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Modeling the Production of Multiple Ecosystem Services from Agricultural and Forest Landscapes in Rhode Island

Tingting Liu, Nathaniel H. Merrill, Arthur J. Gold, Dorothy Q. Kellogg, and Emi Uchida

This study spatially quantifies hydrological ecosystem services and the production of ecosystem services at the watershed scale. We also investigate the effects of stressors such as land use change, climate change, and choices in land management practices on production of ecosystem services and their values. We demonstrate the approach in the Beaver River watershed in Rhode Island. Our key finding is that choices in land use and land management practices create tradeoffs across multiple ecosystem services and the extent of these tradeoffs depends considerably on the scenarios and ecosystem services being compared.

Key Words: climate change, ecosystem services, land use change, SWAT, tradeoff analysis

Over the past century, human-dominated land uses have spread rapidly across landscapes all over the world (Food and Agriculture Organization 2012). In the eastern United States, a major trend is that urbanization is causing both forest and agricultural lands to decline (Zhou et al. 2010). For example, in Rhode Island, urban sprawl has affected landscapes across the state with residential areas spreading further away from the city of Providence (Rhode Island Division of Planning 2006). In addition, the remaining working farm land has become more intensively managed. Combined, these land use and land management changes are leading causes of losses in valuable ecosystem services associated with managed forests and agricultural lands such as provision of clean water, regulation of streamflow, and support of wildlife habitat (Hascic and Wu 2006).

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One challenge associated with enhancing ecosystem services in Rhode Island is that about 90 percent of land in the state is privately owned (Natural Resources Council of Maine 1995). Owners of agricultural and forest land provide private goods in the form of crops and timber. However, they do not have incentives to protect ecosystem services that provide public goods, such as water quality and environmental flow, which is the amount of flow necessary to maintain aquatic habitat. These issues call for public policy to motivate private owners to provide these types of ecosystem services.

Another challenge for decision-makers in designing policies to protect or enhance multiple ecosystem services in a landscape is that they need to make tradeoffs across those services. Conversion of agricultural land into residential and commercial developments may spur regional economic growth and increase a tax base but lead at the same time to even worse water quality and increased flood risk. To inform decision-makers, it is necessary to make a systematic assessment of the potential tradeoffs across multiple ecosystem services that arise as a result of land use and management decisions. However, policymakers often lack the funding and/or expertise to develop methods with which to evaluate complex tradeoffs involving land use changes, land management practices, and the influence of both on valued ecosystem services. One solution would be to adapt existing models and data to provide for characterization of ecosystem services associated with different land uses.

Despite its importance, quantitative information at the landscape scale that is useful for decision-makers remains scarce. Some limited economic research has been done on ecosystem services related to water quality, such as nutrient and sediment loading (Kling 2011, Swallow et al. 2009) but rarely has focused on services related to water quantity, such as environmental flow and flood risk. Moreover, previous studies on ecosystem services have focused on one or two hydrological ecosystem services[1] (Kling 2011, Swallow et al. 2009); few have looked at tradeoffs among multiple services (Nelson et al. 2009, Lautenbach et al. 2010). Lastly, most of the economic studies that used a spatially explicit hydrological model were conducted in the context of Chesapeake Bay (Richardson, Bucks, and Sadler 2008, Tomer and Locke 2011) and the Upper Mississippi River Basin (Wu and Tanaka 2005, Kling 2011). These gaps in the literature arise, in part, from conceptual and computational challenges associated with (i) demonstrating links between choices in land use/management and changes in hydrological regimes and (ii) linking changes in hydrological outcomes to shifts in multiple ecosystem services that benefit people (Korsgaard and Schou 2010).

To address these gaps in the literature, we focus on hydrological ecosystem services—water quantity (environmental flow and flood risk) and quality (nitrogen and phosphorus loads). In some areas, freshwater rivers and streams are stressed by overwithdrawal of water (Watershed Counts 2012). As humans withdraw a growing share of available freshwater, less is available to maintain vital ecosystems. Already, freshwater fish species in Rhode Island are threatened and declining (National Oceanic and Atmospheric Administration National Marine Fisheries Service 2009). Resiliency to flood risk is a critical ecosystem service in Rhode Island and other New England regions, especially in light of expansion of impervious cover that comes with urbanization, which can lead to increased occurrences of flash flooding and the magnitude of precipitation

[1] Hydrological services are water-related ecosystem services that include both quantity and quality of water.

events generated by climate change. The quality of water in lakes for recreation and health risks associated with drinking water are growing concerns in Rhode Island (Rhode Island Department of Environmental Management (RIDEM) (2012). A contribution of this research is our examination of spatial heterogeneity and tradeoffs in provision of multiple ecosystem services within a watershed, which provides information that can assist stakeholders in targeting conservation efforts. This study also is one of the first to examine tradeoffs among hydrological and other ecosystem services in the northeastern United States. In addition to studying the impact of best management practices (BMPs) (the focus of other studies), we examine the impact of shifts in land use from agricultural and forest land to residential development, one of the key stressors to ecosystem services in the region.

Our overall goal is to demonstrate a method for spatially quantifying multiple ecosystem services and potential tradeoffs at the watershed scale. We examine changes in ecosystem services that result from alternative scenarios based on key stressors and factors—land use change, land management practices, and climate change—using an existing hydrological model and data. First, we quantify key hydrological ecosystem services under current land cover, land management, and climatic conditions. Second, we develop seven scenarios based on the key stressors. We simulate the effects of those scenarios on hydrological ecosystem services and crop production in both biophysical and monetary terms. Third, we illustrate how tradeoffs can be examined across ecosystem services that arise from the scenarios when the data set is sufficient to characterize the ecosystem services deemed relevant to land use policy. We also show how such an analysis could be used to identify particular areas within the watershed that can contribute a combination of services that could benefit the watershed as a whole.

One of the challenges in measuring tradeoffs among ecosystem services is ensuring that ecological and hydrological models reflect the complexities, nonlinearities, and dynamic nature of the ecosystem (National Research Council 2005). To infer the effects of land use and management choices with useful spatial detail for decision-makers, we use the Soil and Water Assessment Tool (SWAT), a process-based, spatially explicit hydrological model. Since each piece of land plays an intricate part in the watershed, the stressors have heterogeneous effects on the function of the ecosystem that depend on where changes take place. A caveat to our analysis is that it includes ecosystem services that are relevant only to environmental flow, flood risk, and water quality and does not provide a complete accounting of all private and public benefits and costs associated with land uses in the watershed. However, we show how tradeoffs across selected ecosystem services could be evaluated qualitatively using graphing and mapping methods.

Methodology

We demonstrate our approach using the Beaver River watershed as a case study[2] (Figure 1). Covering about eight square miles in southern Rhode Island, the watershed is lightly developed with only 2.3 percent of land having been converted to residential and commercial uses and more than 90 percent

[2] The Beaver River streamflow monitoring gauge is located at the outlet of the Beaver River watershed in Washington County (Hydrologic Unit 01090005, U.S. Geological Survey Water Resource).

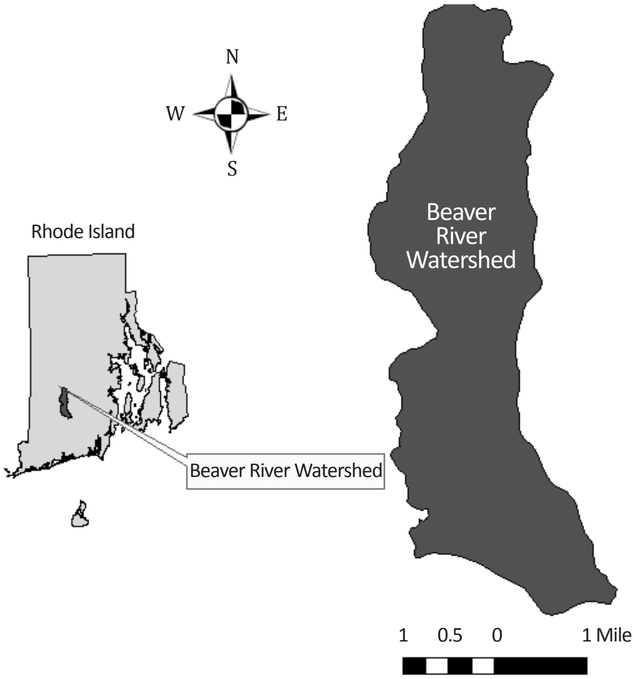

Figure 1. Location Map of the Study Area

Source: Rhode Island Geographic Information System.

remaining as deciduous forest, softwood forest, and mixed forest (Rhode Island Geographic Information System (RIGIS) 2012). Agricultural land uses comprise only about 0.9 percent of the total area. During the past three decades, the amount of agricultural land declined by 1 percent and the amount of deciduous forest declined by 5 percent while conifer and mixed forests increased by about 2 percent and 3 percent, respectively.

The Beaver River watershed is exemplary of a watershed that is important for hydrological ecosystem services such as environmental flow and water quality.[3] It is one of the major tributaries to the Pawcatuck River, beneath which lies a supply of groundwater that serves as the sole source of drinking water for more than 60,000 local residents (The Nature Conservancy 2012). Additionally, it supports roughly 70 percent of Rhode Island's globally imperiled species, including the ringed boghaunter dragonfly (*Williamsonia lintneri*) (The Nature

[3] The Beaver River watershed is comprised of first- through third-order streams that represent headwater tributaries of a larger watershed. These low-order streams account for approximately 60–80 percent of total stream length within most watersheds (Leopold, Wolman, and Miller 1995, Shreve 1969) and typically drain 70–80 percent of the total watershed area (Sedell et al. 1990, Meyer et al. 2001). Given their location and abundance within a stream network, headwater streams significantly contribute to the hydrological, physical, chemical, and biological integrity of downstream waters (Meyer et al. 2001, Nadeau and Rains 2007, Vannote et al. 1980). In New England, these headwater streams provide spawning and nursery grounds for coldwater fisheries and anadromous fish. Further downstream, riverine functions and values are frequently dominated by the effect of dams, reservoirs, and point sources of pollution. The ecosystem functions of headwater streams such as those found within the Beaver River watershed are most influenced by land use and nonpoint pollution, factors that are simulated by models such as SWAT.

Conservancy 2012). However, we acknowledge that a limitation of focusing on a small watershed such as the Beaver River is that we are not capturing the effects of the scenarios on ecosystem services in areas further downstream. Externalities may occur not only at other locations downstream but also at different points in time.

Soil and Water Assessment Tool Model

We use SWAT, a spatially explicit hydrologic model, to quantify the effect of key stressors on hydrological ecosystem services in the Beaver River watershed. Developed by the U.S. Department of Agriculture's (USDA's) Agricultural Research Service, SWAT is a process-based watershed-scale model to simulate the quality and quantity of surface water and groundwater and predict the environmental impacts of changes in land use, land management practices, and climate. Compared to other hydrological models, SWAT has proven to be an effective tool for assessing water resource and nonpoint-source pollution problems for a wide range of scales and environmental conditions across the globe (Gassman et al. 2007). Moreover, it has been widely used to simulate the impacts of changes in land use, land management practices, and climate on the quality and quantity of surface water and groundwater. Importantly, in a recent study, Rabotyagov et al. (2010) found that SWAT generated site selections for a reverse auction more cost-effectively than selections from the Universal Soil Loss Equation (USLE) and Modified Universal Soil Loss Equation (MUSLE). One advantage of SWAT is that the model can be calibrated and validated to actual observations. This process allows SWAT to better reflect the physical process of water and pollutant flux in a watershed, an advantage in simulating environmental impacts of changes in land use/management and climate change. SWAT also has an advantage over other models in that it uses readily available data, can operate in large-scale basins, has the ability to simulate long periods of time, and has a history of success (Arnold and Fohrer 2005).The Beaver River watershed is at the lower bound of the range of watershed sizes for which SWAT is suitable (Srinivasan 2009).

Data

We compiled data from multiple sources to derive parameters that control the hydrologic process in SWAT. We used the U.S. Geological Survey's (USGS's) twelve-digit hydrologic unit codes and national hydrography data set plus a thirty-meter digital elevation model from the National Aeronautics and Space Administration (NASA) ASTER Global Digital Elevation Map to provide watershed configuration and topographic parameter estimation. For land use and land cover data, we used 2003/04 RIGIS land use and cover data. The soil map from the soil survey geographical database and slope and other attributes were obtained from the USDA Natural Resources Conservation Service (NRCS) (2009).[4] Daily precipitation data and maximum and minimum daily temperature

[4] The land use and cover data set is based on true-color digital orthophotography captured in 2003/04 at a two-foot-per-pixel resolution. The minimum mapping unit is 0.1 hectare for soil survey geographic soil polygons, 20 meters for the national hydrography data set, and 5 feet for the lake and pond data set.

data for 1961 through 2010 were collected at the Kingston Weather Station in Rhode Island.[5]

Definition of Hydrologic Response Units

The land use/cover, topographic, and soil data were compiled using ArcGIS and ArcSWAT.[6] We delineated 31 subbasins (see Appendix Figure 4).[7] Each subbasin was further subdivided into hydrologic response units (HRUs) that represented portions of subbasins that possessed unique combinations of land uses, soil types, and slopes. To define HRUs, we adopted a land use threshold of 10 percent, which limited the land use to categories that covered at least 10 percent of the subwatershed. Since agricultural land in the Beaver River watershed falls below that threshold but is an important part of this study, we retained HRUs with agricultural land. In addition, we created new HRUs for septic systems (no sewage treatment) based on population density (medium density residential area equaled 2.0 dwellings per acre; medium-low density residential area equaled 0.5 dwellings per acre). This resulted in 372 HRUs that were comprised of forest, agricultural, residential, septic system, and other land use types.

SWAT Calibration and Validation

Calibration and validation for the SWAT model followed an automated method developed by Arnold and Allen (1999) and was based on land use/cover for 2003 and 2004. Each SWAT simulation was executed for 1987 through 2010. This period included a three-year "warm up" period (1987–1989), a calibration period (1990–1999), and a validation period (2000–2010). The modeled streamflow for 1990–1999 was then compared to observed data on historical water discharges from the USGS gauge located at the outlet of the watershed.[8] Details of the sensitivity analysis are provided in the Appendix.

Graphical comparison of simulated versus observed monthly flows for the calibration period (1990–1999) showed that the model predicted average monthly flow reasonably well (see Appendix Figure 1). Moreover, the statistics for overall fit indicated that the model tracked average monthly flow trends during the validation period satisfactorily. The R-square of simulated versus measured monthly average streamflow was 0.78 and the Nash-Sutcliffe coefficient was 0.77.

In addition to calibrating the overall flow, which is the standard approach, we calibrated both tails of the distribution (lowest 5 percent, lowest 10 percent, highest 5 percent, and highest 10 percent of streamflow) to the observed data using a seven-day moving average (see Appendix Table 1). Based on benchmarks set by Moriasi et al. (2007), the overall simulation of the extreme events was satisfactory. For example, based on PBIAS (percent bias), which measures the average tendency of the simulated data to be larger or smaller than the

[5] Kingston weather station (374266) is located at latitude 41.4906 and longitude –71.5414 (U.S. Historical Climatology Network 2012).

[6] ArcSWAT is an ArcGIS extension and graphical user interface for SWAT developed by USDA's Agricultural Research Service.

[7] The watershed outlet (sampling site) is located on the right bank of the river ten feet downstream from Beaver River Bridge on State Highway 138 in Richmond (USGS).

[8] USGS 01117468, Beaver River near Usquepaug, Rhode Island.

observed counterpart (Gupta, Sorooshian, and Yapo 1999), our calibration of the seven-day moving average for tails of the distribution was categorized as "very good" for the lowest 5 percent and lowest 10 percent of the streamflow distribution. The calibration for peak flow was "good" for the highest 10 percent and "satisfactory" for the highest 5 percent of the streamflow.

Ecosystem Services and Their Indicators

For any study of ecosystem services, it is important to choose an appropriate set of indicators that can represent the services that are critical to maintaining human welfare and ecological integrity. We used simulated water discharges and nutrient loadings from SWAT simulations to calculate alternative indicators of environmental flow, flood risk, and water quality. Here we describe the indicators for each ecosystem service.

Environmental flow is the volume of streamflow needed to sustain ecosystems in downstream receiving wetlands, aquatic organisms, and the overall health and vitality of a river system (USGS 2012). Alterations in land use, differing management practices, and climate change can shift hydrology and hence the aquatic ecosystem by changing physical habitats and disrupting the natural connectivity of habitats (James et al. 2012). Many species may be influenced by altered flow regimes. In particular, species are sensitive to the timing of low-flow and extreme events. The issue of low environmental flow has become more and more critical in Rhode Island and elsewhere due to large uptakes of water to meet increasing demands for water (RIDEM 2012).

Since there is no single indicator for environmental flow, we follow the hydrology literature and measure environmental flow using four indicators that are complementary (James et al. 2012, Armstrong et al. 2004, Richardson 2005). Two widely used indicators are 7Q10 (seven consecutive days of low flow with a ten-year return frequency) and 30Q1 (thirty consecutive days of low flow with a one-year return frequency). In comparing scenarios using these two indicators, we used Scenario 1 (baseline) as the benchmark, a reasonable proxy for a fully forested watershed.

Although 7Q10 and 30Q1 describe the magnitude of changes in extreme (low probability but high impact) events, they do not provide information on how frequently such events may occur, which is correlated with how damaging such changes may be for aquatic habitat. Hence, we followed an approach by the U.S. Fish and Wildlife Service and used two additional indicators developed by USGS and RIDEM that set thresholds below which the aquatic ecosystem might be threatened: the Rhode Island aquatic base flow (RIABF) method and the New England aquatic base flow method (Armstrong et al. 2004, Richardson 2005). We counted the number of days per month during the 20-year study period (1990–2010) that the watershed's median streamflow was below the threshold to determine the percentage of below-threshold days per month (see Table 3 and Appendix Table 3). The same percentages were calculated using the New England aquatic base flow method (see Appendix Table 6).

We employed several indicators to measure flood risk: one-year floods, two-year floods, and ten-year floods (Table 2), which represent the largest streamflow annually, every two years, and every ten years on average.

Water quality was measured by total annual loading of nitrogen (N) and phosphorus (P). SWAT allows users to quantify nutrient loadings at the subbasin level as well as at the outlet of the watershed. We used both in the tradeoff analysis. As an extension, we also used a benefit-transfer method to value the impacts of changes in land use and management practices in monetary terms to reflect people's preferences across different ecosystem services.

Table 1. Seven Scenarios

Scenario	Land Use Change	Crop	Practices	Climate Change
Scenario 1: Baseline	Status quo			
Scenario 2: Conventional agriculture	Forest ⟶ Agricultural[1]	Corn silage	Conventional management	
Scenario 3: BMP agriculture			Best management practices, including reduction in fertilizer and a winter cover crop (rye)	
Scenario 4: Biofuel	Forest ⟶ Agricultural[1]	Corn	Conventional management	
Scenario 5: Suburban medium density	Forest ⟶ Residential[2] (medium)			
Scenario 6: Suburban medium-low density	Forest ⟶ Residential[2] (medium low)			
Scenario 7: Climate change	Status quo			Coupled General Circulation Model 3.1/T47

[1] We changed forest land with soil type suitable for farming to agricultural land use.
[2] We changed forest land with soil type suitable for construction to residential land use.

Table 2. Water Quantity and Quality Statistics from the Seven Scenarios

	Environmental Flow		Flood Period			Nutrient Loading	
	7Q10	30Q1	1-Year	2-Year	10-Year	Total N	Total P
	(cubic meters per second)		(cubic meters per second)			(kg/ha)	
Scenario 1: Baseline	0.025	0.043	2.114	2.803	5.838	24.626	0.483
Scenario 2: Conventional agriculture	0.021	0.037	2.081	2.839	5.718	157.142	1.037
Scenario 3: BMP agriculture	0.022	0.037	2.097	2.789	5.757	70.411	0.676
Scenario 4: Biofuel	0.022	0.038	2.101	2.794	5.74	42.656	0.464
Scenario 5: Suburban medium density	0.087	0.124	6.752	8.674	12.62	197.515	2.765
Scenario 6: Suburban medium-low density	0.041	0.068	3.805	5.294	8.557	205.666	1.169
Climate change baseline*	0.026	0.039	6.61	8.45	15.24		
Scenario 7: Climate change scenario*	0.022	0.037	7.42	8.98	22.58		

* Climate change scenarios were created using monthly averages and SWAT's WXGEN weather generator to create daily runs for SWAT input.

Land Use Change and Climate Change Scenarios

With the calibrated hydrological model, we investigated seven scenarios that reflect potential stressors to ecosystem services from this watershed (Table 1) and then ran SWAT from year 1987 to year 2010, which included a three-year warming-up period. Daily streamflow and nutrient loadings were simulated at the outlet of the watershed.[9] To do so, we created three new digital maps of projected land uses (Scenarios 2–6) and applied changes to the weather input to simulate climate change impacts (Scenario 7). The alternative scenarios were intended to illustrate the direction and extent to which ecosystem services would change. By using scenarios involving drastic land use/management changes, we illustrate the upper bounds and likely direction of potential changes in ecosystem services. The percentage of area in the watershed in each land use category under each scenario is shown in Appendix Table 2.

[9] Please refer to footnote 6.

Table 3. Average Percent of Days each Month below the Requirement of the Rhode Island Aquatic Base Flow

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Scenario 1: Baseline	22.1	42.7	25.2	25.5	46.1	65.2	42.3	37.1	22.0	10.0	10.5	11.1
Scenario 2: Conventional agriculture	22.4	43.2	27.9	26.8	48.5	69.7	44.0	38.5	25.7	11.6	11.5	12.9
Scenario 3: BMP agriculture	24.0	43.4	28.2	27.3	51.6	68.2	43.1	37.7	25.5	11.0	11.7	13.4
Scenario 4: Biofuel	22.1	43.4	27.3	26.8	47.7	67.0	42.6	37.6	25.2	11.8	11.5	12.9
Scenario 5: Suburban medium density	26.1	42.7	23.9	20.5	34.5	46.0	28.5	17.3	12.0	5.8	8.2	13.1
Scenario 6: Suburban medium-low density	20.3	38.2	19.8	19.7	32.4	49.3	33.5	28.4	19.2	8.7	10.7	12.3

Notes: The percentage of days below the threshold is averaged over 20 years. Results for Scenario 7 (climate change) are not reported since these values were calculated based on simulated daily flows. The climate change effects are simulated by imposing monthly changes to the weather so the simulated daily flows are not reliable.

Table 4. Modeled Average Monthly Changes in Climate: 1980–2000 versus 2045–2065

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Precipitation % change millimeters	6.9	–4.0	35.7	10.4	0.5	8.5	–33.7	–7.9	–9.9	0.4	33.8	19.0
Maximum temperature change degrees Celsius	2.1	0.7	4.2	3.0	2.4	2.6	2.3	2.0	2.4	3.2	2.4	2.4
Minimum temperature change degrees Celsius	2.5	1.3	4.2	3.3	2.4	2.3	2.6	2.3	2.5	3.0	2.8	2.1

Note: These changes were calculated from two 20-year runs of the CGCM3.1/T47 model. These were then applied to observed monthly average precipitation and temperature.

Scenario 1: Baseline. This scenario used status quo land cover (land use 2003/04), land management, and climatic data. More than 97 percent of the watershed was covered by forests (Appendix Table 2).[10]

Scenario 2: Conventional Agriculture. Under this scenario, all of the forest land that had soil attributes suitable for cultivation was converted to agricultural land. As a result, 16 percent of the forests were converted. We assumed that corn silage was planted on the new agricultural land.

Scenario 3: BMP Agriculture. This scenario assumed the same land use conversion as Scenario 2 but, in addition, we imposed a set of BMPs. Based on the literature and the expert opinion of an agricultural extension specialist in Rhode Island, we included reduced fertilizer application and a rye cover crop in winter (Arabi et al. 2008, Burdett 2010) as BMPs. Corn silage was assumed to be planted on farm land.

Scenario 4: Biofuel. We assumed the same land use conversion as Scenario 2, but corn suitable for biofuel was planted instead of corn silage. This scenario is relevant because farmers in Rhode Island, following the trend in the rest of the United States, have started to produce corn for ethanol fuel.[11] There are two major differences between these two types of corn that could affect water quantity and quality. Only half of the above-ground plant biomass is harvested in corn production whereas 90 percent is harvested for corn silage. In addition, corn provides more leaf cover at certain times than corn silage.

Scenario 5: Suburban Medium Density.[12] Under this scenario, we converted all the forest land that had soil properties that made it suitable for development to residential land use (about 54 percent of the watershed) to medium density residential (2.0 dwellings per acre).

Scenario 6: Suburban Medium-Low Density. This scenario assumed the same land use conversion as Scenario 5, but forest land was converted to medium-low density residential development (0.5 dwellings per acre).

Scenario 7: Climate Change. We examined the impact of climate change by assuming the baseline land use in 2003/04 (same as Scenario 1) (Appendix Table 2). From various climate change models available, we chose downscaled, bias-corrected model runs of a general circulation model (CGCM3.1/T47) because its fine resolution of one-eighth degree was more appropriate given the small size of our watershed (as opposed to the two-degree raw output from the same model). These model runs were conducted under the SRES A2 emission scenario, which implied a doubling of carbon dioxide (CO_2) concentrations by 2038 (Mearns et al. 2005, Pachauri 2007).[13] The downscaled data were made available by the World Climate Research Program's (WCRP's) bias-corrected, downscaled Coupled Model Intercomparison Project phase 3 (CMIP3) climate projections archive (Maurer et al. 2010).

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[10] Crop growth is simulated in SWAT using the modeling approach from the Erosion Productivity Impact Calculator (EPIC) (Williams, Jones, and Dyke 1984). EPIC allows for variation in growth for different plant species and variation due to climate and growth conditions (Neppel et al. 2002). Crop types and their biomasses (such as the canopy and its maximum leaf index) influence evapotranspiration and surface runoff and the speed of those processes.

[11] For example, Sodco, Inc. in southern Rhode Island started to grow corn for fuel in 2009.

[12] During the past couple of decades, there has been a 78 percent increase in residential development in Rhode Island with a decline in both agricultural and forest land (Archetto and Wang 2012). Though some of the scenarios we created were drastic, they simulated what could happen if current trends continue.

[13] The model runs were conducted as part of the World Climate Research Program's (WCRP's) Coupled Model Intercomparison Project phase 3 (CMIP3) multi-model data set.

To reflect simulated changes in temperature and precipitation, we followed the delta method suggested by Stone (2003) and the Intergovernmental Panel on Climate Change (IPCC) (2012). We extracted monthly differences in degrees Celsius and ratios for precipitation between modeled past data (1980–2000) and predicted future data (2045–2065). These simulated changes implied an increasing average maximum and minimum temperature for all months (with a range of 2–4 degrees) and a decrease in summer rainfall (range of 7–33 percent) (Table 4). We applied these differences to the observed monthly data, which we then used as inputs to the calibrated SWAT model to estimate hydrological outputs and crop yields. We then used two 20-year SWAT runs to compare differences in the relevant hydrological indicators from both periods.

Results of Scenario Simulations

The scenarios demonstrate the effects of land use/management choices clearly and verify the theoretical relationships expected (Table 2). Increased amounts of impervious surfaces lead to increasing surface runoff and result in larger floods and increased environmental flows (Allan and Castillo 2007). A reduction in the fertilizer application rate (kilograms (kg) per hectare (ha)) or adoption of other BMPs (Meals, Dressing, and Davenport 2010, Park et al. 1994) induces less nutrient loading. Conversion of forested land to agricultural land (Scenarios 2–4) results in a reduction of the environmental flow indicators. For example, converting 16 percent of the watershed from forest to corn silage fields (Scenario 2) decreases 7Q10 from 0.025 cubic meters per second (cms) to 0.021 cms, which is a 16 percent reduction in the environmental flow. Similarly, 30Q1 decreases from 0.043 cms to 0.037 cms, a 14 percent reduction. Changes in environmental flow indicators such as 7Q10 and 30Q1 reflect a drier extreme (lower low flows) with potentially detrimental effects for aquatic habitat (Richardson 2005).

We find that a conversion from forest land to crop land results not only in increases in magnitude but also in more frequent extreme dry events (Table 3). This effect is larger in the drier summer months of May, June, and July. In June, for example, a 16 percent conversion of the watershed from forest to corn silage resulted in an average of 4.5 percent more days that failed to meet the minimum threshold required to maintain the aquatic habitat. In contrast to the environmental flow indicators, the flood risk indicators showed only a minor effect under these scenarios, decreasing slightly in magnitude by 1 percent or remaining the same (Table 2).

Conversion from forest land to crop land has more drastic implications for water quality than water quantity (Table 2). Increased nitrogen and phosphorus is a result of nutrient runoff from agricultural land. Not surprisingly, converting large areas of forested land to agriculture results in increased concentrations of both nitrogen and phosphorus. In contrast to conventional agricultural practices (Scenario 2), implementing BMPs (Scenario 3*)* reduces these loadings by almost half. For example, the total nitrogen loading falls from 157 kg/ha to 70 kg/ha and total phosphorus loading drops from 1 kg/ha to 0.68 kg/ha.

Interestingly, growing corn instead of corn silage (Scenario 4) results in a significant reduction in total nutrient loads (Table 2). For example, compared to the scenario with BMPs (Scenario 3), total nitrogen loading drops from 70 kg/ha to 42 kg/ha and total phosphorus loading falls from 0.68 kg/ha to 0.46 kg/ha. This may reflect the difference in fertilizer applied (less is used to grow corn than corn silage)[14] and how much biomass is left on the ground after harvest. Only half of the above-ground plant biomass is harvested in corn production whereas 90 percent is harvested for corn silage.

Next, the results of the suburban scenarios (5 and 6) show that the urbanization trend could have an impact on our ecosystem services of interest (Table 2). The increase in impervious surfaces and conversion of forest cover lead to increases in base flow as measured by the environmental flow indicators. This comes at the expense of an increase in flood risk. For example, the 7Q10 indicator is 2.5 times larger while the two-year flood indicator is more than twice as large when forested land is developed into medium density residential. While an increase in environmental flow may be beneficial, development comes at the cost of water quality as well. Nitrogen and phosphorus loads increase greatly with development and with medium-low residential density without sewage systems (Scenario 6).

We apply projected changes in climate (see Table 4) to create the climate change scenario (Scenario 7) and find that climate change will reduce environmental flows during summer months and raise the flood risk in the winter (Table 2). Modeled changes in average daily flow by month are shown in Appendix Figure 2. Due to both decreased summer rainfall and additional evapotranspiration stemming from higher daily temperatures, environmental flows as measured by 7Q10 are projected to decrease by around 12 percent, resulting in the historically low-flow months of summer becoming drier and leading to even smaller environmental flows. Winter precipitation is predicted to increase by as much as 33 percent in some months. Flood events measured by high daily flows are also predicted to increase. For example, a current ten-year flood event may happen every 7 years, a two-year flood every 1.6 years, and a one-year flood every 0.6 years under the climate change scenario. These general results are consistent with other studies of climate change for the Northeast using an ensemble of climate models (Hayhoe et al. 2008).

It is worth noting that the climate model's ability to reproduce observed magnitudes, timings, and durations of precipitation events is susceptible to the high interannual variability of precipitation. For instance, any trends calculated as beginning or ending during multi-year drought events would change the results substantially (Hayhoe et al. 2006). Our results should be interpreted as the effects of a plausible series of precipitation events under a climate change scenario. Since the changes are based on deviations between modeled past and future monthly means, the changes in our indicators reflect only a mean shift in the observed precipitation distribution.

Valuation of Ecosystem Services

We next evaluated the impacts of stressors and land management practices in monetary terms to reflect people's preferences for different ecosystem services. A common metric of value makes a tradeoff analysis between varying goods and services easy to conduct and aggregate (Kumar et al. 2010). We resorted to the existing valuation literature and used a simple benefit-transfer method. Although benefit transfer may not be the most accurate valuation approach, it

[14] In Scenario 3 (BMP agriculture), we applied manure at 150 pounds of nitrogen per acre and 60 pounds of phosphorus per acre. This amount was significantly more than the amount applied in Scenario 4 (biofuel), which used the default value applied as 31.19 pounds of nitrogen per acre and no phosphorus.

has the advantage of being a less costly way to capture the relative importance of ecosystem services using a common scale and often is used as a screening technique at an early stage of policy analysis (King and Mazzotta 2000). Although we refrained from computing total net value from each scenario because we were not capturing the value of all ecosystem services, our results can be used to compare tradeoffs among alternative scenarios and serve as a pre-assessment of future policy scenarios.

Corn. Following an approach taken by the U.S. Fish and Wildlife Service, we assumed a constant of \$6.25 per bushel based on 2012 prices (USDA 2012). Following Snyder (2011), we priced corn silage at \$1.46 per bushel. We assumed that the profitability for both corn and corn silage was 22 percent (Ibendahl 2012).

Environmental Flow. Karanja et al. (2008) estimated WTP to maintain environmental flow was \$13 per year per person. Based on their study, we assumed that all Washington County residents were willing to pay \$0.03 per day to maintain the environmental flow needed to protect rare wildlife species in the watershed. According to the RIABF (Appendix Table 3), we can calculate people's WTP for 20 years to maintain the environmental flow by multiplying \$0.03 by the number of days the flow dropped below the RIABF threshold. Then we multiply that result by the number of residents living in Washington County based on U.S. census data (126,563) and divide by 20 years. In this way, we determine an approximate estimate of the benefit of the environmental flow per year.

Flood Risk. Based on historical peak flow data, we assumed that a streamflow of 250 cubic feet per second was the threshold for a flood event. To estimate the damage cost from a flood at the outlet of the Beaver River watershed, we started with the average flood insurance premium in Richmond, Rhode Island, which was \$1,717 per year for both buildings and contents in 2012 dollars (National Flood Insurance Program 2012). Dividing by a 10 percent probability of a flood event (based on historic streamflow observations), we estimated the expected damage from flooding for each household as \$17,170. Based on the number of households in a two-mile radius at the watershed outlet, we assumed, for simplicity, that 4,000 residents (1,300 households) would be affected by a flood event. We then multiplied the total damage cost per flood event by the number of predicted flood events under each scenario.

Water Quality. We took into account the effects of nitrogen and phosphorus on drinking water and recreation. Van Grinsven et al. (2010) estimated that the health cost of nitrate in drinking water at \$3.38 per kg. Birch et al. (2011) estimated the cost of damage to recreational use of an estuary due to eutrophication at \$6.38 per kg. Thus, for the total damage cost of nitrogen, we used \$10.14 per kg in 2012 U.S. dollars. For the damage cost from phosphorus, we used the estimated damage cost function for both drinking water treatment and recreation losses (Ancev et al. 2006).[15]

Residential Development. We used the per-acre vacant land price (without buildings) and the annual interest earned from selling the land as a proxy for the return from residential development by modifying the approach of Lubowski et al. (2002, 2008). The per-acre vacant land price was calculated by dividing the lands' assessed tax value by the number of acres in a lot. The median for vacant land was \$143,800 per acre for medium density residential development and

[15] Total cost was estimated by the damage cost function $\mathbf{D(Z)} = 585{,}446.9 - 59.93Z + 0.0015Z^2$ (Z denotes the average phosphorus concentration).

$71,500 per acre for medium-low density development in 2010 in Richmond, Rhode Island. Based on assumptions about land use changes in suburban residential development, $366,977,600 and $182,468,000 respectively would be the instantaneous benefits.[16] By combining that information with data on real interest rates (The World Bank 2012), we estimated the annual return from residential development as $35,156,454 for medium density and $17,480,434 for medium-low density residential development.

Comparison of Ecosystem Service Values across Scenarios

In contrast to the changes in indicators of ecosystem services examined earlier, the valuation exercise reveals the relative magnitude of changes and the tradeoffs across scenarios (Table 5). Our results for the agricultural scenarios suggest that increases in profit from growing corn dominate losses from smaller environmental flows and degraded water quality (rows 1 to 3). For example, in the conventional agriculture scenario (Scenario 2), conversion to corn silage creates an additional profit from crops of $65 million relative to the baseline. That amount far outweighs the monetary losses associated with environmental flow ($253,479) and larger losses from additional nitrogen ($2.7 million) and phosphorus ($0.063 million) compared to the baseline. By imposing BMPs (Scenario 3) and growing corn instead of corn silage for biofuel (Scenario 4), the results show a much smaller loss from nutrient loading.

Our results also indicate that the increase in costs associated with flood damage will be much larger under the suburban scenarios and will far outweigh the benefits of environmental flows (Table 5, rows 4 and 5). With conversion to agricultural land, the probability of flood is 5 percent each year. However, the risk increases to 10 percent in the medium-low density scenario and 75 percent for medium density residential development. For the suburban scenarios, the cost of flood damage is large because of an increase in nutrient loads. However, given our assumptions, the benefit from residential development outweighs the benefits lost in ecosystem services.

Tradeoff Analysis

In applications, it is important for policymakers to understand the extent to which tradeoffs and heterogeneity exist in providing ecosystem services within the watershed. Understanding heterogeneity in ecosystem services across different parts of a study area is essential for government agencies and conservation groups where the goal is to enhance multiple ecosystem services under a fixed budget. Although we lacked sufficient data to provide a complete accounting of tradeoffs among all policy-relevant ecosystem services in the watershed that could be influenced by the scenarios, we can illustrate how tradeoffs can be evaluated when a sufficient supply of data is available.

We took two approaches to assessing the tradeoffs. First, we examined the heterogeneity and tradeoffs within a watershed by measuring ecosystem service indicators for each of the 31 subbasins, graphing the distribution of two ecosystem services at a time, and comparing them across the six scenarios. We then focused on the conventional agriculture scenario (Scenario 2) and extended a mapping approach by Swallow et al. (2009) to examine the

[16] In Scenario 5 (medium density development) and 6 (medium-low density development), we assumed that there would be a 2,552-acre increase in residential development.

heterogeneity and tradeoffs within the watershed visually. We characterized the level of ecosystem service in each subbasin as "high" ("low") when the value exceeded (was less than) the median value of the 31 subbasins.

Results: Tradeoffs across Scenarios

We show the tradeoffs among ecosystem services considered in our analysis for the scenarios at the watershed level in Figures 2 through 4 and Appendix Figure 3. Each point represents a unique subbasin with a combination of crop yield (vertical axes) and 7Q10 (horizontal axes, Figure 2), two-year flood (Appendix Figure 3), and total nitrogen and phosphorus loading (Figures 3 and 4).

Several findings are interesting. First, the extent of heterogeneity depends on the ecosystem service. For example, under the baseline scenario (Scenario 1), variations by subbasin between crop yield and environmental flow (Figure 2, panel 1) or flood risk (Appendix Figure 3, panel 1) are small. However, we observe relatively large variability in total annual nitrogen loading. Some subwatersheds that have similar crop yields differ in the amount of nitrogen loading (Figure 3, panel 1). These results imply that subbasins have inherently different characteristics in generating some types of ecosystem (dis)services such as total nitrogen loading even without stressors or changes in land management practices. As an example, in the baseline scenario, subbasins 17 and 18 have about the same agricultural land use (Appendix Figures 6 and 7), but there is a big difference in nitrogen loading, suggesting that factors such as soil types, slopes, and other intrinsic characteristics influence nutrient loading. These findings are consistent with another tradeoff analysis that used different policy scenarios (Lautenbach et al. 2010).

Table 5. Annual Benefit in 2012 U.S. Dollars of Ecosystem Services from Alternative Scenarios Relative to the Baseline

				Nutrient Loading		
	Crop Profit	Environmental Flow	Flood Damage	Damage from N	Damage from P	Housing Value
Scenario 2: Conventional agriculture	$65,400,754	–$253,479	$0	–$2,744,532	–$62,544	$0
Scenario 3: BMP agriculture	$26,958,467	–$278,648	$0	–$948,251	–$22,225	$0
Scenario 4: Biofuel	$13,137,433	–$176,177	$0	–$373,418	–$2,213	$0
Scenario 5: Suburban medium density	$163,211	$891,672	–$14,422,800	–$3,580,695	–$232,951	$35,156,454
Scenario 6: Suburban medium-low density	$22,703	$735,270	–$1,030,200	–$3,749,510	–$76,880	$17,480,434

Note: Housing value is not an ecosystem service but it is included for comparison and tradeoff analysis purposes.

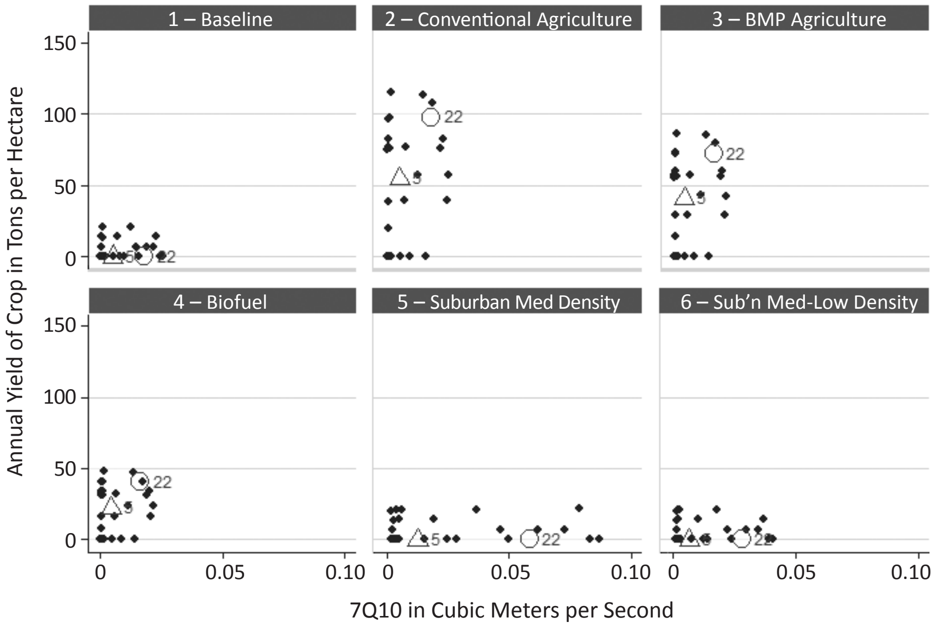

Figure 2. Tradeoff between Crop Yield and Environmental Flow in Different Scenarios

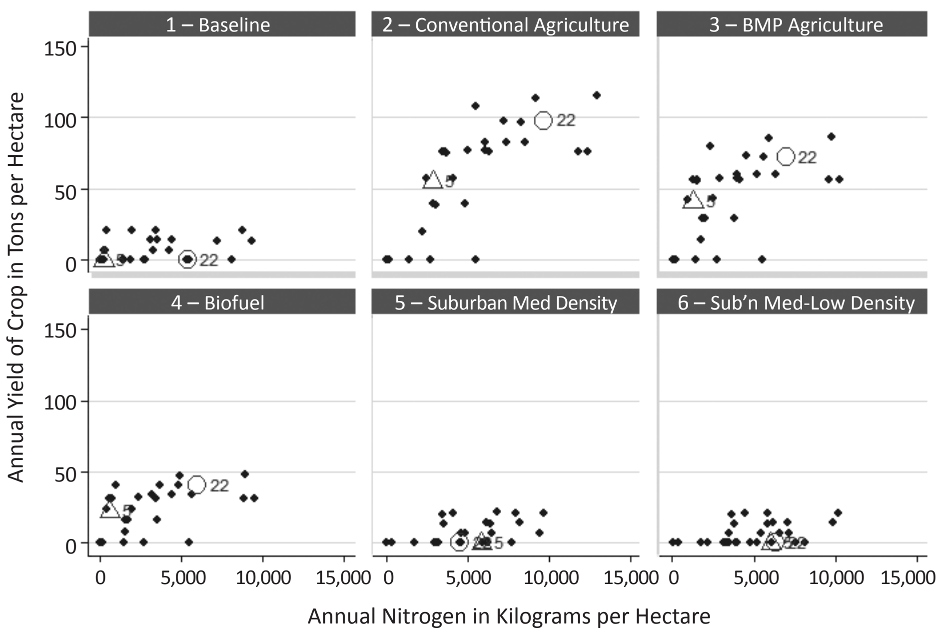

Figure 3. Tradeoff between Crop Yield and Annual Nitrogen Loading in Different Scenarios

Moreover, the extent of tradeoffs among the subset of ecosystem services considered in our analysis depends on which services are compared and on the stressors and land management practices involved. We find limited tradeoffs between crop yield and environmental flow or flood risk (Figure 2 and Appendix Figure 3), but there is a clearer tradeoff between crop yield and total nutrient loading (Figures 3 and 4), especially under the agricultural scenarios (Scenarios 2–4).

These tradeoffs are driven not only by differences in the area converted to agriculture or suburban uses (decided based on soil type suitability) but also by yields and subbasin characteristics that cause some subbasins to generate more nitrogen and phosphorus than others. As an illustrative example, we compared subbasins 5 and 22, both of which involve a conversion to crop land of about 21 percent under the agricultural scenarios (Figure 4). However, even with the same proportion of subbasin committed to crop land, subbasin 22 generates significantly more phosphorus loading than subbasin 5 while at the same time generating larger crop yields. The large difference in nutrient loading and crop yields does not come from the area of agricultural land since they have the same percentage of conversion and adopt the same management practices (fertilizer applied, timing of planting and harvesting, etc.). Other subbasin characteristics make subbasin 22 more prone to phosphorus loading (Figure 4, Scenario 2–4). For nitrogen, subbasins 5 and 22 are not good examples since nitrogen loadings for the two are noticeably different even in the baseline case. This may be because the baseline for subbasin 22 includes septic systems, which contribute to higher nitrogen loading. However, by carefully examining

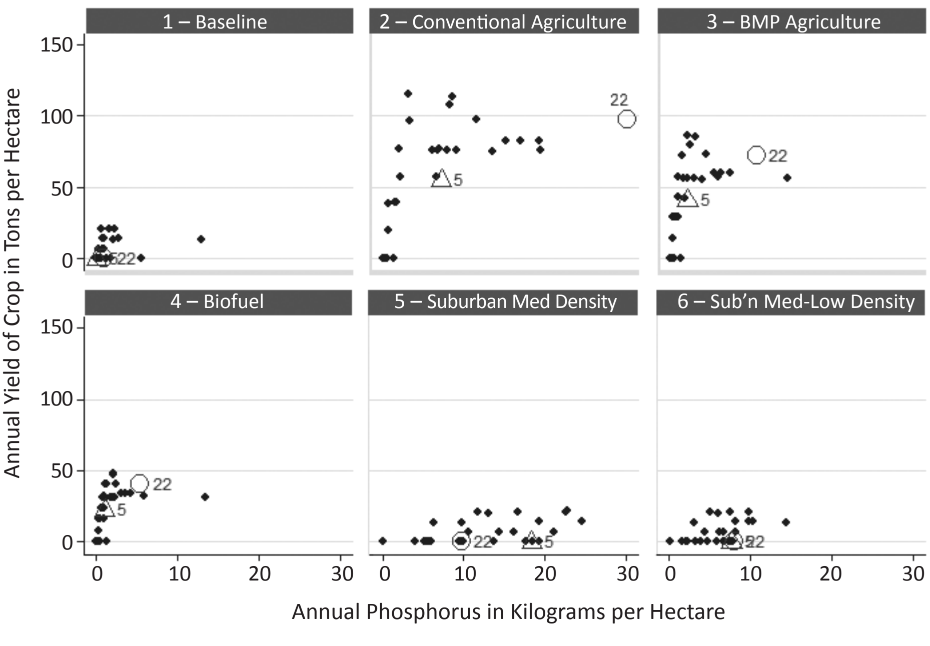

Figure 4. Tradeoff between Crop Yield and Annual Phosphorus Loading in Different Scenarios

the change in nitrogen loading under the traditional agricultural scenario, we found that subbasin 22 also is more prone to nitrogen loading despite the difference shown in Figure 3 (Scenario 1–2).

Likewise, in the suburban scenarios (Scenario 5), subbasins 3 and 28 respond very differently in terms of nitrogen and phosphorus loadings after converting nearly the same amount of land to medium density residential use (Appendix Figures 8 and 9). This difference in the simulated impact is due mostly to differences in inherent characteristics of the subbasins, such as distance from the river of septic systems and soil types, rather than simply to differences in the amount of land converted to suburban use.

These plots also confirm the general tradeoffs found in reviewing the scenarios with our raw indicators from Table 2. For instance, changing land use from forest to agriculture (Scenario 2 and 3) increases the crop yield significantly but decreases the environmental flow for most of the subbasins. Implementing BMPs decreases crop yields but increases the environmental flow compared to the conventional scenario.

This observed difference in influence of long-term drivers (land use change, land management) on ecosystem services in two relatively close subbasins such as 5 and 22 leads us to conclude that there is important heterogeneity among subbasins within the watershed. One can explore this further by modeling ecosystem service tradeoffs measured over the whole watershed under one scenario. Next, we investigate the heterogeneity of the subbasins' provision of ecosystem services under the conventional agriculture scenario as an important first step toward targeting the pieces of the watershed that are most important for supplying particular ecosystem services.

Tradeoffs in the Conventional Agriculture Scenario

Our mapping exercise further clarifies that there are tradeoffs geographically in deciding where to prioritize conservation investments (Figure 5). We illustrate this point using the conventional agriculture scenario (Scenario 2). To get the "biggest bang for the buck," one strategy for agencies is to target subbasins that currently have low environmental flow, high flood risk, and high nitrogen and phosphorus concentrations but are capable of generating high crop yields. For illustration purposes, Figure 5 gives four combinations of ecosystem services.[17] For example, agencies may prioritize subbasins with high crop yields and low environmental flow (panel a). However, subbasins with relatively low environmental flows do not have high flood risk (panel b). Hence, decision-makers would face a tradeoff between protecting environmental flow and mitigating flood risk. As another example, agencies may target subbasins that have high crop yields and high nitrogen concentrations. Although many of these subbasins also have high phosphorus concentration, some of the basins with high phosphorus concentrations (panel d) actually have low nitrogen concentrations (panel c). This implies that intrinsic site variables (such as soil attributes and slope) cause the difference in these two forms of nutrient loading. This finding is potentially useful for stakeholders when deciding where and how to target conservation efforts depends on the ecosystem services of interest.

[17] This case study demonstrated five ecosystem services and resulted in 26 unique combinations of ecosystem services.

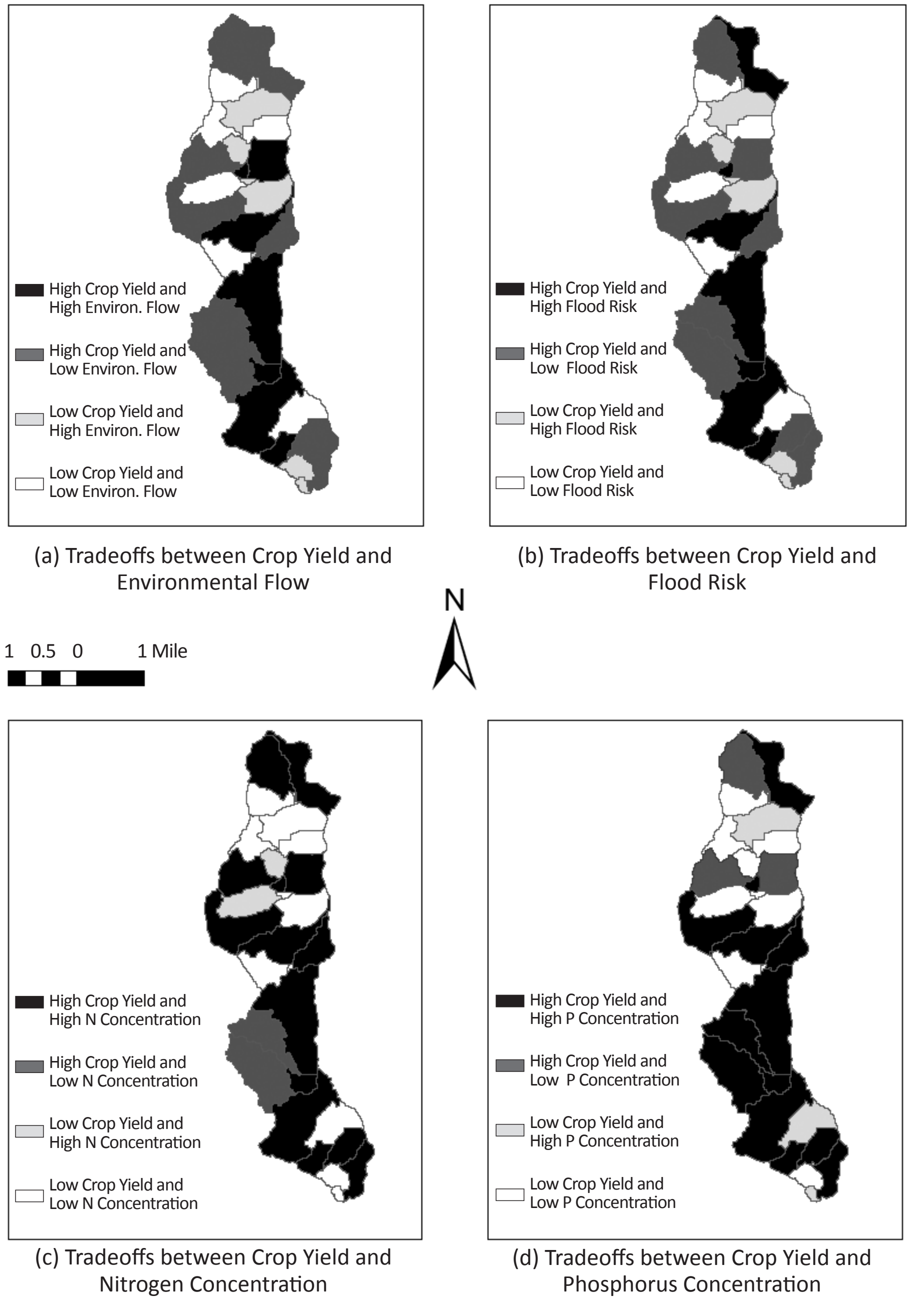

Figure 5. Tradeoffs in Ecosystem Services in the Beaver River Watershed

Discussion and Conclusions

We examined a watershed that sits on an increasingly valuable and vulnerable rural-urban fringe. With pressure for local food production, the value of the land for agriculture increasingly will be weighed against suburban residential development. Both of these possible land uses will result in changes in ecosystem services such as flood resilience and habitat base flows, which are the primary subject of this research. The scenarios were chosen to demonstrate the effects of land use, management practices, and climate change on multiple ecosystem services.

We illustrated one way to simulate the impact of stressors and BMPs on ecosystem services using an existing process-based hydrological model and data. The temporal and spatial details of the stressors, land management practices, and climate and the hydrological outputs are important in studies of hydrological ecosystem services because where and when things happen influence the effect of those changes on ecosystem services. However, we made several simplifying assumptions in the hydrological modeling. For example, there may be more irrigation with expansion of agricultural land and more wells drilled for drinking water with residential development. The hydrological modeling can be improved by incorporating those factors.

The climate change scenario highlights an additional potential stressor on hydrological ecosystem services. Due to uncertainty in modeling of precipitation in climate models, additional research is needed to account properly for possible changes in the variability of future precipitation events. However, we can start to explore the effect that land use choices will have when they occur in a plausible future climate scenario. When we combine the crop silage scenario (2) with climate change (Scenario 7), it is evident that there is no simple linear interpretation of the effects of land use and climate change taken together. For instance, although environmental flow is predicted to decrease under Scenario 2 (–40 percent) and under climate change (Scenario 7, –10 percent), the combined effect is not additive (–17 percent). Additional work must be done to more fully understand the implications of land use change on the resilience of a watershed to future climate conditions. Similarly, when we combine medium density residential development (Scenario 6) with climate change (Scenario 7), we see a doubling of the magnitude of a ten-year flood. Under Scenario 6 alone, there was only a 60 percent increase in the same flood measure.

Although we provide only a crude measure of values, our valuation method reveals some important relationships that put the tradeoffs between ecosystem services in perspective. Of the three agricultural scenarios, conventional practices generate the highest crop yields and thus the greatest benefit when taking into account the cost of damage from decreased environmental flow and increased nutrient loading. In the suburban scenarios, the cost of flood damage far exceeds the benefits gained from greater environmental flow even without taking into account the cost of damage from nutrient loading. By valuing multiple ecosystem services under different scenarios using a benefit-transfer method, policymakers can compare monetary tradeoffs for different choices and target the critical ecosystem services that most concern them. However, due to the large set of possible ecosystem service values, we can obtain only gross estimates for values from multiple ecosystem services.

Our analysis was conducted to illustrate a method by which to characterize the influence of changes in land use and management on ecosystem services using existing hydrological models. We acknowledge that our analysis only includes relevant ecosystem services and does not provide a complete accounting of all private and public benefits and costs associated with land uses in the watershed examined (others include timber production, biodiversity, carbon sequestration, and crop pollination). Any application of our method would need to include the ecosystem services deemed relevant to the land uses and policy context of interest.

Despite these caveats, our case study provides a starting point for stakeholders to begin to take into account both physical and monetary aspects of multiple ecosystem services in the decision-making process. The graphical and mapping approaches may assist them as they choose among many competing land use and land management options.

References

Allan, J.D., and M.M. Castillo. 2007. *Stream Ecology: Structure and Function of Running Waters* (2nd ed.). Dordrecht, the Netherlands: Springer.

Ancev, T., A.L. Stoecker, D.E. Storm, and M.J. White. 2006. "The Economics of Efficient Phosphorus Abatement in a Watershed." *Journal of Agricultural and Resource Economics* 31(3): 529–548.

Arabi, M., J.R. Frankenberger, B.A. Engel, and J.G. Arnold. 2008. "Representation of Agricultural Conservation Practices with SWAT." *Hydrological Processes* 22(16): 3042–3055.

Archetto, G., and Y.Q. Wang. 2012. "Forty Years of Urbanization and Changing Landscape in Rhode Island." Presented at the 27th Annual Landscape Ecology Symposium, "Informing Decisions in a Changing World," Newport, RI.

Armstrong, D.S., T.A. Richards, and G.W. Parker. 2004. *Evaluation of Streamflow Requirements for Habitat Protection by Comparison to Streamflow Characteristics at Index Streamflow-gaging Stations in Southern New England*. U.S. Department of the Interior / U.S. Geological Survey, Washington, D.C.

Arnold, J.G., and P.M. Allen. 1999. "Automated Methods for Estimating Baseflow and Ground Water Recharge from Streamflow Records." *Journal of the American Water Resources Association* 35(2): 411–424.

Arnold, J.G., and N. Fohrer. 2005. "SWAT2000: Current Capabilities and Research Opportunities in Applied Watershed Modeling." *Hydrological Processes* 19(3): 563–572.

Birch, M.B.L., B.M. Gramig, W.R. Moomaw, O.C. Doering III, and C.J. Reeling. 2011. "Why Metrics Matter: Evaluating Policy Choices for Reactive Nitrogen in the Chesapeake Bay Watershed." *Environmental Science and Technology* 45(1): 168–174.

Burdett, H. 2010. Farm Management and Extension Specialist, Kingston, Rhode Island. Personal communication regarding BMPs (December).

Food and Agriculture Organization. 2012. "Land Use" web page. Available at www.fao.org/nr/land/use/en (accessed March 2012).

Gassman, P.W., M.R. Reyes, C.H. Green, and J.G. Arnold. 2007. *The Soil and Water Assessment Tool: Historical Development, Applications, and Future Research Directions*. Center for Agricultural and Rural Development, Iowa State University. Available at http://publications.iowa.gov/archive/00005419/01/paper_12744.pdf (accessed February 2012).

Gupta, H.V., S. Sorooshian, and P.O. Yapo. 1999. "Status of Automatic Calibration for Hydrologic Models: Comparison with Multilevel Expert Calibration." *Journal of Hydrologic Engineering* 4(2): 135–143.

Hascic, I., and J.J. Wu. 2006. "Land Use and Watershed Health in the United States." *Land Economics* 82(2): 214–239.

Hayhoe, K., C. Wake, B. Anderson, X.Z Liang, E. Maurer, J. Zhu, J. Bradbury, A. DeGaetano, A.M. Stoner, and D. Wuebbles. 2008. "Regional Climate Change Projections for the Northeast USA." *Mitigation and Adaptation Strategies for Global Change* 13(5): 425–436.

Hayhoe, K., C.P. Wake, T.G. Huntington, L. Luo, M.D. Schwartz, J. Sheffield, E. Wood, B. Anderson, J. Bradbury, A. DeGaetano, T.J. Troy, and D. Wolfe. 2006. "Past and Future Changes in Climate and Hydrological Indicators in the U.S. Northeast." *Climate Dynamics* 28(4): 381–407.

Ibendahl, G. 2012. *Economics for Corn Production*. Agricultural Communications Service, University of Kentucky, Lexington. Available at www.ca.uky.edu/agc/pubs/id/id139/economics.pdf (accessed May 2012).

Intergovernmental Panel on Climate Change. 2012. *Managing the Risks of Extreme Events and Disasters to Advance Climate Change Adaptation. A Special Report of Working Groups I and II of the Intergovernmental Panel on Climate Change*. Cambridge UK: Cambridge University Press.

James, C.A., J. Kershner, J. Samhouri, S. O'Neill, and P.S. Levin. 2012. "A Methodology for Evaluating and Ranking Water Quantity Indicators in Support of Ecosystem-based Management." *Environmental Management* 49(3): 1–17.

Karanja, F., N. Reid, and O. Cacho. 2008. "Economic Valuation of Ecosystem Services from Environmental Flow Provision in the Gwydir Catchment, North-western NSW, Australia." In *IAIA08 Conference Proceedings*. Available at www.iaia.org/iaia08perth/pdfs/concurrentsessions/CS7-11_biodiversity_Karanja.pdf (accessed May 2012).

King, D.M., and M. Mazzotta. 2000. "*Ecosystem Valuation*" web page. Available at www.ecosystemvaluation.org (accessed February 2012).

Kling, C.L. 2011. "Economic Incentives to Improve Water Quality in Agricultural Landscapes: Some New Variations on Old Ideas." *American Journal of Agricultural Economics* 93(2): 297–309.

Korsgaard, L., and J.S. Schou. 2010. "Economic Valuation of Aquatic Ecosystem Services in Developing Countries." *Water Policy* 12(1): 20.

Kumar, P., M. Verma, M.D. Wood, and D. Negandhi. 2010. *Guidance Manual for the Valuation of Regulating Services*. Nairobi, Kenya: United Nations Environment Program. Available at http://hqweb.unep.org/pdf/Guidance_Manual_for_the_Regulating_Services.pdf (accessed May 2012).

Lautenbach, S., M. Volk, B. Gruber, C.F. Dormann, M. Strauch, and R. Seppelt. 2010. "Quantifying Ecosystem Service Trade-offs." In *Proceedings of the International Environmental Modeling and Software Society 2010 International Congress on Environmental Modeling and Software Modeling for Environment's Sake.* Ottawa, Canada.

Leopold, L.B., M.G. Wolman, and J.P. Miller. 1995. *Fluvial Processes in Geomorphology*. Mineola, NY: Dover Publications.

Lubowski, R.N. 2002. "Determinants of Land-use Transitions in the United States: Econometric Analysis of Changes among the Major Land-use Categories." Ph.D. thesis, Harvard University, Cambridge, MA.

Lubowski, R.N., A.J. Plantinga, and R.N. Stavins. 2008. "What Drives Land-use Change in the United States? A National Analysis of Landowner Decisions." *Land Economics* 84(4): 529–550.

Maurer, E.P., H.G. Hidalgo, T. Das, M.D. Dettinger, and D.R. Cayan. 2010. "The Utility of Daily Large-scale Climate Data in the Assessment of Climate Change Impacts on Daily Streamflow in California." *Hydrology and Earth System Sciences* 14(6): 1125–1138.

Meals, D.W., S.A. Dressing, and T.E. Davenport. 2010. "Lag Time in Water Quality Response to Best Management Practices: A Review." *Journal of Environmental Quality* 39(1): 85–96.

Mearns, L.O., R. Arritt, G. Boer, D. Caya, P. Duffy, F. Giorgi, W.J. Gutowski, I.M. Held, R. Jones, R. Laprise, L.R. Leung, J. Pal, J. Roads, L. Sloan, R. Stouffer, E.S. Takle, and W. Washington. 2005. "North American Regional Climate Change Assessment Program." In *Proceedings of the 85th American Meteorological Society Annual Meeting.* San Diego, California.

Meyer, J.L., and J.B. Wallace. 2001. "Lost Linkages and Lotic Ecology: Rediscovering Small Streams." In *Ecology: Achievement and Challenge: The 41st Symposium of the British Ecological Society*. Orlando, Florida. Available at www.cabdirect.org/abstracts/20023178157.html (accessed March 2012).

Moriasi, D.N., J.G. Arnold, M.W. Van Liew, R.L. Bingner, R.D. Harmel, and T.L. Veith. 2007. "Model Evaluation Guidelines for Systematic Quantification of Accuracy in Watershed Simulations." *Transactions of the ASABE* 50(3): 885–900.

Nadeau, T.L., and M.C. Rains. 2007. "Hydrological Connectivity between Headwater Streams and Downstream Waters: How Science Can Inform Policy." *Journal of the American Water Resources Association* 43(1): 118–133.

National Flood Insurance Program. 2012. "The Cost of Flooding" web page. Available at www.floodsmart.gov/floodsmart/pages/flooding_flood_risks/the_cost_of_flooding.jsp (accessed May 2012).

National Oceanic and Atmospheric Administration, National Marine Fisheries Service. 2009. "Species of Concern" web page. Available at www.nmfs.noaa.gov/pr/pdfs/species/riverherring_detailed.pdf.

National Research Council. 2005. *Valuing Ecosystem Services: Toward Better Environmental Decision-making*. Washington, DC: The National Academies Press.

Natural Resources Council of Maine.1995. "Public Land Ownership by State" web page. Available at www.nrcm.org/documents/publiclandownership.pdf (accessed February 2012).

Natural Resources Conservation Service. 2009. "Web Soil Survey" web page. Available at http://websoilsurvey.nrcs.usda.gov/app/HomePage.htm (verified October 29, 2009).

Nelson, E., G. Mendoza, J. Regetz, S. Polasky, H. Tallis, D.R. Cameron, K.M.A. Chan, G.C. Daily, J. Goldstein, P.M. Kareiva, E. Lonsdorf, R. Naidoo, T.H Ricketts, and R. Shaw. "Modeling Multiple Ecosystem Services, Biodiversity Conservation, Commodity Production, and Tradeoffs at Landscape Scales." *Frontiers in Ecology and the Environment* 7(1): 4–11.

Neppel, J., T. Sunday, R. Cruse, M. Braster, and T. Jacobsen. 2002. "Modeling Switchgrass Production Effects on Runoff Water Quality." Unpublished report, Iowa State University. Available at http://iowaswitchgrass.com/__docs/pdf/Modeling%20Switchgrass.pdf (accessed February 2012).

Pachauri, R.K. 2007. *Climate Change 2007: Synthesis Report. Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Intergovernmental Panel on Climate Change, Geneva, Switzerland.

Park, S.W., S. Mostaghimi, R.A. Cooke, and P.W. McClellan. 1994. "BMP Impacts on Watershed Runoff, Sediment, and Nutrient Yields." *Journal of the American Water Resources Association* 30(6): 1011–1023.

Rabotyagov, S., T. Campbell, M. Jha, P.W. Gassman, J. Arnold, L. Kurkalova, S. Secchi, H. Feng, and C.L. Kling. 2010. "Least-cost Control of Agricultural Nutrient Contributions to the Gulf of Mexico Hypoxic Zone." *Ecological Applications* 20(6): 1542–1555.

Rhode Island Department of Environmental Management. 2012. "Rhode Island Freshwater Lakes and Ponds: Aquatic Invasive Plants and Water Quality Concerns" web page. Available at www.dem.ri.gov/programs/benviron/water/quality/surfwq/pdfs/lakes012.pdf (accessed February 2012).

Rhode Island Division of Planning. 2006. "Land Use 2025" web page. Available at www.planning.ri.gov/landuse/121/landuse2025.pdf (accessed February 2012).

Rhode Island Geographic Information System. 2012. Available at www.edc.uri.edu/rigis (accessed on February 13, 2012).

Richardson. 2005. "Modified Aquatic Base Flow (RI-ABF) for Rhode Island." Rhode Island Department of Environmental Management Office of Water Resources. Available at www.dem.ri.gov/programs/benviron/water/withdraw/pdf/riabf.pdf (accessed March 2011).

Richardson, C.W., D.A. Bucks, and E.J. Sadler. 2008. "The Conservation Effects Assessment Project Benchmark Watersheds: Synthesis of Preliminary Findings." *Journal of Soil and Water Conservation* 63(6): 590–604.

Sedell, J.R., G.H. Reeves, F.R. Hauer, J.A. Stanford, and C.P. Hawkins. 1990. "Role of Refugia in Recovery from Disturbances: Modern Fragmented and Disconnected River Systems." *Environmental Management* 14(5): 711–724.

Shreve, R.L. 1969. "Stream Lengths and Basin Areas in Topologically Random Channel Networks." *The Journal of Geology* 77(4): 397–414.

Snyder, Cindy. 2011. "Calculating Corn Silage Value." *AG Weekly*, March 8.

Srinivasan, R. 2009. "Introduction to the Soil and Water Assessment Tool." Texas A&M University, College Station, TX.

Stone, M.C., R.H. Hotchkiss, and L.O. Mearns. 2003. "Water Yield Responses to High and Low Spatial Resolution Climate Change Scenarios in the Missouri River Basin." *Geophysical Research Letters* 30(4): 1186.

Swallow, B.M., J.K. Sang, M. Nyabenge, D.K. Bundotich, A.K. Duraiappah, and T.B. Yatich. 2009. "Tradeoffs, Synergies, and Traps among Ecosystem Services in the Lake Victoria Basin of East Africa." *Environmental Science and Policy* 12(4): 504–519.

The Nature Conservancy. 2012a. "Nature Conservancy—Protecting Nature, Preserving Life" web page. Available at www.nature.org (accessed February 2012).

The Nature Conservancy. 2012b. "Rhode Island Nature Conservation, Environment Issues—The Nature Conservancy" web page. Available at www.nature.org/ourinitiatives/regions/northamerica/unitedstates/rhodeisland/index.htm (accessed February 2012).

The World Bank. 2012. "Real Interest Rate" Web page. Available at http://data.worldbank.org/indicator/FR.INR.RINR (accessed May 2012).

Tomer, M.D., and M.A. Locke. 2011. "The Challenge of Documenting Water Quality Benefits of Conservation Practices: A Review of USDA-ARS's Conservation Effects Assessment Project Watershed Studies." *Water Science and Technology* 64(1): 300–301.

U.S. Department of Agriculture National Agricultural Statistics Service. 2012. "Prices Received for Corn by Month" web page. Available at www.nass.usda.gov/Charts_and_Maps/Agricultural_Prices/pricecn.asp (accessed May 2012).

U.S. Geological Survey. 2012. "Welcome to the USGS—U.S. Geological Survey" web page. Available at www.usgs.gov (accessed December 2011).
U.S. Historical Climatology Network. 2012. "Kingston Weather Data." Carbon Dioxide Information Analysis Center, Oakridge, TN. Available at http://cdiac.ornl.gov/epubs/ndp/ushcn/ushcn_map_interface.html. (accessed December 2010).
U.S. Fish and Wildlife Service. 2012. "Questions and Answers on the New England Flow Policy." USFWS, Washington, DC. Available at www.fws.gov/newengland/pdfs/Flowpolicy.pdf (accessed on February 13, 2012).
Van Grinsven, H.J.M., A. Rabl, and T.M. de Kok. 2010. "Estimation of Incidence and Social Cost of Colon Cancer due to Nitrate in Drinking Water in the EU: A Tentative Cost-Benefit Assessment." *Environmental Health* 9(1): 58.
Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell, and C.E. Cushing. 1980. "The River Continuum Concept." *Canadian Journal of Fisheries and Aquatic Sciences* 37(1): 130–137.
Watershed Counts. 2012. "2012 Report" web page. Available at www.watershedcounts.org/documents/WatershedCountsReport2012.pdf (accessed December 2011).
Williams, J.R., C.A. Jones, and P.T. Dyke. 1984. "The EPIC Model and Its Application." In *Proceedings of the International Symposium on Minimum Data Sets for Agrotechnology Transfer*. Available at http://oar.icrisat.org/812/1/RA_00071.pdf#page=121 (accessed December 2012).
Wu, J., and K. Tanaka. 2005. "Reducing Nitrogen Runoff from the Upper Mississippi River Basin to Control Hypoxia in the Gulf of Mexico: Easements or Taxes?" *Marine Resource Economics* 20(2): 121.
Zhou, Y., Y. Wang, A.J. Gold, and P.V. August. 2010. "Modeling Watershed Rainfall-Runoff Relations Using Impervious Surface-area Data with High Spatial Resolution." *Hydrogeology Journal* 18(6): 1413–1423.