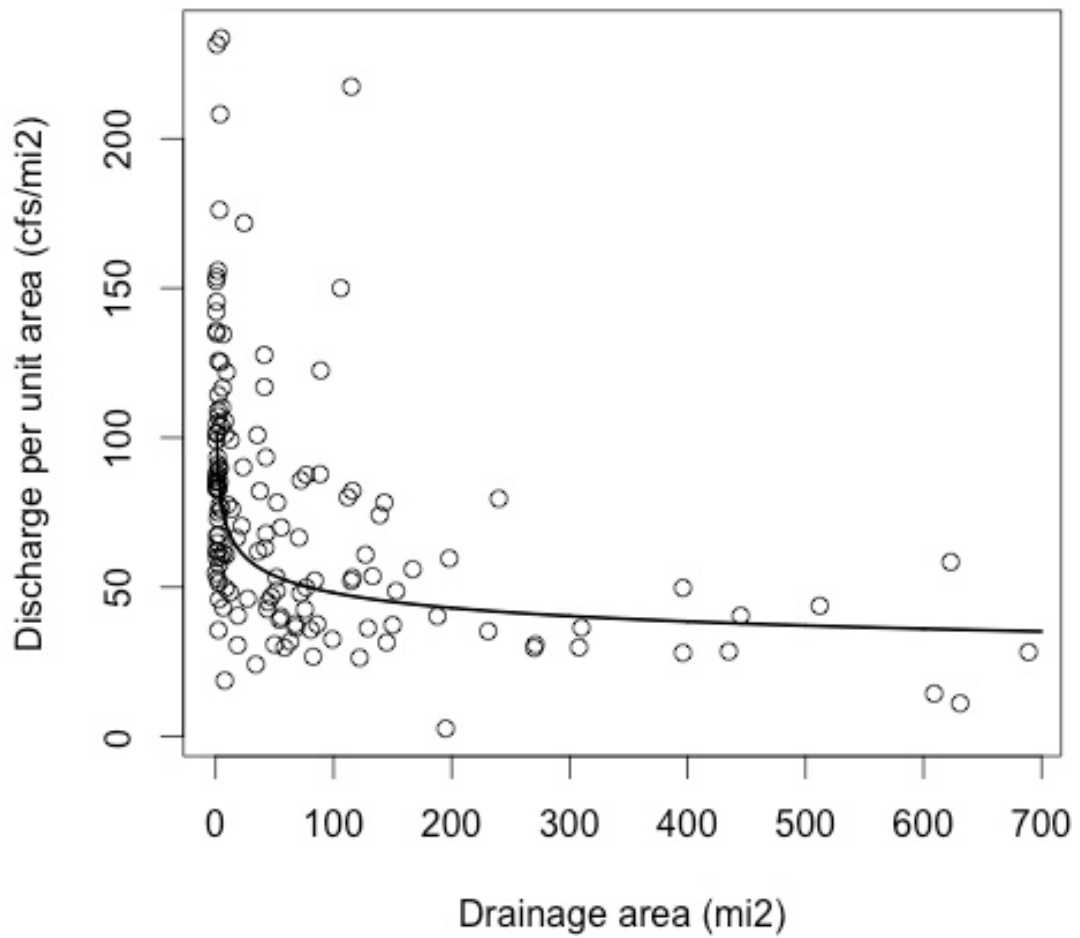


Fig. S2.9 Cross Section of the Otter Creek channel and floodplain at the downstream gauge with modeled flood elevations and the range of data used to determine the rating relationship.

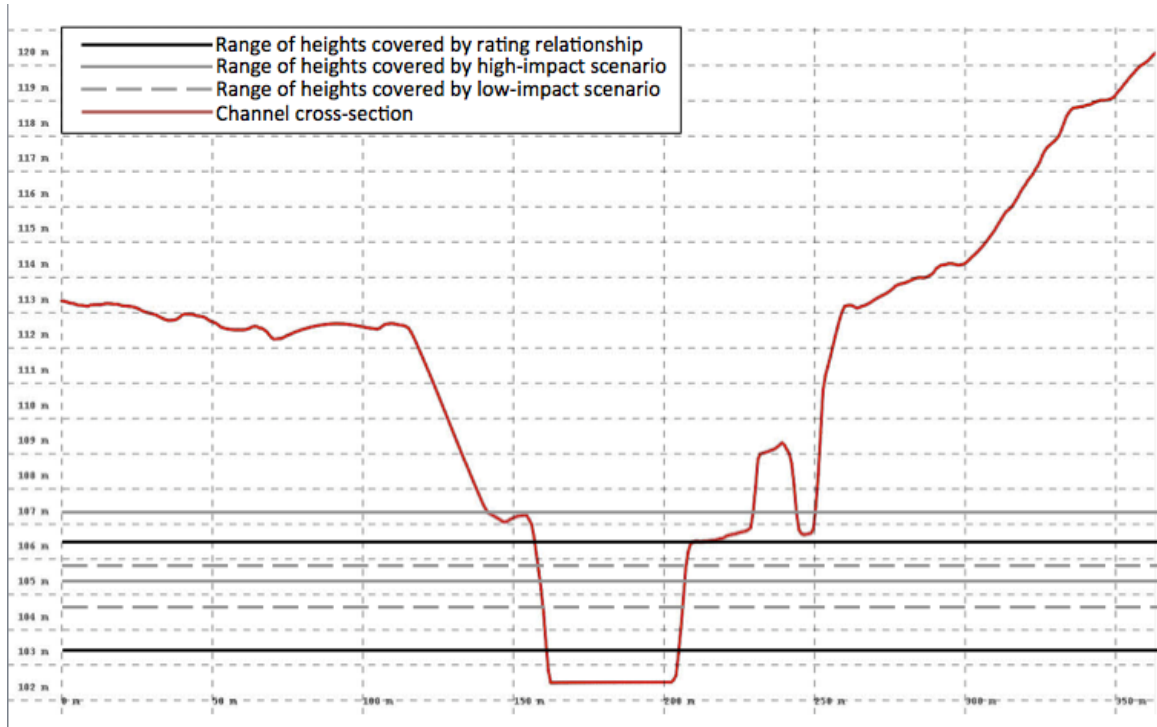
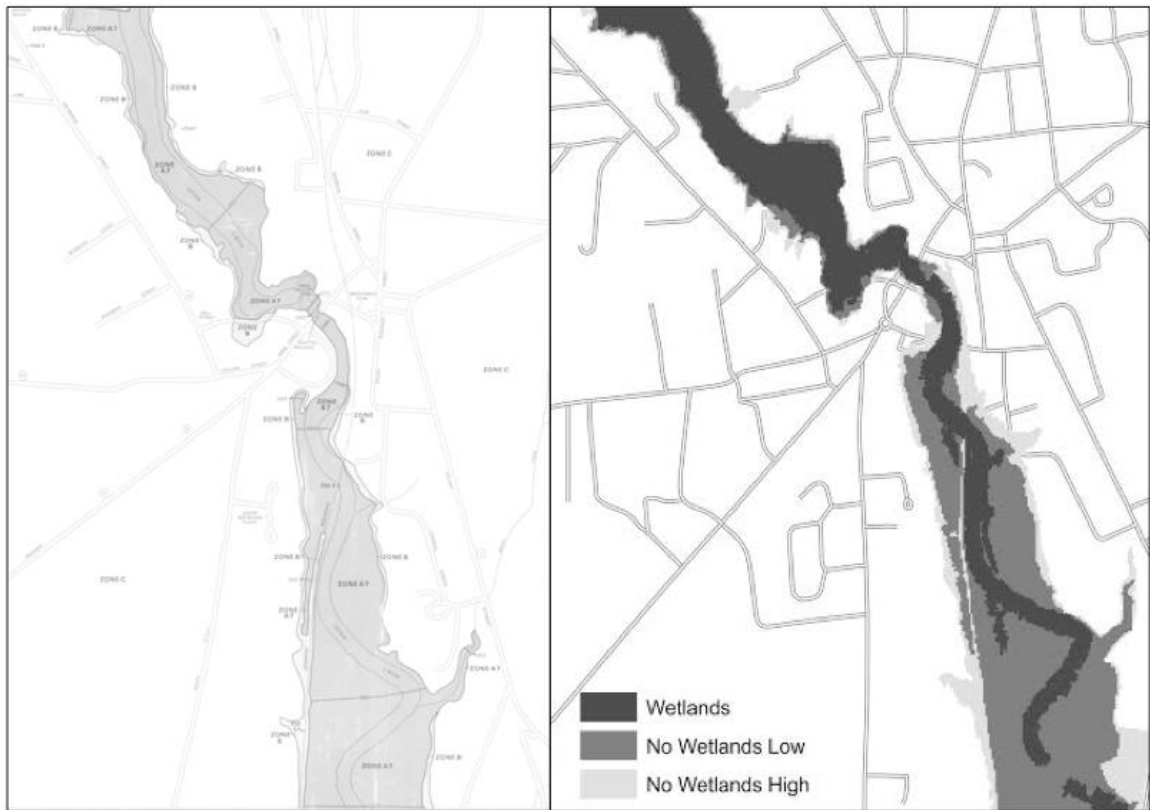


Fig. S2.10 Comparison of modeled flood extents to FEMA's flood insurance rate map for Middlebury [77].



CHAPTER 3: BENEFICIARY DEMAND IMPACTS THE HUMAN WELLBEING AND BIODIVERSITY OUTCOMES OF ECOSYSTEM SERVICE CONSERVATION

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Abstract

Ecosystem service conservation can contribute to human well-being and biodiversity conservation. The quantification of ecosystem services has focused on the biophysical supply of services with less emphasis on the role of demand from human beneficiaries, yet only when both occur to ecosystems benefit people. Here, we quantify the impact of demand on the human and biodiversity benefits of conserving ecosystem services. Using Vermont as a model landscape, we map three ecosystem services - flood mitigation, crop pollination, and nature-based recreation - and identify conservation priorities for each. We find that supply serves as a poor proxy for benefit because demand changes the spatial distribution of ecosystem services. Conservation that targets ecosystem services alone captures little biodiversity. However, when biodiversity and ecosystem services are jointly targeted, biodiversity outcomes are increased by 150% with just a 13% reduction in ecosystem services on average. Demand does not consistently reduce biodiversity outcomes; priority areas for supply and benefit captured roughly equal biodiversity co-benefit for all services. We conclude that incorporating demand is critical to efficiently protecting the benefits people derive from nature, and that doing so does not reduce biodiversity co-benefit.

Keywords

Ecosystem services, biodiversity conservation, spatial planning, beneficiaries

Introduction

Ecosystem services are the many benefits that nature provides to people, such as provisioning food, protection from storms, and cultural and spiritual values (Daily et al., 1997; Millennium Ecosystem Assessment, 2005). Widespread environmental change and degradation have decreased the capacity of ecosystems to provide non-market ecosystem services and to support biodiversity (Millennium Ecosystem Assessment, 2005).

Conservation organizations now recognize the importance of ecosystem services, and increasingly target them alongside biodiversity (Bateman et al., 2013; Ruckelshaus et al., 2013). Many hope that the human focus of ecosystem services will result in increased support for conservation, and thus an increase in the resources available to protect both ecosystem services and biodiversity (Balmford et al., 2002). However, others voice concern that allocating conservation resources to ecosystem services decreases the resources available for biodiversity conservation (Luck et al., 2012; McCauley, 2006; Reyers et al., 2012). It is critical that conservation organizations target their efforts to efficiently achieve ecosystem service goals while minimizing biodiversity losses.

Conservation planning requires information about which places are most important in providing ecosystem services and biodiversity (Kovacs et al., 2013; Withey et al., 2012). For ecosystem services, this involves both supply (i.e., the ecosystem functions that can potentially benefit people) and demand (groups of people who would benefit from that supply) (Fisher et al., 2009; Yahdjian et al., 2015). For example, riparian wetlands can dissipate flood peaks but this function only becomes a service if there are people

downstream who benefit from reduced flooding (Watson et al. 2016). If ecosystem service supply is used as a proxy for benefits, conservation projects may protect the supply of ecosystem services in places where people cannot access them. Some efforts map supply as a proxy when determining which places are most important for ecosystem service, because the data and models to do so are more readily available (Egoh et al., 2009; Lin et al., 2017; Maes et al., 2012). However, efficiently conserving ecosystem services requires understanding the spatial relationship between where ecosystem services are supplied, where people exhibit demand for them, and how services flow from sources of supply to sources of demand to produce benefits (Amy M. Villamagna, Paul L. Angermeier, Elena M. Bennett, 2013; Bagstad et al., 2014; Schröter et al., 2014/1; Serna-Chavez et al., 2014/4). Hereafter, we use “supply”, “demand”, and “benefit” to denote these concepts.

Incorporating demand into ecosystem service quantification may also impact the co-benefits to biodiversity provided by the places identified as most important in terms of ecosystem service (Balvanera et al., 2014). Benefit may be less tightly linked to biodiversity than supply (Cardinale et al., 2012), precisely because of this added human focus. Demand may weaken the functional link (Mitchell et al., 2013) and the spatial concordance (Reyers et al., 2012; Ricketts et al., 2016) between services and biodiversity, and in doing so reduce the biodiversity co-benefits from conserving ecosystem services.

Few efforts explicitly quantify the impact of demand on the spatial distribution of ecosystem services (Verhagen et al., 2016; Wolff et al., 2015/8). We modeled conservation priorities in Vermont, USA for three ecosystem services with and without incorporating demand in order to answer three questions. First, how does incorporating demand shift the spatial distribution of ecosystem services? Second, how much benefit is captured by conservation targeting supply? Third, how does demand alter the biodiversity co-benefits of ecosystem service conservation?

Methods

Overview

We answered these questions by following two basic steps. First, we mapped three ecosystem services - flood mitigation, crop pollination, and nature-based recreation - in terms of the biophysical supply of the service, and then as benefit (the interaction of supply and demand). We compared the distributions of supply and benefit for each ecosystem service to assess how demand affects the distribution of ecosystem services.

Second, we simulated optimal networks of conserved lands for each ecosystem service supply, ecosystem service benefit, and biodiversity using the optimization program Marxan (Ball, I.R., H.P. Possingham, and M. Watts., 2009). We then compared the effectiveness of the resulting networks in capturing both ecosystem service benefits and biodiversity.

Quantifying Ecosystem Services

Using Vermont as a case study, we quantified ecosystem service supply, demand, and benefit for three locally important ecosystem services: flood mitigation, nature-based recreation, and crop pollination (Table 3.6). Our model landscape comprised 4462 hexagonal polygons, each 5.85 km² in area, approximately the average size of existing conserved lands in Vermont (mean=6.7km², median=10.1km² ((The Nature Conservancy, 2012). We aggregated supply, benefit, and biodiversity to the hexagon scale by taking the sum of all contained pixels.

Flood Mitigation

Flood outcomes are determined by the quantity and timing of water entering river channels, and by the hydraulic properties of a river's channel and floodplain. Quick-flow is the portion of water that moves quickly to a channel via surface runoff or interflow, and is the portion of runoff likely to generate a flood. We quantified flood mitigation supply as the retention of quick-flow by natural land-cover types relative to pasturelands (the dominant anthropogenic land-cover class in our study area). Channel and floodplain effects are beyond the scope of this work. We quantified quick-flow using the InVEST monthly water yield model (Sharp et al., 2014). This model estimates quick-flow as the portion of runoff with a residence time of hours to days, as a function of soil type, topography, precipitation, and land-cover. It adapts a curve number approach (Mockus, 2004) to a pixel resolution and a monthly time step, and has been shown to effectively approximate the proportion of rainfall that runs off as quick-flow across the continental

U.S. (Andrew J. Guswa, Perrine Hamel, Kate Meyer, 2017), and New England (Blumstein and Thompson, 2015). We parameterized the model (Table S2.8) to represent the generation of quick-flow from rainfall events onto saturated soils (ARCI conditions, (Mockus and Hjelmfelt, 2004)), and then produced a supply index by calculating standardized quick-flow for each pixel on a zero to one scale. The curve number approach is not appropriate for snow. Historically, Vermont has not received rainfall in winter months, but in recent years rainfall has occurred year round, although winter months remain snow dominated. We calculated our supply index with and without winter months included. The resulting indices were essentially identical (Figure S3.15), so in the subsequent analyses we use the 12 month supply index.

We defined demand for flood mitigation as the number of downstream buildings at risk of flooding. We overlaid a spatial dataset of buildings (E911 Board, 2013) and a dataset of floodplain areas (Sangwan and Merwade, 2015) in ArcGIS (ESRI (Environmental Systems Resource Institute), 2012) to identify at risk buildings. We then used the InVest DelinateIT model (Sharp et al., 2014) to delineate the watershed draining to each floodplain polygon that contained buildings. We assigned a “demand” score to each pixel where each structure equated to one unit of demand which was distributed evenly to each pixel in its drainage. For example, a single home with a 10-pixel drainage would place 0.1 units of demand on each pixel in its drainage. The per pixel demand for flood mitigation service was calculated as the sum of demand from all downstream structures at risk of flooding. We standardized all demand scores on a scale of zero to one.

The relative importance of each pixel in mitigating floods was taken as the product of supply and demand. This multiplicative effect represents the interaction of supply and demand to produce benefit; if either supply or demand is zero, benefit is also zero. By taking the unweighted product of supply and demand we assume both are equally important in determining ecosystem service benefit. Our results are highly insensitive to this assumption. (Figure S2.16). All calculations were performed at a 30 m resolution.

Nature-based Recreation:

We quantified recreation benefit as the visitation rate from nature-based recreants. To do this, we used geo-tagged photos on the website Flickr (www.flickr.com) to estimate visitation as a function of several different characteristics of conserved lands using an existing model of recreation services for Vermont (Sonter et al., 2016). We divided the predictor variables used to estimate recreation service into three sets: 1) landscape attributes (forest cover, slope, opportunities to swim, and opportunities to ski); 2) a demand variable (mean population density within a 25 km radius); and 3) development attributes of publicly accessible protected areas (trail density, area of the conserved land). We estimated each landscape attribute and demand variable for all hexagons, and assigned all hexagons the mean trail density of existing conserved lands under the assumption that they could be developed as a typical protected area. Then we applied the regression model developed by Sonter et al. 2016 to estimate visitation to each hexagon.

We predicted visitation with the demand predictor included in the model to quantify benefit, and again without it to quantify supply.

Crop pollination:

We used existing information on the abundance and demand for wild pollinators (Koh et al., 2016), that were based on a published model of wild bee abundance (Lonsdorf et al., 2009), the U.S. National Agricultural Statistics Service Cropland Data Layer (USDA-NASS, n.d.), and expert opinion based habitat suitability of wild bees across the U.S. We used Koh et al. (2016) estimates of wild bee abundance as our measure of supply, and their map of pollinator-dependent crops as our measure of demand. We then quantified pollination benefit as the number of wild bees foraging on pollinator-dependent crops by clipping bee abundance to the extent of pollinator dependent crops at a thirty-meter resolution.

Quantifying Biodiversity Value

To quantify biodiversity, we used BioFinder, a statewide map of conservation priorities for biodiversity provided by the Vermont Agency of Natural Resources (Austin et al., 2013). BioFinder is already in use by conservation groups in Vermont, so the relationship between its priorities and ecosystem services has direct management relevance.

BioFinder combines 21 different datasets to identify “high priority ecosystems, natural communities, habitats, and species.” We used the combined priority score, which was determined as the weighted sum of the scores from each component, as the biodiversity measure in our analyses. This dataset does not represent biological richness per se, but it

does represent prioritization of different locations in terms of their value for biodiversity conservation. For instance, interior forest blocks, connectivity blocks, riparian wildlife connectivity, surface waters and riparian areas, and physical landscape diversity are included as biofinder components because maintaining these features is likely to conserve the majority of Vermont's species at landscape scales. Other components relate to specific aspects of diversity at the community scale, such as rare species, vernal pools, and rare natural communities (Austin et al., 2013).

Costs of Conservation

We used land value to approximate the relative costs of conservation. For roughly 50% of our study area, public tax records of property values could be associated with digitized parcel maps. We developed a regression model to estimate the remaining unknown land values. Because land values are spatially correlated, we built a generalized additive model with socioeconomic predictors and a spline smoother for spatial location (Bivand, 2008). We found that distance to cities, median household income, predominant land-cover, density of built structures, road density, and the presence of urban centers explained just over 50% of the variation in log transformed land costs ($r^2=0.532$, $df=16$, all coefficients significant at $p<0.05$). The spline term significantly improved the model (approximate $p<2.2e^{-16}$, all coefficients significant at $p<0.05$) (Figure S3.17). We used the predicted log transformed land cost as an index of relative costs of conservation. We use log-transformed costs because the extremely large variation in untransformed land costs likely does not apply to conservation investments; it is driven by the density and

prevalence of developed land in each hexagon, and these developed areas are unlikely to be targeted by conservation efforts. Further, untransformed land costs varied much more widely than did ecosystem service supply and benefit, and their variation otherwise overwhelmed the differences between the two when selecting optimal conservation priorities.

Comparison of supply and benefit:

We compared the frequency distributions of supply and benefit for each ecosystem service at the hexagon scale using a two-sided Kolmogorov-Smirnov test. We tested the cross-autocorrelation of the supply and benefit of each ecosystem service in space using the centered Mantel statistic implemented using the “ncf” package in R (Bjornstad, 2009).

Identifying Conservation Priority Areas:

We performed optimizations to identify priority areas for conservation with and without the influence of demand based on four different targeting scenarios: supply alone, benefit alone, supply and biodiversity, benefit and biodiversity (Table 3.7). We also performed an optimization for biodiversity alone as a control in assessing biodiversity co-benefits. Identifying joint spatial priorities for biodiversity and ecosystem services provides a clearer picture of the opportunities to achieve both than assessing their spatial correlation. Correlations reflect similarities between places with both low and high value, but only

high value areas are relevant in the context of spatial planning for conservation. Even if correlation overall is low, there may still be locations that provide win-win opportunities.

We used the optimization software Marxan (Ball, I.R., H.P. Possingham, and M. Watts., 2009) to identify priority areas for the supply and benefit of each ecosystem service under each targeting scenario (Table 2.7) and for biodiversity alone, for a total of 13 simulations. Marxan uses simulated annealing to approximate optimal conserved lands networks given the value and cost of each unit of analysis, by minimizing the objective function:

Equation 3.1:

$ObjFun_{min} =$

$$Land\ Cost_{(x,y)} + \lambda (Protection\ target - Protection\ achieved)_i + Cost\ constraint$$

Where:

Land Cost = the sum of our land cost index for all hexagons within the selected priority areas

i = the conservation features being targeted (in our case biodiversity, ecosystem service supply, or ecosystem service benefit)

Protection target = the target amount of a conservation feature that the optimization seeks to achieve.

Protection achieved = the amount of a conservation feature held within the selected priority areas

λ = the “species penalty factor” for missing a conservation feature’s protection target: essentially a weighting of the importance of each conservation feature. We set equal weights for biodiversity and ecosystem service

Cost constraint = a penalty for exceeding a user defined cost constraint. We set

this penalty high enough such that the solution never exceeded our constraint.

We set a cost constraint that allowed approximately 15% of the landscape to be selected, and then set protection targets that were impossible to reach given that constraint (50% of statewide supply, benefit, or biodiversity), such that minimizing the objective function above never involved exceeding the cost threshold, and always involved maximizing the protection of conservation features within that constraint.

We performed 500 runs for each simulation, and used the reported “best solution” as our set of priority area (Ball, I.R., H.P. Possingham, and M. Watts., 2009). In effect, this process identified priority areas for ecosystem service and biodiversity as though we redesigned conserved lands today based on these criteria, and set aside approximately the same amount of land area that is currently protected.

We summed the values of ecosystem service and biodiversity across all hexagons within priority areas. We compared the amount biodiversity and ecosystem service within priority areas for each optimization scenario to assess the impact of demand on ecosystem service and biodiversity co-benefits. We compared single factor optimizations to multi-factor optimizations to assess the potential for achieving biodiversity and ecosystem service goals simultaneously.

Results

Demand shifts ecosystem services

Demand shifts the spatial distribution of each ecosystem service (Figure 3.11A), although supply and benefit were highly correlated for nature-based recreation (Figure 3.11A; pollination $r_s=-0.13$, flooding $r_s=0.26$, recreation $r_s=0.95$, $p<2.2e^{-16}$ in all cases). The frequency distribution also differed between supply and benefit in all three cases, but this difference was much smaller in the case of recreation (Figure 3.11B; pollination $D=0.95$, flooding $D=0.80$, recreation $D=0.20$, $p<2.2e^{-16}$ in all cases). These differences in spatial and frequency distributions were reflected in the conservation priority areas. These areas were similar for supply and benefit in the case of nature-based recreation, but noticeably different for flood mitigation and crop pollination (Figure 3.11C).

Supply as a proxy for benefit

In the case of crop pollination and flood mitigation, priority areas directly targeting benefit captured much more benefit than did priority areas for supply (Figure 3.12). Priority areas composed of only about 12.2% and 14.4% of the landscape captured 50% and 90% of benefit in the cases of flood mitigation and crop pollination respectively. In contrast, for nature-based recreation these two strategies captured the same amount of benefit and only 17% of benefit could be captured given our budget constraint.

Biodiversity co-benefit

Across all three services, single factor optimizations captured on average 24% of the biodiversity that could be captured by targeting biodiversity directly with the same budget constraint. In the case of flood mitigation, we find that priorities for benefit capture less biodiversity co-benefit do those for supply (Figure 3.13). Priority areas for benefit and supply contain similar biodiversity co-benefit for pollination and recreation. For all services, multi-factor optimization improved biodiversity co-benefit. Across the six possible comparisons, targeting ecosystem service and biodiversity jointly increased the amount of biodiversity within priority areas by 150% on average while reducing ecosystem service by just 13%.

Multi-factor optimization also shifted the spatial distribution of priority areas relative to single factor optimizations (Figure 3.13: blue vs. red). For flood mitigation and pollination benefit, the new places selected in multi-factor optimizations included locations that were relatively important for both biodiversity and benefit, but also the places that were most important for biodiversity regardless of how much benefit they contained (Figure 3.14). For flood mitigation, multi-factor optimization priority areas for benefit were distributed across the full range of benefit and biodiversity importance (Figure 3.14e) whereas priority areas for supply were more concentrated in the upper fifty percentiles for both benefit and biodiversity (Figure 3.14b). For crop pollination many locations fell within the top fifty percentiles for both supply and service, and many of these were included as conservation priorities (Figure 3.14a), whereas there were

relatively few locations that were important for both benefit and biodiversity, and as a result priority areas were bi-modally distributed in places highly important for one or the other (Figure 3.14f).

Discussion and Conclusions

Demand shifts the spatial distribution of ecosystem service benefit relative to supply. These differences have implications for conservation efforts that seek to benefit both people and biodiversity. Supply is a poor proxy for benefit, and targeting supply does not capture more biodiversity than targeting benefits directly. Single-factor priority areas for supply and benefit alike capture little biodiversity. However, joint targeting greatly improves biodiversity outcomes with minimal losses of ecosystem service. In sum, our results indicate that incorporating demand increases the efficiency of ecosystem service conservation at capturing benefits without reducing biodiversity outcomes.

The differences between supply and benefit maps reflect the relative distributions of supply and demand, as well as the distances over which benefits can flow to beneficiaries. Demand affects the distributions of ecosystem services in two distinct ways (Figure 3.11).

- 1) Concentration: In the cases of flood mitigation and crop pollination, demand

concentrates ecosystem service into select places on the landscape such that small portions of supply are highly valuable, whereas most supply provides little ecosystem service (Figure 3.11, A-B).

2) Spatial Shift: In the case of recreation, the frequency distribution of supply was very similar to that for benefit, and demand only slightly shifted the spatial distribution of ES towards population centers (Figure 3.11, E-F).

Supply is concentrated if it provides greater benefit when it is nearby demand or in a small service shed. Service-sheds are the areas that benefit a source of demand (Mandle et al., 2015; Tallis et al., 2012). When service sheds vary in size, the marginal impact of losing a given quantity of supply will be highest in small service-sheds that do not have much supply to start with (Fisher et al., 2008). For example, the size of a service-shed for flood mitigation is the size of the watershed draining to a cluster of at-risk buildings. These watersheds varied widely in size, but ecosystem service was most concentrated in small watersheds. In the case of pollination, supply only provides benefit when it is very nearby demand; The flow of pollination services is limited by the flight distance of bees, which is very small compared to the statewide scale of our analysis. As a result, pollinators only provide benefit when they are very close to crops that require insect pollination.

In the case of recreation, demand spatially shifted service slightly towards population centers, without altering the frequency distribution of benefit relative to supply. We expect this type of spatial shift to occur where service flows extend far enough to connect all sources of supply to some source of demand. In the case of recreation in Vermont, flows of service extend across most of the extent of our analysis because recreants in Vermont are willing to travel to obtain recreational opportunities, which are generally available within a two-hour drive. As a result, any location that supplied recreational opportunities provided some benefit to people even though places nearby population centers benefited people more. At its extreme, for some ecosystem services all sources of supply may provide equal benefit. This would be the case for climate regulation; carbon sequestered in forests affects climate globally (Bonan, 2008; Cramer et al., 2004).

When demand concentrates ecosystem service benefits, choice of ecosystem service measure has important impacts on conservation priorities. In these cases, policies and management actions that spatially target supply are much less effective in safeguarding benefits than efforts that target benefits directly by accounting for demand (Figure 3.12). In the spatial shift case, where service flows connect all sources of supply to a source of demand, supply may serve as an acceptable proxy for benefit. Although further study is needed to test the generalizability of these two different cases, our results indicate that understanding the spatial dynamics between ecosystem supply and demand can inform when quantifying demand is critical (concentration), and when doing so will result in smaller efficiency gains in conservation planning for ecosystem services (spatial shift).

When demand concentrates ecosystem service, spatial planning can be particularly efficient because actions taken on a small portion of the landscape will have disproportionately large ecosystem service benefits. This efficiency gain is true in any case of spatial targeting where ecosystem service value is unevenly distributed. However, demand makes the distribution of value even more uneven by concentrating it around beneficiaries (Figure 3.11 B and D), thus potential efficiency gains are larger for benefits than for supply (Figure 3.13).

Programs that target ecosystem service alone are unlikely to provide high levels of biodiversity. For example, it is not safe to assume that a program targeted to restore hydrologic function of forested headwaters for flood mitigation will occur in places important for biodiversity by chance (Figure 3.13). Many conservation organizations have begun to target ecosystem services in addition to biodiversity (Reyers et al., 2012), which is better represented by our multi-factor optimizations than single-factor optimization scenarios. Jointly targeting biodiversity and ecosystem service through a multi-factor optimization doubles biodiversity outcomes relative to single factor optimizations with minimal impact on ecosystem services (Figure 3.13). For example, if a conservation organization sought out opportunities to protect places important for nature-based recreation and for biodiversity conservation, our analysis indicates such opportunities exist (Figures 3.13, Figure 3.14). This does not mean that there is no biodiversity tradeoff in targeting ecosystem services; any time additional targets are

added, there will be some tradeoff. In our case, equally weighting service and biodiversity in multi-factor optimization caused a larger tradeoff for biodiversity (31% reduction relative to a single factor optimization for biodiversity) than for service (13% reduction relative to single factor optimizations for service).

Demand does not consistently exacerbate this tradeoff. Human demand is the component of ecosystem services makes them distinct from other measures of ecological health or function (Fisher et al., 2009), and is the source of concern that targeting ecosystem services will shift conservation priorities towards human-dominated landscapes (Reyers et al., 2012). For one of three services (flood mitigation), the single-factor optimization for supply outperformed single factor optimizations for benefit in terms of biodiversity, and all multi-factor optimizations for benefit captured roughly the same amount of biodiversity as the comparable multi-factor optimization for supply. Because benefit and supply are distinguished by the incorporation or omission of demand, this indicates that the human-focused component of ecosystem service, which critical in efficiently capturing benefits to people, does not reduce biodiversity outcomes.

Incorporating demand may also provide an opportunity to simultaneously conserve ecosystem service and biodiversity. The biodiversity gains of joint optimizations were not always achieved by conserving places that are important for both biodiversity and service (Figure 3.14). When service is concentrated, a large portion of service value can be captured in relatively little area, allowing the remaining budget to conserve areas of high

biodiversity regardless of ecosystem services (Figure 3.14 e,f). While this result is sensitive to the budget constraint, our fifteen percent constraint approximates the land area currently protected in Vermont and is likely to be reasonable in many conservation contexts. In effect, even when unit-by-unit co-occurrence of ecosystem service and biodiversity is low, and conservation resources are limited, both can be effectively protected through spatial planning. This opportunity arises as a result of demand concentrating service value, and occurs even when demand concentrates value in places that are less important for biodiversity (flood mitigation).

Large efficiency gains can be achieved when information on the spatial distribution of ecosystem service and biodiversity value is available. Conservation efforts fall short of effectively safeguarding the benefits from nature to people unless they consistently incorporate demand, the people-focused half of that relationship, in spatial planning. Incorporating demand will allow efforts to be targeted towards the places that benefit people the most, and will not reduce the biodiversity co-benefits of these actions. While conserving ecosystem service may not always efficiently capture biodiversity, joint targeting of biodiversity and ecosystem services improves biodiversity outcomes with only minimally reducing ecosystem service outcomes. In sum, incorporating demand is critical to safeguarding nature's benefits to people, doing so does not reduce the biodiversity co-benefit of ecosystem service conservation.

Acknowledgements

I would like to thank Donna Rizzo, Lisa Mandle, Charlie Nicholson, and Nitin Singh for their thoughtful feedback, which improved the quality and clarity of this manuscript. I would also like to thank the Lintilhac Foundation, the USDA McIntire-Stennis award #2014-32100-06050 to the University of Vermont, and the Rubenstein School of Natural Resources for funding that supported this research.

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Tables

Table 3.6 Ecosystem service supply and benefit as defined in our analysis.

| | Supply | Benefit |
|--------------------------------|--|---|
| Flood Mitigation | Retention of quick-flow by natural ecosystems relative to pasture, the dominant anthropogenic landscape. | Retention of quick-flow weighted by the number of downstream structures in a flood risk area. |
| Nature-based Recreation | Visitation by recreants as a function of natural landscape features. | Visitation by recreants as a function of landscape features and surrounding population density. |
| Crop Pollination | Wild bee abundance | Number of wild bees foraging on pollinator-dependent crops. |

Table 3.7 Summary of the optimization scenarios used in our analysis

| <i>Optimization Scenarios</i> | | | |
|-------------------------------|--------------|----------------------------------|-----------------------------------|
| <i>Optimization</i> | | <i>De-coupled from Demand</i> | <i>Linked to Demand</i> |
| <i>Single factor:</i> | Biodiversity | <i>Supply</i> | <i>Benefit</i> |
| <i>Multi-factor:</i> | | <i>Supply & Biodiversity</i> | <i>Benefit & Biodiversity</i> |

Table S3.8 Input data and parameterization of the InVest Seasonal Water Yield model.

| Model Input | Data Source |
|---|--|
| Average monthly precipitation | We downloaded 30 year monthly precipitation normals available at an 800m resolution from the PRISM Climate Group. These normals covered the period from 1981-2010 at the time of download (PRISM Climate Group, 2012). |
| Monthly reference evapotranspiration | Reference evapotranspiration data was derived from CCIGAR's globally available data on potential evapotranspiration (Trabucco and Zomer, 2009). |
| Land-cover | Land-cover data was derived from the national landcover dataset (2011) (Homer et al., 2015). |
| Hydrologic soil group | Hydrologic soil group obtained from SSURGO soils data (USDA Natural Resources Conservation Service, n.d.). No data values were assigned the value C because this hydrologic group was the most common within Vermont (comprised a larger total area than any other hydro-group). Open water pixels were assigned to group D. |
| Curve numbers for each soil type/land-cover combination | <p>We adopted standard curve numbers for each NLCD land-cover class and soil hydrologic group under wetter antecedent runoff conditions (ARC III) (Victor Mockus, 2004) as follows:</p> <p>(NLCD classification - NEH Cover description treatment (Mockus, 2004))</p> <p>Developed open space - Open space, good condition</p> <p>Developed low intensity - Residential districts: lot size 1/4 acre</p> <p>Developed, medium intensity - Residential districts: lot size 1/8 acre or less</p> <p>Developed, high intensity - Urban districts: commercial and business</p> <p>Barren land - Bare soil</p> |

| | |
|--|---|
| | Deciduous forest - Woods good, condition |
| | Evergreen forest - Woods good, condition |
| | Mixed forest - Woods good, condition |
| | Shrub scrub - Brush-forbes-grass mixture, good condition |
| | Herbaceous - Brush-forbes-grass mixture, good condition |
| | Hay/pasture - Pasture, grassland, or range-continuous forage for grazing, good condition |
| | Cultivated crops - Straight row- good condition |
| | Woody wetlands - Woods, good condition |
| | Emergent herbaceous wetlands - Woods, good condition |

Figures

Fig. 3.11 A) The spatial distribution of supply and benefit for crop pollination, flood mitigation and nature-based recreation, B) Density distribution of supply and benefit for each, and C) conservation priority areas identified via single-factor optimizations.

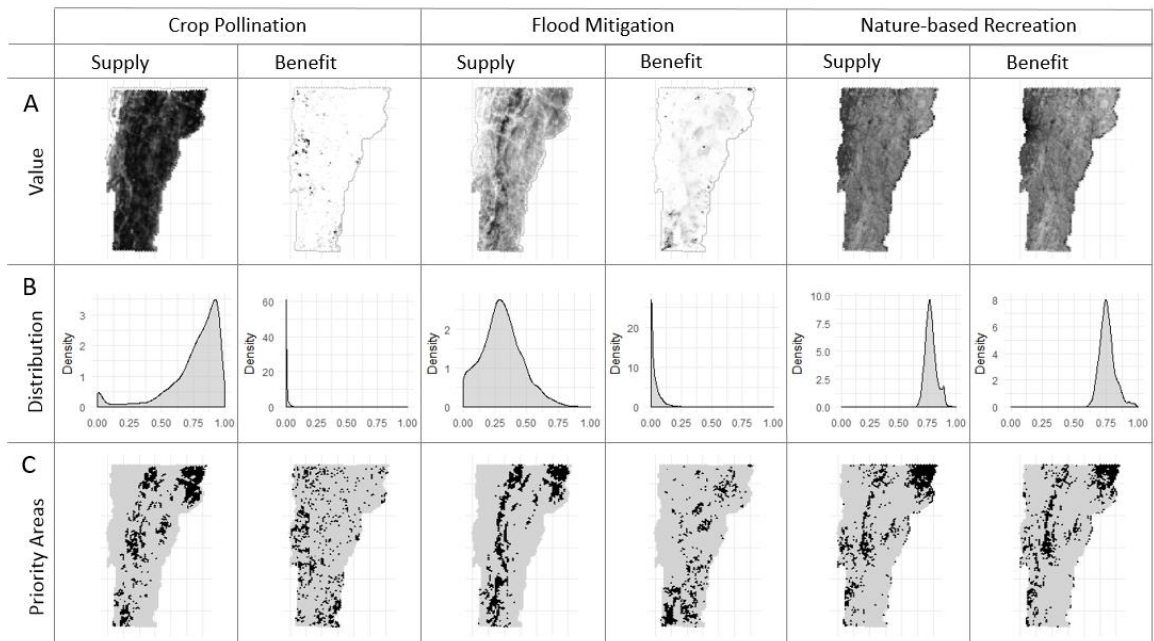


Fig. 3.12 Proportion of benefit in priority areas targeting supply and benefit for crop pollination, flood mitigation, and nature-based recreation.

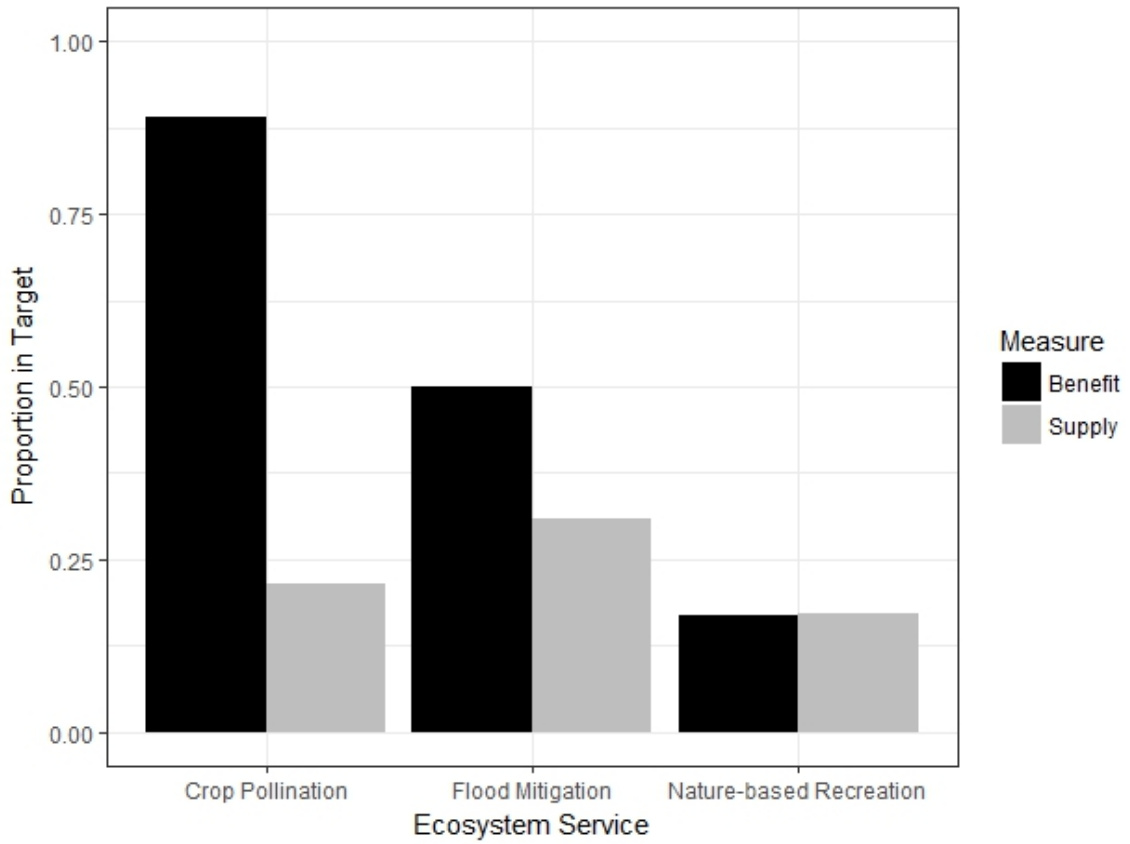


Fig. 3.13 The ecosystem service and biodiversity contained within priority areas for single factor and multi-factor optimization strategies (top). Maps of ecosystem service priority areas (bottom). Single factor optimizations are shown in blue, multi-factor optimizations are shown in red, and locations that were within the conservation target for both multi-factor and single factor optimizations are shown in purple.

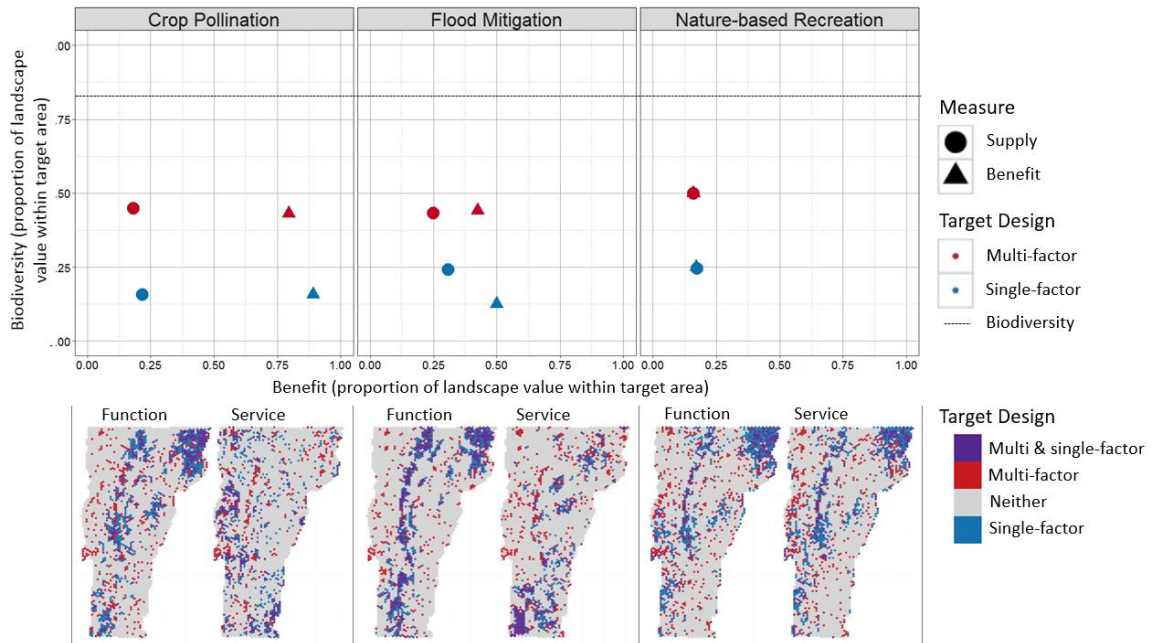


Fig. 3.14 The percentile rank of each unit of analysis in terms of return on investment for ecosystem service (supply or benefit) on the x axis, and biodiversity on the y axis (return on investment was calculated as the ratio of ecosystem service or biodiversity value to conservation cost).

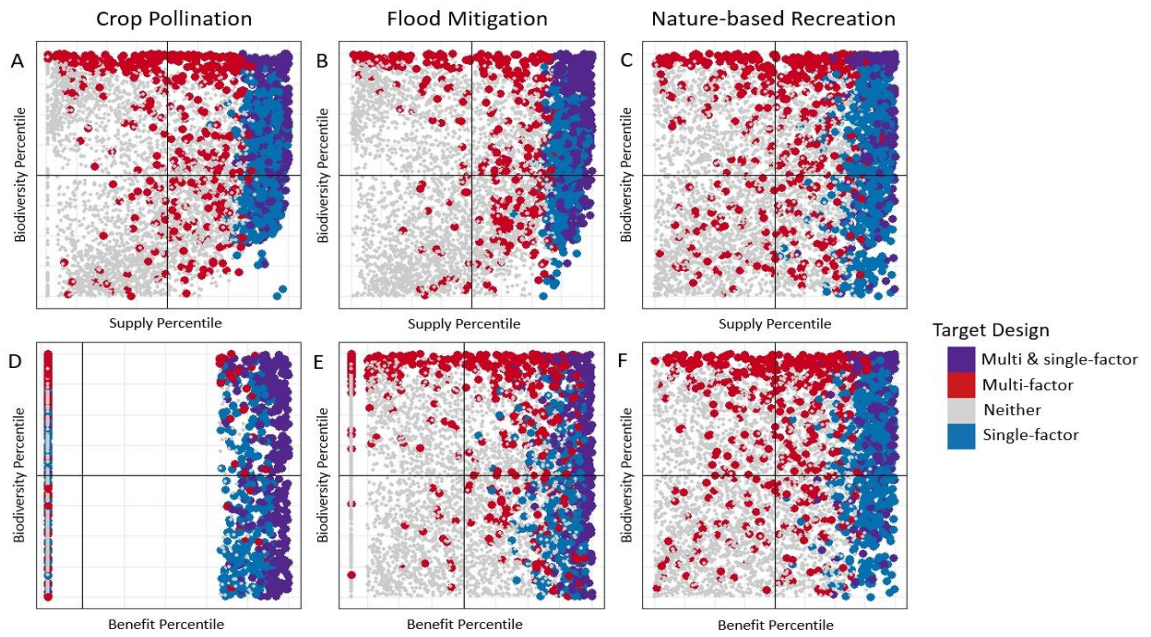


Fig. S3.15 The sensitivity of our flood mitigation supply results to including winter months in the model.

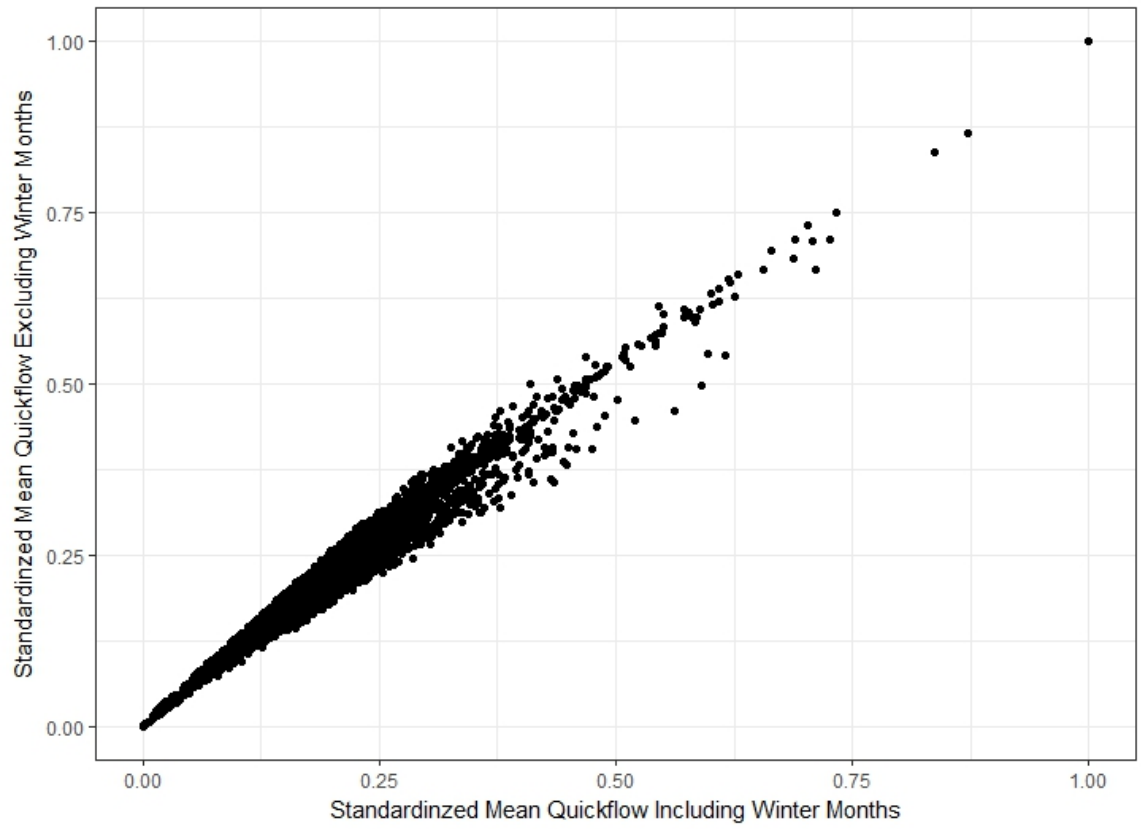


Fig. S3.16 The sensitivity of our flood mitigation service results to the assumption that supply and demand are equally important in determining benefit. When assign demand is one half (grey) and one tenth (black) the weight of supply, the major conclusions about the differences between supply and benefit, and the biodiversity and benefit value of priority areas hold.

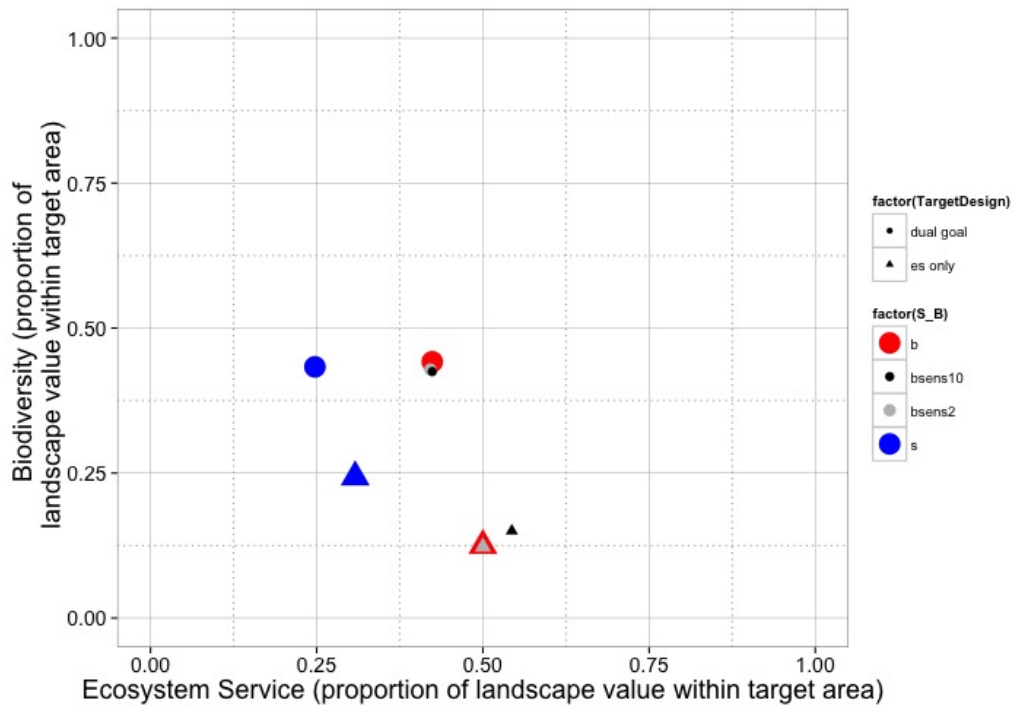
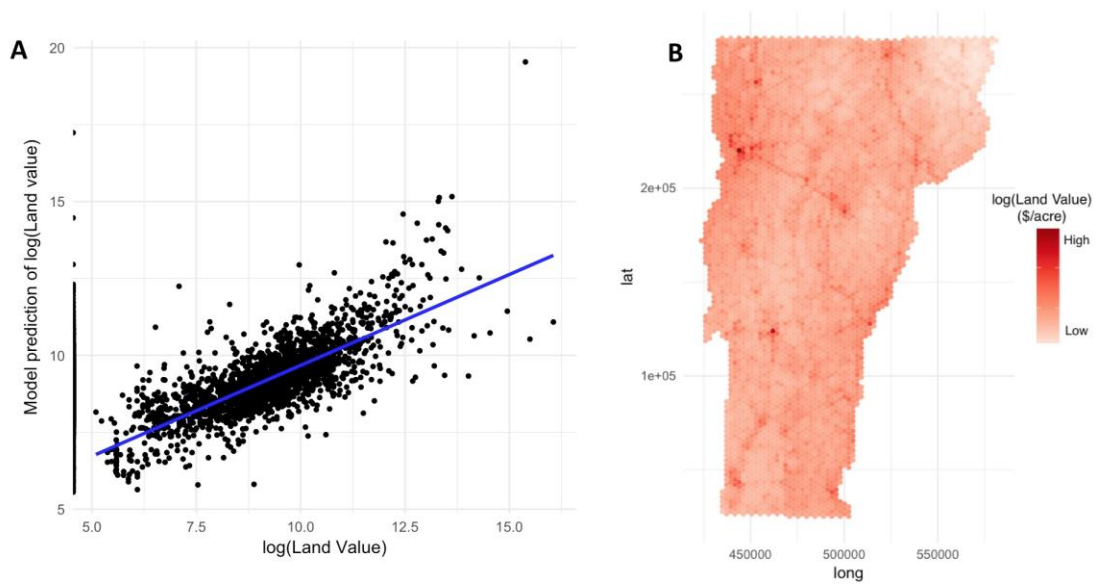


Fig. S3.17 Damage probability functions. Grey diamond: wetlands scenario

($D=e^{10.55757}p^{-0.48927}$, $p= 4.367e-07$, $r^2= 0.9646$), Open black circles: no-wetlands

low scenario ($D=e^{12.02817}p^{-0.16884}$, $p= 0.1119$, $r^2= 0.2851$), Filled black circles:

no-wetlands high scenario ($D=e^{13.11465}p^{-0.07055}$, $p= 0.5626$, $r^2= 0.04361$).



CHAPTER 4: CONSERVING BIODIVERSITY AND ECOSYSTEM SERVICES

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Abstract

Conservation organizations increasingly target ecosystem services, the benefits from nature to people, alongside more traditional biodiversity goals. The net effect of this ecosystem service focus on biodiversity remains unclear, but depends on the biodiversity co-benefits of projects targeting ecosystem services, and the effect of an ecosystem service frame on conservation budgets. Using Vermont, USA as a model landscape, we identify optimal conservation networks for four taxonomic groups, four ecosystem services, and all possible combinations of each. We then assess the biodiversity and ecosystem service value contained in each conserved network, its cost, and its overlap with every other network. We find that overlap varies widely across services and taxa, but that priorities for multiple services contain higher levels of biodiversity than priorities for a single service. Meeting ecosystem service goals alongside those for biodiversity requires a 13% increase in conservation budgets relative to meeting biodiversity goals only. Conserving ecosystem services and biodiversity separately is much less cost effective than conserving them jointly. We conclude that ecosystem services are likely to have a net positive impact on biodiversity, especially when ecosystem service priority is determined using a broad suite of different services.

Introduction

Supporting the wellbeing of a growing human population while avoiding biodiversity loss is a central challenge of sustainable development [1–3]. Ecosystem services (ES) are the benefits that people derive from nature [2,4]. Development-driven environmental degradation is rapidly eroding both biodiversity [5–8] and those ecosystem services whose value is not captured in markets [2,9,10]. By making explicit the link between the well-being of people and nature, ES have the potential to serve as common ground for human development and conservation [2]. As a result, conservation organizations and governments are increasingly prioritizing ecosystem services [11–14].

How will an ES focus impact biodiversity? Considerable debate has arisen among the conservation community over whether an ES approach is undercutting or bolstering traditional biodiversity goals [15]. On the one hand, the resources once allocated specifically to protecting nature for its own sake are now being used to protect the parts of nature that have the highest utility to people. While setting aside natural areas for ES conservation may not have direct negative impacts on biodiversity [16], targeting conservation towards ES will capture less biodiversity than targeting biodiversity directly, such that tradeoffs are likely [17–19]. On the other hand, the human focus of ecosystem services may reframe the importance of nature and increase support for conservation, thereby increasing the resources available to protect natural areas [18,20]. The net effect of ecosystem service conservation on biodiversity thus hinges on two things: First, how much biodiversity co-benefit is achieved in the process of conserving

ecosystem services? Second, will the ecosystem services concept sufficiently boost the resources available to conservation to compensate for the tradeoff involved in sharing conservation budgets between two goals.

The empirical evidence assessing the biodiversity co-benefits of ecosystem service conservation have produced mixed results [21–24] despite a strong body of evidence establishing a mechanistic link between biodiversity and ecological function [23,25,26]. Existing studies of spatial concordance have shown promise of win-win situations in some cases [27–29], that planning jointly for services and diversity can facilitate achieving both targets with minimal increases in cost [12,30], and that the land use scenarios that perform best for ecosystem services also perform well in terms of biodiversity [12]. But other assessments have warned that spatial overlap is low in many contexts [17], or have found correlations that are positive but weak [31].

Priority areas that optimize outcomes for multiple ecosystem services may show a higher degree of spatial coincidence with biodiversity than priority areas for individual services. Functionally, the influence of increasing diversity on any given service levels off at relatively low diversity levels [32,33], however each service is associated with a different set of species [34], such that more diversity is required to support a breadth of services [35]. Further, places that are important in terms of multiple services may capture more biodiversity than “hotspots” for a single service if ecosystem services are distributed differently and are thus weakly correlated to each other [29]. Thus biodiversity and

ecosystem services may be more spatially coincident when they are defined broadly in terms of many species and services than when they are defined narrowly as a single service or taxonomic group. We know of no effort that explicitly tests this hypothesis.

The financial resources for conservation are scarce; the money available to do conservation is insufficient to reach biodiversity goals [19,36–38]. As a result, ES must increase conservation budgets in order to draw from them without presenting a biodiversity tradeoff. The size of the necessary budget increase will depend on the biodiversity co-benefits of projects targeting ES. For instance, this increase may need to be large if ES projects tend to target human-influenced landscapes [20] that have low biodiversity value. Yet conservation projects that include ES tend to attract more funding than conservation projects that do not, and this funding comes from a wider variety of sources, not all of which were prominent in supporting biodiversity-focused conservation efforts [20]. This indicates that ES is drawing new money for conservation, and thus may have net neutral or even positive impact on the funding available for conservation. In order to understand the biodiversity consequence of an ES focus in conservation, we need to know the budget increase needed for biodiversity to break even once ES goals are added.

We investigate these questions using Vermont, U.S.A. as a case study. Vermont is a primarily forested state in the Northeastern Highlands ecoregion [39]. Vermonters broadly recognize the value of this predominantly natural landscape in terms of cultural

identity [40], agriculture [41], and climate resilience [42]. Following the global pattern, many conservation organizations in the state have begun to incorporate ecosystem services in their mission statements and conservation actions. Using Vermont as a case study, we identify optimal conserved lands for four taxonomic groups, four ecosystem services, and all possible combinations of each. We then assess the biodiversity and ES value contained in each conserved lands network, its cost, and its spatial coincidence with each other network. This allows us to address four questions that are crucial to understanding merit of simultaneously conserving ES and biodiversity:

Q1: What is the potential for conservation to simultaneously protect biodiversity and ecosystem services?

Q2: Is this potential greater for projects that target a breadth of taxa and ES than for those that target individual ES and taxa?

Q3: If ES targets are added to those for biodiversity, how much must conservation budgets increase to avoid reductions in the amount of biodiversity conserved?

Q4: If ES conservation is achieved separately from biodiversity conservation, what is the efficiency cost relative to achieving the goals jointly?

Methods

Overview

We identified optimal conservation priorities given a budget constraint for four different taxa: birds, mammals, reptiles, and amphibians, and four different ES: flood mitigation, aboveground carbon storage, crop pollination, and nature-based recreation. We then measured the spatial coincidence of conservation priorities for each as percentage overlap. We also identified optimal conservation priorities for all possible combinations of one to four taxa and one to four ES, and measured the overlap between priority areas for these combinations. This allowed us to assess how increasing the number of ES and taxa affected the degree of spatial coincidence.

In order to determine the additional cost of conserving ES alongside biodiversity, we set conservation targets for each ES and taxa and identified conservation priority areas that could meet these targets at minimal cost. We followed two different methods for including ecosystem services alongside with biodiversity: “dual targeting,” implemented as a formal joint optimization of ES and taxa, and “independent efforts” implemented as the union of the single-factor optimizations for each. We compare the cost requirement of dual targeting to independent efforts for biodiversity to assess the additional resources needed to meet ES goals with no net loss of biodiversity. We then compare the cost requirement of dual targeting and independent efforts to assess the cost efficiencies of including ES within the purview of conservation.

Data Sources

We obtained raster datasets of species distributions from United States Geological Survey's GAP Dataset [43]. This dataset is available at a thirty-meter resolution, and included four amphibian species, one hundred and ninety-four bird species, twenty-six species of mammals, and ten species of reptiles within the state of Vermont. We obtained published maps of ecosystem service for crop pollination, flood mitigation, and nature based recreation [44], and data on aboveground carbon storage from remotely sensed data available at a 30m resolution [45].

We estimated the cost of conservation based on a published index of conservation costs [44]. This data was originally published as an index that represented log transformed land costs. We back transformed those values to obtain approximate land values statewide. We expect land value to overestimate the true cost of conservation because most recent conservation has occurred via the purchase of easements, which is cheaper than acquiring land outright, and because this dataset represents average values at a ~5km² resolution; these averages thus include urbanized areas that are likely to have very high land values, but are unlikely to be of high conservation importance. Although we do not expect land value to strictly represent conservation costs, it does represent the opportunity cost of alternative uses of the land, and so we do expect land values to scale with conservation costs, i.e. to represent differences in the relative costs of conservation across space.

Identifying Priority Areas for Conservation

We used the optimization software Marxan [46] to identify priority areas for each taxonomic group and ecosystem service based on the above information on biodiversity, ecosystem services, conservation costs. Marxan uses simulated annealing to approximate optimal conserved lands networks that meet a conservation target at minimal cost. It produces two different outputs that indicate conservation importance: the irreplaceability index, which is calculated as the number of runs in which a unit was included in the optimal network, and the best conservation network from all runs, where the best network is the one that minimizes the following objective function:

Equation 4.1:

$ObjFun_{min} =$

$$Land\ Cost_{(x,y)} + \lambda (Protection\ target - Protection\ achieved)_i + Cost\ constraint$$

Where:

Land Cost = the monetary cost of conserving all hexagons within the selected priority areas

i = the conservation features being targeted (in our case this included all combinations of birds, mammals, reptiles and amphibians, and all combinations of flood mitigation, crop pollination, carbon storage, and nature-based recreation)

Protection target = the target amount of a conservation feature that the optimization seeks to achieve.

Protection achieved = the amount of a conservation feature held within the selected priority areas.

λ = the “species penalty factor” for missing a conservation feature’s protection target.

Cost constraint = a penalty for exceeding a user defined cost threshold.

For each individual ecosystem service and taxonomic group, and for all possible combinations of two three, and four ecosystem services and taxonomic groups, we performed 500 iterative model runs to approximate optimal conservation solutions.

In order to assess overlap of priority areas (questions 1 and 2), we created priority areas that maximized value for each conservation feature given a cost constraint. We implemented this using the objective function above by setting a cost threshold that allowed for approximately 15% of the landscape to be selected as priority areas. We set targets for each conservation feature (50% of statewide value) that were impossible to reach given that constraint, and set a cost threshold penalty so high that the optimal solution never exceeded the cost threshold. Maximizing value within a cost constraint represents the budget-limited process of spatial planning for conservation, and results in priority areas that are approximately equal in area for each conservation feature. This is important because otherwise the amount of overlap will reflect the total amount of land selected within priority areas as well as the overlap of those features.

In order to determine the budget increase needed to have a net neutral impact on biodiversity (questions 3 and 4), we also identified the least cost means of meeting

conservation targets. To implement this optimization problem we removed the cost constraint from the above objective function, and set our conservation targets at twenty percent of all habitat for non-threatened species, 40% of all habitat for threatened or endangered species, and 40% of total statewide ecosystem service value for each ecosystem service. We then took the best solution from the 500 runs for each simulation as the most cost-efficient way of meeting the relevant conservation targets. We calculated the total cost of each as the sum of the cost for all included units of analysis, and compared the costs of networks that included ecosystem services to otherwise equivalent networks that did not.

Quantifying Overlap

We measured the overlap of best networks as the ratio of the area that was included in both the ecosystem service and the biodiversity network to the mean area of those networks:

Equation 4.2:

$$A_{ES} \cap A_{BD} / ((A_{ES} + A_{BD})/2)$$

Where:

A_{ES} is the area of the best network for ecosystem services
and A_{BD} is the area of the best network for biodiversity

We compared these overlaps to the null expectation (from [31]), calculated as :

Equation 4.3:

$$A_{ES} * A_{BD} / A_{Total}$$

Where:

A_{Total} is the combined area of all units of analysis

To assess the effect of the number of ES and taxonomic groups on the overlap between ecosystem service and biodiversity priorities, we measured overlap of best networks for all possible combinations of one, two, three, and four taxonomic groups to one, two, three, and four ecosystem services. This resulted in 196 different overlap ratios, although sample size was unevenly distributed (Table 4.9)

Results

Q1: Spatial Coincidence of ES and Taxa

The average pairwise overlap between ES and taxa is 47%. This is high compared to a null expectation but lower than the 62% average overlap between taxa and the 49% average overlap between ES (Fig. 4.18d). Overlap varies widely across ES-taxa pairs. Birds and reptiles overlap less with ecosystem services than do mammals and amphibians (Figure 4.18b). Flooding and pollination overlap less with biodiversity than do recreation and carbon (Figure 4.18b).

Q2: Impact of the Number of ES and Taxa

The overlap between biodiversity and ecosystem service priorities increases as the number of ES used in defining priority areas increases. (Fig. 4.19a). Overlap also increases with the number of taxa up to three taxa, and then levels off. These overall trends also hold true for each ES (Fig. 4.19b) and taxon (Fig. 4.19c) individually. The overlap of the best network for all four ES with the best network for all four taxa, is 60%.

Q3-Q4: Cost Effectiveness of Joint Targeting

The cost of meeting all biodiversity targets equates to approximately three percent of the summed cost of all units of analysis (Figure 4.20). This least cost network included 43% of all units (because most of the selected units were low-cost). Meeting ecosystem service targets was less costly.

Reaching targets for all four ES and all four taxa through joint targeting required a 12% increase in cost relative to meeting biodiversity targets alone (Fig. 4.20). On average conserving a single ES in addition to a single taxon through joint efforts had a 13% higher cost than only conserving a single taxon. Across all pairwise combinations of a single ES and a single taxon, this cost increase ranged from 0% to 83% (Table S.410).

On average, conserving a single ES in addition to a single taxon through separate efforts had a 33% higher cost than conserving a single taxon only. Across all pairwise combinations of a single ES and a single taxon, this cost increase ranged from 8% to 128% (Table S.410). Reaching targets for all four ES and all four taxa through separate

efforts required a 45% higher cost than only conserving a single taxon (Fig. 4.20). Across all pairwise combinations of a single ES and a single taxon, this cost increase ranged from 8% to 128% (Table S.410).

Discussion and Conclusions

In Vermont, the overlap of conservation priorities for ES and biodiversity is high relative to a null expectation (Fig. 4.18c) - however it varies widely across service-taxa pairs (Fig. 4.18a, b). Mirroring this result, the budget increase needed to meet an ES target in addition to an existing taxonomic target is just thirteen percent on average (Fig. 4.20). For some ES-taxon pairs a <1% budget increase is required (e.g. for birds and recreation, or reptiles and carbon), but others require the budget to almost double (e.g. amphibians and flooding) (Table S4.10). As a result, projects seeking to conserve specific taxa and ES in Vermont may benefit from a high degree of spatial coincidence, or may find little opportunity to efficiently pursue these goals together. It is important therefore to quantify tradeoffs and identify potential win-win locations on a project by project basis.

The overlap of biodiversity and ES improves as the number of ecosystem services and the breadth of taxa used in defining conservation priorities increases (Fig. 4.20). This has important conservation implications: whereas projects that seek to safeguard a particular ecosystem service may not protect much biodiversity in the process (Fig. 4.18, overlap as low as 0.16), efforts that aim to protect a wide suite of ecosystem services are likely to protect more biodiversity even when this benefit is not explicitly sought out (Fig. 4.19,3

ES overlaps from 0.54 to 0.60). The 0.60 overlap between priority areas for all four ES and all four taxa is comparable to the 0.62 average overlap between taxonomic groups, indicating that in the case of Vermont, USA, adding ecosystem services to the goals pursued by conservation presents tradeoffs no more severe than those already faced by conservation organizations.

Why might ES show a higher degree of overlap with biodiversity than the null expectation? Part of the answer is low-cost areas, which represent opportunities to achieve a relatively high return on investment for all conservation features (Fig S4.21). Several other studies have established the importance of conservation cost in determining optimal conservation outcomes [38,47–52]. Our results indicate costs may in part determine the severity of tradeoffs between biodiversity and ecosystem services as well. Vermont is a small, relatively homogenous state with many wide-ranged species. As a result, biodiversity importance varied less across space than did the relative costs of conservation. In places with high ecological heterogeneity or endemism, or highly uneven demand for ecosystem services, priority areas for biodiversity and ecosystem services may show a weaker response to conservation cost.

Although we find a relatively high spatial coincidence, some tradeoff will occur any time a fixed budget is spread across a widening set of objectives [18]. We estimate that conservation budgets would need to increase by 13% in order to meet targets for ES in addition to those for biodiversity (Fig. 4.20). This implies that in the context of Vermont

reframing conservation around benefits to people must increase the resources garnered for conservation on the order of about 13% in order to avoid negative consequences for biodiversity. Beyond this level, we would expect ES conservation to have net positive impacts on resources available for biodiversity conservation. While we do not have evidence that this budgetary increase has occurred or that it is driven by an ES framing in the case of Vermont specifically, an increase of this size seems feasible. As a point of comparison, the inflation-adjusted annual revenue from contributions and grants for the World Wildlife Fund, The Nature Conservancy, Conservation International, and the Wildlife Conservation Society rose between one and thirty-four percent between fiscal years 2011 and 2015, the interval over which many of these organization reframed themselves around ES. Further, within the Nature Conservancy projects that include ES have been shown to attract more than four times as much funding as projects that do not [20].

Although there has been significant debate about whether ecosystem services should draw from conservation budgets [15], there is consensus that ES are important to maintain. They are critical to human well-being [2,53], their value often exceeds the cost of protecting them [38,53], and we are losing them at an alarming rate [10,54]. Given this agreement about their importance, our analysis indicates that there are significant efficiency gains associated with leveraging the existing framework and mechanism of biodiversity conservation to conserve ES. If land is kept in natural cover to maintain ecosystem services, and separately to maintain biodiversity, the combined resource

requirements of these two sets of natural areas equate to almost a 50% increase monetary cost and spatial extent relative to protecting biodiversity alone. This is much less efficient than achieving the same ES and biodiversity outcomes through joint targeting, which can be accomplished with a 13% budget increase (Fig. 4.21).

In sum, we find that the spatial coincidence of biodiversity and ecosystem services in Vermont is generally high, but is also quite variable. Understanding the specific tradeoffs faced by particular conservation projects is therefore critical to efficiently achieving these two goals at once when specific services and taxa are targeted. On the other hand, we find that spatial priorities for multiple services contain high levels of biodiversity, even when they are selected without explicitly seeking a biodiversity co-benefit. Furthermore, the financial costs of achieving ecosystem service goals within the framework of biodiversity conservation are low compared to additional funding that an ES framing can provide. By contrast, the efficiency cost of pursuing these two goals separately is quite high. Although there will certainly be cases where stark tradeoffs occur between biodiversity and ecosystem services, our results indicate that ecosystem service conservation is more likely to boost biodiversity outcomes than to undermine them.

Acknowledgements

I would like to thank Jesse Gourevitch, Courtney Hammond, Lindsay Barbieri, and Allison Adams for their thoughtful feedback on the figures and presentation of this research. I would also like to thank the Lintilhac Foundation, the USDA McIntire-Stennis

award #2014-32100-06050 to the University of Vermont, and the Rubenstein School of Natural Resources for funding that supported this research.

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Tables

Table S4.9 Sample size for overlap means

| | 1 Taxon (4 combinations) | 2 Taxa (6 combinations) | 3 Taxa (3 combinations) | 4 Taxa (1 combination) |
|--------------------------|-----------------------------|----------------------------|----------------------------|---------------------------|
| 1 ES (4 combinations) | n=16 | n=24 | n=12 | n=4 |
| 2 ES (6 combinations) | n=24 | n=36 | n=18 | n=6 |
| 3 ES (3 combinations) | n=12 | n=18 | n=9 | n=3 |
| 4 ES (1 combination) | n=4 | n=6 | n=3 | n=1 |

Table S4.10 Costs required to meet targets for all pairwise combinations of one ES and one taxon through joint targeting and through separate efforts.

| A) Cost of joint optimizations | | | | | | | |
|--|-------------------|------------------|------------------|------------------|------------------|------------------------|-------------------------|
| | Biodiversity Only | With Flooding | With Carbon | With Recreation | With Pollination | Mean Cost With Service | Mean % Increase in Cost |
| Birds | \$17,133,595,058 | \$18,678,616,721 | \$17,447,424,996 | \$17,345,615,121 | \$19,030,119,813 | \$18,125,444,163 | 6% |
| Mammals | \$13,344,458,365 | \$14,471,397,616 | \$13,320,413,012 | \$13,380,992,838 | \$15,717,490,581 | \$14,222,573,512 | 7% |
| Reptiles | \$12,083,430,111 | \$14,592,920,056 | \$12,131,545,488 | \$12,134,737,079 | \$15,099,462,059 | \$13,489,666,171 | 12% |
| Amphibians | \$6,488,629,961 | \$11,871,738,400 | \$7,349,708,666 | \$6,855,337,352 | \$11,602,380,365 | \$9,419,791,196 | 45% |
| | | | | | | | 13% |
| B) Area requirement of joint optimizations | | | | | | | |
| | Biodiversity Only | With Flooding | With Carbon | With Recreation | With Pollination | Mean Area With Service | Mean % Increase in Area |
| Birds | 10231.65 | 10436.4 | 10243.35 | 10255.05 | 10623.6 | 10389.6 | 2% |
| Mammals | 10097.1 | 10325.25 | 10108.8 | 10085.4 | 10512.45 | 10257.975 | 2% |
| Reptiles | 7078.5 | 9260.55 | 7294.95 | 7283.25 | 9430.2 | 8317.2375 | 18% |
| Amphibians | 3814.2 | 7265.7 | 5270.85 | 4563 | 7493.85 | 6148.35 | 61% |
| | | | | | | | 12% |
| C) Combined cost of separate efforts | | | | | | | |
| | Biodiversity Only | With Flooding | With Carbon | With Recreation | With Pollination | Mean Cost With Service | Mean % Increase in Cost |
| Birds | \$17,133,595,058 | \$21,783,716,169 | \$18,758,842,201 | \$18,483,026,046 | \$23,609,680,615 | \$20,658,816,258 | 21% |
| Mammals | \$13,344,458,365 | \$17,831,209,211 | \$14,578,609,640 | \$14,465,724,884 | \$20,394,874,802 | \$16,817,604,634 | 26% |
| Reptiles | \$12,083,430,111 | \$17,472,042,181 | \$14,053,545,495 | \$13,873,574,109 | \$19,672,843,056 | \$16,268,001,210 | 35% |
| Amphibians | \$6,488,629,961 | \$13,148,798,353 | \$8,956,484,916 | \$8,915,903,391 | \$14,800,166,681 | \$11,455,338,335 | 77% |
| | | | | | | | 33% |
| D) Combined area requirement of separate efforts | | | | | | | |
| | Biodiversity Only | With Flooding | With Carbon | With Recreation | With Pollination | Mean Area With Service | Mean % Increase in Area |
| Birds | 10231.65 | 13144.95 | 12162.15 | 11951.55 | 13624.65 | 12720.825 | 24% |
| Mammals | 10097.1 | 12700.35 | 11471.85 | 11419.2 | 13127.4 | 12179.7 | 21% |
| Reptiles | 7078.5 | 11015.55 | 9950.85 | 9822.15 | 11863.8 | 10663.0875 | 51% |
| Amphibians | 3814.2 | 8611.2 | 6955.65 | 6973.2 | 8944.65 | 7871.175 | 106% |
| | | | | | | | 39% |

Figures

Fig. 4. 18 A) Maps of irreplaceability indices for all ES and taxa individually, and their pairwise combinations. B) Pairwise overlap of best networks C) Observed versus expected overlap between biodiversity and ecosystem services, compared with the overlap among taxa and among ES.

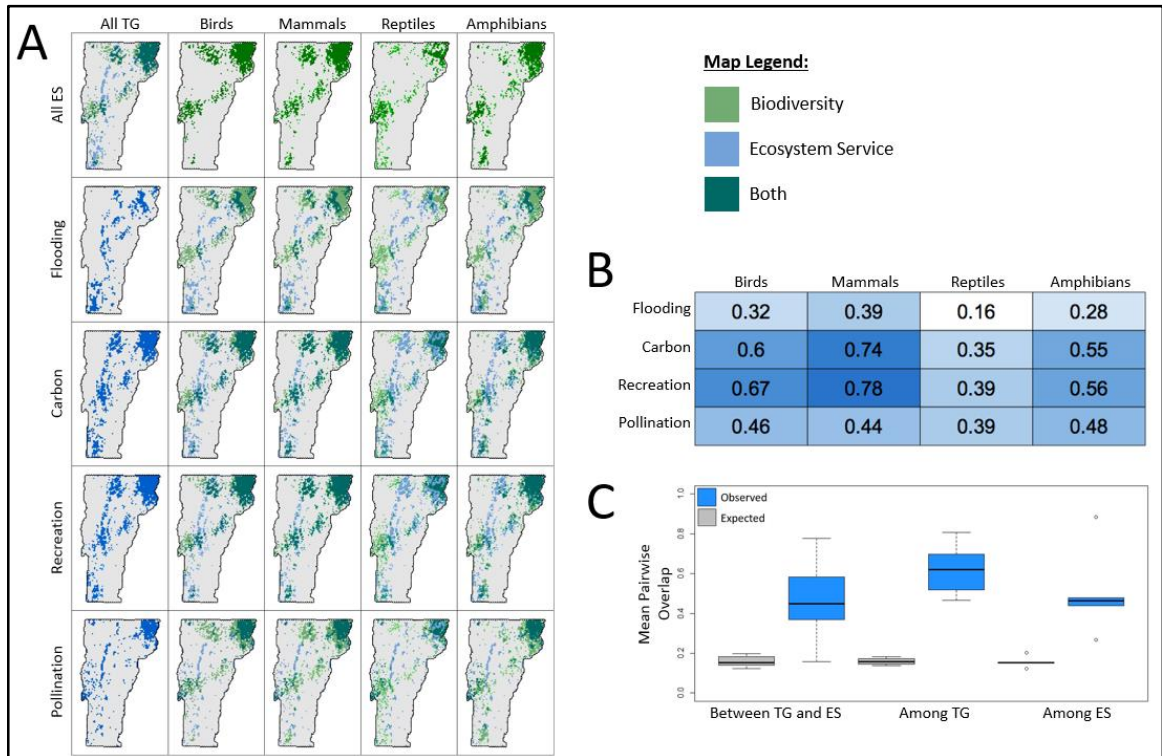


Fig. 4.19 The effect of increasing the number of taxa and services used in defining biodiversity and ecosystem service priorities, respectively, on the overlap between the best conservation networks for each.

| A | | | | | B | | | | |
|----------|---------|--------|--------|--------|-------------|---------|--------|--------|--------|
| | 1 Taxon | 2 Taxa | 3 Taxa | 4 Taxa | | 1 Taxon | 2 Taxa | 3 Taxa | 4 Taxa |
| 1 ES | 0.47 | 0.51 | 0.53 | 0.52 | Flooding | 0.28 | 0.31 | 0.33 | 0.33 |
| 2 ES | 0.51 | 0.55 | 0.58 | 0.57 | Carbon | 0.56 | 0.6 | 0.63 | 0.62 |
| 3 ES | 0.53 | 0.57 | 0.6 | 0.59 | Recreation | 0.6 | 0.66 | 0.69 | 0.68 |
| 4 ES | 0.54 | 0.58 | 0.61 | 0.6 | Pollination | 0.44 | 0.45 | 0.46 | 0.46 |

| C | | | | |
|----------|-------|---------|----------|------------|
| | Birds | Mammals | Reptiles | Amphibians |
| 1 ES | 0.51 | 0.59 | 0.32 | 0.47 |
| 2 ES | 0.56 | 0.63 | 0.35 | 0.51 |
| 3 ES | 0.58 | 0.65 | 0.37 | 0.53 |
| 4 ES | 0.59 | 0.66 | 0.38 | 0.54 |

Fig. 4.20 The monetary cost required to meet conservation targets for biodiversity, ecosystem services, and to achieve both by joint targeting and through separate efforts. Cost is shown as a proportion of the total cost for all units of analysis.

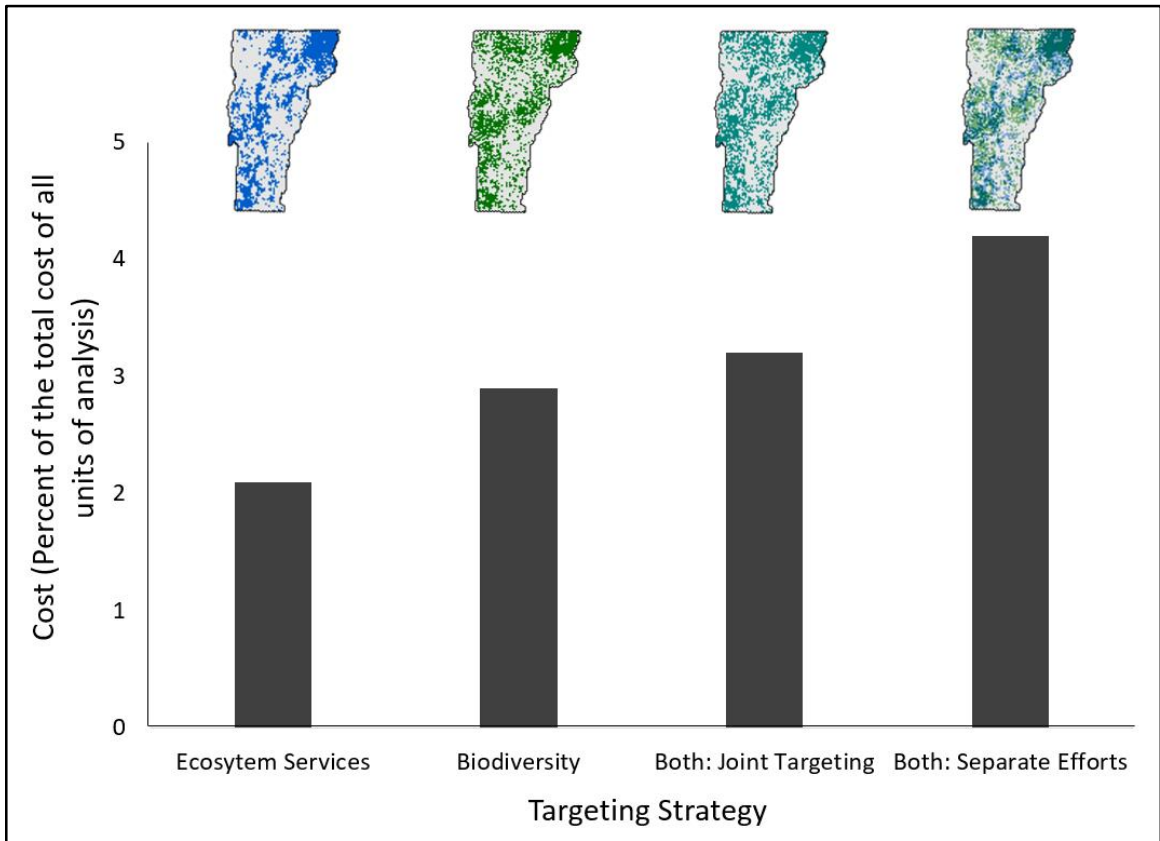
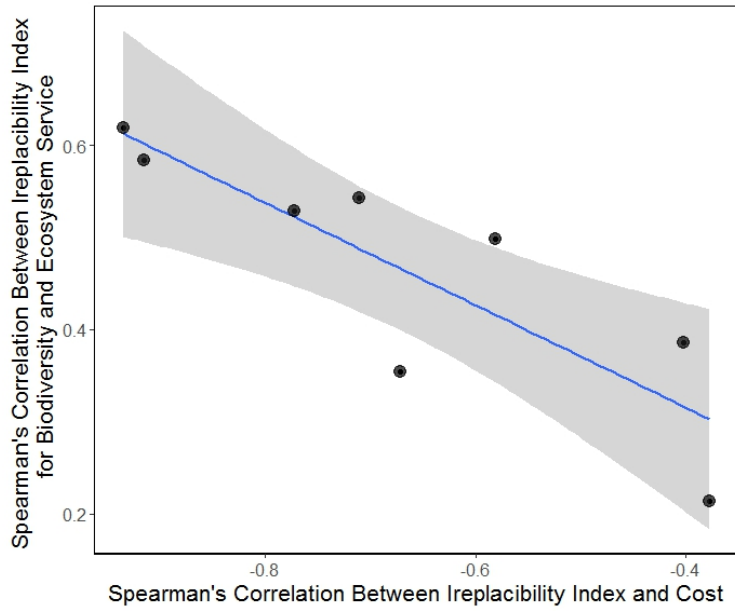


Fig. S4.21 Conservation cost explains 68% of the variation in the correlation between individual ecosystem services and biodiversity, and between individual taxonomic groups and ecosystem services ($p=0.007$, $n=8$, $f=16.02$, $6df$).



CHAPTER 5: SCENARIOS OF FUTURE LAND-COVER CHANGE AND FLOOD MITIGATION IMPACTS IN VERMONT, USA.

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Abstract

Scenario planning is a useful tool for incorporating complexity and uncertainty in conservation planning. Here, we present a set of five scenarios that represent unique visions Vermont in the year 2060, with associated land cover maps for each. We couple land cover simulations to a spatially explicit model of flood mitigation ecosystem services. We show that the uncertainty encapsulated by our scenarios is large enough to change the outcome of ecosystem service threat assessment: some scenarios resulted in increasing threat to flood mitigation services, while others were associated with potential reductions in flood risk. Given this breadth of possible futures, we assess different strategies for targeting conservation in terms of their capacity to impact places with high future flood risk. We find that targeting conservation based on the present day distribution of demand for flood mitigation captures more future flood risk than targeting ecosystem service supply. Future work evaluating the impact of these scenarios on other ecosystem services will shed light on the tradeoffs and opportunities of future land cover change in Vermont.

Introduction

There is increasing evidence that human activities are threatening the resilience of earth's ecosystems and the resilience of the human communities that depend on them [1,2]. In the past centuries, people have rapidly altered the earth's climate [3], land surface [4], biological diversity [5–9], and ecosystem service provision [10,11]. The extent of human impacts has accelerated since the mid 20th century [12,13] to such an extent that a new geologic era, the Anthropocene, has been proposed [14]. Yet these human driven changes interact via complex feedbacks with each other and with human responses to them [15]. The consequence is that we know the future will be fundamentally different from the present, but specifics about what the future will be like are highly uncertain [16,17]. Scenario planning has been presented as a powerful tool for making decisions under conditions of high uncertainty. Originally developed for military strategy [18] and adopted by business strategists [19,20], scenario planning is now applied in a wide variety of contexts including of sustainable development [17] conservation planning [18] and ecosystem services [21]. Scenarios can account for social ecological feedbacks, and represent a plausible account of what the future might look like *if* a certain path is taken, rather than assigning a probability to a particular land-cover trajectory. Exploratory scenarios are not forecasts or predictions, but rather plausible stories about the future given a set of assumptions about policy, economic drivers, and ecological processes [22,23]. Collectively, sets of exploratory scenarios are used to bracket future uncertainty, and thus to inform decisions in light of a future that we cannot predict, but can seek to understand and manage.

One of the most impactful ways we are changing our planet is by altering the composition and pattern of the earth's land surface. Land-cover change has occurred largely in the pursuit of a small set of ecosystem services whose value are captured in markets [4,10] such as food production [24]. Land-use and land-cover change are critical determinants of climate [25], biodiversity [26] and ecosystem service outcomes [10]. As a result, understanding where land-cover is most likely to change is critical for implementing efficient conservation and land use policies. However, we face high uncertainty when trying to predict future land-cover changes [27] and ecosystem service outcomes [21]. Predicting land-cover change is problematic because land-cover outcomes are the product of complex social ecological systems dynamics, and respond to a diverse set of interacting drivers [28–30]. For example, ecosystem services are in part functions of land-cover, and land-cover is largely driven by human pursuit of these services. Because the two interact via complex feedbacks, ecosystem service and land-cover outcomes are ambiguous, and cannot be assigned a probability distribution [21]. As such, it is not safe to assume that future land-cover change will resemble the patterns of land-cover changes observed in the recent past [31]. This is particularly true when the dominant drivers of land use change are shifting.

Despite these difficulties, understanding future land-cover change is important in making sound ecosystem service decisions. The benefit from any given ecosystem service is a product of the ability of landscape to supply that service, demand for the service, and

connections between supply and demand via ecosystem service flows [32–37]. While recent progress has been made in incorporating both supply and demand into current assessments of ecosystem service [38–40], addressing future changes in ES supply and demand, together, remains a key challenge. Many existing efforts to quantify ecosystem service outcomes of future scenarios assume that supply is likely to be the dominant factor in ecosystem service change [41,42]. However, the human population is growing [43,44], and the patterns of development that will occur to accommodate this increasing population, and therefore the future distribution of beneficiary demand, is highly uncertain. Here, we present a set of five scenarios for the state of Vermont designed to capture the uncertainty in future land-cover change. We evaluate the consequences of these different land-cover futures using flood mitigation ecosystem services as an example. Specifically, we evaluate whether the land-cover uncertainty our scenarios represent is large enough to change outcomes in two decision making contexts: ecosystem service threat assessment and conservation planning for ecosystem services.

Threat assessments require understanding the current state of ecosystem service supply and demand, as well as temporal trends in how they will change [45]. Maron et al (2017) present a threat assessment framework to evaluate the threat level of ecosystem services with regards to two thresholds: demand exceeding supply and ecosystem service “extinction” [45] based on the state and trends of ES supply and demand. Essentially, this provides a framework for assessing whether the amount of ecosystem service on a landscape is sufficient, and whether it is likely to remain so.

For conservation planning, we need to know where on the landscape ecosystem services are most important now, and where they will be most important in the future. There have been great strides in identifying current ecosystem service priorities by adapting existing conservation planning tools, incorporating conservation costs and threats, assessing tradeoffs and synergies between multiple services, and in accounting for demand when quantifying service [32,46–55]. However, uncertainty regarding the future means that conservation strategies based on the present can produce sub-optimal results in the long term [56]. This presents a challenge: what is the best strategy for targeting actions today so that they will most effectively benefit future generations?

One option could be to target conservation towards places that have the greatest potential to benefit people, rather than targeting conservation towards the places where benefits are currently the highest. The former case is achieved by targeting present day service supply [36], whereas the latter is achieved by taking both supply and demand into account.

Research to date has established that incorporating demand can make conservation planning efforts much more efficient in the near term [38–40]. However, large uncertainty in land-cover indicates large uncertainty in the future distribution of people, and thus demand for ecosystem services. In cases where this uncertainty in demand is sufficiently large, targeting supply may be an effective hedging strategy for protecting ecosystem services in the long run.

Vermont has a long history of extensive land use change. Following European settlement Vermont's forested landscape was almost completely denuded of forest cover. Since the mid 1800's, natural forest regeneration on abandoned agricultural lands has transformed this landscape again such that forests now cover almost 80% of the state's land area [57,58]. For the first time in over a century, Vermont is now experiencing modest decreases in forest cover [59]. These land use changes included widespread alterations of the state's river corridors, which were designed to reduce flooding locally but can exacerbate flooding downstream [60,61]. This legacy of river hardening is now compounded by climate-change driven increases in the frequency and severity of flooding [62]. Using flood mitigation services as an example, we demonstrate the use of these scenarios for ecosystem service threat assessment, and for conservation planning in light of uncertainty in the future patterns of demand for ecosystem services. This allows us to address the questions:

- 1) Given stakeholder-defined scenarios, what is range of possibilities for Vermont's future landscape, and how far does this deviate from a business and usual trend?
- 2) What are the impacts on flood mitigation across this range of scenarios?
- 3) Where can we expect to see increases or decreases in natural cover and flood risk that are robust to differences among scenarios?
- 4) Is targeting ecosystem service supply an effective hedging strategy in light of uncertainty in the future distribution of ecosystem service demand?

Methods

Developing Scenario Narratives

We developed a workshop that led stakeholders through a structured process with the goal of envisioning different trajectories that Vermont might take in the future, and the impacts of those trajectories on the landscape. Our group of stakeholders represented various organizations involved in conservation and land use policy in Vermont: The Nature Conservancy Vermont, The Vermont Land Trust, The Agency of Natural Resources, Vermont Fish and Wildlife Department, The DEC Watershed Management Division, the Department of Forests Parks and Recreation, Milone and MacBroom Consulting, the VT Division of Emergency Management and Homeland Security, and the Vermont Department of Tourism and Marketing.

Our process for generating scenario narratives followed the methods of the Global Business Group [20,63], which has been widely applied in corporate [19] and nonprofit settings [64], and was employed by the U.S. National Park Service [65] and the Global Millennium Ecosystem Assessment [11]. First, stakeholders described outcomes from the landscape that were most important to them personally, and to the organization they represented. Second, we asked stakeholders to brainstorm key drivers of landscape change. Third, we prioritized these drivers based on their impact on the landscape, and their degree of uncertainty. We selected our top two drivers according to these criteria as axes which defined four scenario spaces. We then broke into groups to describe each of

the four scenarios in terms of the drivers that were not used to define the two axes, and in terms of important outcomes stakeholders identified at the beginning of the process.

Modeling Land-Cover Based on Past Trends

We simulated land use change for each of the four scenarios produced in the workshop using Dinamica EGO [66,67]. We obtained land-cover data from the National Land-cover Dataset (NLCD) for the years 2001, 2006, and 2011 [68]. The NLCD classifies U.S. land-cover into 16 different land-cover classes at a 30-meter resolution, and is based on LANDSAT satellite imagery. We simplified these 16 land-cover classifications into six more general land-cover classes: developed land, agriculture, forest, shrub/scrub, and wetlands. We then calibrated transition rates for 14 possible land-cover transitions from 2001 to 2011. Transitions out of developed land and out of wetlands experienced too few land-cover changes to find any statistically significant weights of evidence, and were omitted from the simulation.

To calibrate the spatial allocation of land-cover transitions, we calculated conditional probabilities between spatially explicit predictor variables and each transition using a Bayesian weights of evidence approach [69]. The weight of evidence represents the influence of each predictor on the likelihood of a transition. These weights are then combined to calculate spatially explicit probabilities of each transition under the assumption that predictors are independent. We employed sixteen predictors of land-cover change, including: landscape attributes (slope [70], distance from roads, highway

density, floodplains [71], wetland classification [72], farmland classification [73], distance from cities), variables related to land use planning and regulation (designated growth areas [74], distance from designated growth areas, potential wetland restoration sites [75], enrollment in the current use program [76], conservation status [77], owner type [77]), and social-demographic data (population density, population growth, and median household income at the census tract level [78]). We calibrated weights of evidence based on the period from 2001-2006. We performed pairwise tests of the independence assumption and removed landscape attributes in all cases where the Crammer coefficient was greater than 0.4, ensuring no spatial autocorrelation between correlate variables. This can result in different sets of variables being used for each transition; each transition also has a unique weight of evidence for each variable.

We then validated our model in two ways: First, we simulated land-cover change from 2006 to 2011 based on 2001-2011 transition rates and 2001-2006 weights of evidence, and compared simulated transitions to observed transitions using fuzzy logic and an exponential decay function across a range of neighborhood window sizes [69]. This validates our ability to simulate observed land-cover patterns at different spatial resolutions based on calibrated weights of evidence and transition rates. Secondly, we measured the proportion of observed transitions as a function of the modeled probability of that transition. This allowed us to assess our probability maps independent from the quantity of simulated changes.

Modeling Land-cover Based on Scenario Narratives

We then modified the calibrated weights of evidence and transition rates achieved above to generate land-cover change simulations that reflected each scenario narrative in terms of its major divergences from recent patterns. An explanation of the major parameter changes for each scenario are described fully in the Appendix. We also implemented a fifth “Business as Usual” scenario where we simulated land-cover change forward to 2060 with the calibrated parameters from our 2001-2011 baseline. In order to simulate change from 2011-2061, we iterated the modified transition rates for five 10-year time steps assuming that rates of change would be constant across each time step. Simulating changes via five 10-year time steps allowed our model to incorporate feedbacks between land-cover changes in our simulation without adding the prohibitive computational load of an annual time step. To produce probability maps across the fifty-year time period, we translated ten-year transition rates into a fifty year transition rate as:

Equation 5.1:

$$i_t = (1+i)^t - 1$$

Where:

i is the transition rate per time step

t is the number of time steps (5)

i_t is the transition rate for t time steps

Modeling Flood Outcomes

We modeled flood outcomes, for each of the five scenarios and for current land-cover, in a three step process. First, we quantified demand for flood mitigation ecosystem services

as the amount of developed and agricultural land in flood-prone areas. We adapted the methods from Watson et al 2017 [40], which quantified demand for flood mitigation as buildings within flood-prone areas, assumed each of these buildings counted equally as a single unit of demand, and attributed that demand to the landscape by dividing it evenly among all upstream pixels. We alter this method in three ways. Because our land-cover simulations do not include the precise locations of buildings in the future, we attribute one unit of demand to each pixel with a developed land-cover class instead. Second, Watson et al (2017) do not account for flood impacts on agriculture. To incorporate flood impacts on agriculture we attribute one unit of demand to each pixel of agriculture as well. Because impacts to agriculture are fundamentally different than impacts to built infrastructure, we report agricultural and development demand separately. Finally, Watson et al (2017) delineate drainage areas to attribute demand to the landscape at 30m pixel resolution. Here, we aggregate demand and attribute it to drainage areas at the resolution of HUC12 watersheds based on the national hydrography dataset [79]. We subtract demand based on 2011 land-cover from demand in each scenario to report changes in demand for flood mitigation services.

Second, we estimated changes in supply of flood mitigation ecosystem services as the change in quick-flow. For all five scenarios and for 2011 land-cover, we estimated quick-flow using the InVEST seasonal water yield model, which adapts a curve number approach to a pixel scale to estimate the portion of rainfall that runs off as quick-flow at a monthly time step [80]. We then subtract 2011 quick-flow from the scenario quick-flow

and report the magnitude of this change as the change in the supply of flood mitigation (where an increase in quick-flow is a decrease in service supply, and vice versa). The supply of flood mitigation services can be thought of as the marginal contribution of natural land-cover types to a reduction in quick-flow, and because our scenarios differ from the baseline only in terms of land-cover, an increase or decrease in quick-flow represents a change in the supply of flood mitigation ecosystem service.

Ecosystem service supply and demand can also be represented in a risk hazard framework. Within this framework the amount of development and agriculture in flood-prone areas is termed exposure, and quick-flow can be used as a proxy for flood hazard. Risk is then calculated as the product of hazard and exposure:

Equation 5.2:

$$\text{Flood Risk} = ((\text{Exposure}_{\text{Ag}} + \text{Exposure}_{\text{Dev}})/2) * \text{Hazard}$$

This multiplicative effect is conceptually sound within both the risk-hazard [81] and ecosystem service [40] frameworks. In an ecosystem services framework, when either demand or supply is zero, benefit is also zero. In a risk-hazard framework, when there is either risk or hazard are lacking, then there is no risk. We also calculated changes in ecosystem service benefits over time. Flood mitigation benefit is the marginal contribution of ecosystems to a reduction in flood risk. Because our scenarios differ only

in terms of land-cover, changes in risk equate to ecosystem service gains and losses. We calculate change in benefit as:

Equation 5.3:

$$\Delta\text{Flood Mitigation Benefit} = - (\text{Flood Risk}_{\text{Scenario}} - \text{Flood Risk}_{\text{Baseline}})$$

Assessing Ecosystem Service Threat

For each scenario, we applied the threat assessment framework of Maron et al. (2017), which categorized threats to ecosystem services based on the state and trends of supply and demand, the ratio between supply and demand, and the threshold where supply is considered to have met demand. To apply this framework, we assume that flood mitigation service is currently “stable but undersupplied”. Recent devastating flood events in Vermont and the historic loss of natural floodplain functions [60] provide evidence that flood mitigation is currently undersupplied in Vermont, i.e.: the supply of this service is insufficient to meet demand. Efforts promoting flood resiliency [82,83] and regulation aiming to prevent wetland loss [84] indicate that the supply of flood mitigation may be stable currently.

While our index based approach to quantifying flood mitigation supply, demand, and benefit allows us to assess whether supply and demand are increasing or declining separately, it does not allow us to assess whether supply “meets” demand, or to quantify changes in the ratio of supply and demand when they increase or decrease together.

Where this lead to ambiguities in assessing ecosystem service threat, we present the possible range of threat levels.

Targeting Ecosystem Service Supply v.s. Accounting for Demand

To assess whether targeting supply is an effective hedging strategy in conservation planning for ecosystem services over long time horizons, we identify the top 15% of all HUC 12 watersheds in terms of ecosystem service benefit (accounting for the current distribution of demand), and in terms of ecosystem service supply, as conservation priorities according to these two different targeting strategies. In order to calculate current ecosystem service supply (as opposed to future changes in that supply as described above), we implemented a counterfactual 2011 scenario where all natural land-cover was converted to agriculture, the most common anthropogenic land-cover in Vermont according to our reclassification of the National Land-cover Dataset. We then calculated the total ecosystem service supply in 2011 as the difference between quick-flow for the actual 2011 land cover and the hypothetical quick-flow that would occur in the absence of natural land-cover types. We calculated ecosystem service as the product of supply and demand:

Equation 5.4:

$$\text{Flood Mitigation Service} = ((\text{Demand}_{\text{Ag}} + \text{Demand}_{\text{Dev}})/2) * \text{Supply}$$

Finally, we assumed that ecosystem service conservation is most important in places with high flood risk, and calculated the mean flood risk in ecosystem service, and service supply priority areas for each scenario.

Results

Scenario Narratives

The two primary drivers of change selected to define our scenario space were the scale of governance, economy, and community; and the strength of proactive policy (Fig. 5.22). As a result, two scenarios (Ironic Hyper-locality and Self Sufficient Vermont) were defined by increasingly localized community and economy, such that state level politics and economy were most important in determining Vermont's trajectory. The other two (Skyscrapers in the Champlain Valley and Laissez Faire) were defined by increasingly globalized community and economy: Vermonters benefited from goods and services provided by a global market but were vulnerable to global market dynamics. Likewise, two were defined by strong governance and innovative markets favoring sustainable development (Skyscrapers in the Champlain Valley and Self-Sufficient Vermont), whereas two were defined by traditional market forces unrestrained by environmental governance (Laissez Faire and Ironic Hyper-locality). The five scenarios, including the four resulting from this process and a business as usual, each represent a distinct land-cover trajectory with unique positive and negative impacts on outcomes Vermonters value. None of these scenarios are predictions, but rather we expect that they collectively will bound the range of possible land-cover change until the year 2060.

Laissez Faire

Under this scenario Vermont's future is heavily influenced by national and global trends in a world where our food, energy, economic, and government systems, as well as human and natural communities, are increasingly interconnected over very broad scales. These close connections with global systems provide some opportunities for the state: grey infrastructure needs are addressed through the continued availability of federal funding, the Vermont brand supports a thriving four season outdoor recreation industry and draws more and more tourism dollars to the state, and conservation efforts are coordinated across state and national borders. However, this outside influence moves Vermont in a direction that is less defined by Vermonters' values. Federal policies, societal priorities outside of the state, and global economic drivers gain influence to the detriment of municipal government. Regional and global markets promote industrial agriculture, large-scale dairy, and energy importation on a grid powered by large scale renewables and fossil fuels alike. In this scenario these national and global forces tend to promote sprawl band development, and continue to subsidize development in floodplains despite repeated flood damage.

Skyscrapers in the Champlain Valley

In this scenario Vermont is also highly connected to global markets for food and energy, and local food and forest product markets diminish. However, strong and proactive governance leverages this connectivity to create new opportunities as the import of forest

products, food and energy from outside of the state relieves pressure on the Vermont landscape: Forest management focuses less and less on timber and other forest products and more on the protection of natural communities; In this scenario riparian corridors are restored and smart growth principles promote high density development on former agricultural lands which are no longer under pressure to produce. Energy is imported from out of the state and many VT hydro projects are dismantled to restore aquatic community connectivity. These benefits to natural communities increase the recreation and ecotourism potential of forests and riparian efforts. This is leveraged towards increasing out of state tourism to compensate for the diminished economic output in other sectors. Movement out of flood zones is incentivized in advance, rather than in response, to flood events.

Self-sufficient Vermont

This scenario is also characterized by strong and proactive policies, but occurs in a world where the scale of community and economy is increasingly localized rather than expanding. Vermont develops strong and more insular markets for food, energy and forest products, and thrives because of its resilience to external market fluctuations and political shifts. Almost every home in Vermont is powered by their own renewables, and supplemented by community level grid powered by larger proactively cited renewable developments. Population is clustered into quasi self-reliant units of clustered houses surrounded by private and collectively owned farms. Fifty percent of food is produced within Vermont by a network of small, highly diversified farms. Dairy is an outlier, and

is greatly diminished out of environmental necessity. Motivation and resources for conservation increase, but these are put towards supporting a strong working landscape rather than an explicit focus on the preservation of natural communities.

Ironic Hyper-locality

In this scenario, as in the Self Sufficient Vermont scenario, the scale of community decreases. However in this case Vermont becomes increasingly insular under weak policy and governance structures. Economic forces drive inequality in the state: commodification and market based land prices increase the wealth of a small number of highly affluent Vermonters. The Vermont brand and landscape are both affected by this gentrification. Land ownership is concentrated in the hands of these lucky few, and land posting is prevalent. This wealthy class supports a thriving high-end local food market, and bolster conservation by purchasing easements surrounding their private lands. Outside of these large blocks of private land sprawl band development expands to accommodate continued pockets of poverty. Conventional transportation is the norm, and energy is brought in on a mixed grid as well as being powered by individual owned small scale solar and hydroelectric. With this growing wealthy demographic comes increased private investment in easements on private forested land and growing demand for a niche local food market. Landscape aesthetics and quality of life for the wealthiest Vermonters are prioritized to the detriment of economic prosperity and public access to natural areas.

Land-cover Trends and Validation

Our analysis shows that currently Vermont is experiencing very modest land-cover change. Transition rates for all land-cover transitions were low from 2001-2011. The highest transition rate was from shrub/scrub to forest (0.19% change per year), and the most significant land-cover change in terms of total areal extent was for the transition from forest back to shrub/scrub (142 km² between 2001 and 2011). These transitions primarily represent timber harvest and forest regrowth. No other land-cover changes exceeded a transition rate of 0.1% annually, or comprised more than a 10km² extent during the ten-year baseline period. This highlights that agriculture and development have been expanding very slowly.

Despite establishing significant weights of evidence relationships for all other land-cover transitions, minimum fuzzy similarity between simulated and observed 2011 land-cover remains below 0.15 up to a 50 pixel window size. Maximum fuzzy similarity at this window size approaches 0.5, but minimum similarity is a better estimate of fit because even randomly distributed changes can produce high maximum similarities [69].

Although our fuzzy similarities were low, we found that land-cover change was more likely to occur in places where our modeled probability of transition was high (Fig. 5.24), indicating that our probability maps do appropriately capture spatial patterns of land-cover change, although our land-cover simulation does not (Fig. 5.23).

Scenarios of Land-cover Change

The five scenarios differ in terms of land-cover composition and pattern (Fig. 5.25). Because Vermont's landscape is currently dominated by forest cover, very large changes in transition rates were necessary to noticeably change the overall composition of the landscape, and as a result many differences appear subtle at a statewide scale. This is particularly true for Ironic Hyper-locality, which is similar to the business as usual scenario. However, where changes did occur, they were distinct among scenarios (Fig. 5.25). Most generally, Laissez Faire was characterized by an increase in developed land, Skyscrapers in the Champlain Valley by forest regrowth and wetland restoration, and Self Sufficient Vermont by the expansion of agriculture and timber harvest.

Consequences for Flood Risk

The modeled differences among scenarios affected flood risk. Scenarios differed in terms of demand for flood mitigation from developed areas (Fig. 5.26a), demand from agricultural areas (Fig. 5.26b), and in terms of the supply of flood mitigation by ecosystems (Fig. 5.26c). Three scenarios result in an increase in demand for flood mitigation and a simultaneous decrease in their supply, exacerbating flood risk (Business as Usual, Laissez Faire, Self Sufficient Vermont), one resulted in an overall decrease in risk despite moderate loss of ecosystem service supply (Ironic Hyper-locality), and one resulted in increasing supply and decreasing demand for flood mitigation (Skyscrapers in the Champlain Valley).

Ecosystem Service Threat Assessment

Flood mitigation service will be increasingly threatened if a future resembling the Laissez Faire or Self Sufficient Vermont scenarios occurs. In these cases, the supply of ecosystem service decreases as demand simultaneously increases (Fig. 5.26). We are thus able to ascertain that the ratio of supply to demand is declining, and categorize flood mitigation service as “critically endangered” according to Maron et al.’s (2017) threat categorization (Fig. 5.27). In contrast, the Skyscrapers in the Champlain Valley scenario depicts a future where this ecosystem service is in recovery: the supply of the service increases moderately, but demand for the service also decreases so that overall supply would be much closer to meeting demand than today (Fig. 5.27). Ironic Hyper-locality represents a mix of these two outcomes; in this scenario, people respond to extensive flood impacts (a near term lack of ecosystem service) by moving outside of flood-prone areas. The result is that flood mitigation falls far short of meeting demand (i.e., is threatened) for part of the time between the present day and 2060, but by 2060 this service is less threatened than today despite small decreases in service supply (Fig. 5.27).

Targeting Conservation

The location of natural cover loss varied greatly across scenarios. Similarly, the amount and spatial pattern of changes in demand and supply for flood mitigation varied widely such that very few watersheds saw an increase in risk across a majority of scenarios. However, we could identify isolated opportunities to reduce risk robust to the variation among scenarios (Fig. 5.28).

Targeting the most important watersheds on the basis of ecosystem service benefit resulted in a different set of prioritized watersheds than targeting ecosystem service supply alone. Targeting conservation priorities based on current patterns of demand captured areas with higher future flood risk than targeting supply in all scenarios, despite the fact that demand changed markedly across scenarios (Fig. 5.26a-b). This suggests that targeting supply is a poor hedging strategy against uncertainty in the future distribution of demand.

Discussion and Conclusions

The breadth of trajectories represented by our scenarios illuminate that future land-cover patterns may deviate fundamentally from recent trends. The scenarios envisioned as plausible by our group of stakeholders differed substantially from each other and from the business as usual scenario, both qualitatively and in terms of quantitative land-cover outcomes. As such, the probability of land-cover change under the business as usual scenario does not nearly capture the spectrum of land-cover change envisioned as plausible in our scenarios. This uncertainty about the future is both a challenge and an opportunity in the conservation planning process. On the one hand, conservation and land use planners cannot be sure of the threats and risks they face, which makes it more difficult to target investments. On the other hand, their decisions will influence which scenario, or scenarios, the future of Vermont most resembles.

Our land-cover simulation illustrates that it is difficult to accurately capture the specific locations of change by calibrating a simulation model to past changes. Our fuzzy similarity analysis indicates that we were not able to predict real changes based on our weights of evidence (Fig. 5.22). This is because there were very few land-cover changes relative to the number of pixels with nearly equal high probabilities of change. As a result, a small number of simulated changes were allocated randomly across a much larger number of candidate pixels. Much of what drives land-cover change comes down to individual decisions that are not predictable, and appear stochastic in a simulation model. For instance, land-cover change may be more likely when parcels of land have recently been inherited and split among a younger generation, and when forested parcels have reached their 30 year rotation and are ready to be harvested for timber. A few events such as these may drive the majority of changes for a particular land-cover transition when so few changes are happening on the landscape overall.

It has long been recognized that “fast” drivers of land-cover change (such as discrete human behaviors) make land-cover change prediction difficult, even as “slow” determinants like our predictor variables allow us to constrain and identify patterns [16,85]. Other types of models, namely agent based models, are designed to capture these individual human behaviors directly [86]. The statistically-driven type of land-cover change model that we employ here is designed to capture the overall patterns in land-cover that emerge from “slow” drivers like environmental gradients and demographic trends. Our validation indicates that our model captured these patterns well. For all land-

cover transitions, observed changes were more likely in places where modeled probabilities of change were high (Fig. 5.23). This demonstrates that our probability maps capture current patterns in land-cover change, and are a sound baseline for establishing exploratory scenarios.

Our scenarios diverged widely from each other, but is this range of possibility large enough that it would change the decisions of conservation and land use planning organizations thinking about the future? We find that our scenarios are divergent enough to differ in term of their ecosystem service threat assessment outcomes. Under a business as usual scenario threat to flood mitigation services remains relatively stable, but across all scenarios threat level could range from critically endangered to least concern by 2060 (Fig. 5.28). Our threat assessment only represents changes that result from land-cover change. Vermont is experiencing an increase in the frequency and intensity of flooding due to climate change [62]. We do not account for climate change in our analysis, but it would manifest in our framework as an increase in flood hazard and exacerbate flood risk. The threat range represented by our scenarios is thus an underestimate of threat.

Our analysis demonstrates several challenges in applying ecosystem service threat assessment. In the case of flood mitigation, there is not a clear threshold where supply meets demand. Rather, identifying such a threshold would require defining an “acceptable” frequency and severity of flooding. Secondly, we quantify flood mitigation using an index based approach that allows us to assess relative increases and decreases in

supply and demand, and to combine them into an index of ecosystem service by weighting supply and demand equally. However, our approach does not allow us to quantitatively equate one unit of supply to one unit of demand. As a result, we cannot determine how the ratio of supply and demand is changing in cases where both are increasing or decreasing. This is true of our Ironic Hyper-locality scenario, which involves reductions in both supply and demand (Fig. 5.26). These challenges are likely to apply broadly in applying the ecosystem service threat assessment we adopted [45].

Our scenarios represent diverging outcomes in terms of the spatial patterns of ecosystem service losses as well as their overall degree of threat. This uncertainty presents a real challenge in terms of conservation planning and land use policy. Understanding how likely it is that conservation features will be lost if they are not protected is crucial to targeting conservation investments efficiently [87]. However, we find that there are very few opportunities to make conservation investments that are robust to the spectrum of land-cover change that our scenarios represent. There are few places that experience natural cover loss under all, four, or even three of our scenarios. Similarly, there are very few watersheds that experience increases in flood risk across most scenarios. However, those watersheds that do see risk increases in four or five scenarios are very likely to be sound investments. These watersheds are different than the ones with the greatest current flood risk (Fig. 5.28-5.29), indicating that the dominant patterns in future changes may not be clearly reflected by current landscape patterns.

This presents a challenge: what is the best strategy for targeting actions today so that they will most effectively benefit future generations? One option could be to target conservation towards places that have the greatest potential to benefit people, rather than targeting conservation towards the places where benefits are currently the highest. The former case is achieved by targeting present day service supply, whereas the latter is achieved by taking both supply and demand into account. Research to date has established that incorporating demand can make conservation planning efforts much more efficient in the near term [36]. However, large uncertainty in land-cover indicates large uncertainty in the future distribution of people, and thus demand for ecosystem services. In cases where this uncertainty in demand is sufficiently large, targeting supply may be a more effective strategy for protecting ecosystem services in the long run. However, our analysis indicates that in Vermont, uncertainty in the spatial distribution of future demand is sufficiently small such that targeting conservation efforts based on the current distribution of demand outperforms targeting supply in all scenarios. Because our scenarios are quite divergent from each other, this suggests that incorporating demand remains important in efficiently conserving ecosystem services over time horizons of approximately 50 years.

The narratives of our scenarios also present several interesting dynamics outside of our quantitative assessment of flood outcomes. The desirability of different scenarios can not be judged solely by their ecosystem service outcomes. For example, the Skyscrapers in the Champlain Valley and Ironic Hyper-locality scenarios essentially represent two

different pathways towards the same flood mitigation outcome: demand decreases drastically though local emigration out of flood zones in both cases. However, in the former this emigration occurs gradually and proactively with the assistance of public and nonprofit sector funding via buyouts of flooded homes and flood easements on agricultural land. In the latter people are forced out of their homes following floods because insurance coverage and public programs that subsidize rebuilding in flood zones are reduced. The Skyscrapers in the Champlain Valley Scenario is clearly more desirable in this light.

Secondly, our scenarios did not include any case where increased demand for flood mitigation services was accompanied by an increase in their supply. This would be hypothetically possible if an expansion of development and agriculture in floodplains occurred simultaneous to a re-naturalization of these areas outside of floodplains. This is essentially the pattern that Vermont has experienced for the past 150 years, thus the absence of this outcome among our scenarios is a clear indication that the land-cover trajectory of Vermont is shifting. That this is not the pattern in the business as usual scenario is likely because these changes are approaching their full possible extent: much of Vermont's current agriculture and development occurs in floodplains, and much of the remainder of the state is already forested, thus only very small increases in floodplain development and non-floodplain forest regeneration are possible.

Finally, rather than arriving at one most desirable scenario, Skyscrapers in the Champlain Valley and Self Sufficient Vermont act as two different visions of a sustainable Vermont landscape. We demonstrate here that the Skyscrapers in the Champlain Valley scenario outperforms Self Sufficient Vermont in terms of flood mitigation outcomes, but the Self Sufficient Vermont narrative represents the prioritization of a different set of ecosystem services: food production, timber harvest, and cultural services associated with Vermonters' pride in a "working landscape". That the land-cover implications of these two narratives are so different (Fig. 5.25) indicates that there may be fundamental tradeoffs between these different visions of sustainability in Vermont. In light of this tension, future work assessing the implications of these scenarios across a range of other ecosystem services may be highly valuable in quantifying the tradeoffs presented by these two competing visions, and in identifying opportunities to simultaneously pursue the positive aspects of each.

In conclusion, we present a set of scenarios that represent four unique visions of Vermont's future, each of which diverges from a business as usual scenario. Our application of these land-cover scenarios to flood mitigation services illuminates how their use in planning for ecosystem services. The uncertainty encapsulated by our scenarios is large enough to change the outcome of ecosystem service threat assessment. We also demonstrate that the importance of incorporating demand in conservation planning is robust to this uncertainty. Applying these scenarios to a broader suite of ecosystem services will help to determine vulnerabilities and opportunities with regards

to other services individually, and also to determine the tradeoffs among services presented by future land-cover change.

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Figures

Fig. 5.22 Summary of scenario space as defined by the two primary drivers of change identifies by our group of knowledge brokers: The scale of community (in red) and strength of proactive policies (in blue). Black labels are the endpoints of these two axes. Grey labels indicate the four resulting scenarios.

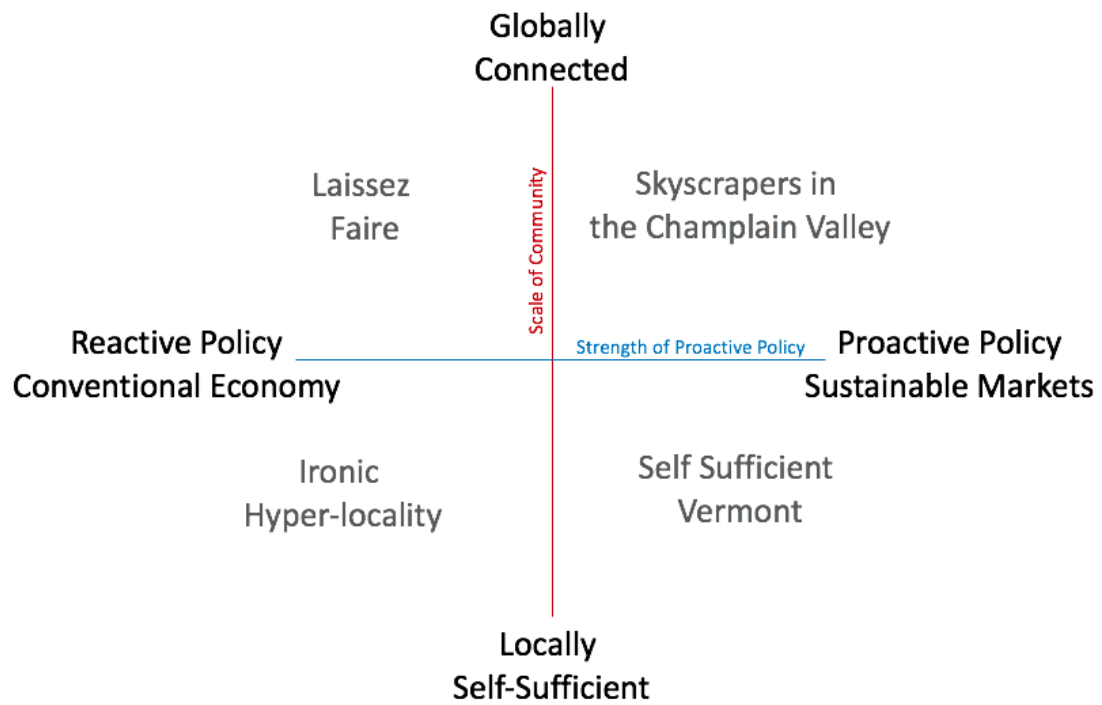


Fig. 5.23 Fuzzy Correlation between simulated and observed 2011 land-cover as a function of window size. Fuzzy similarities represent the two-way comparisons between actual and observed transitions. The minimum similarity represents the similarity obtained by comparing simulated changes against actual ones. The maximum similarity represents the similarity obtained by comparing actual changes against simulated ones. Together these bound the similarity in spatial patterns between simulated and actual land cover changes, but in the latter case high similarities can be found even when comparing against a random map, and as a result minimum similarity is likely a better indicator [69].

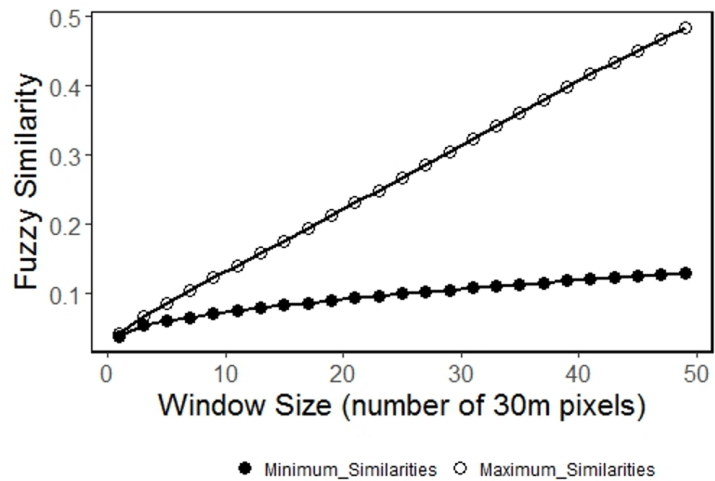


Fig. 5.24 Validation of probability maps for each land-cover transition. Plots show the number of actual land-cover changes during the 2001-2011 reference period (on the y-axis) as a function of modeled probability of land-cover change. Curves shown are loess regressions and shaded areas depict 95% confidence intervals.

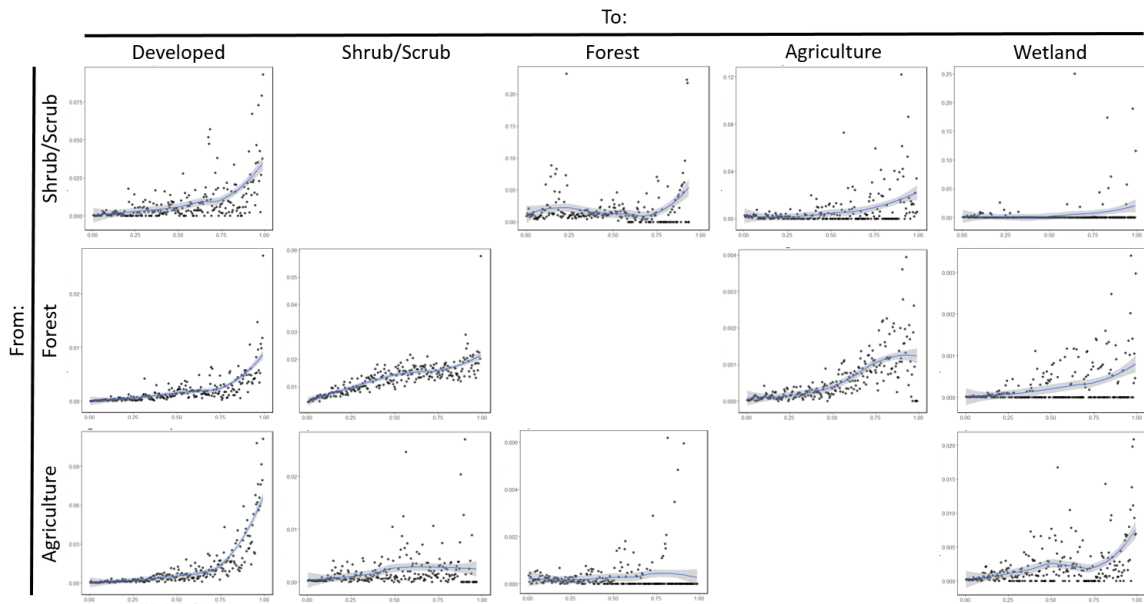


Fig. 5.25 2060 land-cover composition, final landscape maps, and maps of land-cover changes under each scenario.

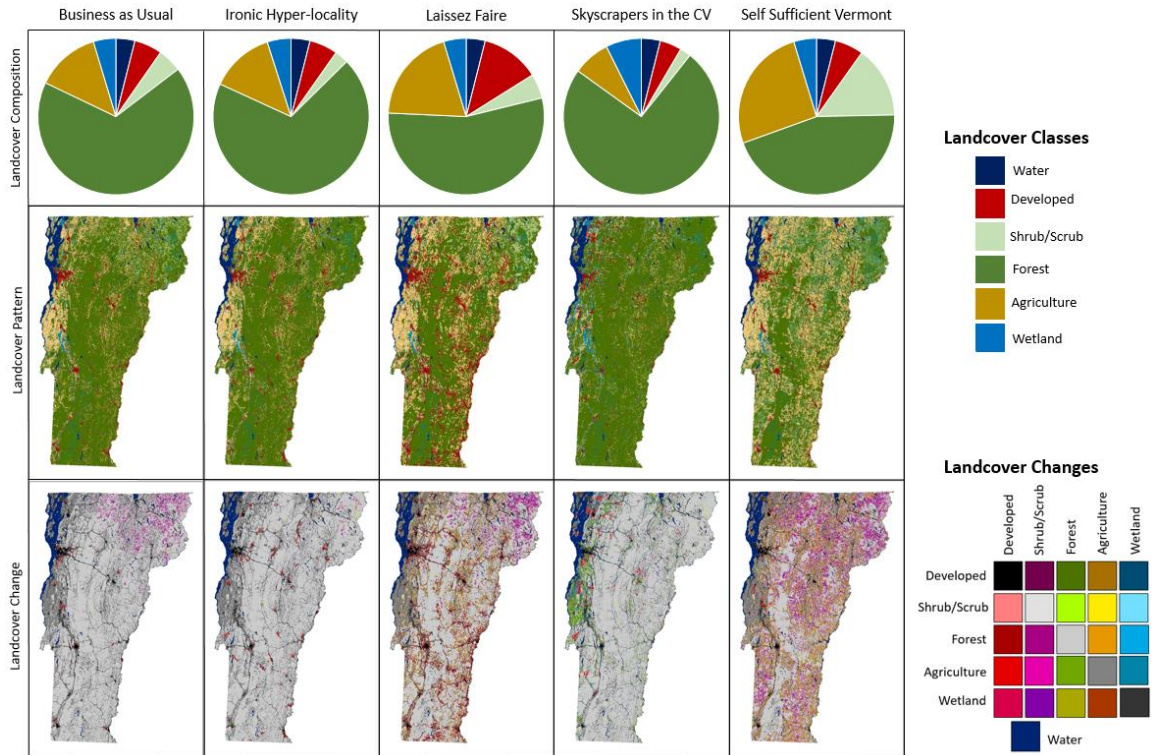


Fig. 5.26 Flood mitigation outcomes of each scenario. A) The spatial distribution of changes in development demand. B) The spatial distribution of changes in agricultural demand. C) The spatial distribution of changes in quick-flow (where an increase in quick-flow represents a decrease in the supply of flood mitigation) and D) the cumulative statewide degree of change in each.

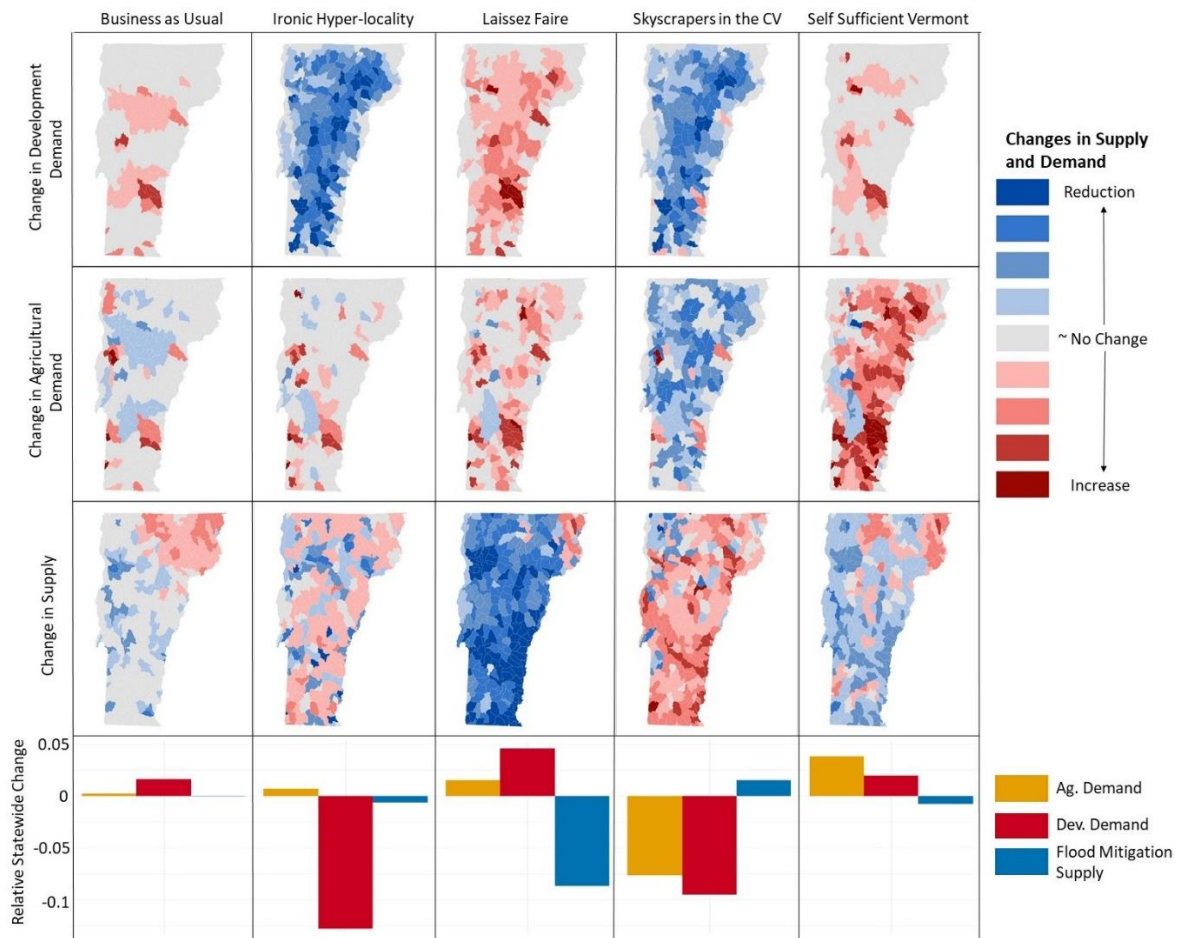


Fig. 5.27 Threat assessment of flood mitigation ecosystem services in each scenario.

The table is adapted from [45].

| Threat Category | Definition | |
|--------------------------|--|---|
| Functionally extinct | Service is no longer supplied in the region and is practically unrecoverable | |
| Dormant | Service is no longer supplied in the region but is potentially recoverable | |
| Critically endangered | Current levels of demand exceed supply and the ratio of supply to demand is declining or is expected to decline | Self Sufficient Vermont Laissez Faire Business as Usual |
| Endangered | Current levels of demand exceed supply; ration of supply to demand is stable but supply is declining | |
| Stable but undersupplied | Current levels of demand exceed supply; neither supply nor ratio of supply to demand is declining | Skyscrapers in the Champlain Valley Ironic Hyper-locality |
| Vulnerable | Ratio of supply to demand is declining or expected to decline such that supply is likely to be insufficient to meet demand with a set time horizon | |
| Least Concern | Supply currently meets or exceeds demand, and does not meet the criteria for Vulnerable | |

Fig. 5.28 The number of scenarios in which the Vermont landscape is projected to experience A) natural cover loss at a 30m resolution and B) increased flood risk at a watershed scale.

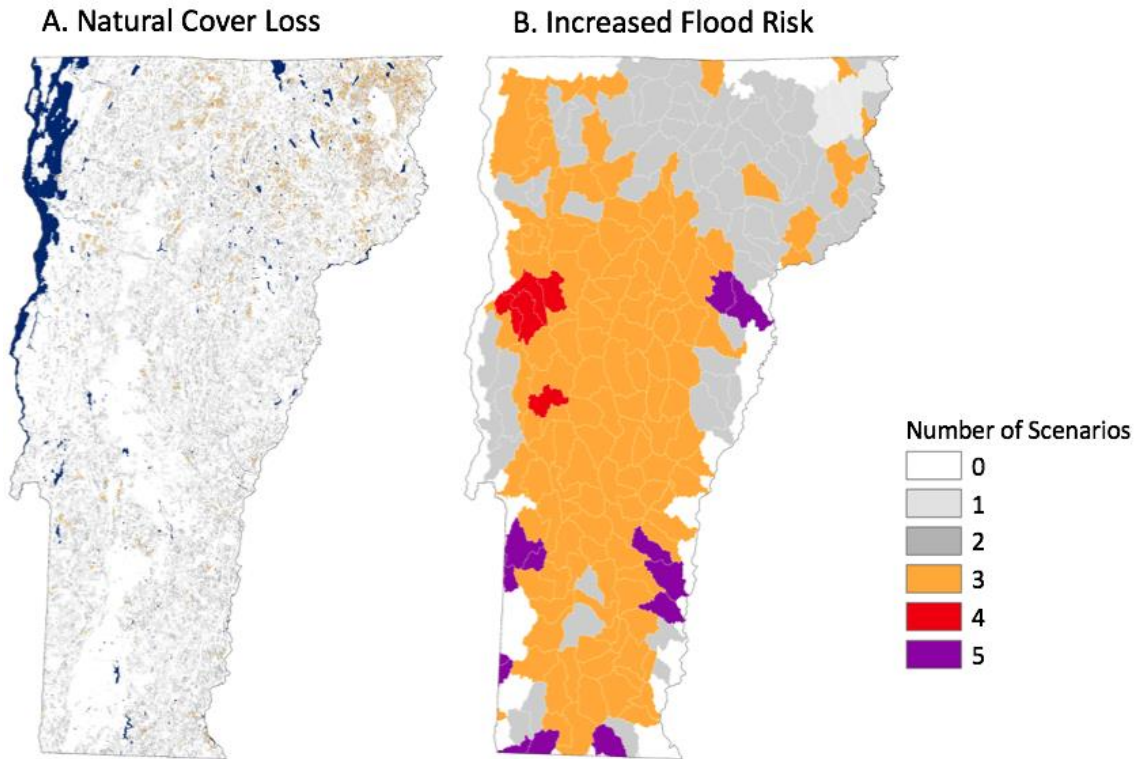
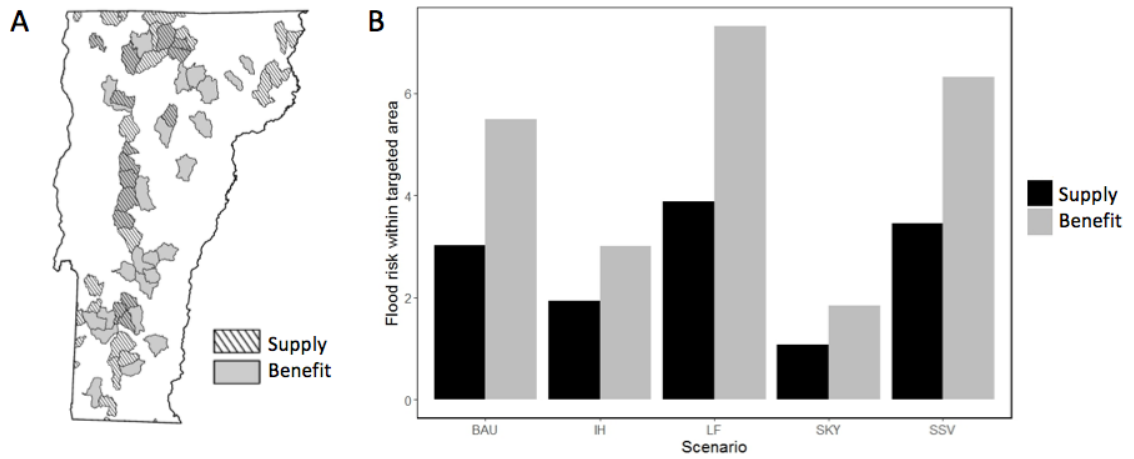


Fig. 5.29 Targeting present day supply, V.V. targeting present day benefit. A) Map of the watersheds identified as priorities based on 2011 flood mitigation supply and benefit and B) The relative flood risk in the year 2060 of watersheds prioritized based on 2011 flood mitigation supply and benefit.



CHAPTER 6: CONCLUSIONS

In this dissertation, I begin by quantifying the economic value of a single ecosystem service, flood mitigation, to the town of Middlebury, Vermont. I estimate that the mean annual value of upstream wetlands and floodplains is between \$126,000 and \$459,000, and that these wetlands reduced damages from Tropical Storm Irene by over ninety percent. Most broadly, this indicates that overlooking the value of ecosystem services when making land use decisions can have large consequences for human well-being. The challenge that emerges from this conclusion is to safeguard those places that are currently providing substantial benefits to people. In the remainder of the dissertation, I address three questions that are each important to efficiently targeting conservation for ecosystem services.

Ecosystem services have two different components: the biophysical supply of services by landscapes, and the demand for services by human beneficiaries. Ecosystems only benefit people when sources of supply are connected to sources of demand via ecosystem service flows. However, many efforts to identify spatial priorities for ecosystem services focus primarily or exclusively on supply. In the second chapter, I compare the ecosystem service and biodiversity benefits provided by priority areas when those areas account for demand, and when they do not. I find that demand shifts the spatial distribution of ecosystem services such that supply serves as a poor proxy for real benefits. I also find that targeting the places that benefit people the most does not exacerbate trade-offs between biodiversity and ecosystem services.

In the third chapter, I look more closely at the tradeoffs involved in targeting ecosystem services alongside biodiversity. I find that the spatial coincidence between specific taxa and ecosystem services varies widely, and is largely determined by whether optimal conservation targets for each are value seeking or cost minimizing. I also find that the overlap of biodiversity and ecosystem services improves as the number of ecosystem services and the breadth of taxa used in defining conservation priorities increases, such that organizations broadly targeting ecosystem services are likely to achieve significant biodiversity co-benefits, even if they do not explicitly seek them out. Furthermore, the financial costs of achieving ecosystem service goals in addition to biodiversity conservation are likely low compared to the additional funds garnered by framing conservation around human benefits. Overall, we find that ecosystem service conservation is more likely to boost biodiversity outcomes than to undermine them.

Finally, I generated four scenarios about what Vermont's landscape might look like in the future, and coupled these spatially explicit models of land-cover and flood mitigation ecosystem service. This revealed that we face high uncertainty when identifying future conservation priorities for flood risk, and that the current spatial distribution of flood risk is not a good indicator of future risk increases. I test whether the primary conclusion from Chapter 2, that incorporating demand is critical to spatially targeting benefits, holds true over longer time spans. I find that this conclusion is robust to the degree of uncertainty encapsulated by scenarios envisioned as plausible by a group of Vermont stakeholders.

Collectively, this body of work points to several policy implications for the state of Vermont. Planning efforts should consider how to reduce flood risk. In my second chapter, I find that protecting naturally functioning floodplains in the state is a cost effective means of promoting flood resilience. We do not measure the potential effect of restoring the hydrologic function of floodplains where it has been lost. This is a potential opportunity, although it would be much more costly than protecting existing floodplains. Vermont currently benefits from predominantly forested watersheds outside of floodplain areas, and there is little room for reducing runoff by increasing natural cover in headwaters. Thus protecting hydrologic function of floodplains before it is lost is likely to be a unique opportunity to reduce flood risk at low cost. A second key opportunity for reducing flood risk is minimizing floodplain development wherever possible. In our scenario analysis, we find the increases and decreases in flood hazard, in the form of floodplain development, play a large role in determining whether flood risk will increase or decrease into the future.

This body of work also points to some of the opportunities and challenges that Vermont will face in pursuing a sustainable future. In Vermont, conservation efforts that seek to simultaneously protect biodiversity and multiple ecosystem services that flow from relatively natural landscapes (e.g. flood mitigation, nature-based recreation, crop pollination, and carbon storage) are likely to find efficient ways of doing so. This is likely a particularly large opportunity in Vermont: because a majority of the state is forested

and the state does not comprise a large diversity of ecosystem types, there is more room to find win-win opportunities than there would be in a location with higher endemism or less remaining natural land. On the other hand, efforts that seek to protect a single ecosystem service will need to be careful to locate specific opportunities for biodiversity co-benefit. Furthermore, our stakeholders voiced two different visions of Vermont becoming more sustainable. These two visions involved different, and often contradictory, land cover trajectories. Future planning efforts should consider how to balance Vermont's goals of local food, energy, and forest products production with the desire to safeguard the state's natural communities.

Overall, this body of research highlights the importance of having good spatial information when doing conservation planning for multiple criteria. However, where this information is lacking, our results point to two potentially helpful heuristics for identifying conservation opportunities. First, optimal solutions for almost all criteria tended to concentrate around low cost areas, thus cost minimizing is a relatively good heuristic in Vermont. Secondly, optimal conservation areas for multiple ecosystem services and biodiversity tended to resemble Vermont's current set of conserved lands, indicating that the areas adjacent to existing conserved lands are likely to be good investments.

Theoretically, this body of research indicates that efficiently and effectively conserving ecosystem services is critical to a more sustainable land system. Efficiently safeguarding

the benefits from nature to people requires placing equal emphasis on the human and biophysical aspects of ecosystem services, and does not necessarily involve steep tradeoffs for biodiversity. Targeting ecosystem service conservation for future generations will be more difficult than identifying efficient conservation targets in the short run, and the scenarios I present here can serve as a tool in identifying future vulnerabilities and priorities. My hope is to pair these scenarios with a broader set of ecosystem services in the future.

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APPENDIX: Modeling Land-cover Change Based on Scenario Narratives

Self Sufficient Vermont

Amount of Agriculture

To model this scenario, we adopt the “omnivore’s delight” vision of agricultural production from “A New England Food Vision” [88]. This scenario involved a diet where approximately half of the footprint of Vermonters diets falls within the New England. This would require approximately 15% of the landscape to be used for agricultural production, a three-fold increase on its current ~5% across New England. According to “omnivore’s delight”, this large increase in the amount of New Englander’s food footprint that occurs regionally is possible will a relatively small increase in ag land due to a diet shift that hinges on consuming less dairy. Our Self sufficient Vermont scenario specifies that that agriculture will continue to thrive in Vermont, but that dairy production will decrease. This focus on regionalizing food footprints by shifting diets away from dairy thus make the “omnivore's delight” scenario highly compatible with out “Self Sufficient Vermont” scenario. However, we needed to estimate how much of this regional expansion of ag would occur in Vermont, whose land cover is currently about 13% agricultural. On the one extreme, if we assume that the proportion of agriculture in New England that occurs in Vermont will remain roughly the same, and triple the amount of ag land in the state such that 39% of the state is used for food production. The alternative would be to assume that each state equally shares this burden, and Vermont expands its agricultural system the additional 2% such that 15 percent of the state is used for food production. We take the mean of these two extremes, such that in this scenario

27% of the state is in an agricultural land cover. We accomplish this increase in agricultural land by applying an equal 0.0375 transition rate for the transitions from shrub/scrub to agriculture and forest to agriculture.

Pattern of Agricultural Expansion

In this scenario, agriculture expands on the best agricultural soils. To implement this, we assigned a positive weight of 5 to prime farmland and farmland of statewide importance, a weight of 3 to areas classified as prime farmland if drained, and prime farmland if protected from flooding, a weight of 2 to farmland of statewide importance if drained and farmland of local importance, and a weight of -5 to places that were not prime farmland for all transitions to ag land. We also assumed that in this case there would be more proactive planning of agriculture's footprint, even though that footprint would be large. To implement this in the model we applied a 1.1 patch isometry to ag lands, creating slightly more isometric patches and less random patterns for new agricultural lands than in the business as usual scenario.

Spatial Pattern of Development

In this scenario, proactive environmental policy mindful of the losses of natural communities due to food, energy, and timber production has a stronger influence in concentrating new development around city and village centers. The Vermont Department of Housing and Community Development has developed a dataset of growth

centers consisting of downtowns, town and village centers, neighborhoods and other areas designated for new growth [74]. In the business as usual scenario, the weights of evidence favoring growth centers were positive (2.44 for the transition from shrub/scrub to developed and 4.68 for forest to developed), but negative weights of evidence for areas outside of these growth centers were approximately zero. We assigned a woe of -2 for shrub/scrub to developed and forest to developed outside of growth centers. We also applied a 1.1 patch isometry to new developed lands, creating slightly more isometric patches and less random patterns than in the business as usual scenario.

Forest Products Harvest

The transition from shrub/scrub to forest exceeded the transition from forest to shrub/scrub in the BAU scenario. In the Working Landscape scenario, timber harvest increases such that most demand for forest products are met within the state. To implement this in the model, we increase the transition rates for these transitions are equal, to approximate a situation where forest cutting and regrowth are equal. We also increase the amount of forest to shrub/scrub transition that happens within current use areas by increasing the negative WOE for a forest to scrub transition outside of current use parcels from -2.17 to -3.17, and we increase the positive weight of evidence for this transition within current use areas from 2.77 to 3.77.

Wetland Loss

We assume that there will be no loss of legislative wetlands in this scenario, and set all transition rates from wetland to other land cover types to zero.

Ironic Hyper-locality

Development Pattern:

This scenario stipulates that the amount of developed land does not increase. As Vermont becomes increasingly gentrified as a location for the wealthy elite, poorer demographics cannot move to or thrive in Vermont, and the population increases expected in other scenarios do not occur. However, the pattern of development changes in two important ways:

Sprawl:

Under weak governance forest fragmentation increases as second home owners and wealthy residents develop homes in the most aesthetically appealing places. The poorer population of Vermont is housed in sprawl pattern development. To implement this pattern of development, we decreased the slope of the negative relationship between distance from roads and/or distance from existing developed lands and the woe for a transition from wetlands, forests, or scrub to developed land.

Move out of floodplains:

Secondly, people are forced out of floodplains because the financial structures no longer exist to subsidize rebuilding in flood-plains after floods. To implement this change, we allocated a transition rate of 0.01 for transitions that represent the loss of developed land and regeneration of floodplain forests and wetlands. We also changed the transition rate to 0.01 for shrub/scrub to developed and forest to developed to compensate for these losses. We changed the weights of evidence for floodplains for these transitions accordingly: We assigned a large negative weight of evidence for new floodplain development (-10), and a positive weight of evidence for new development outside of floodplains (5). According to current trends, there are too few transitions out of a developed land cover class to determine any statistically significant weights of evidence explaining the distributions of these transitions. As such, we assigned a weight of evidence of zero (no effect) for all explanatory variables where no logical relationship could be inferred, and made the following assumptions about these transitions:

- Essentially all loss of developed lands would occur within floodplains; we assigned a weight of evidence of -10 outside of floodplains and 5 inside of floodplains.
- Loss of developed land is most likely where population density is decreasing; we applied positive weight of 2 for places experiencing decreases in population density

- In this scenario families are forced to abandon their homes in floodplains because it is not economically feasible to rebuild. As a result this transition will have a larger effect in lower income rural areas in the Northeast Kingdom and southern Vermont. We applied decreasing weights of evidence with increasing median household income (woe=2 for medhhi of 0-30,000, 1 for 30,000 to 50,000, and 0 for 50,000-69,000)
- Basic infrastructure like roads is likely to be rebuilt, whereas private homes and businesses would not. To maintain the integrity of the road system within floodplains, we assigned a -10 weight of evidence for areas within 30m of roads (30m = 1 pixel).
- When developed areas are lost within floodplains, they are most likely to transition to wetlands in identified potential wetland restoration sites. We applied a weight of evidence of 4 for the transition from developed land to wetland for wetland restoration sites.
- When developed areas were lost within floodplains, they were most likely to transition to agricultural lands nearby existing agricultural lands, and applied a decreasing woe with distance from agriculture.

Maintenance of a Rural Aesthetic:

Decreased the 5→4 transition by an order of magnitude, to prevent the forest harvest and regeneration seen in the business as usual scenario, In this scenario the aesthetic components of the landscape, like forested hillsides, will be maintained outside of low income sprawl zones

Laissez Faire

This scenario in many ways represents an exacerbation of market driven landscape changes. As a result, we kept many of the original weights of evidence from the business as usual scenario, with the following changes:

Amount of Development:

To represent the expansion of developed lands we assigned a 0.01 transition rate for the conversion of shrubland and forests to developed land.

Development Pattern:

In this scenario new development occurs in a sprawl band pattern. To implement this, we applied the same weights of evidence for distance to roads and distance to existing developed lands weight for all “to development” transitions as we did for the ironic Hyper-locality scenario, which also involved these relationships weakening to allow for an increase in low density development. To further spur sprawl type development, we allocated 20% of new developed lands to the expansion of existing developed lands, and allowed 80% of these transition to occur as the formation of new developed patches.

Pattern of Agricultural Expansion:

In this scenario agriculture becomes more intensive and the amount of agricultural land increases, although not as much as in the “Self-Sufficient Vermont” scenario. To implement this increase in agricultural lands, we applied 0.01 transition rate for transitions from shrub/scrub and forest to agriculture. On average farm size increases, although some small boutique farms remain. As a point of comparison a farm of about 50 acres, or about 20.5 hectares, counts as a certified small farm operation in Vermont (<http://agriculture.vermont.gov/node/1322>). To implement this, we assumed that 80% of agricultural expansion would occur by expanding existing agriculture, with just 20% implemented as the formation of new patches. We then set a very large patch size (300 ha for from conversion of shrub/scrub and forests to agriculture, and 100 ha for the conversion of developed areas and wetlands) to simulate the formation of large agricultural productions for export outside of the state. The formation of new ag patches was set to a very small patch size (15 ha) to simulate to formation of small boutique farms that sell specialty products at farmers markets. The expansion of urban, shrub, and forest areas into agriculture was set at 60 ha to simulate the loss of medium sized farms.

Skyscrapers in the Champlain Valley

Forest Regrowth and Expansion:

This scenario envisions the widespread expansion or forest regrowth as the product of a strong environmental visions, without the pressures of a localized economy as in the Self-

Sufficient Vermont scenario. To implement, we parameterized this scenario to have no forest loss. We change all transition rates out of forest cover to zero, except for the transition rate from forest to wetland, where we retail the business as usual rate.

Without the pressure of a local food economy, the focus of conservation shifts such that conserved lands no longer act to support small scale agriculture. To implement this we assigned a weight of evidence of 10 for the transition from ag to forest in conserved lands with a gap status of 1-4, and a weight of evidence of 5 for places with a gap status of 39.

Wetlands and Floodplains:

In this scenario there is also a renewed focus on river corridors, riparian connectivity, and wetland restoration. To implement this, we increased all transitions into wetlands by one order of magnitude, and assigned a +15 weight of evidence for new wetlands in identified wetland restoration sites. This approximates a case where all identified wetland restoration sites are converted to wetlands. We also implemented the same weights of evidence and transition rates for the loss of developed lands within floodplains as in the Ironic Hyperlocality scenario with the following alterations:

- o Changes were not income driven in this scenario, so we removed the weight of evidence for median household income.
- o We added a -5 weight of evidence for urban growth in floodplains.

- o We removed the negative weight of evidence associated with distance to roads; this scenario envisions that roads and infrastructure would also be moved outside of flood prone areas where possible.

Development Pattern:

This scenario envisions high density smart growth in former agricultural lands to compensate for the contraction of low density sprawl and the move away from floodplain development. To implement the move out of floodplains, we set the transition rate from developed to all other transitions to 0.01, which is the same as ironic Hyper-locality scenario. We increased the transition rate from agriculture to urban to 0.001 to compensate for the loss of developed lands to natural cover. We assumed that nothing other than agriculture would transition to development except for a few pockets within growth centers (to do this, we applied a 0.00001 transition rate for from shrub/scrub to development within designated growth centers). To ensure that new development would occur in a “smart growth” pattern, we assigned a +10 for new development in designated growth areas. We also changed developed land patch isometry to 1.5, and the patch size to 40ha.

Agriculture:

Without the pressure of a local food economy the extent of agricultural land contracts in this scenario. We increased the transition rate from agriculture to forests by two orders of magnitude (This is the rate increase needed to ensure that a larger total area of ag

converted to forest than the developed land each time step). We concentrated remaining agriculture on the best agricultural soils by assigning prime farmland given a positive weight of 5, farmland of statewide importance a positive weight of 1. All other classifications (e.g. areas that need to be drained or protected from flooding to be good for agriculture, and places listed as only of local importance) were given a -2 weight. We also increased the patch isometry of agriculture to 1.1.