ESSAYS ON THE EFFECTIVENESS OF ENVIRONMENTAL CONSERVATION AND WATER MANAGEMENT POLICIES

A Dissertation

by

MARIANO MEZZATESTA

Submitted to the Office of Graduate Studies of
Texas A&M University
in partial fulfillment of the requirements for the degree of

DOCTOR OF PHILOSOPHY

August 2012

Major Subject: Agricultural Economics

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Approved by:

Co-Chairs of Committee,	David A. Newburn
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ABSTRACT

Essays on the Effectiveness of Environmental Conservation and Water Management Policies.

(August 2012)

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Co-Chairs of Advisory Committee: Dr. David A. Newburn
Dr. Richard T. Woodward

An awareness of the effect of agricultural production on the environment has led to the development of policies to mitigate its adverse effects. This dissertation provides analyses of agri-environmental policies designed to protect environmental assets, as well as analytical decision-making tools useful for conducting policy evaluations.

The first essay employs propensity score matching techniques to estimate the additionality of federal agricultural conservation programs for six conservation practices for farmers in Ohio. Additionality is an important measure of the effectiveness of conservation programs in inducing an increase in the conservation effort of farmers. Results suggest that additionality is positive and statistically significant for all six conservation practices. However, while programs achieve positive additionality for all practice types, a comparison between conservation practices reveals that certain practice types achieve higher percent additionality than others. Such results, coupled with information on the environmental benefits obtained per practice, could prove useful to program managers for improving the effectiveness of conservation programs.

The second essay develops a new methodology to decompose the additionality measure into the two effects induced by conservation programs: expansion versus the new adoption of conservation practices. To do so, the relative contributions of two types of farmers, prior-adopters and new-adopters, are estimated. Results of the decomposition reveal that the additionality for prior-adopters is not significant for all practice types. Instead, additional conservation effort comes from new-adopters adopting new practices. Second, decomposition estimates suggest that practice types with a greater fraction of enrolled farmers that are new-adopters achieve greater percent additionality than those with greater proportions of prior-adopters. This suggests that a farmers' history in conservation adoption has a significant influence on additionality levels.

The final essay analyzes the effect of recent instream flow diversion-guidelines on agricultural water security and streamflows within a decentralized water management regime. Spatially-explicit economic and hydrologic models are integrated to evaluate the tradeoffs between salmon bypass-flows and agricultural water security for three different diversion-guidelines within a northern-California watershed. Results indicate that the most restrictive diversion-guideline provides the greatest protection of bypass-flow days within smaller watersheds; however, within larger watersheds protection is not as significant. Water security, however, decreases sharply under the strict and moderate diversion-guidelines, especially during dry years. Overall, results indicate that greater focus should be given to protecting streamflows in the smallest watersheds, and meeting human water needs during dry years, when agricultural water security is impacted the most.

DEDICATION

A mis Padres.

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NOMENCLATURE

ac-ft	Acre-Foot
ATT	Average Treatment Effect on the Treated
CI	Confidence Interval
CREP	Conservation Reserve Enhancement Program
CRP	Conservation Reserve Program
CSP	Conservation Security Program
EQIP	Environmental Quality Incentives Program
EOA	End of Anadromy
NASS	National Agricultural Statistical Service
NHD	National Hydrography Dataset
POD	Point-of-Diversion
Q_{mbf}	Minimum-Bypass-Flow Threshold
Q_{fmf}	February-Median-Flow Threshold
SWRCB	State Water Resources Control Board
WQTP	Water Quality Trading Program

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CHAPTER I

INTRODUCTION

The impact of the agricultural sector on the environment is well known. Agricultural production, for instance, reduces water quality through nutrient and sediment runoff, impairs instream flows through surface water diversions for irrigation, diminishes natural habitat through deforestation, and potentially leads to an increase in GHG emissions. A growing awareness of the effect of agricultural production on the environment has led to the development of programs and policies to mitigate its adverse effects. For example, the use of markets and subsidy programs to incentivize voluntary, private investment by farmers in environmental stewardship has been gaining popularity for many years (USDA ERS 2009). The importance of instream flows in sustaining aquatic ecosystems has also led to the development of water management policies that restrict surface flow diversions, and thus, reduce the impacts of agricultural production on water resources (Richter et al. 2003). The challenge of reconciling competing needs between agriculture and ecosystems for natural resources requires effective and innovative agri-environmental policies. As the adoption of agri-environmental policies to address environmental concerns increases, analysis of existing policies is crucial to understanding the effectiveness and impacts of such policies, as well as for determining areas for improvement.

The overarching contribution of this dissertation is to provide analyses and

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This dissertation follows the style of *Land Economics*.

insights into agri-environmental programs and policies designed to enhance or protect environmental assets, as well as to introduce analytical decision-making tools that can be useful to policy-makers in conducting policy analyses. The three essays that comprise this dissertation are as follows:

1. Additionality and the Adoption of Farm Conservation Practices;
2. The Decomposition of Additionality; and,
3. The Effect of Instream Flow Policies on Agricultural Water Security and Streamflows.

Essay I, or Chapter II, employs propensity score matching techniques to estimate the additionality (i.e., the average treatment effect on the treated (ATT)) of federal agricultural conservation programs for six conservation practices. Federal agricultural conservation programs, such as the Conservation Reserve Program (CRP) and the Environmental Quality Improvement Program (EQIP), have invested billions of dollars to incentivize farmers to enhance environmental benefits. The effectiveness of such federal cost-share programs depends in part on whether payments induce an increase in conservation effort by farmers (i.e., additionality). Estimates of additionality allow for a more thorough understanding of how incentives in conservation programs alter farmer behavior, and assist in designing programs that cost-effectively enhance environmental benefits. Data on six conservation practices as well as farmer adoption and enrollment decisions within several conservation programs, were obtained from a survey conducted in Ohio.

Essay II, or Chapter III, develops a new methodology to analyze two types of effects that conservation programs can have on farmers. Conservation programs can lead to either the expansion of existing conservation practices, or the new adoption of conservation practices. To achieve this, the additionality measure (ATT) is decomposed into the relative contributions of two types of farmers: prior-adopters and new-adopters. The term “new-adopters” refers to those farmers who adopt a new practice, i.e., those who would have not adopted a conservation practice without the assistance of a program subsidy. On the other hand, “prior-adopters” refers to those farmers who would have adopted the practice even in the absence of a subsidy, and thus, potentially expand the practice as a result of program support. The disaggregation of the ATT provides a more thorough understanding of additionality, and reveals greater insights into cost-share programs. The new methodology is used to decompose the overall additionality estimates obtained in Essay I, which are an aggregate measure of additionality across the two types of farmers.

Essay III or Chapter IV, analyzes the effect of recent instream flow policies, which aim to maintain minimum-bypass-flows for adult salmonid migration, on agricultural water security and streamflows within a decentralized water management regime. Water use conflicts have become a dominant environmental issue, particularly in arid climates, such as the Western U.S., where the listing of endangered species has placed greater pressures on regulatory agencies to protect instream flows. In response, regulatory agencies have adopted instream flow polices requiring that certain levels of streamflow be maintained in an effort to protect flows. While restrictions on surface flow diversions to maintain instream flows can assist in protecting aquatic ecosystems, it is important to understand the inherent tradeoffs between environmental protections and agricultural water security. Within the Western U.S., the challenge of reconciling competing water needs is well exemplified, where flow regime alterations from water management have been a primary driver of ecosystem degradation (Dole and Niemi 2004). Agricultural producers within this area often rely on a decentralized management system, based on groundwater wells and/or privately-owned storage ponds filled with water from run-off and surface water diversions, rather than on more traditional releases from large dams (Merenlender et al. 2008; Grantham et al. 2010; Newburn et al. 2011). Spatially-explicit economic and hydrologic models are integrated to quantify the tradeoffs between losses in ecologically-relevant flow metrics and impacts on agricultural water security under different instream flow policies within a northern-California watershed, accounting for spatial and temporal variation in water availability.

CHAPTER II

ADDITIONALITY AND THE ADOPTION OF FARM CONSERVATION PRACTICES

2.1 Introduction

Federal agricultural conservation programs, such as the Conservation Reserve Program (CRP) and the Environmental Quality Improvement Program (EQIP), have invested billions of dollars to incentivize farmers to enhance environmental benefits. Funding for major USDA conservation programs was approximately 24 billion dollars during the period 2002-2007, and, starting in 2002, the portion allocated to working-lands programs have increased considerably relative to land retirement programs (USDA ERS 2009). The effectiveness of federal cost-share programs depends in part on whether payments induce an increase in farmer conservation effort. In this chapter, propensity score matching methods are used to estimate the level of additionality from enrollment in cost-share programs for six conservation practices.

Propensity score matching estimators were developed by Rosenbaum and Rubin (1983) and are often used for program evaluation to estimate the average treatment effect on the treated (ATT), i.e. the average impact on those who are directly affected by the policy. Matching estimators pair treated and untreated individuals that are similar in terms of observable characteristics in order to correct for sample selection bias induced by nonrandom program enrollment. These methods have been used for program evaluation in several contexts pertaining to conservation. For example, Andam et al.

(2008) analyzed the effect of protected areas in reducing deforestation rates in Costa Rica and found that the rate of deforestation in protected areas was 11% lower than in similar unprotected areas. Matching methods have been used to analyze the effect of land-use policies aimed at reducing farmland loss (Liu and Lynch 2011) and reducing future urban development (Bento et al. 2007; Butsic et al. 2011). Ferraro et al. (2007) used matching methods to analyze the impact of the US Endangered Species Act on species recovery rates and found significant improvements in recovery rates but only when the listing was combined with substantial government funding for habitat protection.

While the studies mentioned above focused primarily on programs or polices that protect against future land-use conversion, federal cost-share programs incentivize the adoption of conservation practices for land restoration. Using regression analysis to analyze the effect of CRP on land retirement, Lubowski et al. (2008) estimate a discrete choice land-use change model with Natural Resource Inventory data where CRP is included as an alternative. They found that approximately 90% of land enrolled under CRP constitutes additional land retirement, implying that CRP significantly increased the likelihood of land retirement. Lichtenberg and Smith-Ramirez (2011) estimated the impact on land allocation of a cost-share program in Maryland using a switching regression model. They found that cost-share funding induced farmers to adopt conservation practices they would not have used without funding; however, it also had the unintended consequence of inducing slippage (i.e., pasture and vegetative cover converted to cropland).

In this chapter, I estimate the level of additionality from enrollment in cost-share programs for six conservation practices. I apply matching estimators to quantify additionality, estimated as the ATT, which equals the average increase in conservation effort of enrolled farmers with funding relative to their counterfactual effort without funding.[1] I analyze conservation adoption and enrollment decisions using data from a farmer survey in Ohio. The survey includes farmer enrollment in major federal conservation programs, such as CRP, EQIP, and others. I estimate the ATT for six conservation practice types: conservation tillage, cover crops, hayfield establishment, grid sampling, grass waterways, and filter strips. Conservation tillage leaves crop residue on fields to reduce soil erosion and runoff. Cover crops provide soil cover and absorb nutrients on cropland when the soil would otherwise be bare. Hayfield establishment retires cropland to a less intensive state to provide habitat and other conservation benefits. Grid sampling improves the efficiency of nutrient application rates to maximize crop yields, while reducing excess fertilizer that potentially would runoff or leach into surrounding water bodies. Grass waterways are located in the natural drainage areas within cropland to reduce soil erosion and gully formation. Filter strips are typically planted grass along stream banks to capture sediment, nutrients, and pesticides from runoff before they enter surrounding water bodies.

The empirical analysis provides two main results. First, the overall ATT for enrollment in cost-share programs is positive and statistically significant for each of the

[1] The term counterfactual refers to what would or might have happened under different conditions. In this study, the counterfactual is what the conservation effort of an enrolled farmer would have been had they not enrolled.

six practice types. That is, cost-share programs induce farmers to increase the average proportion of conservation acreage adopted for all practices. Second, percent additionality is found to vary dramatically between practice types. Percent additionality is defined as the percent increase in the proportion of conservation acreage relative to the total proportion of conservation acreage adopted by enrolled farmers. The percent additionality is highest for hayfield establishment (92.9%), filter strips (89.1%), and cover crops (88.9%), while it is lowest for conservation tillage (20.5%).

The chapter is structured as follows. First, I discuss the propensity score matching method and assumptions. Next, I describe and summarize the data from the farmer survey in Ohio. Thereafter, I provide the estimation results for the ATT, %ATT, and robustness checks. I conclude with policy implications for conservation programs.

2.2 Propensity Score Matching Estimator

In this section, I first formalize the ATT and discuss the identification assumptions. Then, I develop the propensity score matching estimator. The development is mostly standard in the literature, though I follow most closely the presentation of Smith and Todd (2005) and Caliendo and Kopeinig (2008). An indicator variable D is equal to one if a farmer enrolled in a cost-share program to fund the adoption of a conservation practice, and D equals zero if a farmer did not enroll. Further, two potential outcome variables Y_1 and Y_0 are defined for each farmer and practice type. Let Y_1 be the proportion of farm acreage in the conservation practice if a farmer enrolled in a program (D=1), and let Y_0 be the proportion of farm acreage in the conservation practice if the farmer did not enroll (D=0), where $0 \le Y_0 \le 1$ and $0 \le Y_1 \le 1$. Only one of these two outcome variables is observable for any given farmer.

The treatment effect of enrollment in a cost-share program is defined as the increase in the proportion of conservation acreage adopted with program enrollment relative to the proportion without being enrolled, $\tau = Y_1 - Y_0$. Additionality is defined as the average treatment effect on the treated (enrolled) group of farmers

$$ATT = E\left[Y_1 - Y_0 \mid D=1\right] = E\left[Y_1 \mid D=1\right] - E\left[Y_0 \mid D=1\right]. \qquad [2.1]$$

The application of matching estimators to estimate the ATT requires that two identification assumptions be satisfied. The first, often called the unconfoundedness assumption, states that the potential outcome Y_0 must be independent of program enrollment conditional on the set of observable covariates *X*, i.e., $Y_0 \perp D \mid X$ (Heckman et al. 1997). The vector of covariates *X* should affect both the farmer decision on enrollment and the potential outcomes. Rosenbaum and Rubin (1983) demonstrated that if the unconfoundedness assumption is satisfied, then it is also true that Y_0 is independent of program enrollment conditional on the propensity score, i.e., $Y_0 \perp D \mid P$, where the propensity score is defined as the probability of enrollment conditional on *X*, $P = P\left(D=1 \mid X\right)$. The propensity scores are often estimated using discrete choice models, typically a probit or logit model.

The second identification assumption states that for all farmer characteristics *X*, there is a positive probability for both enrolled or non-enrolled farmers, i.e.,

$0 < P(D=1 \mid X) < 1$. This overlap assumption, also known as the common support condition, implies that for each enrolled farmer there exists a positive probability of a match within the group of non-enrolled farmers with a similar set of covariates *X*.

Let H_1 denote the set of *I* enrollees and H_0 the set of *J* non-enrollees that are on common support. Each enrollee and non-enrollee has a vector of characteristics, X_i and X_j, and propensity scores, P_i and P_j, respectively, where *i=1,...,I* and *j=1,...,J*. Propensity scores on the probability of enrollment are estimated from a probit model, such that $P_i = P(D_i = 1 \mid X_i)$ for $i \in H_1$ and $P_j = P(D_j = 1 \mid X_j)$ for $j \in H_0$.

The primary goal of the matching process is to obtain for all $i \in H_1$, a counterfactual estimate, $\hat{Y}_0^i$, of what the enrolled farmer would have done without cost-share funding. The counterfactual estimate, $\hat{Y}_0^i$, is a weighted average

$$\hat{Y}_0^i = \hat{E}\left[Y_0^i \mid P_i, D_i = 1\right] = \sum_{j \in H_0} W(i,j) Y_0^j , \qquad [2.2]$$

where Y_0^j is the observed outcome for the non-enrollee $j \in H_0$.[2] A variety of matching algorithms are available to construct the weights $W(i,j)$ in [2.2] (Guo and Fraser 2010). For example, the propensity score kernel matching uses the non-enrollees in H_0 as matches, and the weights $W(i,j)$ are determined based on a kernel function, a

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[2] The expression $\hat{E}\left[Y_0^i \mid P_i, D_i = 1\right]$ denotes the empirical estimate of $E\left[Y_0^i \mid P_i, D_i = 1\right]$. Refer to Smith and Todd (2005) for further clarification on this expression.

bandwidth parameter, and the differences between P_i and P_j. The weights are normalized so that $\sum_{j \in H_0} W(i,j) = 1$ for each enrolled farmer *i*. The matching estimators for $E[Y_1 \mid D=1]$ and $E[Y_0 \mid D=1]$ in equation [2.1] are

$$\hat{E}[Y_1 \mid D=1] = \frac{1}{I} \sum_{i \in H_1} Y_1^i \qquad [2.3]$$

and

$$\hat{E}[Y_0 \mid D=1] = \frac{1}{I} \sum_{i \in H_1} \hat{Y}_0^i = \frac{1}{I} \sum_{i \in H_1} \sum_{j \in H_0} W(i,j) Y_0^j . \qquad [2.4]$$

Hence, the matching estimator for the ATT in equation [2.1] is

$$A\hat{T}T = \frac{1}{I} \sum_{i \in H_1} \left[Y_1^i - \hat{Y}_0^i \right] = \frac{1}{I} \sum_{i \in H_1} \left[Y_1^i - \sum_{j \in H_0} W(i,j) Y_0^j \right]. \qquad [2.5]$$

2.3 Data in the Farmer Survey

For this study, data is used from a farmer survey conducted in southwestern Ohio within 25 counties in and around the Great Miami River Watershed. The study area is dominated by agricultural uses (83% of land area) particularly for row-crop production in corn, soybeans, and wheat. Typical livestock operations include swine, beef cattle, and dairy. The survey was conducted in 2009 by the Ohio Division of the National Agricultural Statistical Service (NASS) (the questionnaire used to conduct the survey is provided in Appendix C). The sample of farmers was drawn from the NASS master list of farmers, where a random stratified sampling was used to ensure a sufficient number of responses from large commercial farms. The questionnaire was mailed to 2000 farmers with follow-up phone calls. There were a total of 773 survey respondents. However, useable responses varied by practice type depending on whether the farmer completed the questionnaire for each practice type. The questionnaire contained questions on farmer socioeconomic characteristics, farm management and operation, and land quality characteristics.

The questionnaire also included information on the acreage adopted for the following six conservation practices in 2009: conservation tillage, cover crops, hayfield (or grassland) establishment, grid sampling, grass waterways, and filter strips. These six practices are categorized into two groups. First, practices for environmentally sensitive areas (filter strips and grass waterways) are almost exclusively used along stream banks or in natural drainage areas, respectively. Second, field practices (conservation tillage, cover crops, hayfield establishment, and grid sampling) are often adopted as a practice for a significant portion of the cropland.

For each of the six conservation practices, the questionnaire asks whether the farmer received funding from enrollment in any cost-share programs. Federal programs included explicitly in the survey are EQIP, CRP, Conservation Reserve Enhancement Program (CREP), and Conservation Security Program (CSP).[3] The Great Miami River Watershed has a regional water quality trading program (WQTP) (Newburn and Woodward, forthcoming). The WQTP was included in the survey because it similarly

[3] The CSP later changed its name to the Conservation Stewardship Program.

provides cost-share funding for conservation practices. An “other” option was also included in the survey to capture any other federal or state conservation programs not already listed above, such as wetland and grasslands programs.

Table 2-1 reports farmer decisions on conservation practice adoption and program enrollment for the six practice types. Farmer decisions are categorized into three groups: no adoption, adoption without enrollment, and adoption with enrollment. For example, conservation tillage has 97 farmers who did not adopt this practice, 379 farmers who adopted without enrollment (i.e., self-funded), and 87 farmers who enrolled in a cost-share program for this practice. Table 2-1 also provides the average proportion for the conservation acreage adopted relative to the total farm acreage for enrolled farmers and non-enrolled farmers who adopted a practice. Non-enrolled farmers who did not adopt a practice have an average proportion of zero for this practice. The average proportion of conservation acreage adopted by enrolled farmers is greater than that by non-enrolled farmers who adopted a practice for all practices except for grass waterways, where the average proportions are equal (Table 2-1). Notice that the average proportions for the four field practices are much larger than the two environmentally sensitive practices. The reason is that filter strips and grass waterways are solely implemented along stream banks and in natural drainage areas rather than across the entire field, and thus, represent a smaller proportion of total farm acreage.

TABLE 2-1. Farmer Adoption, Enrollment, and Average Proportion of Conservation Acreage Adopted on Total Farm Acreage by Practice Type.

Practice Type	No Adoption	Adoption without Enrollment	Adoption with Enrollment	Total[a]	Average Proportion for Non-Enrolled who Adopted	Average Proportion for Enrolled
Conservation Tillage	97	379	87	563	0.695	0.779
Cover Crops	513	68	24	605	0.197	0.262
Hayfield Establishment	522	53	19	594	0.153	0.287
Grid Sampling	323	161	55	539	0.636	0.749
Grass Waterways	243	137	146	526	0.018	0.018
Filter Strips	395	56	93	544	0.008	0.011

[a] There were a total of 773 survey respondents; however, the number of useable observations varies by practice type due to missing or incomplete survey information.

For the empirical analysis, the treatment group for a given practice type is comprised of farmers who enrolled in any cost-share program for this practice, while the control group is comprised of farmers who did not enroll in any program for this practice. Table 2-2 summarizes farmer enrollment in the cost-share programs. CRP was the dominant funding source for enrolled farmers who adopted grass waterways and hayfield establishment. However, there was not a single dominant funding source for enrolled farmers who adopted conservation tillage, filter strips, cover crops, or grid sampling. Enrollment in the Great Miami WQTP represents only a small fraction of overall enrollment in Table 2-2 because this program was still in a pilot phase in 2009. The CSP rules are known to allow cost-share funding for both new and existing conservation practices. As such, CSP funds may be directed towards subsidizing conservation effort that is not additional. As a robustness check, I examine whether the estimation results for ATT differ significantly between CSP and all other programs, as discussed below in the results section.

TABLE 2-2. Farmer Enrollment in Cost-Share Programs by Practice Type.

Practice Type	CRP	CSP	EQIP	CREP	WQTP	OTHER	TOTAL
Conservation Tillage	24	36	16	1	5	11	93
Cover Crops	2	3	6	0	6	4	21
Hayfield Establishment	13	1	1	2	0	1	18
Grid Sampling	3	21	13	1	2	6	46
Grass Waterways	89	15	10	6	3	15	138
Filter Strips	48	15	8	18	1	8	98
Total	179	91	54	28	17	45	

Note: Total enrollment values in Table 2-2 do not match exactly those in Table 2-1 because certain farmers did not report in which program(s) they enrolled, and certain farmers reported enrolling in more than one program for a practice.

Tables A-1 through A-6 (Appendix A) provide the summary statistics of the covariates, prior to matching, for enrolled and non-enrolled farmers for all practices. T-test statistics on the differences in the covariate sample means for the two groups are also included. For example, for the grid sampling practice, the sample mean of the dummy variable on education exceeding high school is 0.655 for enrolled farmers and 0.413 for non-enrolled farmers, which is significantly different at the 99% level (Table A-4). Other covariates, including farm revenue, farm horizon, acres in grain, and farm size are also statistically different in their means for enrolled and non-enrolled farmers. Similarly, the other five practice types exhibit statistically significant differences in the sample means of several covariates before matching is conducted.

Propensity scores are estimated for each practice type using a probit model, where the dependent variable is the enrollment decision and the covariates X are used as explanatory variables. The estimated probit coefficients for grid sampling are provided in Table 2-3, where the covariates used in the estimation are those in Table A-4. The covariates for grid sampling that are significant are the proportion in grain crop, high slope, and education exceeding high school. The significance of covariates in the probit estimation varied by practice type (refer to Tables A-7 through A-11). For example, for grass waterways the covariates medium and high income, proportion in grain crop, medium slope, high slope, farm size, and stream adjacency are all significant at the 95% level or higher (Table A-10).By contrast, for filter strips, proportion in grain crop, stream adjacency, farm revenue, and education are significant at the 99% level (Table A-11).

The application of propensity score matching requires that the covariates are balanced given the propensity score (Deheija and Wahba, 1999). To test whether the covariates are balanced conditional on the propensity score, the probit model specification for each practice type was evaluated using the balancing algorithm explained in Becker and Ichino (2002). This test divides farmers into strata based on equal intervals of the estimated propensity score. Within each stratum, a test is conducted to assess whether there is a significant difference between the means for each covariate for enrolled and non-enrolled farmers. The probit model specification for each practice type satisfied the balancing test for all covariates.

TABLE 2-3. Estimated Coefficients from Probit Model to Compute Propensity Scores for Cost-Share Enrollment in Grid Sampling.

Variable	Estimated Coeff.	Std. Error
Farm Revenue	0.178	0.229
Farm Horizon	0.737	0.382
Age	0.009	0.011
Experience	-0.009	0.010
Education	0.566**	0.168
Soil Type: Loam or Sandy	0.103	0.188
Medium Income	0.251	0.256
High Income	0.115	0.274
Rented	0.014	0.267
Grain Crops	1.879*	0.844
Medium Slope	0.343	0.234
High Slope	1.473**	0.555
Farm Size	0.209	0.140
Stream	-0.184	0.168
Livestock	0.049	0.178
Constant	-5.871**	1.288
Log Likelihood	-152.439	

Note: Statistical significance: 99% (**), 95% (*).

2.4 Estimation Results

In this section, I provide and discuss the estimation results on additionality for the six conservation practices. Table 2-4 provides the estimates for the overall ATT and %ATT for all practice types. The estimation in Table 2-4 is performed using propensity score kernel matching with the Gaussian kernel type, where the common support requirement is enforced and the kernel bandwidth is 0.06.[4] Covariates were verified to be balanced across matched groups of enrollees and non-enrollees using a two-group t-test that checks for differences in the covariate means across the two groups. All covariates were balanced successfully for all practice types at the 95% level.[5] The standard errors and 95% confidence intervals (CI) were generated using a bootstrap procedure based on 1,000 simulations.[6]

In Table 2-4, the overall ATT is estimated based on equation [2.5]. The %ATT in Table 2-4 is the ratio of the overall ATT in equation [2.5] to the $E\left[Y_1 \mid D=1\right]$ in equation [2.3]

$$\% \, ATT = \frac{ATT}{E\left[Y_1 \mid D=1\right]} \cdot 100 . \qquad [2.6]$$

Note that the overall ATT is equal to $E\left[Y_1 \mid D=1\right]-E\left[Y_0 \mid D=1\right]$, which therefore has an upper bound of $E\left[Y_1 \mid D=1\right]$. The %ATT can be interpreted as the percentage increase in the proportion of conservation acreage relative to the total proportion of conservation

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[4] Two common support conditions are imposed in Stata to reduce poor quality matches. First, I used the common support condition that drops enrolled farmers whose propensity score is higher than the maximum or less than the minimum propensity score of the non-enrolled farmers (control group). Second, I used the 2% trimming condition that drops 2% of the enrolled farmers where the propensity score density of the control observations is the lowest.

[5] Refer to Caliendo and Kopeinig (2008) for more information on the covariate balancing test using a two-group t-test.

[6] An analytical formula for the standard error of the propensity score kernel matching estimator is not available (refer to Abadie and Imbens (2006; 2008)). As such, a bootstrapping procedure is used based on 1,000 random draws from the data set of farmers, with replacement and using the same number of farmers in each draw equal to the number in the original data set. The 95% bootstrapped CI consists of the 26th and 975th largest parameter estimates.

acreage adopted by enrolled farmers. The %ATT is thus equal to the percent additionality.

TABLE 2-4. Average Treatment Effect on the Treated and % ATT using Propensity Score Kernel Matching (Kernel Type: Gaussian, Bandwidth = 0.06).

Conservation Tillage	**Estimate**	**Std. Error**	**95% Bootstrapped CI**	
ATT	0.1600	0.0321	0.0910	0.2166
% ATT	20.5	3.6	12.3	26.5
Matched enrolled farmers = 86, Matched non-enrolled farmers = 476				
Cover Crops	**Estimate**	**Std. Error**	**95% Bootstrapped CI**	
ATT	0.2327	0.0472	0.1449	0.3273
% ATT	88.9	5.7	76.5	94.3
Matched enrolled farmers = 24, Matched non-enrolled farmers = 581				
Hayfield Establishment	**Estimate**	**Std. Error**	**95% Bootstrapped CI**	
ATT	0.2274	0.0660	0.0741	0.3435
% ATT	92.9	5.0	78.3	96.4
Matched enrolled farmers = 18, Matched non-enrolled farmers = 575				
Grid Sampling	**Estimate**	**Std. Error**	**95% Bootstrapped CI**	
ATT	0.5032	0.0514	0.3780	0.5827
% ATT	66.3	4.7	54.8	72.9
Matched enrolled farmers = 54, Matched non-enrolled farmers = 484				
Grass Waterways	**Estimate**	**Std. Error**	**95% Bootstrapped CI**	
ATT	0.0120	0.0023	0.0079	0.0165
% ATT	65.0	6.1	51.3	75.1
Matched enrolled farmers = 144, Matched non-enrolled farmers = 380				
Filter Strips	**Estimate**	**Std. Error**	**95% Bootstrapped CI**	
ATT	0.0101	0.0020	0.0065	0.0141
% ATT	89.1	5.3	75.8	95.7
Matched enrolled farmers = 92, Matched non-enrolled farmers = 451				

The overall ATT is positive and statistically significant for all six practices (Table 2-4). That is, the bootstrapped 95% confidence intervals on ATT for each of the six practice types does not contain zero. This suggests that enrollment in cost-share programs achieves a significantly positive level of additionality for each practice type. The ATT values in Table 2-4 are higher for field practices than those for environmentally sensitive areas. This is not surprising because filter strips and grass waterways are solely focused along stream banks and in natural drainage areas, and thus, represent a smaller proportion of the total farm acreage. Recall that the proportion of conservation acreage adopted by enrolled farmers is less than 0.02 for both filter strips and grass waterways (Table 2-1).

To compare the level of additionality between practice types, the %ATT in equation [2.6] is used. The largest %ATT is found for hayfield establishment, filter strips, and cover crops with 92.9%, 89.1%, and 88.9%, respectively (Table 2-4). Moderate percent additionality was found for grid sampling and grass waterways with %ATT at 66.3% and 65.0%. Conservation tillage had the lowest percent additionality at only 20.5%. In sum, this suggests that while cost-share funding from enrollment in conservation programs achieve a positive ATT for all practice types, certain practice types achieve higher percent additionality than others.

To test whether the %ATT values are statistically different across practice types, I construct bootstrapped confidence intervals of the difference in %ATT for all pairwise combinations of practice types (Table 2-5). For example, the difference in %ATT between cover crops relative to conservation tillage has a 95% bootstrapped confidence interval spanning lower and upper bounds of 55.0 % to 78.7%, respectively. This indicates that cover crops have a much higher %ATT than conservation tillage. Meanwhile, the difference in %ATT between cover crops and hayfield establishment is not statistically significant from zero because the bootstrapped confidence interval spans from -16.0% to 11.9%. When comparing the two practice types for environmentally sensitive areas, filter strips have a statistically larger %ATT than grass waterways.

TABLE 2-5. Bootstrapped 95% Confidence Intervals for Pair-wise Differences in %ATT using Propensity Score Kernel Matching (Kernel Type: Gaussian, Bandwidth = 0.06) (Row minus Column).

	Conservation Tillage	**Cover Crops**	**Hayfield Establishment**	**Grid Sampling**	**Grass Waterways**	**Filter Strips**
Conservation Tillage	-	[-78.7, -55.0]*	[-80.5, -57.6]*	[-55.8, -32.9]*	[-58.1, -28.8]*	[-79.7, -53.6]*
Cover Crops	[55.0, 78.7]*	-	[-16.0, 11.9]	[9.8, 36.5]*	[7.1, 38.1]*	[-13.8, 12.9]
Hayfield Establishment	[57.6, 80.5]*	[-11.9, 16.0]	-	[10.0, 37.5]*	[9.4, 40.8]*	[-12.5, 16.1]
Grid Sampling	[32.9, 55.8]*	[-36.5, -9.8]*	[-37.5, -10.0]*	-	[-14.7, 14.4]	[-37.2, -8.7]*
Grass Waterways	[28.8, 58.1]*	[-38.1 -7.1]*	[-40.8, -9.4]*	[-14.4, 14.7]	-	[-40.2, -7.4]*
Filter Strips	[53.6, 79.7]*	[-12.9, 13.8]	[-16.1, 12.5]	[8.7, 37.2]*	[7.4, 40.2]*	-

Note: * denotes statistical significance of the bootstrapped 95% confidence interval.

When evaluating whether to adopt a conservation field practice, farmers typically consider the impact such a practice would have on factors such as crop yields and operating costs. Hayfield establishment, for instance, would result in a complete loss in grain crop yields for the length of the enrollment contract. Meanwhile, conservation tillage often results in only modest changes in yields and may even lower operating costs stemming from reduced fuel consumption. As such, $E\left[Y_0 \mid D=1\right]$ for conservation tillage should be greater than that for hayfield establishment. Consequently, the %ATT is more likely greater for hayfield establishment than for conservation tillage (refer to equation [2.6]). The results in Table 2-4, showing that %ATT is higher for hayfield establishment than for conservation tillage, is consistent with the expectation that there are higher opportunity costs from losses in yield for hayfield establishment than for conservation tillage, and thus, hayfield establishment achieves greater percent additionality.

2.5 Robustness Checks

As a robustness check to the estimation results presented in Table 2-4, the ATT and %ATT are estimated using a variety of matching estimators that differ in the model specifications on the weights. Specifically, I conduct sensitivity analysis on the estimation results for all combinations of the following specifications: two matching methods (kernel and local linear), two kernel functions (Gaussian and Epanechnikov), and four bandwidths (bandwidths = 0.02, 0.06, 0.1, and 0.15).[7] This yields a total of sixteen different model specifications. The various model specifications provide a tradeoff between bias and variance. For instance, smaller (larger) bandwidth typically results in lower (higher) bias because it provides more weight to controls that are higher

[7] In addition to kernel and local linear matching, nearest-neighbor matching is another commonly used model specification. However, Abadie and Imbens (2006; 2008) explain that bootstrapped standard errors are not valid for nearest-neighbor matching with a fixed number of neighbors, and then further explain that kernel-based matching, for which the number of matches increase with sample size, has estimators that are asymptotically linear, and thus expect that the bootstrapped standard errors provide valid inferences. For this reason, I focus on the kernel and local linear matching estimators.

quality matches, but higher (lower) variance because less information is used to construct the counterfactual for each enrolled farmer. As an example, the estimation results for propensity score kernel matching with the Gaussian kernel function and a bandwidth of 0.02 are provided in Table A-12 (Appendix A); this is analogous to Table 2-4 except the smaller bandwidth of 0.02 is used instead of 0.06.

The main results in Table 2-4 are generally robust across all the model specifications. First, the overall ATT is positive and significant for every practice type across all sixteen model specifications. Second, the %ATT for each practice are similar in magnitude to the results in Table 2-4 across all model specifications, varying generally by less than 5%. The %ATT results also maintain the same ordering for three groups of practices. In Table A-12, for example, the largest %ATT estimates are hayfield establishment (91.1%), filter strips (89.6%), and cover crops (84.8%); the moderate %ATT estimates are grid sampling (66.7%) and grass waterways (65.5%); and the lowest %ATT is conservation tillage (17.3%).

Similarly, I analyzed the robustness of the bootstrapped confidence intervals for the pairwise difference in %ATT between practice types using the same sixteen model specifications on the weights described above. Again as an example, Table A-13 (Appendix A) provides the bootstrapped differences in %ATT when using the propensity score kernel with Gaussian function and bandwidth of 0.02; this is analogous to Table 2-5 with bandwidth of 0.06. The %ATT for conservation tillage is statistically smaller than the other five practices across all sixteen model specifications. For the majority of model specifications, the group of practice types with the largest %ATT estimates (hayfield establishment, filter strips, and cover crops) are significantly greater than the group of practices with moderate %ATT estimates (grid sampling and grass waterways). But the difference in %ATT between the practice types in the largest and moderate groups at times is no longer statistically significant; not surprisingly this occurs particularly for practice types with a smaller number of enrolled farmers and when using the smaller bandwidth at 0.02 (Table A-13). That said, the group of practice types with the largest %ATT has a mean difference in %ATT that is greater than 15% in all cases when compared to the group of practice types with moderate %ATT (Table A-12).

As mentioned above, the CSP rules specifically allow funding for both new and existing practices that could potentially result in lower levels of additionality for this program. As a robustness check, I performed bootstrapped simulations to test whether the estimation results on %ATT were significantly different between CSP and all other programs. Table 2-6 provides the estimation results for each of the following four practice types: conservation tillage, grid sampling, grass waterways, and filter strips. Note that cover crops and hayfield establishment are not included in Table 2-6 because the number of farmers enrolled in CSP for these practices was insufficiently large to perform this test (see Table 2-2). Estimation results in Table 2-6 indicate that the mean %ATT is actually higher in CSP than in all other programs for each of the four practice types; however, in all cases the difference was not statistically significant using the bootstrapped differences.

TABLE 2-6. Bootstrapped 95% Confidence Intervals for the Difference in %ATT between CSP and All Other Programs using Propensity Score Kernel Matching (Kernel Type: Gaussian, Bandwidth = 0.06).

Practice Type	% ATT (CSP Only)	% ATT (Other Programs)	Difference in %ATT	Std. Error	95% Bootstrapped CI	
Conservation Tillage	29.91	18.33	11.58	6.96	-0.93	26.73
Grid Sampling	70.10	66.18	3.92	8.34	-12.43	20.62
Grass Waterways	65.88	65.20	0.68	16.41	-37.26	21.89
Filter Strips	91.66	88.11	3.55	9.20	-15.79	16.96

It should be acknowledged that estimation of the ATT using propensity score matching is based on the unconfoundedness assumption. If there exist unobserved covariates that influence both enrollment and the outcome variables, then the estimated ATT may be biased. Although the unconfoundedness assumption cannot be verified in practice, Rosenbaum (2002) developed a method to test the extent to which a matching estimator is sensitive to hidden bias. Specifically, Rosenbaum's approach assumes that the propensity score, $P(D=1|X)$, is influenced not only by observed covariates *X*, but also by an unobserved covariate. As a result of this unobserved covariate, farmers that are matched based on similar propensity score values, may actually differ in their odds of enrolling by a factor of Γ, where $\Gamma=1$ represents the baseline case of no hidden bias. The higher the level of Γ to which the ATT remains statistically different from zero, the more robust are the estimation results to the potential influence of hidden bias.

Rosenbaum bounds sensitivity analysis is conducted to estimate the extent to which selection on unobservables may bias the estimates of the ATT (see Rosenbaum

2002 and DiPrete and Gangl 2004 for more information). Using this approach, which relies on a signed rank test, I determine the upper bounds on the significance level (i.e., critical p-values) of the ATT for different levels of hidden bias in terms of Γ. Estimation results from the Rosenbaum bounds sensitivity analysis are provided in Table 2-7. The first column provides the Γ values and the second column (sig+) provides the corresponding upper bound on the p-value for the ATT. The critical p-values associated with the largest Γ where the ATT remains statistically significant at the 5% level are enclosed in a box. Filter strips is the most robust to the potential presence of hidden bias, where the estimated ATT remains statistically different from zero at the 5% level for a critical threshold Γ value at 7.3. Conservation tillage, on the other hand, is the least robust to hidden bias, where the critical Γ value is 1.3. The other practice types have moderate to high critical Γ values ranging from 2.2 for hayfield establishment to 6 for cover crops.

In summary, the estimates for ATT and %ATT are robust across the majority of matching model specifications estimated. In addition, for the majority of model specifications, the pairwise difference in %ATT between practice types reveal that the group of practice types with the largest %ATT estimates (hayfield establishment, filter strips, and cover crops) are significantly greater than the group of practices with moderate %ATT estimates (grid sampling and grass waterways), while the %ATT for conservation tillage is consistently smaller than that of all other practices. Finally, the Rosenbaum bounds sensitivity analysis suggests that the ATT estimates vary by practice type in their level of robustness to unobserved selection bias.

TABLE 2-7. Results for Rosenbaum Bounds Sensitivity Analysis.

Conservation Tillage		Cover Crops		Hayfield Establishment		Grid Sampling	
Γ	Sig +	Γ	Sig +	Γ	Sig +	Γ	Sig +
1	0.0043	3	0.0073	1	0.0014	2	0.0002
1.05	0.0072	3.5	0.0120	1.2	0.0040	2.5	0.0012
1.1	0.0114	4	0.0176	1.4	0.0083	3	0.0042
1.15	0.0173	4.5	0.0237	1.6	0.0145	3.5	0.0101
1.2	0.0252	5	0.0302	1.8	0.0226	4	0.0198
1.25	0.0353	5.5	0.0369	2	0.0323	4.5	0.0335
1.3	0.0478	6	0.0437	2.2	0.0435	4.8	0.0435
1.35	0.0629	6.5	0.0506	2.4	0.0558	5	0.0509

Grass Waterways		Filter Strips	
Γ	Sig +	Γ	Sig +
1.4	0.0000	2	0.0000
1.6	0.0001	3	0.0001
1.8	0.0007	4	0.0012
2	0.0033	5	0.0060
2.2	0.0110	6	0.0175
2.4	0.0281	7	0.0374
2.55	0.0498	7.3	0.0450
2.6	0.0590	7.6	0.0535

2.6 Conclusions

Federal cost-share funding for the adoption of conservation practices on working lands have increased considerably starting in 2002. The efficiency of cost-share programs depends in part on the degree to which they provide additional conservation effort. In this chapter, I use propensity score matching to estimate the level of additionality from enrollment in cost-share programs for six conservation practices. Results indicate that the enrollment achieves positive and significant levels of additionality for each of the six practice types. That being said, the percent additionality varies dramatically between practice types. Specifically, the percent additionality is highest for hayfield establishment (92.9%), filter strips (89.1%), and cover crops (88.9%), while it is lowest for conservation tillage (20.5%). While these results are valuable for program managers in evaluating conservation program effectiveness, they provide only part of the analysis that is required. For example, it is important to evaluate not only increases in conservation effort, but also overall improvements in environmental benefits. Having quantifiable information on the actual improvement in environmental benefits per practice, coupled with additionality estimates, would provide greater insight into the effectiveness of conservation programs.

The practice of offering payment incentives to landowners to improve environmental stewardship is growing in popularity. Emerging markets for ecosystem services are being developed that offer payments to landowners to enhance carbon sequestration and water quality via the adoption of agricultural conservation practices. Additionality is a major concern in such programs because it is an essential element of program effectiveness. As the implementation of incentive-based programs increases to address environmental concerns, analysis of existing programs is crucial to determine how much these programs induce increases in conservation effort. In sum, this study helps meet that need by measuring additionality for incentive-based programs.

While Chapter II provides insights into the additionality of conservation programs across a range of conservation practices, this research could be improved in the following areas. Future directions of study include better accounting for the potential correlation that exists between enrollment in conservation programs for different conservation practices, which can lead to a potential bias of the additionality estimates. Second, the estimates of additionality are aggregated across several conservation programs. Greater insight into conservation programs would be obtained if additionality estimates were generated for individual programs rather than as an aggregation across several programs. This analysis will be pursued to the extent that the data permits.

CHAPTER III

THE DECOMPOSITION OF ADDITIONALITY

3.1 Introduction

The estimation of additionality (ATT) in Chapter II of conservation programs is an aggregate measure composed of two effects: the new adoption of conservation practices versus the expansion of existing conservation practices. First, conservation is implemented by farmers who would have not adopted the practice without the subsidy. That is, the subsidy convinces “new-adopters” to use an entirely new practice. Second, the conservation program subsidy can also cause farmers who would have adopted the practice without the subsidy (“prior-adopters”) to expand their use of the practice. The purpose of this chapter is to decompose these two components of the treatment effect and empirically estimate them for federal agricultural conservation programs.

The disaggregation of the ATT into the relative contributions of prior-adopters and new-adopters reveals greater insights into the additionality achieved by cost-share programs. First, the disaggregation of the ATT allows for a comparison of the level of additionality provided by each type of farmer, and, most importantly, whether one farmer type provides greater levels of additionality than the other. Second, the disaggregation also reveals the proportion of enrolled farmers that belong to each farmer type. Overall, the disaggregation of the ATT allows for a more thorough understanding of the additionality achieved by cost-share programs.

The decomposition of the ATT uses matching estimators to determine the likelihood that enrolled farmers are prior-adopters or new-adopters, as well as to estimate the relative contribution for each group to the overall ATT. Two main results are obtained from decomposing the ATT. First, I find that the ATT for prior-adopters is not significant for all practice types, implying that prior-adopters do not significantly expand the proportion of conservation acreage when receiving cost-share funding. Second, decomposition estimates also suggest that the differences in %ATT between practice types are mainly determined by the fraction of enrolled farmers that are prior-adopters and new-adopters. Practice types that are estimated to have a large fraction of new-adopters, such as filters trips and hayfield establishment, exhibit larger values for %ATT.

The chapter is structured as follows. First, I derive the decomposition of the ATT. Second, I derive the estimators for the expected probabilities that enrolled farmers are prior-adopters or new-adopters, followed by the estimators for the ATT of prior-adopters and new-adopters. Thereafter, I provide the estimation results for the decomposition of the ATT and robustness checks. I conclude with policy implications for conservation programs.

3.2 Decomposition of the Propensity Score Matching Estimator

In this section, I derive the decomposition of the ATT and the estimators for each of the decomposed components of the ATT.

3.2.1. Decomposing the ATT for New-Adopters and Prior-Adopters

I define two types of farmers based on their potential outcome Y_0 (refer to Chapter II). Treated farmers are divided into:

- *New-Adopters* are enrolled farmers $(D=1)$ who would have not adopted the practice without funding $(Y_0=0)$.
- *Prior-Adopters* are enrolled farmers $(D=1)$ who would have adopted the practice even in the absence of cost-share funding $(Y_0>0)$.

As discussed in Chapter II, the potential outcome Y_0 for enrolled farmers is not observed and must be estimated. I define as well the probabilities $P(Y_0=0|D=1)$ and $P(Y_0>0|D=1)$, which are the expected probabilities that the enrolled farmers are either new-adopters or prior-adopters, respectively. Given that $Y_0 \geq 0$, it holds that

$$P(Y_0=0|D=1)+P(Y_0>0|D=1)=1. \qquad [3.1]$$

The ATT described in the previous chapter in equation [2.1] can be decomposed into two parts to determine the relative amount of the ATT that is attributable to prior-adopters and new-adopters. Using conditional expectations and probabilities based on Y_0, the ATT can be decomposed into:

$$\begin{aligned} ATT &= P(Y_0=0|D=1)\cdot\left\{E\left[Y_1|Y_0=0,D=1\right]-E\left[Y_0|Y_0=0,D=1\right]\right\} \\ &\quad +P(Y_0>0|D=1)\cdot\left\{E\left[Y_1|Y_0>0,D=1\right]-E\left[Y_0|Y_0>0,D=1\right]\right\}. \end{aligned} \qquad [3.2]$$

The first line represents the portion of the ATT that corresponds to new-adopters, i.e. those for whom $Y_0 = 0$. The term $E\left[Y_1 \mid Y_0 = 0, D = 1\right]$ is the expected proportion of acreage that new-adopters dedicate to the conservation practice when receiving funding. Meanwhile, $E\left[Y_0 \mid Y_0 = 0, D = 1\right]$ is the expected proportion new-adopters dedicate to the practice when not receiving funding, which equals zero by definition. The difference of these two terms equals the expected additional proportion of acreage that new-adopters dedicate to the conservation practice when receiving funding. The second line in [3.2] represents the portion of the ATT that corresponds to prior-adopters , i.e. those for whom $Y_0 > 0$. The difference in the two terms $E\left[Y_1 \mid Y_0 > 0, D = 1\right]$ and $E\left[Y_0 \mid Y_0 > 0, D = 1\right]$ is equal to the expected additional proportion of acreage that prior-adopters dedicate to the conservation practice as a result of receiving funding. The overall ATT in equation [3.2] equals the weighted average of these two expected gains in the proportion of conservation acreage adopted.

To simplify notation, I define the respective ATT for enrolled new-adopters and prior-adopters as

$$\begin{aligned} ATT_n &= E\left[Y_1 \mid Y_0 = 0, D = 1\right] - E\left[Y_0 \mid Y_0 = 0, D = 1\right], \\ ATT_a &= E\left[Y_1 \mid Y_0 > 0, D = 1\right] - E\left[Y_0 \mid Y_0 > 0, D = 1\right]. \end{aligned} \quad [3.3]$$

Similarly, I define the respective expected probabilities that the enrolled farmers are new-adopters or prior-adopters as

$$\begin{aligned} P_n &= P\left(Y_0 = 0 \mid D = 1\right), \\ P_a &= P\left(Y_0 > 0 \mid D = 1\right). \end{aligned} \quad [3.4]$$

The decomposed ATT in [3.2] can be rewritten as:

$$ATT = P_n \cdot ATT_n + P_a \cdot ATT_a \,. \qquad [3.5]$$

Below I derive the estimators for each of the decomposed terms in equation [3.5].

3.2.2. Estimators for the Probabilities of New-Adopters and Prior-Adopters

In this section, I derive the estimators for P_n and P_a in equation [3.4]. I first define a binary variable B_0 to explain how matching estimators are used to derive the estimators for these probabilities. Specifically, B_0 equals one if a farmer would adopt a practice without funding, and zero otherwise, i.e., $B_0 = 1$ if $Y_0 > 0$, and $B_0 = 0$ if $Y_0 = 0$. The probability that Y_0 is greater than zero, P_a, can be expressed in terms of the expectation of B_0

$$P_a = P\left(Y_0 > 0 \mid D = 1\right) = P\left(B_0 = 1 \mid D = 1\right) = E\left[B_0 \mid D = 1\right]. \qquad [3.6]$$

An estimator for $E\left[B_0 \mid D = 1\right]$ can be obtained using a matching estimator, analogous to the approach used on the estimator for $E\left[Y_0 \mid D = 1\right]$ in equation [2.2]. This yields the estimate for P_a, and the estimate for the other probability, P_n, can be obtained using [3.1].

Similar to equation [2.2], the matching estimator for $\hat{B}_0^i$ is the weighted average

$$\hat{B}_0^i = \sum_{j \in H_0} W\left(i, j\right) B_0^j \,, \qquad [3.7]$$

where B_0^j is the B_0 for non-enrollee *j*. Note that $\hat{B}_0^i$ is the estimate of the probability that an enrolled farmer with propensity score P_i is a prior-adopter, such that

$$\hat{B}_0^i = \hat{E}\left[B_0^i \mid P_i, D_i = 1\right] = \hat{P}\left(Y_0^i > 0 \mid P_i, D_i = 1\right). \qquad [3.8]$$

The matching estimator for $E\left[B_0 \mid D = 1\right]$ is then the average of the $\hat{B}_0^i$ for the set of *I* enrollees in H_1

$$\hat{E}\left[B_0 \mid D = 1\right] = \frac{1}{I}\sum_{i \in H_1} \hat{B}_0^i . \qquad [3.9]$$

Consequently, given equations [3.6] and [3.7], the estimator for P_a is

$$\hat{P}_a = \hat{P}\left(Y_0 > 0 \mid D = 1\right) = \frac{1}{I}\sum_{i \in H_1} \hat{B}_0^i = \frac{1}{I}\sum_{i \in H_1}\sum_{j \in H_0} W\left(i, j\right) B_0^j , \qquad [3.10]$$

and the estimator for P_n is obtained by substituting [3.10] into [3.1]

$$\hat{P}_n = \hat{P}\left(Y_0 = 0 \mid D = 1\right) = 1 - \frac{1}{I}\sum_{i \in H_1} \hat{B}_0^i = \frac{1}{I}\sum_{i \in H_1}\left(1 - \hat{B}_0^i\right). \qquad [3.11]$$

3.2.3. Estimators for the ATT of New-Adopters and Prior-Adopters

In this section, I provide the estimators of ATT_n for new-adopters and ATT_a for prior-adopters that are defined in equation [3.3], respectively. I first provide the estimators for the conditional expectations of Y_1, then for the conditional expectations of Y_0, and finally take the difference in these conditional expectations to arrive at the respective estimators for ATT_n and ATT_a.

The estimator for the conditional expectation of Y_1 for new-adopters is

$$\hat{E}\left[Y_1 \mid Y_0 = 0, D = 1\right] = \frac{\sum_{i \in H_1} \left(1 - \hat{B}_0^i\right) Y_1^i}{\sum_{i \in H_1} \left(1 - \hat{B}_0^i\right)}. \qquad [3.12]$$

This estimator is the average value of Y_1 across all *I* enrollees weighted by the estimated probability that the enrollee is a new-adopter, $1 - \hat{B}_0^i$. Likewise, the estimator for the conditional expectation of Y_1 for prior-adopters is:

$$\hat{E}\left[Y_1 \mid Y_0 > 0, D = 1\right] = \frac{\sum_{i \in H_1} \hat{B}_0^i Y_1^i}{\sum_{i \in H_1} \hat{B}_0^i}, \qquad [3.13]$$

which is the estimator for the average value of Y_1 across all *I* enrollees weighted by the estimated probability that the enrollee is a prior-adopter, $\hat{B}_0^i$.

The estimator for the conditional expectation of Y_0 for new-adopters, $E\left[Y_0 \mid Y_0 = 0, D = 1\right]$, equals zero by definition, as noted previously. The estimator for the conditional expectation of Y_0 for prior-adopters is

$$\hat{E}\left[Y_0 \mid Y_0 > 0, D=1\right] = \frac{\hat{E}\left[Y_0 \mid D=1\right]}{\hat{P}\left(Y_0 > 0 \mid D=1\right)} = \frac{\sum_{i \in H_1} \sum_{j \in H_0} W(i,j) Y_0^j}{\sum_{i \in H_1} \sum_{j \in H_0} W(i,j) B_0^j}, \qquad [3.14]$$

where I have substituted $\hat{E}\left[Y_0 \mid D=1\right]$ in equation [2.4] and $\hat{P}\left(Y_0 > 0 \mid D=1\right)$ in equation [3.10] into equation [3.14] above.[8]

After substituting [3.12] into the expression for ATT_n found in equation [3.3] and noting that $E\left[Y_0 \mid Y_0 = 0, D=1\right] = 0$, the estimator for the ATT of new-adopters is obtained

$$A\hat{T}T_n = \frac{\sum_{i \in H_1} \left(1 - \hat{B}_0^i\right) Y_1^i}{\sum_{i \in H_1} \left(1 - \hat{B}_0^i\right)}. \qquad [3.15]$$

Similarly, after substituting [3.13] and [3.14] into the expression for ATT_a in equation [3.3], the estimator for the ATT of prior-adopters is obtained

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[8] Equation [3.14] can be equivalently expressed as $\hat{E}\left[Y_0 \mid Y_0 > 0, D=1\right] = \frac{\sum_{i \in H_1} \sum_{j \in H_0} W(i,j) B_0^j Y_0^j}{\sum_{i \in H_1} \sum_{j \in H_0} W(i,j) B_0^j}$.

Note that $B_0^j Y_0^j = Y_0^j$ in the numerator of Equation [3.14] because $B_0^j = 0$ when $Y_0^j = 0$, and $B_0^j = 1$ when $Y_0^j > 0$.

$$A\hat{T}T_a = \left[\frac{\sum_{i \in H_1} \hat{B}_0^i Y_1^i}{\sum_{i \in H_1} \hat{B}_0^i}\right] - \left[\frac{\sum_{i \in H_1} \sum_{j \in H_0} W(i,j) Y_0^j}{\sum_{i \in H_1} \sum_{j \in H_0} W(i,j) B_0^j}\right]. \qquad [3.16]$$

The estimator for the overall ATT in equation [3.5] is

$$A\hat{T}T = \hat{P}_n \cdot A\hat{T}T_n + \hat{P}_a \cdot A\hat{T}T_a \ , \qquad [3.17]$$

where the proposed estimators for the decomposed parts in equations [3.10], [3.11], [3.15], and [3.16] are substituted into equation [3.17] above. In the section below, I validate that this yields the same expression as the overall ATT in equation [2.5].

3.2.4 Validation of the Estimators for the Decomposition of the ATT

Here I demonstrate that when the proposed estimators for the decomposed parts in equations [3.10], [3.11], [3.15], and [3.16] are substituted into equation [3.5], this yields the same expression as the overall ATT shown in equation [2.5]. To begin, I substitute the estimators on the four decomposed parts from equations [3.10], [3.11], [3.15], and [3.16] into equation [3.5]

$$\begin{aligned}
A\hat{T}T &= \hat{P}_n \cdot A\hat{T}T_n + \hat{P}_a \cdot A\hat{T}T_a \\
&= \left[\frac{1}{I} \sum_{i \in H_1} \left(1 - \hat{B}_0^i \right) \right] \cdot \left[\frac{\sum_{i \in H_1} \left(1 - \hat{B}_0^i \right) Y_1^i}{\sum_{i \in H_1} \left(1 - \hat{B}_0^i \right)} \right] \\
&+ \left[\frac{1}{I} \sum_{i \in H_1} \hat{B}_0^i \right] \cdot \left\{ \left[\frac{\sum_{i \in H_1} \hat{B}_0^i Y_1^i}{\sum_{i \in H_1}^{n} \hat{B}_0^i} \right] - \left[\frac{\sum_{i \in H_1} \sum_{j \in H_0} W(i,j) Y_0^j}{\sum_{i \in H_1} \sum_{j \in H_0} W(i,j) B_0^j} \right] \right\}.
\end{aligned} \qquad [3.18]$$

After using equation [3.10] and cancelling terms, equation [3.18] can be rewritten as

$$A\hat{T}T = \frac{1}{I} \left\{ \sum_{i \in H_1} \left(1 - \hat{B}_0^i \right) Y_1^i + \sum_{i \in H_1} \hat{B}_0^i Y_1^i \right\} - \frac{1}{I} \sum_{i \in H_1} \sum_{j \in H_0} W(i,j) Y_0^j . \qquad [3.19]$$

The first term in brackets in equation [3.19] reduces to the matching estimator for $E\left[Y_1 \mid D = 1 \right]$ in [2.3]. Thus, equation [3.19] yields

$$A\hat{T}T = \frac{1}{I} \sum_{i \in H_1} Y_1^i - \frac{1}{I} \sum_{i \in H_1} \sum_{j \in H_0} W(i,j) Y_0^j = \frac{1}{I} \sum_{i \in H_1} \left[Y_1^i - \sum_{j \in H_0} W(i,j) Y_0^j \right], \quad [3.20]$$

which equals the matching estimator for the overall ATT in equation [2.5].

3.3 Estimation Results

In this section, I provide the estimation results on the decomposed components of the ATT for the six conservation practices. Refer to the data section of Chapter II for information on the data used for the estimation. The estimated average probabilities P_a and P_n for the set of enrolled farmers that are prior-adopters or new-adopters are calculated based on equations [3.10] and [3.11], respectively. Meanwhile, the estimates of ATT_n for new-adopters and ATT_a for prior-adopters are calculated using equations [3.15] and [3.16], respectively. Table 3-1 provides the estimates for the decomposed components for all practice types, as well as the overall ATT and %ATT values from Table 2-4 for reference. The estimates are performed using propensity score kernel matching with the Gaussian kernel type, where the common support requirement is enforced and the kernel bandwidth is 0.06 (refer to Chapter II for more information on the model specification). The standard errors and 95% confidence intervals (CI) were generated using the same bootstrap procedure based on 1,000 simulations.

The components of the decomposed ATT (Table 3-1) show the relative contributions of new-adopters and prior-adopters to the overall ATT, which, in turn, explains the differences in %ATT between practice types. Table 3-1 highlights that ATT_a is less than ATT_n for all practice types. Interestingly, ATT_a is positive but not statistically different from zero for all practices. This result indicates that the cost-share programs have no significant effect on the conservation of those who would have adopted the practice without funding – the additional conservation comes from convincing new-adopters to adopt a conservation practice.

TABLE 3-1. Average Treatment Effect on the Treated and Decomposed Effects for New-Adopters and Prior-Adopters using Propensity Score Kernel Matching (Kernel Type: Gaussian, Bandwidth = 0.06).

Conservation Tillage	**Estimate**	**Std. Error**	**95% Bootstrapped CI**	
ATT	0.1600	0.0321	0.0910	0.2166
% ATT	20.5	3.6	12.3	26.5
P_n	0.1403	0.0191	0.1027	0.1769
P_a	0.8597	0.0191	0.8231	0.8973
ATT_n	0.7449	0.0324	0.6823	0.8090
ATT_a	0.0645	0.0317	-0.0036	0.1190
Matched enrolled farmers = 86, Matched non-enrolled farmers = 476				
Cover Crops	**Estimate**	**Std. Error**	**95% Bootstrapped CI**	
ATT	0.2327	0.0472	0.1449	0.3273
% ATT	88.9	5.7	76.5	94.3
P_n	0.8665	0.0249	0.8060	0.9033
P_a	0.1335	0.0249	0.0967	0.1940
ATT_n	0.2623	0.0453	0.1849	0.3527
ATT_a	0.0403	0.0797	-0.1297	0.1774
Matched enrolled farmers = 24, Matched non-enrolled farmers = 581				
Hayfield Establishment	**Estimate**	**Std. Error**	**95% Bootstrapped CI**	
ATT	0.2274	0.0660	0.0741	0.3435
% ATT	92.9	5.0	78.3	96.4
P_n	0.9007	0.0212	0.8454	0.9326
P_a	0.0993	0.0212	0.0676	0.1560
ATT_n	0.2401	0.0666	0.0933	0.3595
ATT_a	0.1122	0.0875	-0.1192	0.2247
Matched enrolled farmers = 18, Matched non-enrolled farmers = 575				
Grid Sampling	**Estimate**	**Std. Error**	**95% Bootstrapped CI**	
ATT	0.5032	0.0514	0.3780	0.5827
% ATT	66.3	4.7	54.8	72.9
P_n	0.6001	4.6890	0.5113	0.6587
P_a	0.3999	4.6890	0.3413	0.4887
ATT_n	0.7560	0.0415	0.6689	0.8276
ATT_a	0.1237	0.0614	-0.0074	0.2378
Matched enrolled farmers = 54, Matched non-enrolled farmers = 484				

TABLE 3-1. Continued.

Grass Waterways	Estimate	Std. Error	95% Bootstrapped CI	
ATT	0.0120	0.0023	0.0079	0.0165
% ATT	65.0	6.1	51.3	75.1
P_n	0.5771	0.0357	0.5098	0.6456
P_a	0.4229	0.0357	0.3544	0.4902
ATT_n	0.0187	0.0022	0.0146	0.0234
ATT_a	0.0028	0.0029	-0.0027	0.0089
Matched enrolled farmers = 144, Matched non-enrolled farmers = 380				
Filter Strips	**Estimate**	**Std. Error**	**95% Bootstrapped CI**	
ATT	0.0101	0.0020	0.0065	0.0141
% ATT	89.1	5.3	75.8	95.7
P_n	0.8326	0.0310	0.7656	0.8851
P_a	0.1674	0.0310	0.1149	0.2344
ATT_n	0.0114	0.0019	0.0079	0.0154
ATT_a	0.0036	0.0036	-0.00420	0.0103
Matched enrolled farmers = 92, Matched non-enrolled farmers = 451				

Practices for which a large fraction of enrolled farmers are prior-adopters (i.e., P_a is large) typically have a lower %ATT. Consider conservation tillage where ATT_n is 0.75, while ATT_a is only 0.07. The estimated fraction of enrolled farmers for conservation tillage that are prior-adopters, $P_a = 0.86$, is much larger than that of new-adopters, $P_n = 0.14$. Consequently, the overall ATT is small relative to the total proportion of conservation acreage adopted, and thus, the %ATT is relatively low for conservation tillage.

In general, practices where P_n is considerably larger than P_a have higher %ATT values. When comparing the two environmentally sensitive practice types, the fraction of enrolled farmers that are new-adopters for filter strips is $P_n = 0.83$, while for grass waterways $P_n = 0.58$ (Table 3-1). This is the principle reason why the %ATT is larger for filters strips (89.1%) than for grass waterways (65.0%). Similarly, when comparing the four field practices, cover crops and hayfield establishment have larger P_n values than either grid sampling or conservation tillage. As such, the %ATT values for cover crops (88.9%) and hayfield establishment (92.9%) exceed that of either grid sampling (66.3%) or conservation tillage (20.5%).

The heterogeneity in P_a and P_n, and consequently in %ATT, may be related to differences in the onsite net benefits provided by the different practice types. Higher onsite net benefits should increase the likelihood that a farmer would adopt a practice even without funding, increasing the proportion of prior-adopters for this practice type. Consider a comparison of the two environmentally sensitive practice types. Filter strips are typically located along stream banks, and therefore mainly provide offsite benefits in terms of improved water quality by reducing nutrients and sediments from entering downstream water bodies. Grass waterways, in contrast, are typically installed in natural drainage areas within cultivated lands which provide both onsite and offsite benefits. The results in Table 3-1, showing that P_n and %ATT are higher for filter strips than grass waterways, coincide with the expectation that farmers would be less likely to adopt filter strips without cost-share funding.

3.4 Robustness Checks

As in Chapter II, I estimate the decomposed effects using a variety of matching estimators as a robustness check to the estimation results presented in Table 3-1. Refer to the robustness checks section of Chapter II for more information on the sixteen different model specifications used.

The main results in Table 3-1 are generally robust across all the model specifications. First, ATT_a is less than ATT_n for all practice types. In addition, the ATT_a is not statistically different from zero for all practices, except for four model specifications, where the ATT_a for conservation tillage and grid sampling are statistically significant. Second, as discussed in Chapter II, the %ATT for each practice across all model specifications are similar in magnitude, and maintain the same ordering for the three groups of practices. Given that the %ATT is a function of the decomposed components P_n, ATT_n, P_a, and ATT_a (refer to equations [2.6] and [3.5]), this implies that the decomposed components are also generally similar in magnitude across the different model specifications.

As an example, the estimation results for propensity score kernel matching with the Gaussian kernel function and a bandwidth of 0.02 are provided in Table 3-2; this is analogous to Table 3-1, except the smaller bandwidth of 0.02 is used instead of 0.06.

TABLE 3-2. Average Treatment Effect on the Treated and Decomposed Effects for New-Adopters and Prior-Adopters using Propensity Score Kernel Matching (Kernel Type: Gaussian, Bandwidth = 0.02).

Conservation Tillage	Estimate	Std. Error	95% Bootstrapped CI	
ATT	0.1351	0.0338	0.0693	0.2019
% ATT	17.3	3.9	9.5	24.9
P_n	0.1199	0.0211	0.0863	0.1683
P_a	0.8801	0.0211	0.8317	0.9137
ATT_n	0.7401	0.0371	0.6661	0.8146
ATT_a	0.0527	0.0333	-0.0169	0.1168
Matched enrolled farmers = 86, Matched non-enrolled farmers = 476				
Cover Crops	**Estimate**	**Std. Error**	**95% Bootstrapped CI**	
ATT	0.2220	0.0500	0.1364	0.3216
% ATT	84.8	9.1	66.1	95.5
P_n	0.8538	0.0428	0.7465	0.9216
P_a	0.1462	0.0428	0.0784	0.2535
ATT_n	0.2612	0.0455	0.1836	0.3542
ATT_a	-0.0067	0.1032	-0.2218	0.2011
Matched enrolled farmers = 24, Matched non-enrolled farmers = 581				
Hayfield Establishment	**Estimate**	**Std. Error**	**95% Bootstrapped CI**	
ATT	0.2232	0.0677	0.0690	0.3398
% ATT	91.1	9.8	62.6	96.4
P_n	0.8928	0.0412	0.7778	0.9400
P_a	0.1072	0.0412	0.0601	0.2246
ATT_n	0.2423	0.0682	0.0872	0.3668
ATT_a	0.0639	0.1152	-0.2071	0.2612
Matched enrolled farmers = 18, Matched non-enrolled farmers = 575				
Grid Sampling	**Estimate**	**Std. Error**	**95% Bootstrapped CI**	
ATT	0.5067	0.0580	0.3503	0.5788
% ATT	66.7	5.9	50.3	73.4
P_n	0.5751	0.0502	0.4618	0.6618
P_a	0.4249	0.0502	0.3382	0.5382
ATT_n	0.7611	0.0438	0.6652	0.8315
ATT_a	0.1623	0.0737	-0.0323	0.2584
Matched enrolled farmers = 54, Matched non-enrolled farmers = 484				

TABLE 3-2. Continued.

Grass Waterways	Estimate	Std. Error	95% Bootstrapped CI	
ATT	0.0121	0.0023	0.0078	0.0165
% ATT	65.5	6.7	49.8	76.0
P_n	0.5613	0.0418	0.4843	0.6432
P_a	0.4387	0.0418	0.3568	0.5157
ATT_n	0.0190	0.0023	0.0147	0.0237
ATT_a	0.0032	0.0031	-0.0030	0.0093
Matched enrolled farmers = 144, Matched non-enrolled farmers = 380				
Filter Strips	**Estimate**	**Std. Error**	**95% Bootstrapped CI**	
ATT	0.0101	0.0020	0.0064	0.0141
% ATT	89.6	5.7	74.7	95.9
P_n	0.8304	0.0374	0.7431	0.8899
P_a	0.1696	0.0374	0.1101	0.2569
ATT_n	0.0115	0.0019	0.0078	0.0154
ATT_a	0.0035	0.0040	-0.00459	0.0116
Matched enrolled farmers = 92, Matched non-enrolled farmers = 451				

The decomposed estimates found in Tables 3-1 and 3-2 are similar, except for differences in the ATT_a estimates for cover crops and hayfields. The ATT_a for cover crops and hayfields in Table 3-1 are 0.0403 and 0.1122, respectively, while they are -0.0067 and 0.0639, respectively, in Table 3-2. However, none of these estimates are statistically significant. Note also that both cover crops and hayfields are the practices with the smallest number of enrollees; as such, these practices should experience greater variation in the estimates across the different model specifications.

3.5 Conclusions

As stated in Chapter II, the efficiency of cost-share programs depends in part on the degree to which they provide additionality. The estimation of additionality (ATT) in

Chapter II is based on an aggregate measure of the additionality achieved by cost-share programs across two types of farmers: prior-adopters and new-adopters. Decomposing the ATT reveals the relative contributions of prior-adopters and new-adopters to the overall ATT, thus providing additional insights into the additionality of these programs. In this chapter, I develop the decomposition of the ATT, derive estimators for each component of the decomposition, and estimate the decomposed terms for all six conservation practices. The decomposed estimates enhance the insights obtained from the overall ATT estimates found in Chapter II.

Results indicate that the ATT for prior-adopters (ATT_a) was not significant for all practice types, suggesting that program enrollment is not inducing significant management changes for farmers who would have used a practice even in the absence of cost-share funding. Second, decomposition estimates suggest that the differences in %ATT between practice types are mainly determined by the fraction of enrolled farmers that are prior-adopters versus new-adopters. Practice types that have a large fraction of new-adopters, such as filter strips and hayfield establishment, exhibit larger values for %ATT.

Measuring additionality is of importance not only for conservation programs, but for any program or policy where the average treatment effect on the treated (ATT) is of interest. This chapter provides a new methodology that permits the ATT to be decomposed into two effects based on the relative contributions of two types of farmers: prior-adopters and new-adopters. However, this approach is applicable not only to cost-share programs, but to the study of any policy or program where participants can be categorized into two distinct groups. As such, the methodology proposed in this chapter can be used to decompose the ATT estimates for any program or policy, thus enhancing the study of additionality within other programs of interest.

The analysis of additionality in Chapters II and III is limited in the following areas. First, the overall objective of conservation programs is to increase the amount of environmental benefits provided by farmers. Consequently, additionality should be measured ideally in terms of increases in environmental benefits, rather than increases in the percent of conservation acreage. Having quantifiable information on the actual improvement in environmental benefits achieved by enrolled farmers would thus provide greater insight into the effectiveness of conservation programs. Second, the limited information obtained from the survey on the cost-share payments made to farmers per practice limit the estimation of additionality to only increases in conservation effort. However, the estimation of additionality achieved per dollar spent would provide a more interesting analysis of the cost-effectiveness of programs and conservation practices.

CHAPTER IV

THE EFFECT OF INSTREAM FLOW POLICIES ON AGRICULTURAL WATER SECURITY AND STREAMFLOWS*

4.1 Introduction

Water use conflicts have become a dominant global environmental issue, particularly in arid climates, where increasing water demands by growing populations, irrigation for agricultural, and the adverse effects of climate change threaten human water security. At the same time, a growing awareness of the importance of streamflows for sustaining aquatic ecosystems is forcing regulatory agencies to further restrict water uses. Such instream flow protections are concerned not only with the quantity of water, but also the timing of flow releases and water quality.

The challenge of reconciling competing water needs is exemplified in the Western U.S., where flow regime alterations from water management have been a primary driver of ecosystem degradation (Dole and Niemi 2004). Consequently, the protection of instream flows has become a necessity for maintaining ecosystem functions and preserving endangered species. For example, dam operations in many rivers have been modified to meet instream flow requirements for endangered species (Richter and Thomas 2007). An increasing recognition of the importance of flow

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dynamics to aquatic ecosystems (Poff et al. 1997; Bunn and Arthington 2002; Nilsson et al. 2005; Dudgeon et al. 2006) has stimulated the development of methods to set environmental flow standards, and new approaches for modeling hydro-ecological responses to flow alterations (Richter et al. 1997; King and Brown 2006; Poff et al. 2010; Carlisle et al. 2010). However, the complex nature of water systems, which involve the interaction between economic agents, ecosystem processes, and potential regulatory restrictions, has hindered the development of effective water management policies. In order to address these difficulties, researchers have emphasized the need for multi-disciplinary collaboration (Nilsson et al. 2003) and proposed holistic management frameworks for assisting in the development of sustainable water policies (Richter et al. 2003; Richter 2010; Arthington et al. 2010).

While the need to protect flows for aquatic ecosystem preservation is now well established, the consequences of environmental standards on human water security also require consideration. Restrictions on water resources to maintain instream flows have the potential to adversely affect agricultural water security (Woodward and Romm 2001). Several studies have analyzed the economic impacts of meeting instream flow requirements and developed least-cost strategies to mitigate these costs (e.g., Paulsen and Wernstedt 1995; Turner and Perry 1997; Willis and Whittlesley 1998; Green and O’Connor 2001; Ward and Booker 2003; Briand et al. 2008). These studies have generally focused on relatively large rivers in the West, where large dams provide the primary source of water for both agricultural water security and augmented instream flows. However, many agricultural producers located in upland watersheds have no access to releases from large centrally operated dams, often relying instead on stored groundwater or small privately owned onsite reservoirs (i.e., a decentralized water management system) (Deitch et al. 2009a; Grantham et al. 2010; Newburn et al. 2011). Other economic studies have integrated spatially-explicit environmental models with economic models to evaluate the effectiveness of fund allocation strategies focused on improving water quality conditions for endangered fish. Wu et al. (2000) and Watanabe et al. (2006) show that the cost-effectiveness of targeting strategies for riparian restoration depend on the response of ecosystems to restoration efforts, which are influenced by nonlinear, ecological cumulative and threshold effects.

The consequences of instream flow protections on agricultural systems have rarely been assessed within watersheds where the method of water delivery is based on a decentralized water management regime. Although the environmental impact of a single water diversion is typically small, distributed networks of small-scale projects have the potential to cumulatively impair flow regimes and adversely affect aquatic species (McKay and King 2006; Spina et al. 2006; Deitch et al. 2009b). However, the heterogeneous distribution of water diversions across the stream network, and the potential for cumulative impacts, make the evaluation of hydrologic impacts particularly complicated. Similarly, the effects of flow regulations on water users are likely to vary across the landscape. Hence, to analyze the environmental and economic effects of flow regulations, a spatially-explicit approach is needed with sufficient spatial and temporal resolution to characterize seasonal, free-flowing streams and rivers, and the ability to model the impact on small storage ponds for agricultural use.

The goal of this chapter is to analyze the effect of recently adopted instream flow policies aimed at maintaining bypass-flows for adult salmonid migration within coastal California watersheds. As a result of the ESA-listing of salmon, California's State Water Resources Control Board (SWRCB) has become increasingly stringent in approving new appropriative water storage rights (Merenlender et al. 2008; Deitch et al. 2009b), and has established new rules that limit diversions by water users to instances when minimum streamflow thresholds are met. Diversion guidelines were first introduced in 2002, which required that flows exceed the February-media-flow threshold (CDFG/NMFS 2002), and subsequently revised in 2010, requiring that flows exceed a minimum-bypass-flow threshold necessary for maintaining sufficient water depths for adult salmon migration (SWRCB 2010). The recently adopted 2010 policy also provides a decision framework for the SWRCB to begin to approve new appropriative water storage rights, which may lead to the development of additional onsite storage than currently exists. I explore the effects on instream flows and agricultural water security of these recent diversion guidelines and of a prior low regulatory policy. Specifically, I quantify the tradeoffs between losses in ecologically-relevant flow metrics and impacts on agricultural water security for each policy, accounting for spatial and temporal variation in water availability.

In order to evaluate the effects of instream flow policies on streamflows and agricultural water security, as well as the prospect of new reservoir construction, spatially-explicit economic and hydrologic models are integrated. An economic model is developed to predict the location and amount of new onsite water storage. These new reservoirs are included in the landscape in addition to pre-existing reservoirs, which allows us to explore the potential impact of new reservoir development on instream flows and existing water right permit holders. A hydrologic model, developed in a Geographic Information System (GIS), is used to model the impacts on instream flows and agricultural water security. This model, which functions at ecologically relevant spatial (10 meters) and temporal (daily) scales, is able to estimate unimpaired streamflows, propagate the impact of diversions on flows throughout the drainage network, reproduce the various instream flow policies, verify whether streamflows exceed specific thresholds, calculate the amount of water stored within onsite reservoirs, and account for different rainfall years.

To my knowledge, this is the first study that evaluates the effect of instream flow policies on both salmon minimum-bypass-flows and agricultural water security within a decentralized management regime. Previous studies have focused on large river systems that rely on centrally operated dams to maintain instream flows; however, many agricultural producers in upland watersheds have no access to water releases from dams. Streams located in the upper parts of watersheds provide pathways for salmon migration and rearing habitat for juvenile salmon; as such, protecting instream flows in these areas is important. While some studies have explored the effect of diversions on small, free-flowing streams (Merenlender et al. 2008; Deitch et al. 2009a; Deitch et al. 2009b; Grantham et al. 2010), they focus primarily on reductions in streamflow rather than on the impact of diversions on meeting minimum-bypass-flow thresholds. This study provides greater insight into the effect of alternative instream flow policies on both bypass-flows and water security within a decentralized management regime.

4.2 Background

Coastal counties of California, such as Napa, Sonoma, and Mendocino counties, have experienced a large increase in their acreage of wine grape production since 1990 (Merenlender 2000). Due to the large increase in newly planted vineyards, approximately 50% of all water rights requests throughout all of California from 2000 to 2006 are located in watersheds within the north coast wine country. Federal listing under the ESA of coho salmon in 1995 and steelhead trout in 1997 heightened attention on the California State Water Resource Control Board (SWRCB) permitting process within these coastal counties. The study area is focused within the Maacama watershed (~180 km^2), located within Sonoma County, California. Sonama County is composed of Sonoma Valley and part of the Russian River basin (~2,000 km^2). The Maacama watershed is representative of the coastal California regions with recent vineyard expansion and increased regulatory stringency (Figure 4-1).

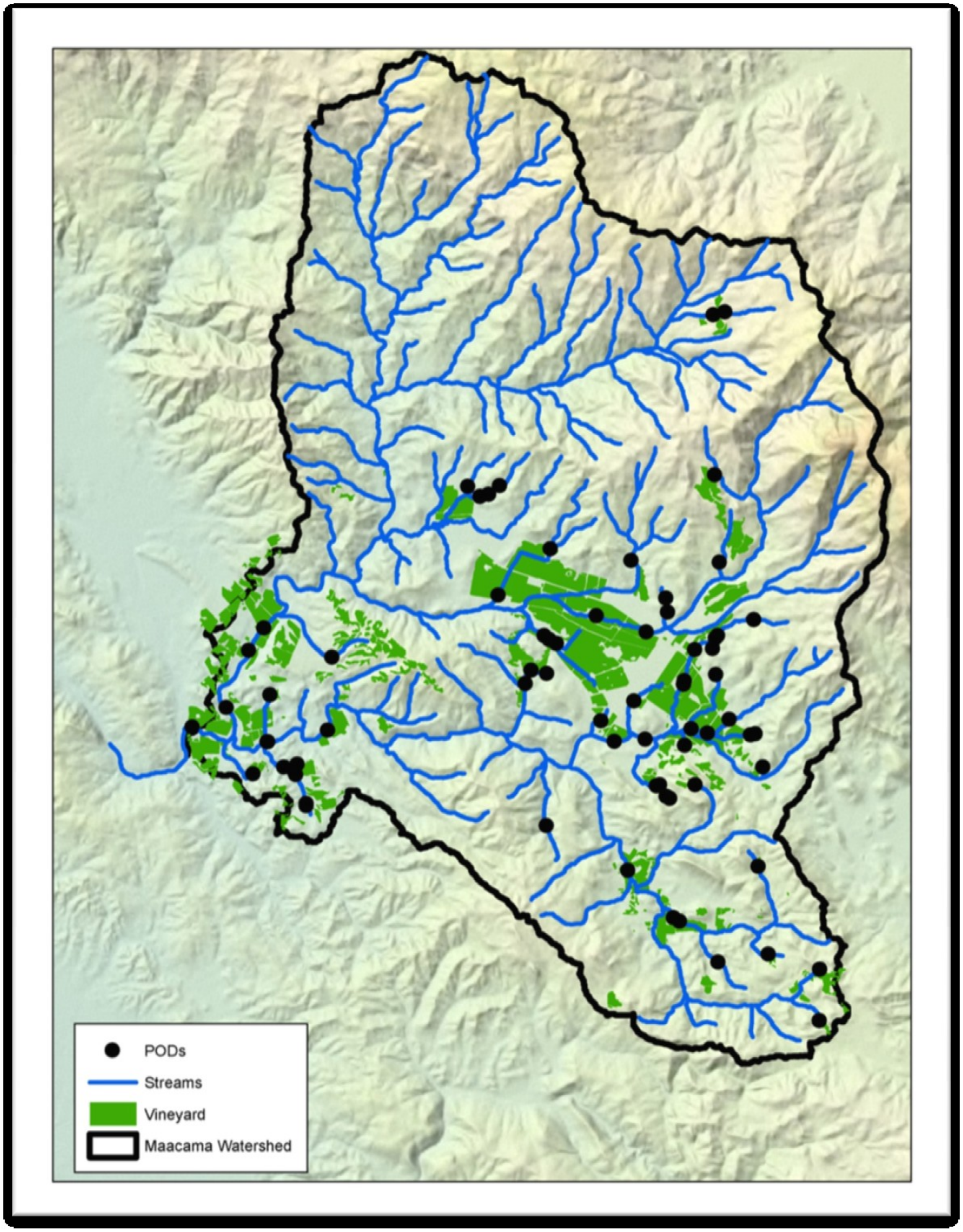

FIGURE 4-1. Map of the Maacama watershed, including the network of streams, vineyards, and points-of-diversion (PODs).

The Maacama watershed is characterized by a Mediterranean-like climate, with the majority of rainfall occurring in the winter months (November-March), followed by a dry period that can last six months and coincides with the grape growing season (April-September). Precipitation is also quite unpredictable, both seasonally and inter-annually, which results in a variable supply of surface water. The upland streams in coastal California watersheds experience a high degree of seasonal variation, in which winter peak flows may be several orders of magnitude greater than base flows during the summer drought period. This causes upland streamflows to peak during the winter months, and then slowly recede through the spring as they reach intermittency by the end of the dry summer.

Water users in the Maacama watershed consist mainly of spatially distributed agricultural producers. Most of the water demand by agriculture is attributed to vineyards, which have water demands for irrigation as well as heat and frost protection. The network of vineyards has a high abundance and density of water diversions and onsite reservoir storage throughout the watershed (Deitch et al. 2009a), as well as growing irrigation needs (Merenlender 2000; Merenlender et al. 2008). Ranchlands also have demand for onsite storage; however, this is mostly for stock water ponds, which have relatively minor amounts of consumptive water use compared to ponds used for vineyard water management.

The Russian River basin supports several salmonids listed under the ESA, which utilize the river and its tributaries for spawning and rearing, including streams within the Maacama watershed (Grantham et al., forthcoming). The longest stream in the Maacama watershed is Maacama Creek (11.7 km long), which is one of five main tributaries that empties into the Russian River. During the winter, vineyards divert flows from streams to fill reservoirs, which can potentially impact bypass flows for adult salmon migration. At the same time, because low-flow periods coincide with peak agricultural water demands, diversions and subsurface groundwater pumping during the summer can accelerate stream drying and potentially limit summer rearing habitat for juvenile fish (Deitch et al. 2009a). Consequently, in an effort to protect instream flows for endangered salmonids, the California SWRCB has denied most landowner requests since 1990 for new appropriative water rights to construct water storage ponds (Merenlender et al. 2008; Deitch et al. 2009b). The State's intention is to maintain winter flows to increase adult fish migration during the winter months (Merenlender et al. 2008).This has resulted in delays in processing new permits and led to a backlog of applications. As a consequence, the need for water storage is probably in excess of the current stock of water storage sites on the landscape.

The SWRCB has also imposed increasingly stringent regulations on winter diversions to protect adult fish migration. The diversion guidelines require that a certain flow threshold be exceeded at the landowner's point-of-diversion (POD) (Figure 4-1) before water can be extracted from a stream to fill an onsite reservoir. Prior to the development of these diversion thresholds, individuals with appropriative water rights were effectively unregulated and could capture all streamflow at a POD to fill a reservoir. I refer to the absence of a diversion-guideline as the unregulated policy. Diversion-guidelines were first implemented in 2002, and were later revised in 2010.

Both of these guidelines restrict diversions to take place between Dec 15th to Mar 31st (referred to as the diversion period or season). The 2002 guidelines (CDFG/NMFS 2002), referred herein as the moderate policy, allow diversions when streamflows exceed the unimpaired February-median-flow (Q_{fmf}) threshold at the POD. Historically, the month of February experiences higher flows relative to other months, and thus provides a reference for setting flow thresholds for adult salmon migration. The more recent North-Coast Instream Flow Policy guidelines (SWRCB 2010), referred to as the strict policy, sets a higher diversion threshold than the moderate policy. This threshold is determined using a formula developed by the SWRCB, which defines the minimum-bypass-flow (Q_{mbf}) necessary for salmon to be able to migrate upstream (SWRCB 2010), thus allowing them the possibility to reach adequate spawning grounds (Merenlender et al. 2008).

Using data from a gauge within Maacama Creek located in the Maacama watershed, a hydrograph for a POD on a headwater stream is provided for a dry rainfall year (1981) and a moderate year (1975) covering the migration period (Oct 1st – April 30th) (Figure 4-2).[9] The hydrograph highlights the variability in flow for a small, upland stream within this area. The two flow thresholds (Q_{mbf} and Q_{fmf}) corresponding to this stream segment are provided in the graph. An important ecological metric is the number of days throughout the migration period that the minimum-bypass-flow threshold is met, such that salmon would be able to migrate upstream through during this period. I refer to

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[9] Table 4-4 provides a summary of the data for twenty precipitation years.

this ecological metric as the number of 'bypass-flow days,' i.e., the number of days that the minimum-bypass-flow threshold (Q_{mbf}) is met at a particular stream segment.

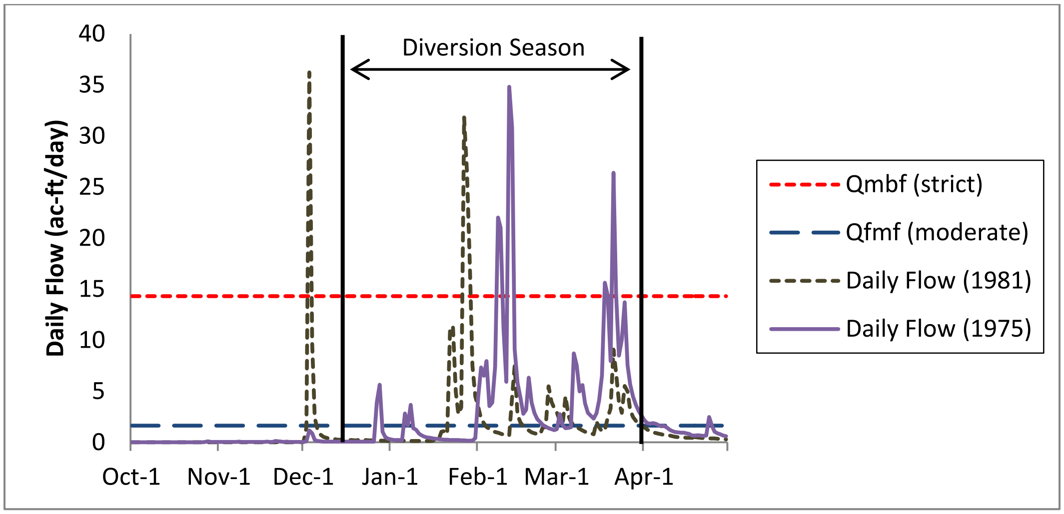

FIGURE 4-2. Hydrograph for a dry and moderate rainfall year (1981 and 1975, respectively) at a POD located on a headwater stream within the Maacama watershed. Included in the hydrograph are the Qfmf (moderate) and Qbmf (strict) diversion thresholds, where the diversion season is from Dec 15th – Mar 31st and the migration period is from Oct 1st - April 30th.

For both rainfall years, streamflow exceeds the Q_{fmf} threshold on more occasions than it does the Q_{mbf} threshold (Figure 4-2). Thus, for these two years, more days are available for diverting streamflow under the moderate policy than under the strict policy. When comparing the dry year (1981) to the moderate year (1975), both the Q_{fmf} and Q_{mbf} thresholds are met with greater frequency during the moderate year than during the dry year. Note also that for the dry year, the first major peak-flow that exceeds both thresholds occurs outside of the diversion period. As such, under both policies, none of this excess flow can be diverted for storage in a reservoir, and must be allowed to continue downstream. The unregulated policy, however, essentially imposes a diversion threshold of zero, and all available streamflow can be diverted throughout the entire migration period.

4.3 Methods

In this section, I introduce the methodological approach utilized in the analysis. I begin with the development of an economic model of landowner decisions for vineyard and reservoir development. This model is used to predict where reservoir construction would likely take places within the Maacama watershed in the absence of restrictions on water right storage permits, i.e., I predict where there exists a need for new onsite storage. These new reservoirs are then included in the hydrologic simulation analysis along with all existing reservoirs within the Maacma watershed.

I then proceed to describe the spatially-explicit hydrologic model used in the simulation analysis. This model simulates the cumulative impact of diversions on streamflows throughout the drainage network, and the diversion restrictions imposed on water users under the different instream flow polices. Finally, I define a set of seven policy scenarios within the hydrologic model, which are used to evaluate the impacts on instream flows and agricultural water security under the different policies and precipitation years.

4.3.1 Economic Model

As a result of the ESA listing of salmonids, the SWRCB placed increasing restrictions on the approval of appropriative water rights for onsite storage within the Maacama watershed and the surrounding northern-California counties. Newburn et al. (2011) analyzed the effects of the listing of salmonid species on landowner behavior in Sonoma County, and showed that vineyard development with onsite reservoirs became significantly less likely after the listing, particularly in upland watersheds with seasonal streams.[10] The recently approved diversion-guidelines, however, provide a framework for approving new water storage permits. Since new reservoir development can affect the water security of permitted water users and instream flows, it is important to identify the need for additional reservoirs throughout the watershed. An economic model is employed to predict the development of additional onsite storage by landowners within the Maacama watershed under the assumption that restrictions on reservoir construction are relaxed.

The economic model consists of a bivariate probit econometric model which characterizes the landowner's joint decision on vineyard and reservoir construction in the period prior to the ESA listing, during a time of lower regulatory stringency on reservoir development. I develop a cross-sectional version of the panel bivariate probit model developed in Newburn et al. (2011). The econometric model developed in Newburn et al. (2011) is a panel bivariate probit model used to study the effect of the

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[10] Refer to Newburn et al. (2011) for a theoretical economic model of the landowner choice between onsite surface water storage and groundwater for a recently developed vineyard without access to dam releases.

ESA listing on landowner behavior before and after the listing. However, since the focus is on modeling reservoir development decisions prior to severe restrictions (i.e., prior to the listing), I rely on a cross-sectional model rather than a panel model.

The econometric model assumes that a landowner makes two discrete choice decisions on land use (build vineyard or not) and water management (build reservoir or not) for the period prior to the ESA listing. The cross-sectional bivariate probit model defines two unobserved latent variables, y_{i1}^* and y_{i2}^*, used to represent the underlying value of vineyard and reservoir development, respectively, on property *i*

$$\begin{aligned} y_{i1}^* &= x_{i1}'\beta_{i1} + u_{i1} \\ y_{i2}^* &= x_{i2}'\beta_{i2} + u_{i2} , \end{aligned} \qquad [4.1]$$

where the parameter vectors are β_1 and β_2 and the error terms u_{i1} and u_{i2} follow a bivariate normal distribution with zero means, unit variances, and correlation ρ. Explanatory variables, x_{i1} and x_{i2}, represent the vectors of time-invariant property attributes, such as geology type, stream access, slope, microclimate and other physical variables, that affect the profitability of vineyard development and reservoir construction. All explanatory variables are included in both the vineyard and reservoir equations within the bivariate probit model. The bivariate probit model specifies the observed outcomes to be

$$\begin{aligned} y_{i1} &= 1 \quad \textit{if } y_{i1}^* > 0, \quad y_{i1} = 0 \quad \textit{otherwise}, \\ y_{i2} &= 1 \quad \textit{if } y_{i2}^* > 0, \quad y_{i2} = 0 \quad \textit{otherwise}. \end{aligned} \qquad [4.2]$$

The reduced-form model shown in [4.1] is based on a structural model that includes the latent variables y_{i1}^* and y_{i2}^* as right-hand side variables. Intuitively, this structural model means that, for a given property, the value of reservoir construction also depends on the value of vineyard development, and *vice versa*. Then, the structural model with simultaneity in the latent variables is reformulated as a reduced-form model based on the exogenous parcel attributes.

The econometric model given by [4.1] is used to estimate the probability of reservoir construction for vineyard landowners within the Maacama watershed that do not have onsite storage. The joint probability function for vineyard and reservoir development is

$$\Pr\left[y_{i1}=1, y_{i2}=1\right]=\Phi\left(x_{i1}'\beta_1, x_{i2}'\beta_2, \rho\right), \qquad [4.3]$$

where $y_{i1}=1$ and $y_{i2}=1$ imply that the landowner develops both vineyard and onsite storage, respectively, Φ is the cumulative density function for the standardized bivariate normal distribution with zero means, unit variances, and correlation ρ. The probability of reservoir construction, conditioned on vineyard development and landowner property attributes, is determined using the following conditional probability

$$\Pr\left[y_{i2}=1 \mid y_{i1}=1, x_{i1}, x_{i2}\right]=\frac{\Pr\left[y_{i1}=1, y_{i2}=1 \mid x_{i1}, x_{i2}\right]}{\Pr\left[y_{i1}=1 \mid x_{i1}, x_{i2}\right]}, \qquad [4.4]$$

where $y_{i2} = 1$ implies that the landowner builds a reservoir, and the probability of development is conditioned on the landowner having a vineyard, $y_{i1} = 1$, and the landowners characteristics x_{i1} and x_{i2}.

Estimation of the Econometric Model

The landowner property is the basic unit of analysis for the econometric model. The Sonoma County Tax Assessor's Office provides the complete map of parcel boundaries, and the assessor's database contains landowner name, current use, and other characteristics for each parcel. The landowner data covers a significant portion of Sonoma County, composed of Sonoma Valley and part of the Russian River basin. Agricultural landowners may own and jointly operate multiple parcels; for example, a vineyard is located on the one parcel and a reservoir on another parcel. Parcel boundaries are combined to create the landowner "property" where adjacent parcels have the same landowner name. Most landowners only have a single parcel for their property. Those properties with limited agricultural use are screened out based on the following criteria: public lands, cities and municipal sewer service areas, assessor codes for non-agricultural land uses (e.g., commercial, industrial, residential), and properties less than 20 acres.

High-resolution aerial photographs for 1973, 1993, and 2006 are used to map and digitize all vineyards and reservoirs in Sonoma Valley and the Russian River basin within a geographical information system (GIS). The photographs in 1993 provide a view of the landscape for the crucial year immediately prior to the species listing of coho salmon in the study area.[11] Consequently, all vineyards and reservoirs established during the 1973-1993 period occur before restrictions on reservoir construction became more severe. The dataset for the 1973-1993 period thus provides landowner development decisions prior to the moratorium on reservoir construction. Meanwhile, vineyards and reservoirs established during the 1993-2006 period occur largely after severe restrictions on reservoir construction were in effect. Combining the vineyard and reservoir GIS datasets with the parcel map, for each landowner property the period of establishment for vineyards and reservoirs is determined, as well as the amount of vineyard acreage and reservoir capacity.[12] Vineyards built without a reservoir are assumed implicitly to rely on groundwater pumping or summer diversions.

The data used to estimate the econometric model [4.1] consists of all landowner reservoir and vineyard construction decisions made in Sonoma County during the period from 1973-1993, prior to severe restrictions on reservoir construction. The dataset is generated as follows. First, I determine the set of undeveloped properties in 1973, i.e., properties that had neither a vineyard nor a reservoir. There are 3,561 properties in Sonoma County that were undeveloped in 1973.Then, for this set of undeveloped properties, the landowner makes a land use decision (develop vineyard or not) and a water management decision (build a reservoir or not) during the period 1973-1993. For

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[11] The coho salmon listing occurred in 1995, and the aerial photos in 1993 were the best available high-resolution imagery data prior to listing within region. According to Sonoma County Agricultural Crop Reports, over 94 percent of the vineyard acreage planted in the period 1993-2006 occurred after the coho listing in 1995.
[12] Reservoir capacity is estimated using a previously determined regression function that relates reservoir volume to reservoir surface area. The surface area of each digitized reservoir is calculated using a tool within GIS.

these properties, I provide the frequency of the four possible development decisions made by landowners during the time period 1973-1993 (Table 4-1). Of the 3,561 properties, there were 85 vineyards built with reservoir and 309 vineyards without reservoirs. Additionally, there were 180 properties with only reservoirs, which are primarily on ranchland for stock watering. There were also 2,987 properties with no development of any kind during this period.

TABLE 4-1. Vineyard and Reservoir Development Outcomes during the 1973-1993 Period.

	Reservoir	No reservoir	Total
Vineyard	85	309	394
No vineyard	180	2987	3167
Total	265	3296	3561

Explanatory variables in the economic model are site characteristics extracted within the GIS for each of the 3,561 landowner properties, such as: geology, riparian access, slope, microclimate, and other variables. Some attributes are explanatory variables that are expected primarily to affect conversion costs or returns for the vineyard development decision. Some variables are expected mainly to affect the landowner's costs or value of water security from building onsite storage, relative to relying only on groundwater or summer diversions.

The following set of landowner property attributes are used in the economic model. Growing degree-days, averaged over the April to October vineyard growing season, serves as a proxy for microclimate.[13] A warmer microclimate may be expected to increase the likelihood for a landowner to build a reservoir to meet higher water demand. Average slope (percent) and elevation (meters) are calculated for each property. Because steeper slopes raise the vineyard establishment costs and lower grape yields, vineyard development is expected to be more likely on areas with lower slope. Reservoirs are also more expensive to build in steep areas. Elevation is used to represent the property location relative to the valley floor. A dummy variable is used to represent whether a given property is situated within the 100-year floodplain. Vineyards and reservoir construction are not restricted in floodplain areas, but they are at greater risk for damage. The distance in kilometers from each property centroid to the nearest major highway is calculated. This variable represents a vineyard landowner's access to markets and population centers within Sonoma County, because all cities are located along these transportation corridors.

Physical variables that represent the landowner's access to adequate ground and surface water supplies during the summer growing season (i.e., water supplies other than onsite storage) are expected to affect development decisions. Groundwater potential is based on the geology type because alluvial areas have much higher groundwater yields than areas with other geology types. Dummy variables are created for the four main geologic types (in order from highest to lowest expected groundwater yield): young alluvium, old alluvium, volcanic, and Franciscan. Young alluvium serves as the baseline

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[13] Temperature data were taken from the Parameter-elevation Regressions on Independent Slopes Model (PRISM), which is created by the Spatial Climate Analysis Service at Oregon State University. Growing degree-days were averaged for each two kilometer grid cell in the region over the growing season (April 1st to September 30th).

geologic type in the regression model. Because the other three geology types have a lower expected groundwater yield, landowners in these water insecure areas are expected to be more likely to build an onsite reservoir.

Riparian access influences the landowner's access to surface water supplies. The State of California maintains riparian rights for those landowners adjacent to rivers and streams. A dummy variable called "mainstem" is created to indicate whether the landowner property is adjacent to either the Russian River mainstem or Dry Creek River, where two large-scale dams are required to release water to maintain stream flows within these rivers. However, the vast majority of landowners are located in smaller upland watersheds outside the influence of large-scale dams. Another dummy variable called "seasonal stream" is used to represent whether a seasonal stream runs through or is adjacent to the landowner property. Riparian access to seasonal streams provides an opportunity to store water during winter peak flows. Lastly, a third dummy variable represents landowners who do not have riparian access to either the mainstem river or seasonal stream (serving as the baseline type). Landowners with riparian access to seasonal streams are expected to have the highest likelihood of reservoir construction because they are water insecure during the summer growing season, but have stream access to fill and store water.

The estimation results for the cross-sectional bivariate probit model [4.1] are provided in Table B-1 in Appendix B. Results suggest that growing degree-days and access to the mainstream increase the likelihood of vineyard construction, while increasing slope decreases the likelihood of construction. With regard to reservoir construction, landowners with access to seasonal streams and lower groundwater yields are significantly more likely to build onsite storage, while increasing slope decreases the likelihood of construction. Elevation also increases the likelihood of reservoir construction. Refer to Newburn et al. (2011) for a more detailed discussion on the effect of the explanatory variables on vineyard and reservoir construction.

Prediction of New Reservoirs

In this section, I predict the reservoir development that would have potentially occurred within the Maacama watershed in the absence of the SWRCB limitations on reservoir construction. I then combine the predicted new reservoirs with the set of reservoirs existing in the 2006 Maacama landscape to obtain a complete map of reservoirs across the watershed. I use the estimated parameters in Table B-1, in Appendix B, and the conditional probability formulation given by [4.4] to estimate the probability of reservoir development for a subset of properties within the Maacama watershed. The subset of properties consists of those landowners who had vineyard and no onsite reservoir by 2006 (i.e., by the end of the period, 1993-2006, after the ESA listing). Onsite storage development by landowners with no vineyard is mostly for stock water ponds, which have relatively minor amounts of consumptive water use compared to ponds for vineyard water management. Consequently, I focus only on landowners with more than one acre of vineyard. Note that the time period after the ESA listing (1993-2006) is shorter than the time period prior to the listing (1973-1993). As such, I adjust the conditional probabilities obtained from [4.4] to account for this shorter time span.[14]

There are 217 properties within the Maacama watershed that are greater than or equal to 20 acres in size. Of these, 59 had more than one acre of vineyard and no onsite reservoir by 2006. For each of these properties, I generate a single random number and, if the number drawn exceeds the estimated conditional probability of reservoir development, I predict that a reservoir will be developed on their property.[15] This random simulation represents one realistic expansion of reservoir development across the Maacama landscape. Although this represents only one possible development scenario, using many reservoir development scenarios was not feasible given the computational demands of conducting the simulations within the hydrologic model. Nonetheless, several development scenarios were evaluated, and the low concentration of new reservoirs across the landscape suggested that the main findings were unlikely to be impacted by the specific random development scenario.

The economic model predicts that, out of the 59 landowners with vineyard and no reservoir storage by 2006, 9 landowners would construct a reservoir. This represents an increase in reservoirs of approximately 15%. New reservoirs are placed at the point within the landowner property with the largest catchment area. The storage capacity for

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[14] The conditional probability of reservoir construction for the period 1993-2006 under the assumption of no limitations on reservoir construction is given by

$$\Pr_{93-06}\left[y_{i2}=1 \mid y_{i1}=1, x_{i1}, x_{i2}\right]=1-\left[\left(1-\Pr\left[y_{i2}=1 \mid y_{i1}=1, x_{i1}, x_{i2}\right]\right)^{(1/20)}\right]^{13},\text{ where}$$

$\Pr\left[y_{i2}=1 \mid y_{i1}=1, x_{i1}, x_{i2}\right]$ is given by [4.4].

[15] The conditional probability of reservoir construction is assumed to be independent across landowner properties.

each new reservoir, in acre-feet, is set equal to two-thirds of the vineyard acreage located on the landowner property.[16]

Mapping of Reservoirs

The storage capacity and the location of the POD for each reservoir, both new and existing, within the Maacama watershed must be specified before the hydrologic model can predict the impact of diversions across the drainage network. The POD for each reservoir in the Maacama watershed is determined as follows: 1) if a stream exists on the landowner property, the POD for the reservoir is assumed to be the stream point within the property closest to the reservoir[17]; 2) if no stream exists on the property, then the POD for the reservoir is assumed to be located at the downstream most point of the reservoir. For the simulations conducted, there are a total of 70 PODs within the Maacama watershed: 61 are pre-existing, and 9 are predicted by the economic model. Table 4-2 provides a summary for new and all (i.e., both new and pre-existing) PODs within the Maacama watershed by catchment area and reservoir capacity. The total storage capacity for all reservoirs within the Maacama watershed is 1,896 ac-ft.

For each reservoir, both new and existing, it is necessary to also identify whether the reservoir is associated with a permitted or unpermitted diversion. This allows us to explore the potential impacts of diversions by unpermitted water users on permitted water users. Landowner compliance on reservoir storage for all existing reservoirs

[16] The approximation that grape production requires two-thirds of an acre-foot of water for irrigation per acre of vineyard was obtained from Lewis et al. (2008).
[17] Streams on properties were identified using the National Hydrography Dataset (NHD) provided by the U.S. Geological Survey.

within the Maacama watershed is assessed from the SWRCB water right permit data. Permit locations were overlaid with the landowner property map to determine which landowners with existing reservoirs in the Maacama watershed have an approved appropriative right for onsite storage.[18] Landowners with existing reservoirs that have an approved appropriative right for onsite storage are classified as having permitted diversions. Those landowners with existing reservoirs that do not have an approved appropriative right are classified as having unpermitted diversions. All reservoirs predicted by the economic model are considered as unpermitted diversions as well since they have yet to obtain an approved appropriative right for storage. Of the 70 PODs in the Maacama watershed, 35 are associated with permitted diversions, and 35 with unpermitted diversions, including the 9 reservoirs predicted by the economic model, all of which are considered unpermitted diversions (Figure 4-3). A summary of the 70 PODs by catchment area and reservoir capacity, broken down by permitted and unpermitted diversions, is provided in Table 4-3.

[18] Registered riparian rights and appropriative rights for diversion were similarly assessed for each landowner; however, these water rights do not allow for storage during the winter rainy season for later use during the growing season.

TABLE 4-2. Number of New and All PODs by Catchment Area and Reservoir Capacity.

New	Catchment Area (miles squared)				
Reservoir Capacity (ac-ft)	< 1	1-10	>=10	**Grand Total**	**Total Capacity (ac-ft)**
<10	3	0	0	3	17.33
10-50	3	1	1	5	77.33
>50	0	1	0	1	88.00
Grand Total	6	2	1	9	182.66
Total Capacity (ac-ft)	60.00	103.33	19.33	182.66	

All	Catchment Area (miles squared)				
Reservoir Capacity (ac-ft)	< 1	1-10	>=10	**Grand Total**	**Total Capacity (ac-ft)**
<10	28	4	3	35	189.49
10-50	17	4	3	24	591.64
>50	8	2	1	11	1114.45
Grand Total	53	10	7	70	1895.58
Total Capacity (ac-ft)	1441.80	328.34	125.44	1895.58	

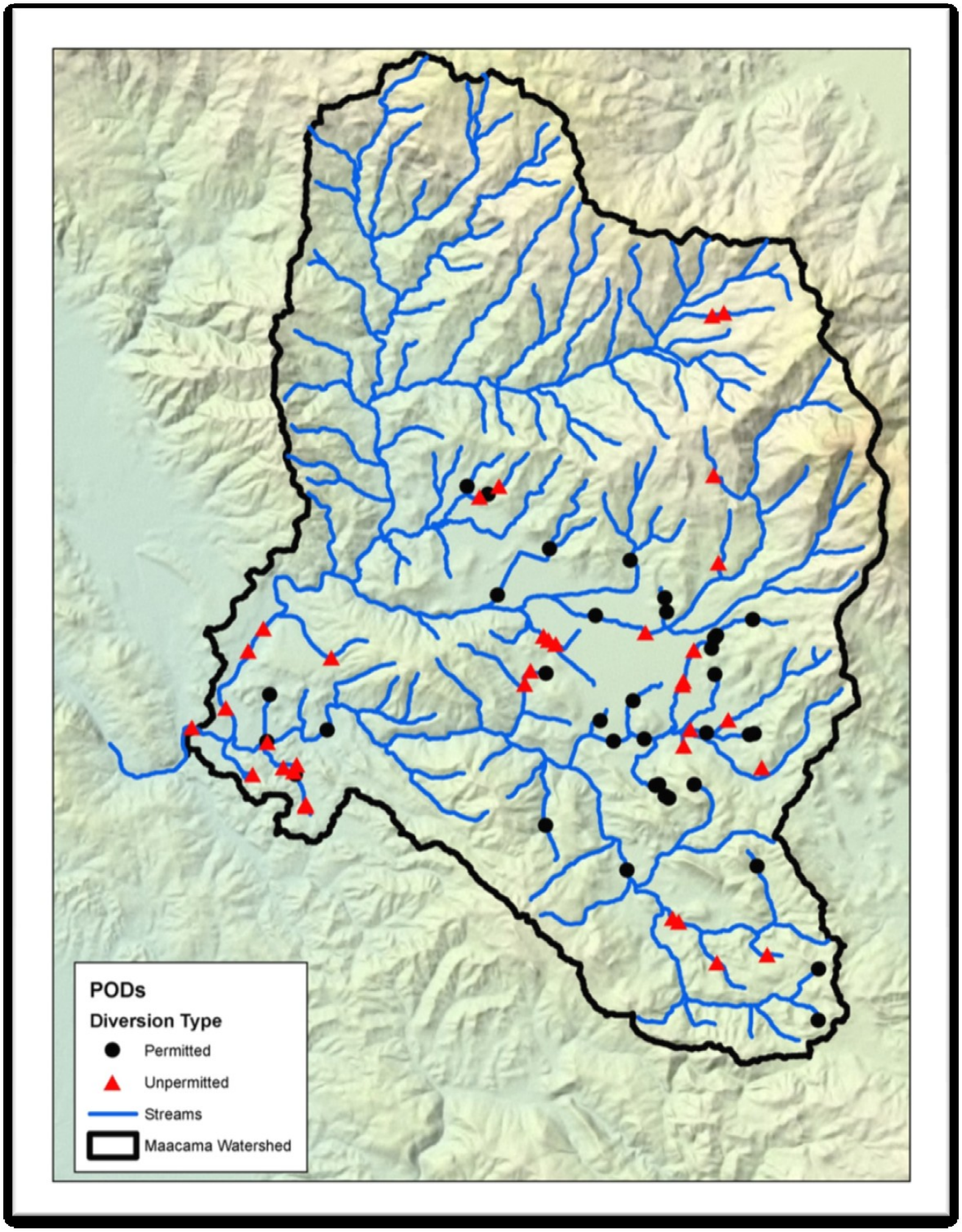

FIGURE 4-3. Map of permitted and unpermitted diversions in the Maacama watershed.

TABLE 4-3. Number of PODs (Both Permitted and Unpermitted) by Catchment Area and Reservoir Capacity.

Permitted	**Catchment Area (miles squared)**				
Reservoir Capacity (ac-ft)	< 1	1-10	>=10	**Grand Total**	**Total Capacity (ac-ft)**
<10	14	2	1	17	75.31
10-50	6	3	0	9	244.59
>50	7	1	1	9	931.54
Grand Total	27	6	2	35	1251.44
Total Capacity (ac-ft)	974.95	213.92	62.57	1251.44	
Unpermitted	**Catchment Area (miles squared)**				
Reservoir Capacity (ac-ft)	< 1	1-10	>=10	**Grand Total**	**Total Capacity (ac-ft)**
<10	14	2	2	18	114.18
10-50	11	1	3	15	347.05
>50	1	1	0	2	182.91
Grand Total	26	4	5	35	644.14
Total Capacity (ac-ft)	466.85	114.42	62.87	644.14	

4.3.2 Hydrologic Model

The hydrologic model is a spatially-explicit watershed model that allows for the analysis of cumulative impacts on streamflows at ecologically relevant scales resulting from diversions by a distributed network of water users (Merenlender et al. 2008; Grantham et al., 2010). Specifically, it is designed to propagate streamflow impacts at a daily time scale downstream through the drainage network associated with diversions used to fill onsite reservoirs. The model is capable of running under various precipitation years, estimating streamflow impairments, determining whether daily expected streamflows meet specific flow thresholds, calculating the amount of water stored within reservoirs, and imposing different regulations on diversions.

In order to model stream discharge at all points throughout the drainage network, inputs to the hydrologic model consist of a 10-meter digital elevation model (DEM) for the Maacama watershed, which is used to construct the drainage network of streams within the watershed. Daily streamflow within the drainage network is estimated using twenty years of gauge data (1961-1980) from a historical USGS gauge on Maacama Creek. These data are scaled by catchment area and precipitation ratios in order to estimate flows within all streams segments in the drainage network for the twenty precipitation years. The twenty years of USGS gauge data for Maacama Creek are ranked based on the total recorded annual flow (Table 4-4).

An end of anadromy (EOA) GIS layer is used by the hydrologic model to establish the location on a stream considered physically inaccessible to adult salmon migration, even under unimpaired conditions. More specifically, the SWRCB defines an EOA point as a point on a stream segement where there exists a gradient that is of a continuous longitudinal slope of 12%, or greater, over a distance of 330 feet (SWRCB 2010). Because salmon cannot migrate to places above EOA points, the number of days that the salmon minimum-bypass-flow threshold (Q_{mbf}) is met on a stream (i.e., the number of bypass-flow days) is only ecologically relevant for streams segments located below the EOA. The hydrologic model thus segregates stream segments that are above and below the EOA.

TABLE 4-4. Precipitation Years (1961-1981) used for Hydrologic Simulations for Maacama Creek Ranked by Total Annual Flow.

Rank	Year	Rank	Annual Flow (ac-ft)
Driest	1977	1	1,642
	1976	2	8,548
	1972	3	18,151
	1964	4	23,485
Dry	1981	5	34,783
	1979	6	38,220
	1968	7	41,694
	1962	8	47,795
	1966	9	50,365
Moderate	1975	10	50,365
	1971	11	60,102
	1980	12	70,898
	1973	13	73,000
	1963	14	74,563
Wet	1965	15	84,936
	1967	16	90,953
	1978	17	95,050
	1969	18	95,861
	1970	19	97,990
Wettest	1974	20	121,131
Mean			58,977
Std. Dev.			32,984

The hydrologic model calculates the bypass-flow threshold, Q_{mbf}, for every stream segment and POD in the drainage network as follows (SWRCB 2010):

i. If the catchment area upstream of the stream segment, or POD, is 1 square mile or smaller, then the bypass-flow threshold is given by

$$Q_{mbf} = 9.0Q_m \; . \qquad [4.5]$$

ii. If the catchment area upstream of the stream segment, or POD, is greater than 1 square mile, then the bypass-flow threshold is given by

$$Q_{mbf} = 8.8 Q_m \left(DA \right)^{-0.47} , \qquad [4.6]$$

where Q_{mbf} equals the minimum-bypass-flow in cubic-feet-per-second needed for salmon bypass, Q_m equals the mean annual unimpaired flow in cubic-feet-per-second, and *DA* equals the watershed drainage area, or catchment area, in square miles upstream of the stream segment or POD. The hydrologic model compares the estimated mean daily flow at a stream segment or POD to the corresponding bypass-flow threshold to determine whether the bypass-flow threshold is met.

The modeling of diversion guidelines by the hydrologic model requires that the following be defined for each reservoir: its storage capacity, the location of its POD, and whether the diversion is permitted or unpermitted. With all of the above specified, it is then possible to impose any of the three diversion policies (unregulated, moderate, and strict) at each POD. For the unregulated policy, all water users are not subject to diversion thresholds, and can capture all streamflow at a POD during any day throughout the migration period, from Oct 1st to April 30th. Once a reservoir reaches its capacity, all flow at the POD is assumed to continue downstream, and no further diversions are simulated.

Under the moderate policy, I assume that flows must exceed the February-median-flow (Q_{fmf}) threshold at the POD before diversions are allowed. The hydrologic model calculates the Q_{fmf} for every stream at each POD using the daily February flows for twenty years of data from the historical USGS gauge on Maacama Creek (Table 4-4), which are scaled accordingly based on catchment area. Under this policy, diversions are only on days within the diversion period, from Dec 15th to Mar 31st.

For the strict policy, the model distinguishes between PODs located below and above the EOA. If a POD is located above an EOA point, then diversions are allowed if streamflow exceeds the February-median-flow threshold (i.e., the same threshold as under the moderate policy). However, if a POD is located below an EOA point, then flow must exceed the bypass-flow threshold (Q_{mbf}) at the POD. The bypass-flow threshold is larger than the February-median-flow threshold, and thus allows for less water extraction, and generally for fewer diversion days, than the moderate policy (Figure 4-2). The strict policy also caps the cumulative storage capacity of PODs above EOA at ten-percent of the seasonal flow volume at the EOA. As with the moderate policy, diversions under the strict policy are limited to be within the diversion period.

4.3.3 Policy Scenarios

The hydrologic model is used to study the impact of diversions, instream flow policies, and precipitation years on instream flows and agricultural water security. I define a set of seven policy scenarios within the hydrologic model, which allow for a comparison of impacts on instream flows and agricultural water security across different policies and precipitation years. The seven policy scenarios are defined in Table 4-5.

TABLE 4-5. Scenario Numbers and Descriptions of Policy Scenarios for Permitted and Unpermitted Diversions.

		Policy for Permitted Diversions			
		none allowed	strict	moderate	unregulated
Policy for Unpermitted Diversions	none allowed	1	2	3	4
	strict		5		
	moderate			6	
	unregulated				7

Scenario 1 is the baseline scenario, where streamflows are unimpaired by diversions (i.e., no diversions are allowed). Scenarios 2-4 impose the strict, moderate, and unregulated policies, respectively, on permitted diversions only (i.e., no diversions by unpermitted water users are allowed). Scenarios 5-7 impose the strict, moderate, and unregulated policies, respectively, on both permitted and unpermitted diversions. All seven scenarios in Table 4-5 are run in the hydrologic model for each of the twenty precipitation years, resulting in twenty simulations per policy scenario and a total of 140 simulations.

As a result of diversions throughout the drainage network, total water flow through stream segments that are downstream of PODs, or that contain a POD, will be potentially reduced. To measure and compare the impact of diversions on instream flows across the different policy scenarios, two ecological metrics are used. The first metric is the number of days that the salmon bypass-flow threshold (Q_{mbf}) is met at each stream segment (i.e., the number of bypass-flow days). The hydrologic model compares the estimated mean daily flow predicted for each stream segment under a policy scenario to the corresponding bypass-flow threshold, and counts the number of days that the threshold is met at each stream segment throughout the migration period. The unimpaired scenario (scenario 1) provides the number of bypass-flow days for every stream segment in the absence of diversions. Thus, the impacts of diversions under each policy scenario can be measured by the observed losses of bypass-flow days relative to the unimpaired scenario.

The second ecological metric I define for every stream segment in the drainage network is the percent loss in streamflow, or equivalently, the percent of flow that is diverted from a stream segment. For each policy scenario, the percent loss in streamflow for a stream segment equals the percent reduction in the total amount of flow relative to the unimpaired scenario. The reduction is measured relative to the total amount of flow throughout the migration period.

I also measure the water security of a landowner based on the percent of the reservoir capacity that is filled by diversions at the end of the high flow season on April 30th. This metric serves as a proxy for water security and allows us to compare these values across different landowners.

4.4 Results

In this section, I present the results of the simulation analysis. The overall focus of the analysis is to understand the relative influence of instream policies on bypass-flows and agricultural water security within a decentralized water management regime. I begin by analyzing the natural spatial and temporal variation in the number of bypass-flow days in the absence of diversions. I then explore the impact of agricultural diversions on streamflows by measuring the losses in bypass-flow days under the various diversion guidelines. This allows us to evaluate how much each policy reduces the impact of diversions on bypass-flow days. I evaluate these impacts both spatially and temporally to understand the heterogeneity of diversion impacts on streams. The effect of diversions on flow is also measured in terms of the percent reduction in streamflow, which provides additional information on the impact of diversions, and regulations, on instream flows.

Afterwards, I turn my attention to agricultural water security. I evaluate the agricultural water security of water users, both spatially and temporally, for each policy by simulating the effect that each policy has on the percent of each farmer's storage that is filled I also analyze how percent storage varies by reservoir size to determine if impacts vary by reservoir capacity. The cumulative impacts of unpermitted diversions on downstream permitted water users are also analyzed. I end the section by quantifying the tradeoffs between losses in bypass-flow days and agricultural water security under the different instream flow polices.

4.4.1 Instream Flow Impacts

In Table 4-6, the average number of bypass-flow days for impacted stream segments (i.e. streams segments downstream of a POD or containing a POD) that are below EOA, by precipitation year, are provided for the unimpaired (1), strict (5), moderate (6), and unregulated (7) policy scenarios.[19] The average number of bypass-flow days across all years is provided at the bottom of the table. Recall that a bypass-flow day occurs when flow within a stream segment exceeds the Q_{mbf} threshold (Figure 4-2). The unimpaired scenario (1) provides the upper bound on the average number of bypass-flow days across all streams segments (23.37 days) because diversions are not allowed for this scenario. The strict scenario (5) follows closely behind with 23.35 days, the unregulated scenario (7) is next with 22.82 days, and the moderate scenario (6) has the smallest average number of bypass-flow days (22.79 days).

The number of bypass-flow days for streams segments across the Maacama watershed are mapped for the dry year, 1981 (ranked 5), to illustrate the spatial heterogeneity of bypass-flow days (Figure 4-4). From the map, the stream segments in the upper reaches of the watershed are observed to have less bypass-flow days than those in the lower reaches. This is due to the fact that the bypass-flow threshold does not increase proportionally with the stream drainage area (refer to equation [4.6]); as such, flows downstream with larger drainage areas are more likely to meet the bypass-flow threshold than those further upstream. For example, streams in the upper parts of the watershed have between 4-6 bypass-flow days (highlighted in red) compared to streams further below with bypass-flow days up to 43 days (Figure 4-4).

[19] All averages for bypass-flow days and percent loss in streamflow are weighted by the length of the stream segments, i.e., weighted averages are calculated based on stream segment lengths.

TABLE 4-6. Average Bypass-Flow Days for Impacted Streams below EOA by Precipitation Year for the Unimpaired (1), Strict (5), Moderate (6), and Unregulated (7) Policy Scenarios.

		Average Bypass-Flow Days per Scenario			
Rank	**Year**	**Unimpaired**	**Strict**	**Moderate**	**Unregulated**
Driest	1977	0.000	0.000	0.000	0.000
	1976	1.058	1.048	0.913	0.979
	1972	4.559	4.558	4.082	4.023
	1964	7.912	7.902	7.305	7.233
Dry	1981	13.165	13.158	12.600	12.813
	1979	16.146	16.136	15.523	15.675
	1968	17.829	17.781	17.195	17.232
	1962	19.601	19.555	18.915	19.135
	1966	18.448	18.438	17.845	17.586
Moderate	1975	22.798	22.755	21.999	22.163
	1971	25.773	25.756	24.995	25.098
	1980	29.317	29.219	28.575	28.725
	1973	31.379	31.367	30.881	30.938
	1963	28.044	28.040	27.541	27.567
Wet	1965	28.834	28.829	28.497	28.324
	1967	39.402	39.398	38.880	38.594
	1978	36.590	36.578	35.781	36.101
	1969	40.302	40.288	39.472	39.503
	1970	34.626	34.622	34.136	33.875
Wettest	1974	51.639	51.635	50.654	50.821
Average		23.371	23.353	22.789	22.819

The total number of bypass-flow days depends not only on total annual flow, but also on the timing, size and the number of peak-flows. For example, a water year that experiences a few large peak-flows followed by many low-flows can achieve less bypass-flow days than a water year that has many medium-sized flows. This occurs,

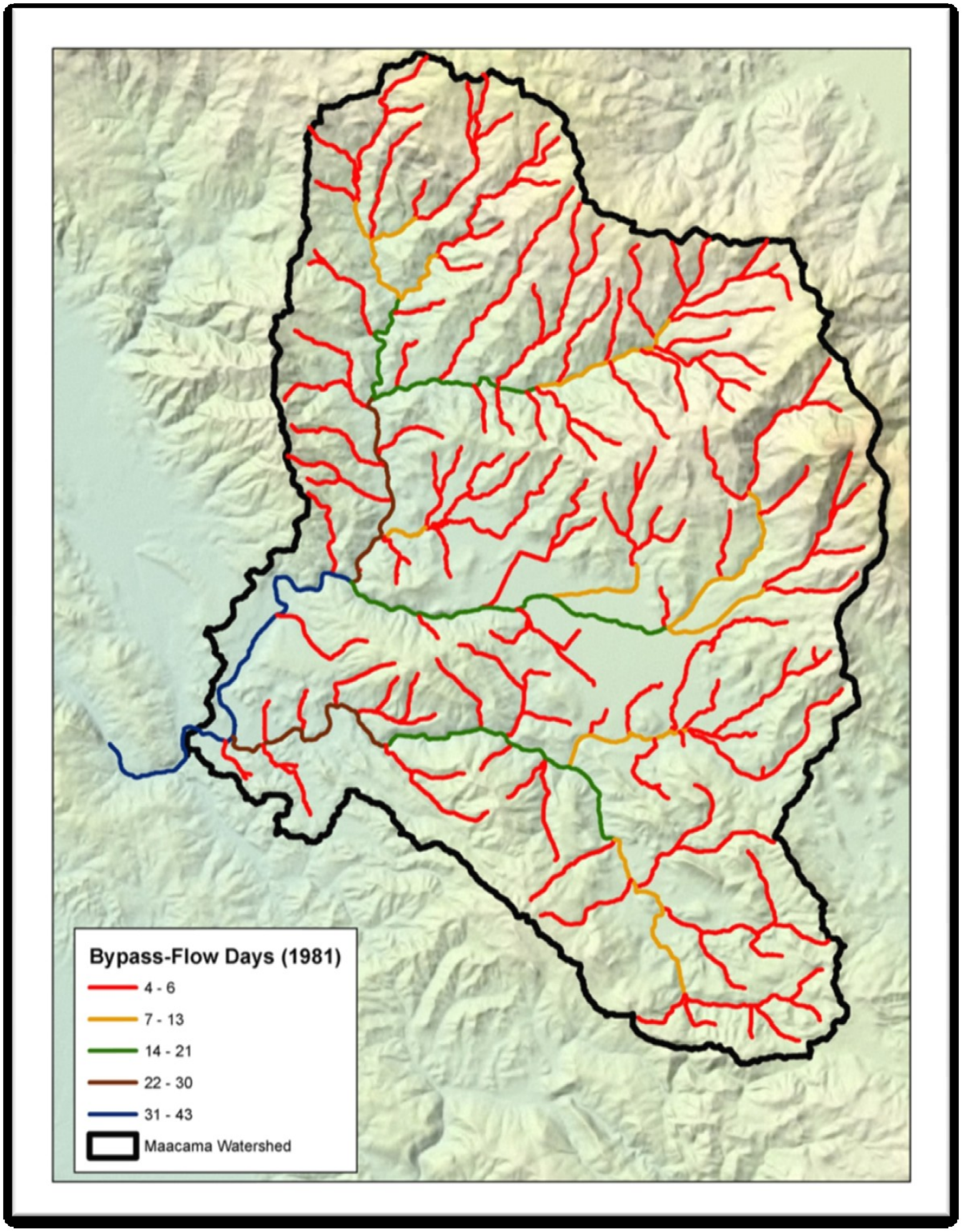

FIGURE 4-4. Map of bypass-flow days for the unimpaired scenario (1) for all streams segments in the Maacama watershed for the dry year 1981 (ranked 5).

for instance, when precipitation follows the pattern of 1973 (ranked 13), which led to more unimpaired bypass-flow days on average (31.38 days) than the year 1965 (ranked 15) with 28.83 days, even though the year 1965 had a total annual flow greater than that of year 1973 (Table 4-6). When flows are really low, such as during the driest year 1977, the unimpaired number of bypass-flow days is zero because flows never exceed the Q_{mbf} threshold within any stream segment.

For each policy scenario and precipitation year, I calculate the loss in average bypass-flow days relative to the unimpaired scenario (1) to determine the ecological impact of diversions under each regulatory regime (Table 4-7). The largest loss occurs for the moderate policy, where average bypass-flow days decrease from 23.37 days to 22.79 days, for an average loss of 0.58 days (a decrease of 2.49%). The unregulated policy achieves a slightly smaller decrease relative to the unimpaired scenario of 2.36%, while the strict policy results in an average loss in bypass-flow days of only 0.08%. Overall, however, average losses in bypass-flows days for the alternative policy scenarios are not that different.

TABLE 4-7. Average Loss in Bypass-Flow Days Relative to the Unimpaired (1) Scenario for Impacted Streams below EOA by Precipitation Year for the Strict (5), Moderate (6), and Unregulated (7) Policy Scenarios.

		Average Loss in Bypass-Flow Days per Scenario		
Rank	**Year**	**Strict**	**Moderate**	**Unregulated**
Driest	1977	0.000 (N/A)	0.000 (N/A)	0.000 (N/A)
	1976	0.001 (0.93)	0.145 (13.67)	0.078 (7.41)
	1972	0.001 (0.02)	0.477 (10.46)	0.536 (11.80)
	1964	0.010 (0.12)	0.607 (7.67)	0.680 (8.59)
Dry	1981	0.006 (0.05)	0.565 (4.29)	0.351 (2.67)
	1979	0.011 (0.07)	0.623 (3.86)	0.471 (2.92)
	1968	0.048 (0.27)	0.635 (3.56)	0.597 (3.35)
	1962	0.046 (0.23)	0.686 (3.50)	0.465 (2.37)
	1966	0.009 (0.05)	0.602 (3.26)	0.861 (4.67)
Moderate	1975	0.043 (0.19)	0.798 (3.50)	0.634 (2.78)
	1971	0.017 (0.07)	0.778 (3.02)	0.675 (2.62)
	1980	0.098 (0.33)	0.742 (2.53)	0.592 (2.02)
	1973	0.012 (0.04)	0.498 (1.59)	0.441 (1.40)
	1963	0.004 (0.01)	0.503 (1.80)	0.477 (1.70)
Wet	1965	0.005 (0.02)	0.337 (1.17)	0.510 (1.77)
	1967	0.004 (0.01)	0.522 (1.33)	0.807 (2.05)
	1978	0.012 (0.03)	0.809 (2.21)	0.489 (1.34)
	1969	0.014 (0.04)	0.830 (2.06)	0.800 (1.99)
	1970	0.004 (0.01)	0.491 (1.42)	0.751 (2.17)
Wettest	1974	0.004 (0.01)	0.986 (1.91)	0.818 (1.58)
Average		0.018 (0.08)	0.582 (2.49)	0.552 (2.36)

Nonetheless, average losses in bypass-flow days are heterogeneous and vary by the catchment area of stream segments. Streams segments within smaller watersheds are the least likely to be impacted by diversions, but they experience the largest impacts (Table 4-8). For instance, under the unregulated policy, losses in bypass-flow days for impacted stream segments within the smallest watersheds experience the largest decrease of 8.95%, while streams in larger watersheds lose 1.73% and 1.32%, respectively. However, most streams segments within small catchment areas (<1 mile squared) are unaffected by diversions: 57.29 miles of the total 77.71 miles (73.7%) in the watershed are unaffected by diversions (Table 4-8). This is because stream segments with small catchment areas are in the upper reaches of the watershed, and are less likely to be downstream of a POD, or to contain a POD. Meanwhile, all stream segments with catchment areas greater than 10 square miles are impacted because they are either downstream of a POD or contain a POD. The largest percent decrease in bypass-flow days occurs under the moderate policy within small catchment areas, with a loss of 10.22%. On the other hand, the strict policy provides the greatest protection within small catchment areas, with a loss in bypass-flow days of only 0.49%.

To further illustrate the spatial pattern of losses in bypass-flow days, I map bypass-flow days lost within the Maacama watershed for the year 1981for the unregulated policy (7) (Figure 4-5). The map clearly shows that the majority of streams segments in the upper reaches of the watershed are unaffected by diversions because they are upstream of PODs. However, most stream segments with larger catchment areas are downstream of PODs or contain a POD. Maps for losses in bypass-flow days under the strict and moderate policy scenarios are provided as well (Appendix B, Figures B-2 and B-3, respectively).

TABLE 4-8. Average Bypass-Flow Days for the Unimpaired Scenario (1) and Loss in Bypass-Flow Days by Catchment Area for the Strict (5), Moderate (6), and Unregulated (7) Policy Scenarios Aggregated by Precipitation Years (only impacted streams below EOA).

	Catchment Area (miles squared)			
Scenario	**<1**	**1-10**	**>=10**	**Average**
Unimpaired (1)	8.569	19.104	44.315	23.371
Loss Strict (5)	0.042 (0.49)	0.007 (0.04)	0.005 (0.01)	0.018 (0.08)
Loss Moderate (7)	0.876 (10.22)	0.413 (2.16)	0.470 (1.06)	0.582 (2.49)
Loss Unregulated (9)	0.767 (8.95)	0.331 (1.73)	0.586 (1.32)	0.552 (2.36)
Stream Length	**<1**	**1-10**	**>=10**	**Total**
Length (miles) Impacted Streams	20.42 (26.28)	22.90 (54.36)	19.10 (100.00)	62.41 (44.93)
Length (miles) Unaffected Streams	57.29 (73.72)	19.22 (45.64)	0.00 (0.00)	76.51 (55.07)
Length (miles) All Streams	77.71	42.12	19.10	138.92

Note: Values in parenthesis equal the percent loss in average bypass-flow days relative to the unimpaired scenario, and the percentage of total stream length that is impacted and unaffected.

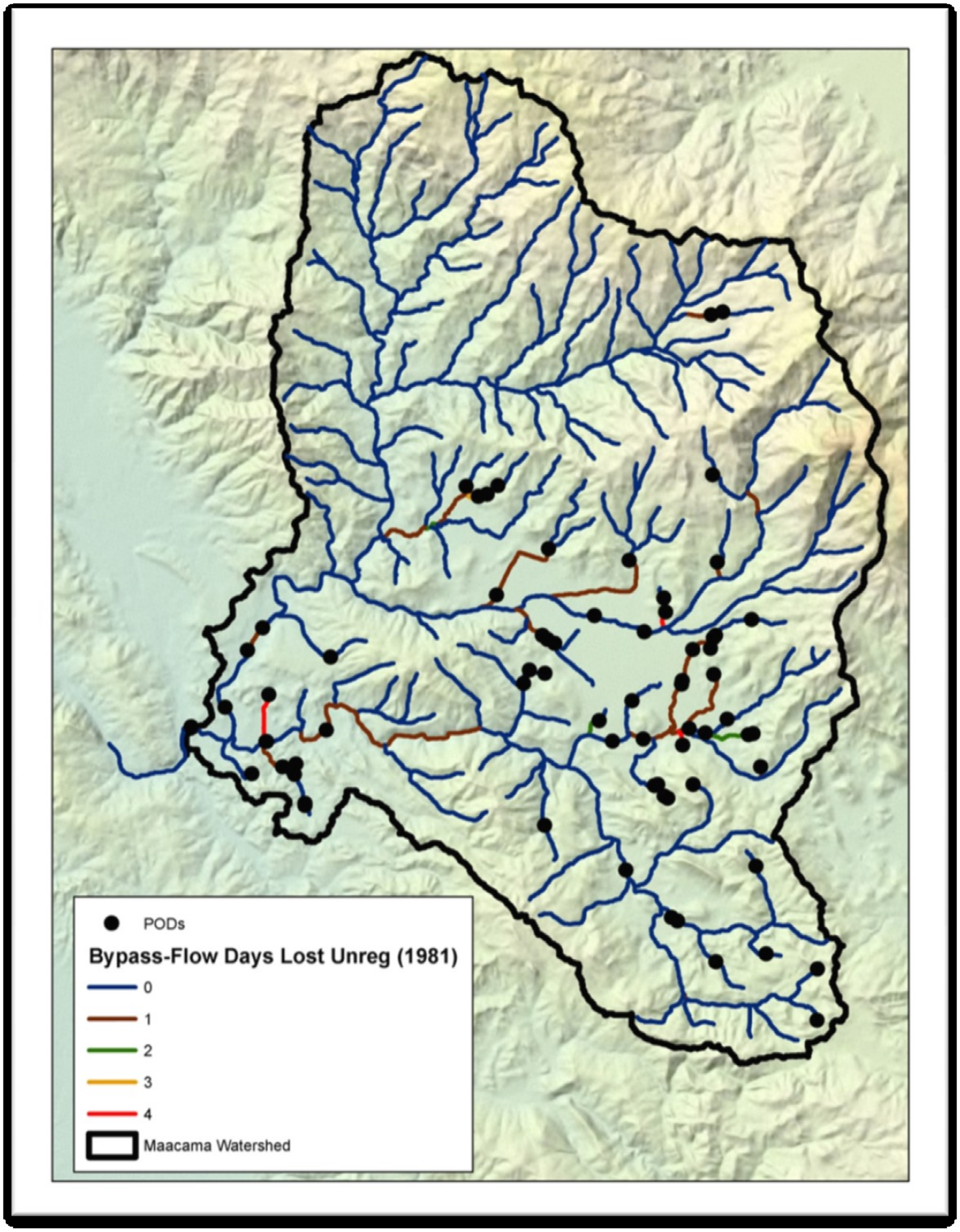

FIGURE 4-5. Map of losses in bypass-flow days under the unregulated policy scenario (7) for streams segments in the Maacama watershed for the dry year 1981 (ranked 5).

I plot the average percent loss in streamflow, for all impacted stream segments, across the twenty precipitation years, to better understand the impact of diversions on streamflows (Figure 4-6).[20] Losses in flow are relative to the entire amount of flow measured for the migration period between October 1st and April 30th. The largest differences in the amount of percent flow diverted across the various policy scenarios occur during the driest years (those years in the lowest quartile, ranked 1-4). For the driest year (1977), the moderate and strict policy scenarios lead to a zero percent loss in stream flow because the diversion thresholds are never met for any day during the migration period. However, for the unregulated policy, where diversions are not restricted, the average loss in streamflow for the driest year is close to 30% compared to the unimpaired scenario. The percent loss in flow for the moderate and unregulated policies converges in year 1979 (ranked 6), which implies that these two policies divert approximately the same amount of flow for most non-dry years (those above the first quartile). The strict policy, however, diverts less flow than the moderate and unregulated scenarios for most years except the wet years (the highest quartile years).

[20] The reduction in average percent loss in streamflow that is observed for the strict policy for year 1971 (ranked 11) in Figure 4-6 is due to the fact that the several large peak-flow events exceeding the Q_{mbf} threshold for this year occur prior to the diversion season (Dec 15th – Mar 31st).

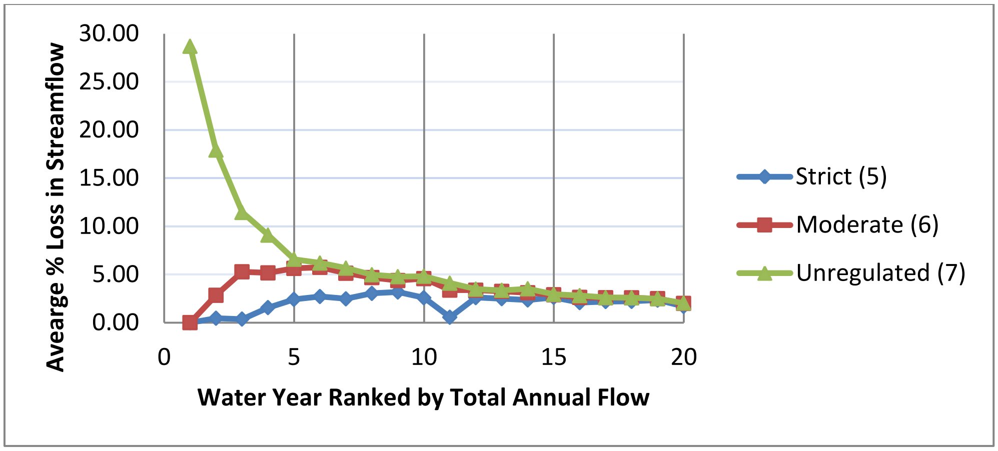

FIGURE 4-6. Average percent loss in streamflow for the strict (5), moderate (6), and unregulated (7) policy scenarios across precipitation years for impacted streams both above and below EOA.

The percent loss in streamflow is heterogeneous across the watershed, where most streamflow impacts occur in the larger watersheds, while most stream segments in smaller watersheds remain unaffected by diversions (Table 4-9). For instance, approximately 82% (93.65 miles of the 114.62 miles in the watershed) of streams segments are unaffected by diversions in catchment areas less than 1 square mile. However, the largest impacts are observed in the smaller watersheds. For the unregulated policy, the average loss in streamflow for impacted streams within small watersheds is 11.99%, compared to 3.93% and 3.55% for streams with catchment areas between 1-10 and greater than 10 miles squared, respectively. Percent loss within smaller watersheds is considerably smaller under the strict policy than either of the two other policies.

TABLE 4-9. Average Percent Loss in Streamflow by Catchment Area for the Strict (5), Moderate (6), and Unregulated (7) Policy Scenarios Aggregated by Precipitation Years (only impacted streams, both above and below EOA).

	Catchment Area (miles squared)			
Scenario	**<1**	**1-10**	**>=10**	**Average**
Strict (5)	3.502	1.306	1.166	1.995
Moderate (7)	6.742	2.098	1.889	3.581
Unregulated (9)	11.993	3.930	3.547	6.499
Stream Length	**<1**	**1-10**	**>=10**	**Total**
Length (miles) Impacted Streams	20.96 (18.29)	22.90 (49.26)	19.10 (100.00)	62.70 (34.94)
Length (miles) Unaffected Streams	93.65 (81.71)	23.59 (50.74)	0.00 (0.00)	117.24 (65.06)
Length (miles) All Streams	114.62	46.48	19.10	180.20

Note: Values in parenthesis equal the percentage of total stream length for impacted and unaffected streams.

Overall, results suggest that the number of unimpaired bypass-flow days varies considerably both spatially and temporally. The strict diversion-guideline provides some benefit to protecting bypass-flow days, although, benefits are mostly within small watersheds. The moderate policy, however, does not provide more protection than unregulated diversions. In dry years, both the strict and moderate policies lead to considerable reductions in the percent loss in streamflow relative to the unregulated policy. However, since bypass-flow thresholds under unimpaired conditions are usually not met during dry years, reducing diversions during dry years does not necessarily lead to more bypass-flow days than under the unregulated policy.

4.4.2 Agricultural Water Security Impacts

As an indicator of agricultural water security, I use the percent storage filled at the end of the high flow season (April 30th). In Figure 4-7, I plot the average percent storage that is filled for reservoirs in the watershed for different precipitation year under the strict (5), moderate (6), and unregulated (7) policies.[21] The largest differences between the policies occur during the driest years (i.e., those in the lowest quartile). This is because the unregulated policy allows for diversions even under the driest years, while the moderate and strict policies place restrictions on diversions during these years (Figure 4-6). In the wet years (i.e., highest quartile), the average percent filled was close to 100 percent under all three policies (Figure 4-7), indicating high water security in wet years even under the strict policy.

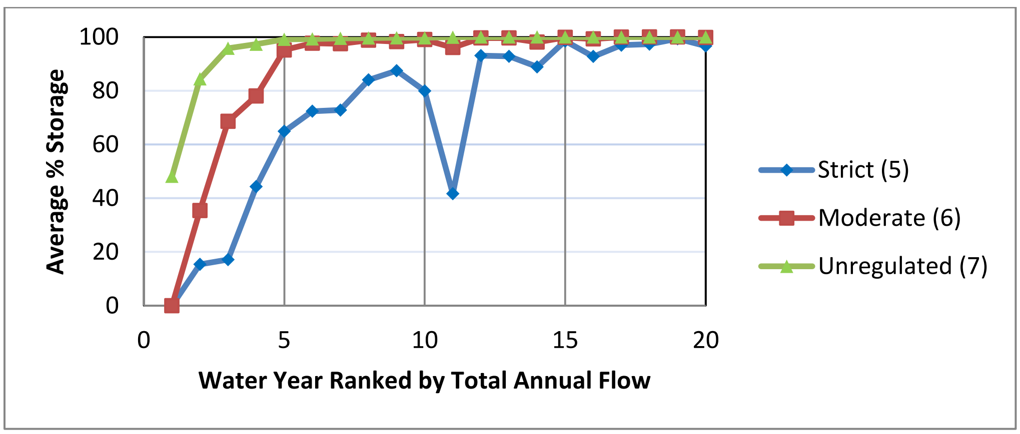

FIGURE 4-7. Average percent reservoir storage filled for policy scenarios strict (5), moderate (6), and unregulated (7) by precipitation year.

The amount of water stored under the strict and moderate policy scenarios depends not only on the amount of total annual flow, but also on the size and timing of

[21] The reduction in average percent loss in streamflow that is observed for the strict policy for year 1971 (ranked 11) in Figure 4-7 is due to the fact that the several large peak-flow events exceeding the Q_{mbf} threshold for this year occur prior to the diversion season (Dec 15th – Mar 31st).

peak-flows. For example, percent storage drops considerably for the strict policy in year 1971 (ranked 11). The reason is that most peak-flows exceeding the bypass-flow threshold in 1971 occur prior to the diversion period that begins on December 15th, and thus, diversions during these early peak-flows are not allowed. The moderate policy, on the other hand, is only slightly affected by this because some flows during the diversion period exceed the February-median-flow (Q_{fmf}) threshold, which is a lower threshold than the Q_{mbf} threshold (Figure 4-2). As such, water security can be impaired under a moderate rainfall year if the timing of peak-flow events occurs outside the diversion period.

The impact of diversions guidelines on water security is heterogeneous and varies by POD location and reservoir size. This heterogeneity is especially observed for the strict policy, where PODs with small catchment areas are impacted the most (Table 4-10). Specifically, PODs with catchment areas less than one square mile experience the greatest reduction in average percent storage (from 95.0% under unregulated conditions, to 66.0% under the strict policy); meanwhile, PODs with large catchment areas (> 10 miles squared) achieve a percent storage of 95% or irrespective of the policy.

TABLE 4-10. Average Percent of Storage Filled by POD Catchment Area for the Strict (5), Moderate (6), and Unregulated (7) Policy Scenarios Averaged across Precipitation Years.

	POD Catchment Area (miles squared)			
Scenario	**<1**	**1-10**	**>10**	**Average**
Strict (5)	66.0	86.2	95.0	71.8
Moderate (6)	86.1	93.5	95.0	88.0
Unregulated (7)	95.0	99.2	100.0	96.1
Total Capacity (ac-ft)	1441.8	328.3	125.4	1895.6

The variation in percent storage for the strict policy is clearly observed in a map for the year 1981 (ranked 5) (Figure 4-8), where PODs located further upstream tend to have lower percent storage values than those further downstream. For the moderate and unregulated policies, variation in percent storage across the watershed for the year 1981 tends to be less pronounced, especially for the unregulated policy (Appendix B, Figures B-4 and B-5, respectively).

As for reservoir size, larger reservoirs have lower water security than smaller reservoirs, especially under the strict policy (Table 4-11). For the unregulated policy, reservoirs with different sizes tend to have similar values for percent storage filled: 97.4% for reservoirs with less than 10 ac-ft in capacity, 96.6% for those between 10-50 ac-ft, and 91.0% for reservoirs greater than 50 ac-ft. However, under the strict policy, large reservoirs above 50 ac-ft in size are the least water secure. Their average percent storage decreases from 91.01% under the unregulated policy, to 53.77% under the strict policy.

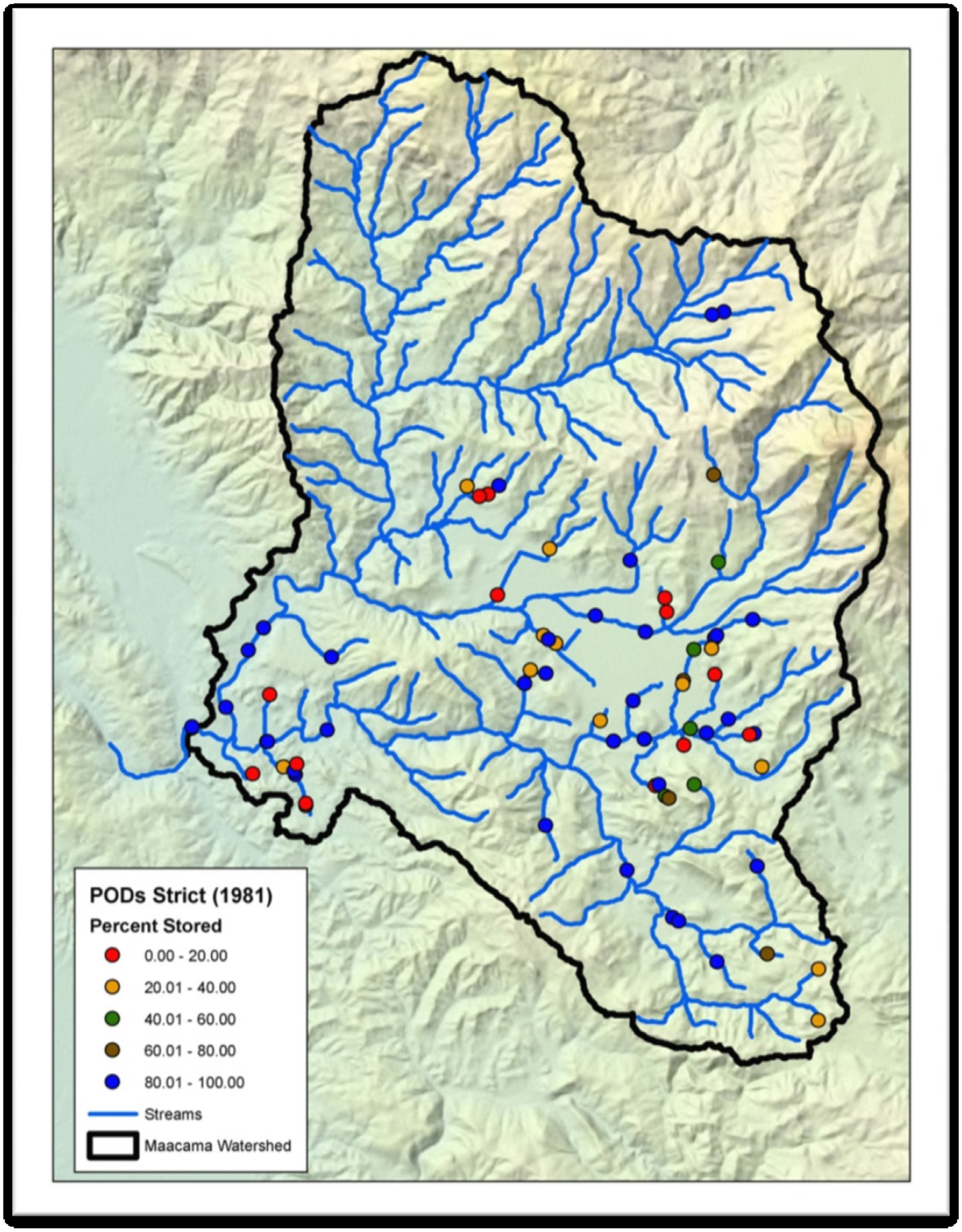

FIGURE 4-8. Map of percent storage filled for reservoirs in the Maacama watershed for the strict policy scenario (5), for the dry year 1981 (ranked 5).

TABLE 4-11. Average Percent of Storage Filled by Reservoir Size for the Strict (5), Moderate (6), and Unregulated (7) Policy Scenarios Averaged across Precipitation Years.

	Reservoir Capacity (ac-ft)			
Scenario	**<10**	**10-50**	**>50**	**Average**
Strict (5)	77.0	72.5	53.8	71.8
Moderate (6)	90.3	88.9	78.9	88.0
Unregulated (7)	97.4	96.6	91.0	96.1
Total Capacity (ac-ft)	189.5	591.6	1114.5	1895.6

The presence of upstream diversions reduces the amount of water available to downstream water users, thus potentially decreasing their water security. I measure the cumulative impact of upstream unpermitted diversions on permitted waters users by calculating the loss in their percent storage filled. This impact is evaluated across all policy scenarios and water years. In Table 4-5, I defined the policy scenarios strict (2), moderate (3), and unregulated (4), where only permitted diversions are allowed to divert (i.e., unpermitted diversions are absent). Hence, these three scenarios can be used to simulate the percent storage filled for each permitted water user in the absence of unpermitted diversions for the different policies, and compared to the strict (5), moderate (6), and unregulated (7) policy scenarios that allow both permitted and unpermitted diversions. I measure the impact on permitted water users under each policy by comparing the values for percent storage filled when unpermitted diversions are not allowed versus allowed. For example, for the set of permitted water users, I calculate the difference in percent storage filled for each reservoir between the strict policy scenarios 2 versus 5. This provides the loss in percent storage filled, under the strict policy, for each permitted water user as a result of unpermitted diversions. Similarly, I perform the difference in percent storage filled for the moderate policy scenarios, 3 versus 4, and the unregulated policy scenarios, 4 versus 7.

The results for the impact analysis of unpermitted diversions on permitted diversions are provided in Table 4-12. Recall that there are a total of 35 permitted diversions and 35 unpermitted diversions within the Maacama watershed (Table 4-3). Overall, the results show that losses in percent storage can be large; however, few permitted water users are affected. The impacts that occur are generally in dry years or under the strict policy. Impacts are observed for seven out of twenty years under the strict policy, for three of the dry years under the moderate policy, and only once for the unregulated policy in the driest year (1977). Impacts are rare for the unregulated policy because the majority of water users are able to meet their water needs even in the presence of upstream diversions. On the other hand, under the strict and moderate policies, upstream diversions can make it more difficult for downstream water users to meet the necessary diversion thresholds to divert water, especially during dry years.

For instance, for the dry year 1976 (ranked 2), only one permitted water user is affected for the strict policy; three permitted water users are affected for the moderate policy; and no permitted water users are affected for the unregulated policy. For this same year, the average loss in percent storage filled for affected permitted water users equals 6.85% for the strict policy, 27.88% for moderate policy, and 0% for the unregulated policy. The largest loss in percent storage equals 42.27%, which occurs for the dry years 1981, 1979, and 1968 under the strict policy, and for the year 1972 for the moderate policy. Impacts vary across water years due to the timing and magnitude of peak-flows.

TABLE 4-12. Cumulative Impacts of Unpermitted Diversions on Permitted Diversions.

		Strict			**Moderate**			**Unregulated**		
Rank	**Year**	number affected	number unaffected	average loss in % storage filled (affected)	number affected	number unaffected	average loss in % storage filled (affected)	number affected	number unaffected	average loss in % storage filled (affected)
Driest	1977	0	35	0.00	0	35	0	2	33	23.60
	1976	1	34	6.85	3	32	27.88	0	35	0.00
	1972	0	35	0.00	1	34	42.27	0	35	0.00
	1964	2	33	12.93	1	34	41.43	0	35	0.00
Dry	1981	1	34	42.27	0	35	0.00	0	35	0.00
	1979	1	34	42.27	0	35	0.00	0	35	0.00
	1968	1	34	42.27	0	35	0.00	0	35	0.00
	1962	0	35	0.00	0	35	0.00	0	35	0.00
	1966	0	35	0.00	0	35	0.00	0	35	0.00
moderate	1975	1	34	28.21	0	35	0.00	0	35	0.00
	1971	2	33	10.75	0	35	0.00	0	35	0.00
	1980	0	35	0.00	0	35	0.00	0	35	0.00
	1973	0	35	0.00	0	35	0.00	0	35	0.00
	1963	0	35	0.00	0	35	0.00	0	35	0.00
Wet	1965	0	35	0.00	0	35	0.00	0	35	0.00
	1967	0	35	0.00	0	35	0.00	0	35	0.00
	1978	0	35	0.00	0	35	0.00	0	35	0.00
	1969	0	35	0.00	0	35	0.00	0	35	0.00
	1970	0	35	0.00	0	35	0.00	0	35	0.00
Wettest	1974	0	35	0.00	0	35	0.00	0	35	0.00

For example, under the strict policy, there are three separate permitted water users that are affected at least once by upstream diversions. Two of these water users are affected in 1964 (ranked 4), but are not affected in the years 1962 (ranked 8) and 1966 (ranked

9). However, both are affected again in 1971 (ranked 11). Recall that the year 1971 experiences a rainfall pattern that leads to a large decrease in overall water security for the strict policy (Figure 4-7). As such, the likelihood of impacts is not based only on the total annual flow for a water year, but also on the timing and size of flows.

There are several reasons why a permitted water user may not be affected by the presence of unpermitted diversions throughout the drainage network: 1) there are no unpermitted diversions located upstream from a permitted water user; 2) the permitted water users achieves 100% of their capacity even in the presence of upstream unpermitted diversions; 3) the permitted water user is not allowed to fill due to policy restrictions on diversions, even when upstream diversions are not allowed; and 4) unpermitted water users are not allowed to fill due to restrictions on diversions, thus causing no impact on downstream permitted water users. The first reason is the most common. The highly branched network of stream segments, and the distribution of reservoirs across the landscape, is such that the majority of PODs for permitted water users are not located downstream from unpermitted diversions (Figure 4-3). Hence, while the reduction in percent storage caused by unpermitted diversions can be large, most permitted water users are not affected. The second reason commonly occurs in wet years, when there is sufficient water to meet the needs of most water users, while the third and fourth reasons are most common during dry years.

4.4.3 Tradeoffs between Bypass-Flows and Agricultural Water Security

I quantify and graph the tradeoffs between losses in bypass-flows and agricultural water security by comparing the average number of bypass-flow days with the average percent storage filled for the unimpaired (1), strict (5), moderate (6), and unregulated (7) policy scenarios (Figure 4-9). Overall, the strict policy provides slightly greater protection of bypass-flow days than the moderate and unregulated policies, but significantly reduces water security relative to these two policies. The strict policy results in an overall percent loss in an average bypass-flow days of 0.08%, while the moderate and unregulated policies lead to reductions in bypass-flow days of 2.49% and 2.36%, respectively (Table 4-7). Meanwhile, for the strict and moderate policies, average percent storage decreases by around 25 and 10 percentage points, respectively, relative to the unregulated policy (Figure 4-9).

Interestingly, the moderate policy provides lower agricultural water security than the unregulated policy, and yields a larger loss in the average number of bypass-flow days (a decrease of 2.49%) than the unregulated policy (2.36%) (Table 4-7). Note, the vertical axis in Figure 4-9 is set to start at 22.70 days rather than 0 days to better highlight the lower number of bypass-flow days for the moderate policy than for the unregulated policy.

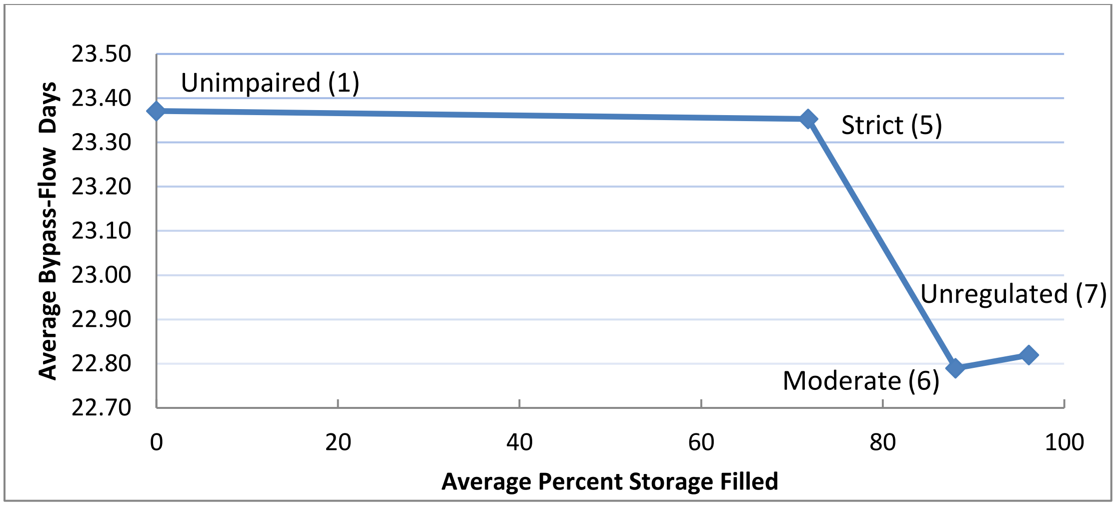

FIGURE 4-9. Average bypass-flow days versus average percent storage filled for the unimpaired (1), strict (5), moderate (6), and unregulated (7) policy scenarios aggregated by precipitation years.

4.5 Discussion

Findings highlight the high spatial and temporal variability of flows in the Maacama watershed, which has important implications for both ecological processes and agricultural water users. Without the presence of any diversions, the number of days in which salmon bypass-flows are met within stream segments of the Maacama watershed is both spatially and temporally heterogeneous. The average number of unimpaired bypass-flow days for streams segments in the upper reaches (< 1 mile squared) is 8.6 days, while for streams segments further downstream (>= 10 miles squared), the number of bypass-flow days is 44.3 (Table 4-8). This spatial variation is well illustrated in the mapping of bypass-flow days across the Maacama watershed (Figure 4-4). Bypass-flow days vary with the precipitation year as well, generally tending to increase with greater annual flow. For the twenty precipitation years, the minimum average number of bypass-flow days is 0, while the largest is 51.6 days (Table 4-6). As such, precipitation patterns significantly influence the variation in the number of bypass-flow days.

Although the impact of diversions on bypass-flow days is larger within small catchment areas than within large watersheds, the extent of impacts on flows is smaller within smaller watersheds than within larger watersheds. For instance, under the unregulated policy, the percent loss in the average number of bypass-flow days within large watershed areas (>=10 miles squared) is 1.32%, while the percent loss in average bypass-flow days within small watersheds (<1 mile squared) is 8.95% (Table 4-8). This highlights that watershed size plays an important role in determining the relative impact of diversions on bypass-flow days. However, at the same time, 73% of streams segments within small watersheds, and below EOA, are not impacted by diversions, while all streams segments within large watersheds (>= 10 miles squared) are impacted (Table 4-8 and Figure 4-5). As such, the impact of diversions on bypass-flow days is greater in small watersheds, but stream segments in the upper reaches are less likely to be affected because most are not downstream of PODs.

Results suggest that the strict diversion-guideline provides spatially heterogeneous improvements in the number of bypass-flow days across the Maacama watershed when compared to the unregulated policy. Across all water years and streams segments, the strict policy yields on average 2.28% (2.36% - 0.08%) (Table 4-7) more bypass-flow days than the unregulated policy. However, within small watersheds, the improvement is 8.46% (8.95% - 0.49%), while within large watersheds, it reduces to

1.31% (1.32% - 0.01%) (Table 4-8). As such, the strict diversion-guideline provides greater protection of bypass-flow days in the upper reaches than in the lower reaches of the watershed.

Meanwhile, the moderate diversion-guideline does not provide an improvement in bypass-flow days across the watershed as a whole. Results suggest that the moderate policy leads to an overall decrease in bypass-flow days of 0.13% (2.36% - 2.49%) (Table 4-7) relative to the unregulated policy. Within small catchment areas (<1 mile squared), the moderate policy leads to an additional decrease in bypass-flow days of 1.27% (8.95% - 10.22%) relative to the unregulated policy, as well as a decrease of 0.43% (1.73% – 2.16%) (Table 4-8) within medium sized watersheds (1-10 miles squared). For the larger watersheds (>= 10 miles squared), however, the moderate policy actually protects bypass-flow days slightly more than the unregulated policy, yielding an improvement of 0.26% (1.32% – 1.06%) (Table 4-8) in bypass-flow days.

The moderate policy does not outperform the unregulated policy due to a combination of the diversion season and the February-median-flow threshold. The moderate policy restricts all diversions to take place during the winter rainy season, from Dec15th to Mar 31st, when most bypass-flow days usually occur. Meanwhile, the unregulated policy allows for diversions to begin on October 1st, prior to the start of the winter rainy season, which provides some protection against reductions in bypass-flow days. The more restrictive diversion season, combined with the fact that the moderate policy only requires that flows exceed the less stringent February-median-flow threshold, not the bypass-flow threshold, lead to a perverse effect: the moderate policy does less to protect bypass-flows than the unregulated policy.

Overall, in dry years, both the strict and moderate policies lead to considerable reductions in the percent loss in streamflow relative to the unregulated policy. However, since bypass-flow thresholds under unimpaired conditions are usually not met during dry years, reducing diversions during dry years do not necessarily lead to significantly more bypass-flow days than under the unregulated policy. For instance, losses in bypass-flow days for the moderate and unregulated policies are quite similar for dry years (ranked 1-4), (Table 4-7 and Figure B-1), even though the percent of flow diverted for these two polices is different for these years (Figure 4-6).

The level of agricultural water security attained by landowners is high under unregulated conditions for the majority of water years. For the drier years (lowest quartile), average percent storage for the year 1976 (ranked 2) is above 80%, while by 1981 (ranked 5), average storage is almost 100% (Figure 4-7). Percent storage is large under dry years because the unregulated policy does not impose limits on diversions. For example, for the driest year (1977), the average streamflow loss under the unregulated policy is close to 30%, while under the moderate and strict policies losses are 0% (Figure 4-6). Water security under unregulated conditions is also relatively homogenous across reservoirs of different catchment areas and sizes, where reservoirs fill on average to above 90% across all categories (Table 4-8 and Table 4-9).

The diversion-guidelines, however, limit the water security of landowners relative to the unregulated policy. The strict policy significantly reduces percent water stored, achieving lower percent storage values than the moderate and unregulated policies for dry and moderate water years (Figure 4-7). The moderate policy yields lower levels of water storage relative to the unregulated policy only during dry years (Figure 4-7). For precipitation years wetter than 1965 (ranked 15), all three policies divert approximately the same amount of flow (Figure 4-6), and thus achieve similar levels of water security in terms of percent storage (Figure 4-7). By and large, the strict policy leads to the largest overall reduction in landowner water security, while both the strict and moderate policies significantly reduce water security during dry years relative to the unregulated policy.

The impact of the diversion-guidelines on water security is heterogeneous and varies by the POD location and reservoir size. PODs with smaller catchment areas (< 1 mile squared) are located in the upper parts of the watershed, where headwater streams are found. Landowners located in these areas are more vulnerable to the adverse effects of regulations than waters users located further downstream. This is because the bypass-flow threshold increases non-linearly with watershed size (equation [4.6]); as such, streamflows are less likely to meet the bypass-flow threshold in the upper reaches of the watershed, placing greater diversion restrictions on waters users in these parts of the watershed. For instance, PODs with a catchment area greater than 10 miles squared fill consistently above 95%, on average, under all three policies (Table 4-10). However, PODs with small catchment areas (<1 mile squared) fill on average to 66% under the strict policy versus 91.4% under the unregulated policy (Table 4-10). Since landowners with larger reservoirs require greater amounts of flow to meet their water needs, the size of the reservoir also affects water security. The average percent storage for large reservoirs above 50 ac-ft in size decreases from 91% for the unregulated policy to 53.8% for the strict policy (Table 4-11). As such, this suggests that water management policies should account for the heterogeneous impacts across landowners.

Unpermitted diversions can lead to a large reduction in the percent storage filled of permitted water users; however, results suggest that impacts are rare. The highly branched network of stream segments, and the distribution of PODs across the landscape, is such that the majority of PODs for permitted water users are not located downstream of unpermitted diversions (Figure 4-3). Hence, the majority of permitted water users are not affected by unpermitted water users. For those located downstream of unpermitted water users, most impacts occur during dry years or under the strict policy. For instance, the greatest number of impacted water users is observed in the year 1976 (ranked 2), where three permitted water users are affected (Table 4-10). On the other hand, impacts occur most frequently under the strict policy, since unpermitted diversions can make it more difficult for downstream permitted water users to meet the bypass-flow diversion threshold. During wet years, however, the impacts of upstream diversions are reduced to zero since there is sufficient water available for all water users to meet their needs.

Results from this study suggest that there exist inherent tradeoffs between instream flow protections and agricultural water security. Both of the diversion-guidelines significantly reduce water storage, especially during dry years. However, the strict policy provides a small improvement in bypass-flow days across the watershed as a whole, while the moderate policy provides no improvement in bypass-flow days. The strict policy achieves an average improvement in bypass-flow days of 2.28% over the unregulated policy (Table 4-7), but results in an average reduction of around 25% in the measure of agricultural water security (Figure 4-9). On the other hand, within smaller watersheds, the strict policy provides greater protection of bypass-flow days, exceeding the unregulated policy in bypass-flow days by 8.46% (Table 4-8). During wet years, the strict policy has a small impact on both bypass-flow days and water security (Table 4-7 and Figure 4-7). Consequently, greater focus should be given to better managing instream flow protection in the smallest watersheds and meeting human water needs during dry years, when agricultural water security impacts are greatest.

4.6 Conclusions

The protection of instream flows has become a necessity for maintaining ecosystem functions and preserving endangered species. Consequently, regulatory agencies are increasingly placing greater restrictions on water users in order to protect instream flows, which can have significant impacts on agricultural water users. It is important to quantify the tradeoff between environmental protection and agricultural water security to better understand the effect of regulations on both instream flows and agricultural water needs. Within decentralized water management systems, this can be challenging due to the spatial and temporal variation in water supply, the dispersed network of water users, the cumulative impacts of diversions across the watershed, and the need to meet instream flow requirements throughout the system of streams. The integrated modeling framework used in this study allows for such a comprehensive analysis.

Results from this study suggest that the minimum-bypass-flow diversion-guideline provides a certain level of protection against the adverse impacts of diversions, while the February-median flow guideline does not. The resulting losses in agricultural water security, however, are significant, especially during dry years under both policies, and under the strict policy for most years. The strict diversion-guideline is most effective in reducing the impact of diversions within the upper parts of the watershed, which provide critical habitat for juvenile salmon. Although both the strict and moderate diversion-guidelines reduce water security the most during dry years, results suggest that within wet years it is possible to protect bypass-flows without limiting agricultural water security. In addition, given the highly branched network of stream segments that exist within a decentralized management system, it may be possible to allow for additional onsite storage to meet agricultural needs without impacting existing water users.

Part of the challenge to achieving the long-term sustainability of ecosystems and agriculture is to develop watershed management policies that effectively balance human and ecosystem needs, especially during drought years, when conflicts appear to be the greatest. This study provides greater insight and quantitative tools to address the complexity of managing environmental protection and human water user needs within unpredictable climates, such as Mediterranean regions. With the effects of global climate change likely to lead to an increase in the variability of fresh water supplies throughout many regions of the world, approaches to analyzing water management policies within variable systems will be needed to better balance human and environmental needs.

A limitation of the hydrologic model is that it models only surface flow diversions and does not account for the effects of groundwater pumping on streamflows. Specifically, the pumping of subsurface flows can potentially affect streamflow levels and thus affect aquatic ecosystems. However, since the need for groundwater resources for irrigation purposes are mostly during the summer months, surface flow diversions represent the principal threat to reductions in streamflows during the migration period. The evaluation of tradeoffs between bypass-flow days and agricultural water security could be improved if a quantifiable relationship between bypass-flow days and salmon abundance were available, as well as the relationship between agricultural water security and its value to farmers. Such information would provide greater insights into the tradeoffs between protecting instream flows and losses in agricultural water security resulting from instream flow policies.

CHAPTER V

SUMMARY AND CONCLUSIONS

A growing awareness of the effect of agricultural production on the environment has led to the development of programs and policies to mitigate its adverse effects. Conservation programs, such as CRP and EQIP, have been developed to incentivize farmers to voluntarily adopt more environmentally friendly practices. In Essay I, or Chapter II, an aggregate measure of additionality (ATT) is estimated for cost-share programs for six conservation practices based on a farmer survey in Ohio. Results suggest that additionality is positive and statistically significant for all six conservation practices, where the ATT values are higher for field practices, such as conservation tillage and cover crops that can be applied across a whole field, than for those practices applied only within environmentally sensitive areas. As such, this difference in ATT values is expected given that practices within environmentally sensitive areas represent a smaller proportion of the total farm acreage. However, while enrollment in conservation programs achieve a positive ATT for all practice types, a comparison between the level of additionality between conservation practices, using the %ATT, reveals that certain practice types achieve higher percent additionality than others. In fact, percent additionality varies dramatically between practice types. The largest %ATT is found for hayfield establishment, filter strips, and cover crops, with a %ATT of around 90%, the practices grid sampling and grass waterways achieved approximately 65%, while conservation tillage had the lowest percent additionality of around 20%. While these results are valuable for program managers in evaluating conservation program effectiveness, they provide only part of the analysis that is required. For example, it is important to evaluate not only increases in conservation effort, but also overall improvements in environmental benefits. Having quantifiable information on the actual improvement in environmental benefits per practice, coupled with additionality estimates, would provide greater insight into the effectiveness of conservation programs.

Meanwhile, in Essay II, or Chapter III, a new methodological approach is developed to disaggregate the additionality measure into two separate effects: the expansion of existing conservation practices versus the new adoption of conservation practices. To do so, the contributions of "prior-adopters" and "new-adopters" to the overall ATT are estimated for cost-share programs for six conservation practices. This consists of estimating the likelihood that enrolled farmers are prior-adopters or new-adopters, as well as the relative contribution for each group to the overall ATT. Results of the decomposition of the ATT reveal several findings. First, the ATT for prior-adopters is not significant for all practice types, implying that prior-adopters do not significantly expand the proportion of conservation acreage when receiving cost-share funding. This suggests that cost-share programs have no significant effect on the conservation of those who would have adopted the practice without funding (i.e., prior-adopters). Second, decomposition estimates suggest that the differences in %ATT between practice types found in Chapter II are mainly driven by the fraction of enrolled farmers that are prior-adopters and new-adopters. Practice types that are estimated to have a large fraction of new-adopters, such as filter strips and hayfield establishment, exhibit larger values for %ATT. Whether a practice has a large fraction of enrolled new-adopters or not is believed to be driven by the opportunity costs associated with adopting the practice. For instance, hayfield establishment would result in a complete loss in grain crop yields for the length of the enrollment contract. Consequently, enrolled farmers should be less likely to adopt hayfield establishment without funding, and thus, a larger fraction of enrolled farmers are new-adopters for hayfield establishment. Overall, this suggests that a farmers' history in conservation adoption has a significant influence on additionality levels.

Regulatory agencies have been faced with the need to protect instream flows to sustain aquatic ecosystems. In response, instream flow polices have been developed that curtail agricultural water diversions, placing greater pressures on agricultural producers to meet their water needs. In Essay III, or Chapter IV, the effects of instream flow policies on agricultural water security and streamflows within a decentralized management regime are analyzed for a watershed in northern-California. Results highlight that watershed size plays an important role in determining the relative impact of diversions on bypass-flow days. The impact of diversions on bypass-flow days is larger in small watersheds; however, stream segments located in the upper reaches are less likely to be affected because most are not downstream of PODs (points-of-diversion). Relative to the unregulated policy, the strict diversion-guideline provides greater protection of bypass-flows days within smaller watersheds; however, within larger watersheds, the amount of protection is not as significant. Meanwhile, the moderate diversion guideline leads to an overall decrease in bypass-flow days when compared to the unregulated policy.

With respect to agricultural water security, percent storage is high for the majority of water years and relatively homogenous across reservoirs of different catchment areas and sizes under the unregulated policy. Water security, however, decreases sharply under the diversion-guidelines. The strict policy significantly reduces percent storage, especially during dry and moderate years, while the moderate policy significantly decreases water security during dry years. In addition, the impact of the diversion-guidelines on water security is heterogeneous and varies by POD location and reservoir size. The impact of unpermitted diversions on permitted water users can also reduce the water security of permitted water users. However, because of the highly branched network of stream segments, and the distribution of PODs across the landscape, the majority of permitted PODs are not located downstream of unpermitted diversions. Hence, most permitted water users are not affected by unpermitted water users. Finally, results from this study suggest that greater focus should be given to better managing streamflow protection in the smallest watersheds, where percent losses in bypass-flow days are greatest, and meeting human water needs during dry years, when impacts on agricultural water security from instream flow policies are largest.

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APPENDIX A

ADDITIONAL RESULTS FOR CHAPTER II

TABLE A-1. Summary Statistics on Covariates for Enrolled and Non-Enrolled Farmers for Conservation Tillage.

		Enrolled (N = 87 farmers)		Non-Enrolled (N = 476 farmers)		
Variable	**Definition**	**Mean**	**Std. Dev**	**Mean**	**Std. Dev**	**Diff in Means**
Farm Revenue	=1 if farm revenue exceeded $250,000 in 2009	0.402	0.493	0.284	0.451	0.119*
Farm Horizon	=1 if farm will be operated by family within the next 5 years	0.897	0.306	0.882	0.323	0.014
Age	age	57.437	10.600	56.300	11.621	1.136
Education	=1 if education exceeds high school	0.540	0.501	0.405	0.491	0.135*
Soil type	=1 if dominant soil texture is clay	0.759	0.430	0.767	0.423	-0.008
	=1 if dominant soil texture is loam or sandy	0.241	0.430	0.233	0.423	0.008
Household Income	=1 if 0% - 10% of household income comes from farming	0.218	0.416	0.210	0.408	0.008
	=1 if 10% - 50% of household income comes from farming	0.299	0.460	0.311	0.463	-0.012
	=1 if more than 50% of household income comes from farming	0.483	0.503	0.479	0.500	0.004
Rented	proportion of farm acreage rented in 2009	0.477	0.337	0.431	0.370	0.046
Grain Crops	proportion of farm acreage devoted to grain crops in 2009	0.942	0.147	0.857	0.269	0.085**

TABLE A-1. Continued.

Variable	Definition	Enrolled (N = 87 farmers) Mean	Enrolled Std. Dev	Non-Enrolled (N = 476 farmers) Mean	Non-Enrolled Std. Dev	Diff in Means
Slope	proportion of farm acreage with slope 0%-2%	0.515	0.368	0.582	0.389	-0.067
	proportion of farm acreage with slope 2%-6%	0.405	0.353	0.366	0.365	0.039
	proportion of farm acreage greater than 6% slope	0.080	0.165	0.052	0.130	0.029
Farm Size	natural log of total farm acreage operated in 2009	6.148	0.986	5.791	1.053	0.357**
Stream	=1 if a river or stream borders or runs through the property	0.644	0.482	0.590	0.492	0.053
Livestock	=1 if managed livestock in 2009	0.402	0.493	0.517	0.500	-0.115*

Note: Statistical significance: 99% (**), 95% (*).

TABLE A-2. Summary Statistics on Covariates for Enrolled and Non-Enrolled Farmers for Cover Crops.

		Enrolled (N = 24 farmers)		Non-Enrolled (N = 581 farmers)		
Variable	**Definition**	**Mean**	**Std. Dev**	**Mean**	**Std. Dev**	**Diff in Means**
Farm Revenue	=1 if farm revenue exceeded $250,000 in 2009	0.375	0.495	0.299	0.458	0.076
Farm Horizon	=1 if farm will be operated by family within the next 5 years	0.875	0.338	0.878	0.328	-0.003
Age	age	52.417	11.244	57.170	11.553	-4.754*
Education	=1 if education exceeds high school	0.500	0.511	0.422	0.494	0.078
Soil type	=1 if dominant soil texture is clay	0.750	0.442	0.768	0.423	-0.018
	=1 if dominant soil texture is loam or sandy	0.250	0.442	0.232	0.423	0.018
Household Income	=1 if 0% - 10% of household income comes from farming	0.125	0.338	0.208	0.406	-0.083
	=1 if 10% - 50% of household income comes from farming	0.208	0.415	0.330	0.471	-0.122
	=1 if more than 50% of household income comes from farming	0.667	0.482	0.461	0.499	0.205*
Rented	proportion of farm acreage rented in 2009	0.452	0.336	0.435	0.362	0.017
Grain Crops	proportion of farm acreage devoted to grain crops in 2009	0.723	0.276	0.881	0.248	-0.158*

TABLE A-2. Continued.

		Enrolled (N = 24 farmers)		Non-Enrolled (N = 581 farmers)		
Variable	**Definition**	**Mean**	**Std. Dev**	**Mean**	**Std. Dev**	**Diff in Means**
Slope	proportion of farm acreage with slope 0%-2%	0.538	0.419	0.568	0.384	-0.030
	proportion of farm acreage with slope 2%-6%	0.383	0.396	0.376	0.361	0.007
	proportion of farm acreage greater than 6% slope	0.080	0.171	0.056	0.135	0.024
Farm Size	natural log of total farm acreage operated in 2009	5.929	0.822	5.854	1.045	0.074
Stream	=1 if a river or stream borders or runs through the property	0.667	0.482	0.596	0.491	0.071
Livestock	=1 if managed livestock in 2009	0.833	0.381	0.468	0.499	0.365**

Note: Statistical significance: 99% (**), 95% (*).

TABLE A-3. Summary Statistics on Covariates for Enrolled and Non-Enrolled Farmers for Hayfields.

		Enrolled (N = 19 farmers)		Non-Enrolled (N = 575 farmers)		
Variable	**Definition**	**Mean**	**Std. Dev**	**Mean**	**Std. Dev**	**Diff in Means**
Farm Revenue	=1 if farm revenue exceeded $250,000 in 2009	0.158	0.375	0.306	0.461	-0.148
Age	age	62.000	9.843	56.887	11.677	5.113
Education	=1 if education exceeds high school	0.579	0.507	0.419	0.494	0.160
Soil type	=1 if dominant soil texture is clay	0.789	0.419	0.769	0.422	0.021
	=1 if dominant soil texture is loam or sandy	0.211	0.419	0.231	0.422	-0.021
Household Income	=1 if 0% - 10% of household income comes from farming	0.263	0.452	0.203	0.403	0.060
	=1 if 10% - 50% of household income comes from farming	0.368	0.496	0.320	0.467	0.048
	=1 if more than 50% of household income comes from farming	0.368	0.496	0.477	0.500	-0.108
Rented	proportion of farm acreage rented in 2009	0.204	0.233	0.446	0.361	-0.241**
Grain Crops	proportion of farm acreage devoted to grain crops in 2009	0.657	0.416	0.888	0.237	-0.230**

TABLE A-3. Continued.

Variable	Definition	Enrolled (N = 19 farmers) Mean	Enrolled Std. Dev	Non-Enrolled (N = 575 farmers) Mean	Non-Enrolled Std. Dev	Diff in Means
Slope	proportion of farm acreage with slope 0%-2%	0.402	0.414	0.575	0.383	-0.173
	proportion of farm acreage with slope 2%-6%	0.472	0.421	0.375	0.361	0.097
	proportion of farm acreage greater than 6% slope	0.125	0.262	0.050	0.122	0.075*
Farm Size	natural log of total farm acreage operated in 2009	5.244	1.268	5.877	1.029	-0.633**
Stream	=1 if a river or stream borders or runs through the property	0.579	0.507	0.593	0.492	-0.014
Livestock	=1 if managed livestock in 2009	0.526	0.513	0.471	0.500	0.055

Note: Statistical significance: 99% (**), 95% (*).

TABLE A-4. Summary Statistics on Covariates for Enrolled and Non-Enrolled Farmers for Grid Sampling.

		Enrolled (N = 55 farmers)		Non-Enrolled (N = 484 farmers)		
Variable	**Definition**	**Mean**	**Std. Dev**	**Mean**	**Std. Dev**	**Diff in Means**
Farm Revenue	=1 if farm revenue exceeded $250,000 in 2009	0.455	0.503	0.264	0.442	0.190**
Farm Horizon	=1 if farm will be operated by family within the next 5 years	0.964	0.189	0.868	0.339	0.096*
Age	age	55.800	11.557	56.791	11.652	-0.991
Experience	years of farming experience	31.091	12.532	32.134	12.964	-1.043
Education	=1 if education exceeds high school	0.655	0.480	0.413	0.493	0.241**
Soil type	=1 if dominant soil texture is clay	0.745	0.440	0.758	0.429	-0.013
	=1 if dominant soil texture is loam or sandy	0.255	0.440	0.242	0.429	0.013
Household Income	=1 if 0% - 10% of household income comes from farming	0.109	0.315	0.219	0.414	-0.110
	=1 if 10% - 50% of household income comes from farming	0.400	0.494	0.320	0.467	0.080
	=1 if more than 50% of household income comes from farming	0.491	0.505	0.461	0.499	0.030
Rented	proportion of farm acreage rented in 2009	0.509	0.345	0.424	0.365	0.086
Grain Crops	proportion of farm acreage devoted to grain crops in 2009	0.967	0.085	0.858	0.269	0.108**

TABLE A-4. Continued.

Variable	Definition	Enrolled (N = 55 farmers) Mean	Enrolled Std. Dev	Non-Enrolled (N = 484 farmers) Mean	Non-Enrolled Std. Dev	Diff in Means
Slope	proportion of farm acreage with slope 0%-2%	0.468	0.361	0.570	0.385	-0.103
	proportion of farm acreage with slope 2%-6%	0.447	0.335	0.375	0.365	0.072
	proportion of farm acreage greater than 6% slope	0.086	0.145	0.055	0.138	0.030
Farm Size	natural log of total farm acreage operated in 2009	6.296	0.910	5.757	1.035	0.539**
Stream	=1 if a river or stream borders or runs through the property	0.564	0.501	0.597	0.491	-0.033
Livestock	=1 if managed livestock in 2009	0.400	0.494	0.490	0.500	-0.090

Note: Statistical significance: 99% (**), 95% (*).

TABLE A-5. Summary Statistics on Covariates for Enrolled and Non-Enrolled Farmers for Grass Waterways.

		Enrolled (N = 146 farmers)		Non-Enrolled (N = 380 farmers)		
Variable	**Definition**	**Mean**	**Std. Dev**	**Mean**	**Std. Dev**	**Diff in Means**
Farm Revenue	=1 if farm revenue exceeded $250,000 in 2009	0.377	0.486	0.268	0.444	0.108*
Farm Horizon	=1 if farm will be operated by family within the next 5 years	0.904	0.295	0.868	0.338	0.036
Age	age	56.404	10.725	57.358	11.825	-0.954
Education	=1 if education exceeds high school	0.479	0.501	0.405	0.492	0.074
Soil type	=1 if dominant soil texture is clay	0.801	0.400	0.739	0.440	0.062
	=1 if dominant soil texture is loam or sandy	0.199	0.400	0.261	0.440	-0.062
Household Income	=1 if 0% - 10% of household income comes from farming	0.226	0.420	0.192	0.394	0.034
	=1 if 10% - 50% of household income comes from farming	0.315	0.466	0.337	0.473	-0.022
	=1 if more than 50% of household income comes from farming	0.459	0.500	0.471	0.500	-0.012
Rented	proportion of farm acreage rented in 2009	0.460	0.336	0.409	0.363	0.051
Grain Crops	proportion of farm acreage devoted to grain crops in 2009	0.941	0.147	0.851	0.279	0.090**

TABLE A-5. Continued.

Variable	Definition	Enrolled (N = 146 farmers) Mean	Enrolled (N = 146 farmers) Std. Dev	Non-Enrolled (N = 380 farmers) Mean	Non-Enrolled (N = 380 farmers) Std. Dev	Diff in Means
Slope	proportion of farm acreage with slope 0%-2%	0.483	0.369	0.620	0.384	-0.136**
	proportion of farm acreage with slope 2%-6%	0.451	0.358	0.333	0.362	0.118**
	proportion of farm acreage greater than 6% slope	0.066	0.149	0.048	0.127	0.018
Farm Size	natural log of total farm acreage operated in 2009	6.141	0.928	5.707	1.072	0.434**
Stream	=1 if a river or stream borders or runs through the property	0.685	0.466	0.561	0.497	0.0124**
Livestock	=1 if managed livestock in 2009	0.411	0.494	0.508	0.501	-0.097*

Note: Statistical significance: 99% (**), 95% (*).

TABLE A-6. Summary Statistics on Covariates for Enrolled and Non-Enrolled Farmers for Filter Strips.

		Enrolled (N = 93 farmers)		Non-Enrolled (N = 451 farmers)		
Variable	**Definition**	**Mean**	**Std. Dev**	**Mean**	**Std. Dev**	**Diff in Means**
Farm Revenue	=1 if farm revenue exceeded $250,000 in 2009	0.452	0.500	0.262	0.440	0.190**
Farm Horizon	=1 if farm will be operated by family within the next 5 years	0.925	0.265	0.863	0.345	0.062
Age	age	56.559	10.097	57.098	11.890	-0.538
Education	=1 if education exceeds high school	0.548	0.500	0.392	0.489	0.156**
Soil type	=1 if dominant soil texture is clay	0.763	0.427	0.769	0.422	-0.006
	=1 if dominant soil texture is loam or sandy	0.237	0.427	0.231	0.422	0.006
Household Income	=1 if 0% - 10% of household income comes from farming	0.194	0.397	0.197	0.398	-0.004
	=1 if 10% - 50% of household income comes from farming	0.323	0.470	0.328	0.470	-0.006
	=1 if more than 50% of household income comes from farming	0.484	0.502	0.475	0.500	0.009
Rented	proportion of farm acreage rented in 2009	0.502	0.328	0.419	0.364	0.083*
Grain Crops	proportion of farm acreage devoted to grain crops in 2009	0.947	0.112	0.856	0.273	0.091**

TABLE A-6. Continued.

		Enrolled (N = 93 farmers)		Non-Enrolled (N = 451 farmers)		
Variable	**Definition**	**Mean**	**Std. Dev**	**Mean**	**Std. Dev**	**Diff in Means**
Slope	proportion of farm acreage with slope 0%-2%	0.574	0.357	0.570	0.391	0.004
	proportion of farm acreage with slope 2%-6%	0.365	0.330	0.377	0.369	-0.012
	proportion of farm acreage greater than 6% slope	0.061	0.118	0.053	0.136	0.008
Farm Size	natural log of total farm acreage operated in 2009	6.120	0.917	5.746	1.068	0.374**
Stream	=1 if a river or stream borders or runs through the property	0.839	0.370	0.517	0.500	0.322**
Livestock	=1 if managed livestock in 2009	0.473	0.502	0.492	0.500	-0.019

Note: Statistical significance: 99% (**), 95% (*).

TABLE A-7. Estimated Coefficients from Probit Model to Compute Propensity Scores for Cost-Share Enrollment in Conservation Tillage.

Variable	**Estimated Coeff.**	**Std. Error**
Farm Revenue	0.218	0.192
Farm Horizon	0.001	0.226
Age	0.007	0.007
Education	0.301*	0.137
Soil Type: Loam or Sandy	0.052	0.160
Medium Income	-0.209	0.194
High Income	-0.282	0.200
Rented	0.020	0.217
Grain Crops	1.022*	0.432
Medium Slope	0.201	0.189
High Slope	1.127*	0.456
Farm Size	0.110	0.108
Stream	0.084	0.142
Livestock	-0.095	0.144
Constant	-3.167**	0.779
Log Likelihood	-226.411	

Note: Statistical significance: 99% (**), 95% (*).

TABLE A-8. Estimated Coefficients from Probit Model to Compute Propensity Scores for Cost-Share Enrollment in Cover Crops.

Variable	Estimated Coeff.	Std. Error
Farm Revenue	0.001	0.281
Farm Horizon	-0.383	0.334
Age	-0.017	0.010
Education	0.248	0.216
Soil Type: Loam or Sandy	0.106	0.248
Medium Income	0.044	0.349
High Income	0.437	0.322
Rented	0.001	0.336
Grain Crops	-0.981*	0.418
Medium Slope	0.072	0.295
High Slope	0.343	0.673
Farm Size	0.193	0.160
Stream	0.018	0.221
Livestock	0.686**	0.250
Constant	-1.709	0.959
Log Likelihood	-86.500	

Note: Statistical significance: 99% (**), 95% (*).

TABLE A-9. Estimated Coefficients from Probit Model to Compute Propensity Scores for Cost-Share Enrollment in Hayfields.

Variable	Estimated Coeff.	Std. Error
Farm Revenue	0.077	0.353
Age	0.012	0.011
Education	0.299	0.231
Soil Type: Loam or Sandy	0.021	0.279
Medium Income	-0.054	0.304
High Income	0.020	0.311
Rented	-0.696	0.413
Grain Crops	-0.685	0.395
Medium Slope	0.430	0.310
High Slope	1.393*	0.647
Farm Size	-0.067	0.148
Stream	-0.035	0.236
Livestock	-0.010	0.242
Constant	-1.868	0.958
Log Likelihood	-71.929	

Note: Statistical significance: 99% (**), 95% (*).

TABLE A-10. Estimated Coefficients from Probit Model to Compute Propensity Scores for Cost-Share Enrollment in Grass Waterways.

Variable	**Estimated Coeff.**	**Std. Error**
Farm Revenue	0.146	0.174
Farm Horizon	0.058	0.209
Age	-0.007	0.006
Education	0.142	0.127
Soil Type: Loam or Sandy	-0.200	0.149
Medium Income	-0.341	0.179
High Income	-0.472**	0.184
Rented	-0.215	0.205
Grain Crops	1.068**	0.379
Medium Slope	0.616**	0.172
High Slope	0.962*	0.478
Farm Size	0.228*	0.099
Stream	0.284*	0.129
Livestock	-0.148	0.135
Constant	-2.59**	0.693
Log Likelihood	-279.606	

Note: Statistical significance: 99% (**), 95% (*).

TABLE A-11. Estimated Coefficients from Probit Model to Compute Propensity Scores for Cost-Share Enrollment in Grass Waterways.

Variable	**Estimated Coeff.**	**Std. Error**
Farm Revenue	0.580**	0.194
Farm Horizon	0.203	0.251
Age	0.004	0.007
Education	0.373**	0.142
Soil Type: Loam or Sandy	-0.099	0.168
Medium Income	-0.118	0.206
High Income	-0.331	0.213
Rented	0.143	0.228
Grain Crops	1.657**	0.503
Medium Slope	-0.032	0.199
High Slope	0.333	0.546
Farm Size	-0.045	0.103
Stream	0.919**	0.160
Livestock	0.128	0.156
Constant	-3.509**	0.854
Log Likelihood	-210.595	

Note: Statistical significance: 99% (**), 95% (*).

TABLE A-12. Average Treatment Effect on % ATT using Propensity Score Kernel Matching (Kernel Type: Gaussian, Bandwidth = 0.02).

Conservation Tillage	**Estimate**	**Std. Error**	**95% Bootstrapped CI**	
ATT	0.1351	0.0338	0.0693	0.2019
% ATT	17.3	3.9	9.5	24.9
Matched enrolled farmers = 86, Matched non-enrolled farmers = 476				
Cover Crops	**Estimate**	**Std. Error**	**95% Bootstrapped CI**	
ATT	0.2220	0.0500	0.1364	0.3216
% ATT	84.8	9.1	66.1	95.5
Matched enrolled farmers = 24, Matched non-enrolled farmers = 581				
Hayfield Establishment	**Estimate**	**Std. Error**	**95% Bootstrapped CI**	
ATT	0.2232	0.0677	0.0690	0.3398
% ATT	91.1	9.8	62.6	96.4
Grid Sampling	**Estimate**	**Std. Error**	**95% Bootstrapped CI**	
ATT	0.5067	0.0580	0.3503	0.5788
% ATT	66.7	5.9	50.3	73.4
Matched enrolled farmers = 54, Matched non-enrolled farmers = 484				
Grass Waterways	**Estimate**	**Std. Error**	**95% Bootstrapped CI**	
ATT	0.0121	0.0023	0.0078	0.0165
% ATT	65.5	6.7	49.8	76.0
Matched enrolled farmers = 144, Matched non-enrolled farmers = 380				
Filter Strips	**Estimate**	**Std. Error**	**95% Bootstrapped CI**	
ATT	0.0101	0.0020	0.0064	0.0141
% ATT	89.6	5.7	74.7	95.9
Matched enrolled farmers = 92, Matched non-enrolled farmers = 451				

TABLE A-13. Bootstrapped 95% Confidence Intervals for Pair-wise Differences in %ATT using Propensity Score Kernel Matching (Kernel Type: Gaussian, Bandwidth = 0.02) (Row minus Column).

	Conservation Tillage	**Cover Crops**	**Hayfield Establishment**	**Grid Sampling**	**Grass Waterways**	**Filter Strips**
Conservation Tillage	-	[-82.0, -46.4]*	[-81.9, -45.1]*	[-58.0, -30.7]*	[-60.5, -29.1]*	[-82.6, -55.4]*
Cover Crops	[46.4, 82.0]*	-	[-26.0, 24.9]	[0.6, 40.1]*	[-2.9, 39.5]	[-22.9, 15.0]
Hayfield Establishment	[45.1, 81.9]*	[-24.9, 26.0]	-	[-2.3, 41.1]	[-3.8, 40.7]	[-26.0, 16.1]
Grid Sampling	[30.7, 58.0]*	[-40.1, -0.6]*	[-41.1, 2.3]	-	[-18.9, 16.0]	[-40.6, -8.8]*
Grass Waterways	[29.1, 60.5]*	[-39.5, 2.9]	[-40.7, 3.8]	[-16.0, 18.9]	-	[-41.7, -5.9]*
Filter Strips	[55.4, 82.6]*	[-15.0, 22.9]	[-16.1, 26.0]	[8.8, 40.6]*	[5.9, 41.7]*	-

Note: * denotes statistical significance of the bootstrapped 95% confidence interval

APPENDIX B

ADDITIONAL RESULTS FOR CHAPTER III

TABLE B-1. Bivariate Probit Model on Vineyard and Reservoir Development for the Period 1973-1993.

Variable		
Vineyard development equation	**Coefficient**	**Standard Error**
Average slope	-0.0293**	0.0042
Growing-degree days	0.1992**	0.0387
Elevation (x1000)	-0.1351	0.3303
Floodplain	0.1247	0.1335
Distance to nearest highway	-0.0282	0.0174
Riparian access [a]		
Mainstem	0.6964**	0.1743
Seasonal stream	-0.0322	0.0676
Geology type [b]		
Old alluvium	0.0288	0.1133
Volcanic	-0.2695*	0.1333
Franciscan	-0.2568	0.1371
Constant	-3.8563**	0.6857
Reservoir construction equation	**Coefficient**	**Standard Error**
Average slope	-0.0360**	0.0036
Growing-degree days	0.0261	0.0164
Elevation (x1000)	0.6673*	0.2650
Floodplain	0.1025	0.1681
Distance to nearest highway	-0.0188	0.0129
Riparian access [a]		
Mainstem	-0.1072	0.2468
Seasonal stream	0.3453**	0.0731

TABLE B-1. Continued.

Reservoir construction equation	Coefficient	Standard Error
Geology type [b]		
Old alluvium	0.5555**	0.1593
Volcanic	0.5332**	0.1732
Franciscan	0.6867**	0.1828
Constant	-1.9657**	0.3274
ρ	0.3440**	0.0457
N = 3561	Ln L= -1784.15	

* Significant at the 5% level; ** significant at the 1% level
[a] Riparian access baseline type = No stream access
[b] Geology baseline type = Young alluvium

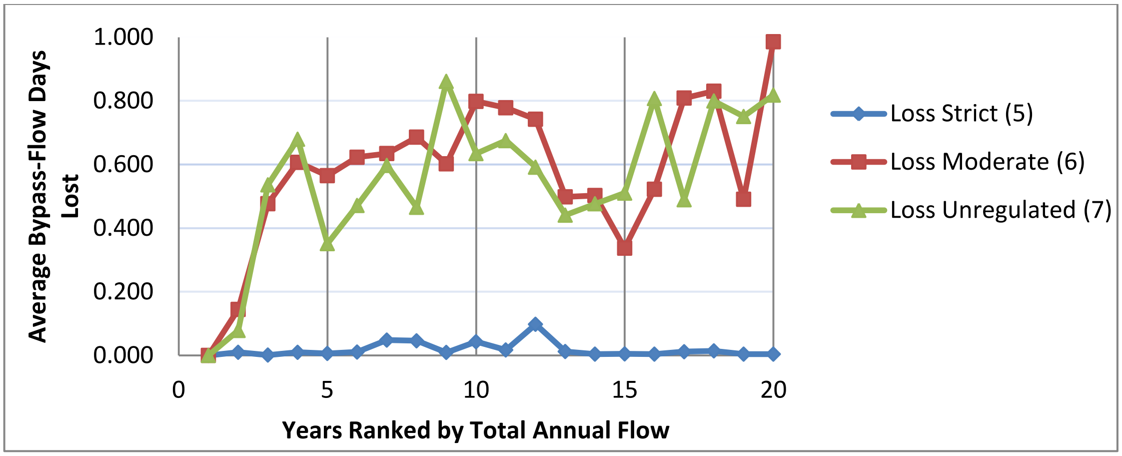

FIGURE B-1. Average loss in bypass-flow days across precipitation years for scenarios 5 (Strict), 6 (Moderate), and 7 (Unregulated) (only impaired streams below EOA).

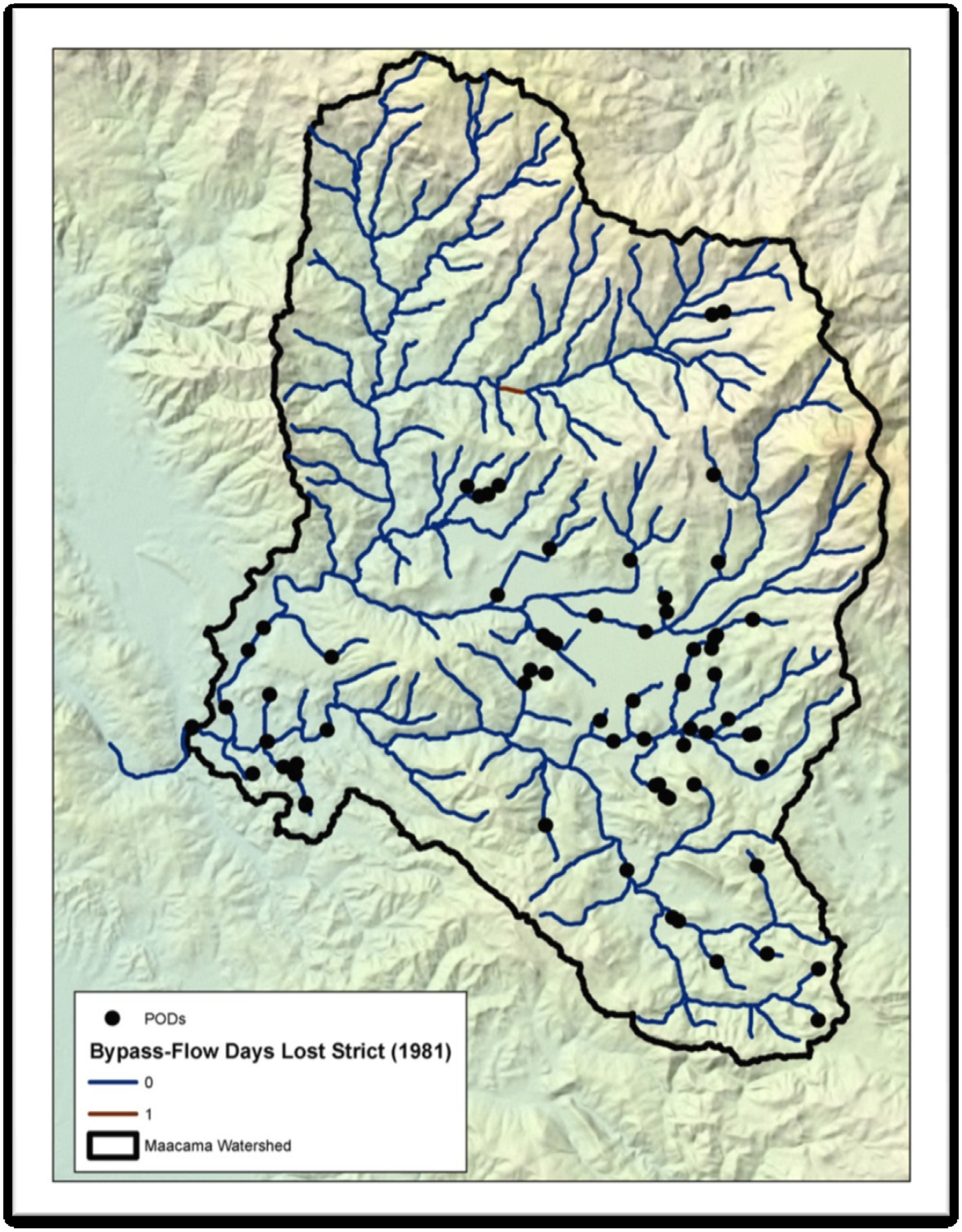

FIGURE B-2. Map of losses in bypass-flow days under the strict policy scenario (5) for streams in the Maacama watershed for the dry year 1981 (ranked 5).

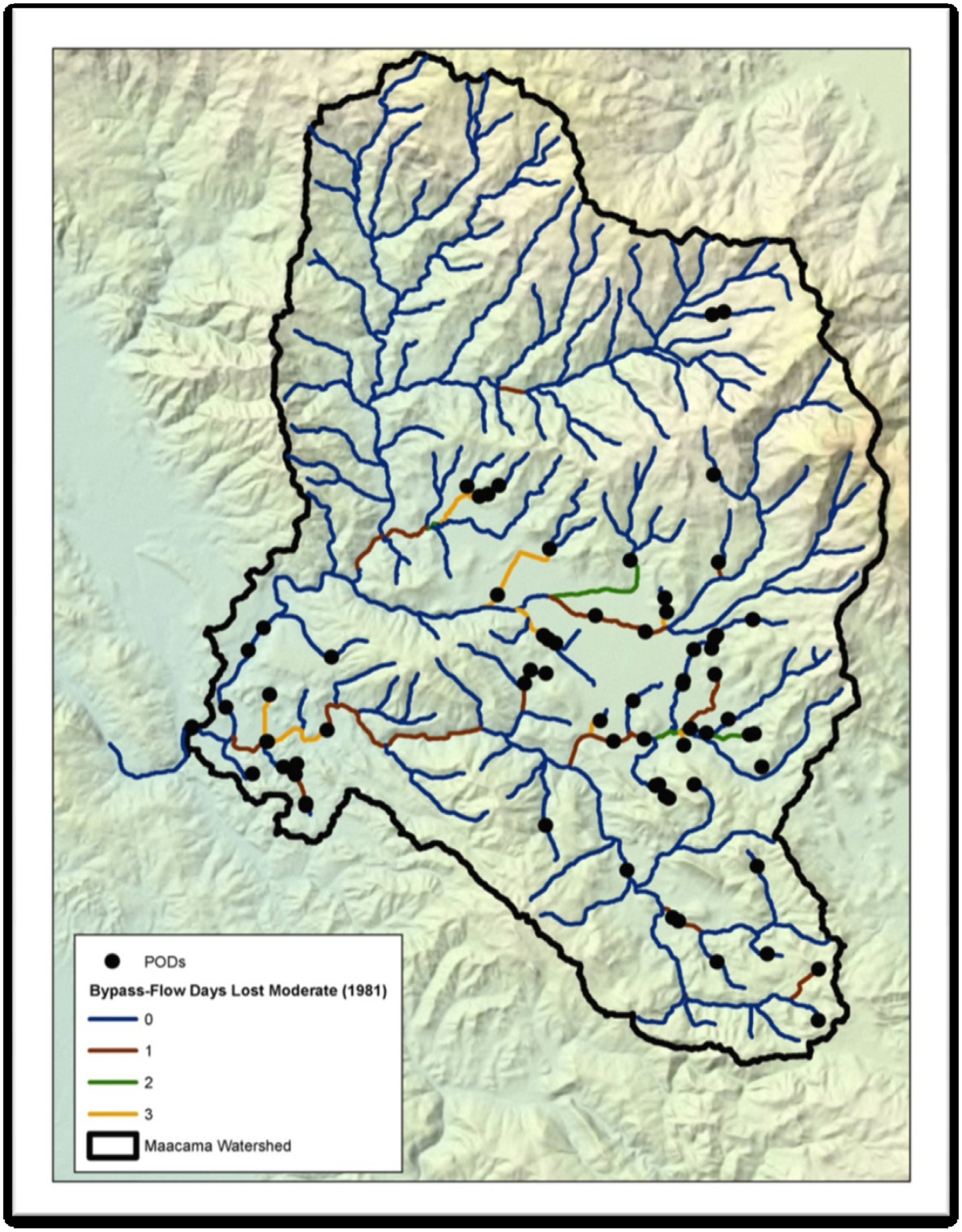

FIGURE B-3. Map of losses in bypass-flow days under the moderate policy scenario (6) for streams in the Maacama watershed for the dry year 1981 (ranked 5).

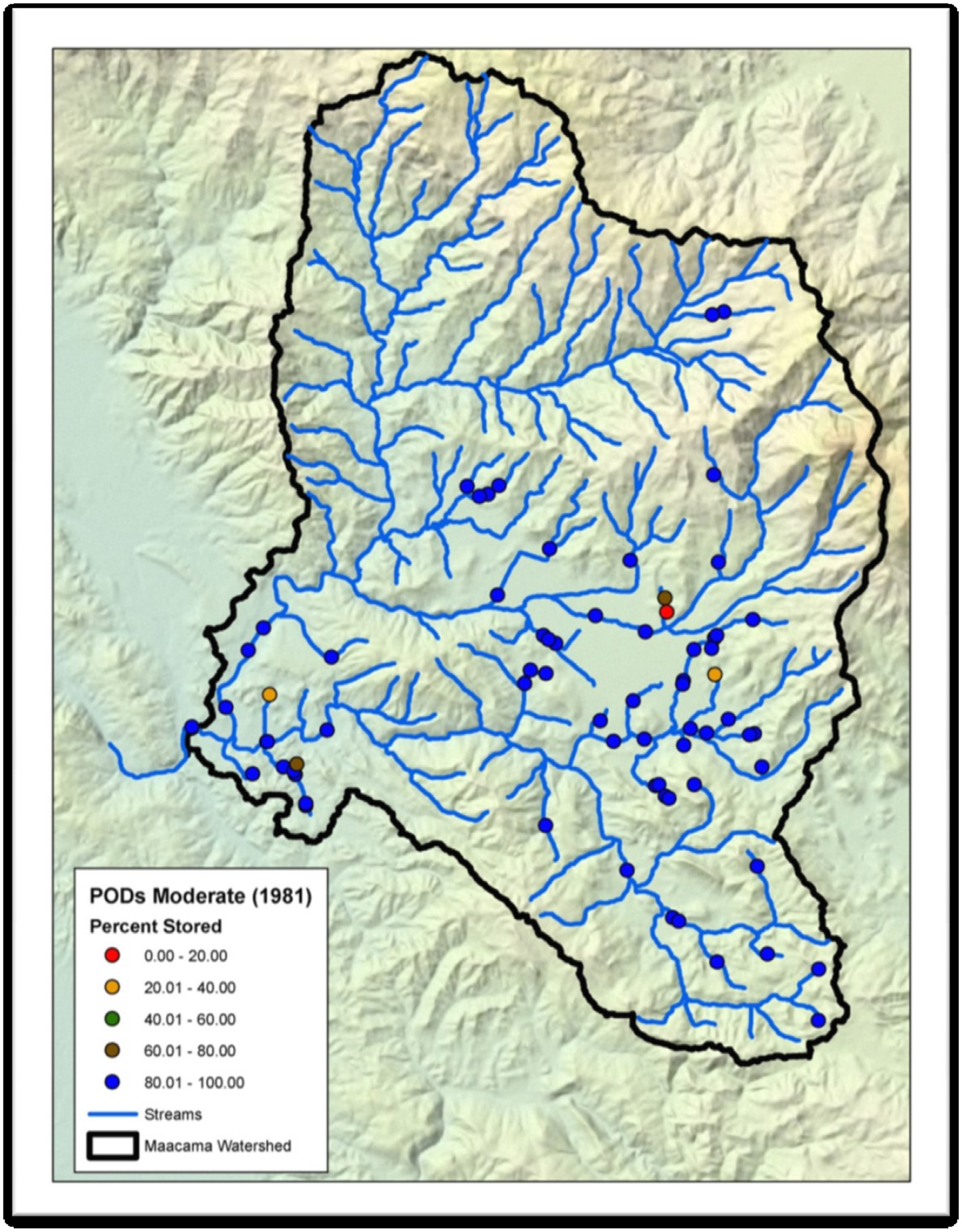

FIGURE B-4. Map of percent storage filled for reservoirs in the Maacama watershed for the moderate policy scenario (6), for the dry year 1981 (ranked 5).

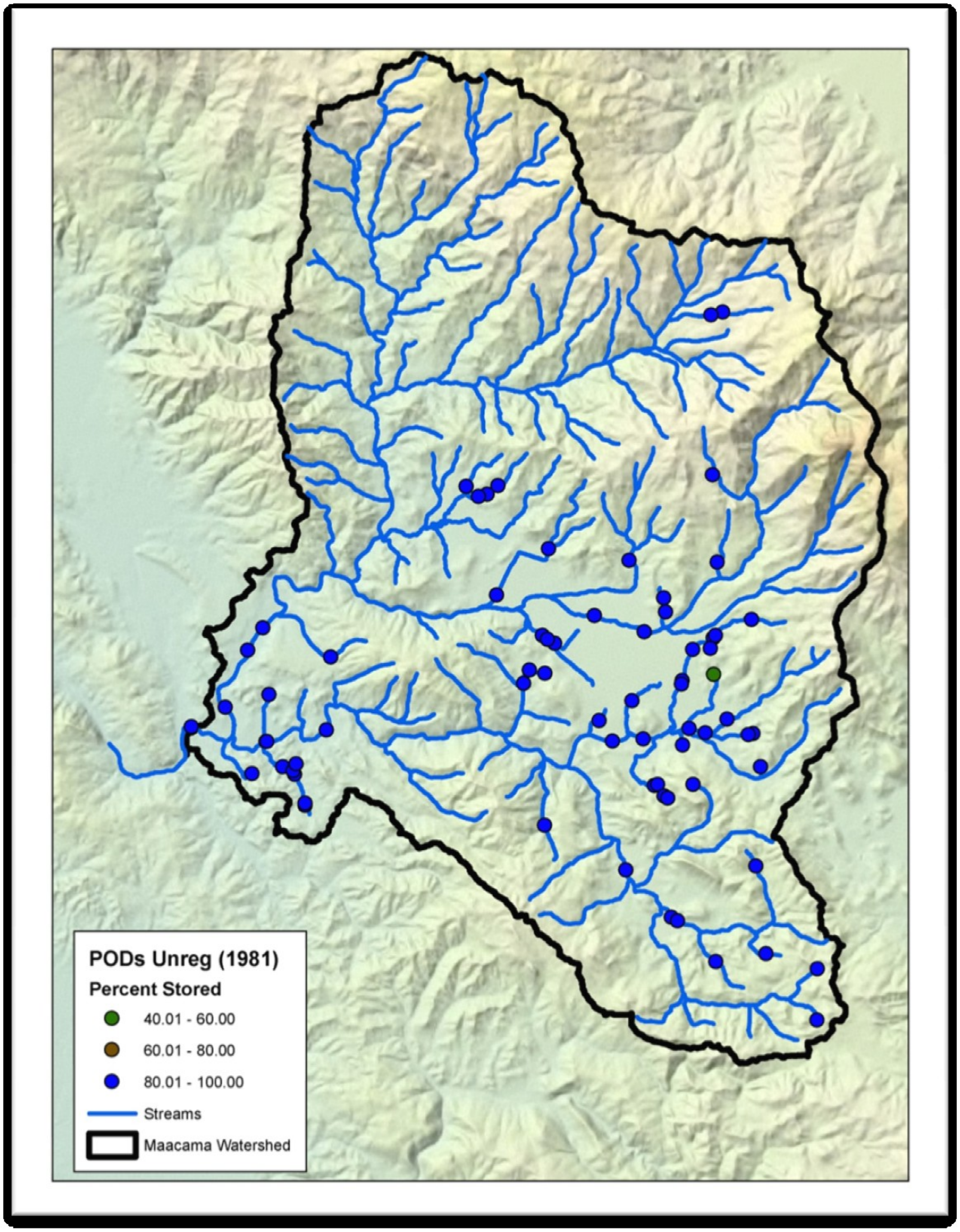

FIGURE B-5. Map of percent storage filled for reservoirs in the Maacama watershed for the unregulated policy scenario (7), for the dry year 1981 (ranked 5).

APPENDIX C

SURVEY INSTRUMENT

Dear Ohio Farm Operator,

The Ohio Field Office of the National Agricultural Statistics Service (NASS) is conducting a survey of Ohio farmers on behalf of The Ohio State University (OSU) and Texas A&M University (TAMU). This survey aims to learn about conservation practices, including those paid for entirely by farmers and those receiving cost-share support. There is growing interest in the relationship between agricultural practices and water quality improvements. This study will provide important information to guide future policies and programs.

We ask that the principal farm operator answer this survey. The survey is expected to take about 15 minutes to complete. Your participation is voluntary, and you may discontinue at anytime. Please return it in the enclosed postage paid return envelope.

As a token of our appreciation, **one respondent will be chosen at random to receive a $200 gift card from Home Depot**. If you complete the survey and are selected as the winner, the NASS will send you the gift card prize that we will provide. You do not need to do anything else to be eligible for the prize. In addition, completing the survey is a benefit to you since you are helping policy makers and farm leaders make better decisions about designing conservation programs in the future. Even if you have never participated in a cost-share program, your response is extremely valuable to provide accurate information on the range of farming practices.

The information you provide will be completely confidential. The Ohio Field Office of the NASS will conduct the survey and no identifying information that can be linked to your individual farm will be provided in the data files given to the researchers at OSU and TAMU. The results from this study will be reported only in aggregate form, such that you and your farm can not be individually identified in any research results. If you have any questions about the risks associated with this survey you can contact The Ohio State University Office of Responsible Research Practices at 614-688-8457.

We thank you for your time and effort in answering the survey. Our contact information is provided below if you have any questions regarding this survey.

Sincerely,

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Instructions

We ask that the principal decision-maker of the farm business answer this survey. We would appreciate that you answer each question with the answer you believe is most representative for your farm.

1. Did you operate a farm business in 2005? ☐ Yes ☐ No
2. Did you operate a farm business in 2009? ☐ Yes ☐ No

If your answer is NO to *either* question 1 or question 2, please stop here and return the uncompleted survey in the enclosed envelope. Postage is paid by the survey project.

Section A: Farmer Characteristics

A1. What is your age? ____________

A2. How many years have you operated a farm? ____________

A3. What was the last year of school you completed?

☐ Did not graduate from high school
☐ High school graduate or GED
☐ Attended college, but did not complete degree
☐ Completed technical school/community college
☐ College graduate
☐ Masters or doctoral degree

A4. Five years from now, which of the following do you think will be most likely?

☐ I will still be operating the farm.
☐ The farm will be operated by one or more relatives (children or other relative).
☐ The farm will be operated by non-related farmer.
☐ The farm will be converted into non-farm use
☐ Do not know

Section B: Farm Operations

B1. Where is most of your farm located?

County: ____________________

Township: __________________

Zip Code: __________________

B2. Is your farm located in the GREAT MIAMI RIVER WATERSHED (including the subwatersheds of the Lower Miami, the Mad, the Upper Miami, or the Stillwater Rivers)?

☐ Yes ☐ No ☐ Don't know

B3. Of the acres in your farming operation, how many were owned and how many were rented?

	Acres in **2009**	Acres in **2005**
Owned by you	________________	________________
Rented from others (cash or share rent)	________________	________________

B4. Of the acres in your farming operation, how many acres were used for each of the following?

Land Use	**Acres in 2009**	**Acres in 2005**
Grain Crops (corn, soybean, wheat etc.)	______________	______________
Hay, forage or pasture	______________	______________
Other crops	______________	______________
Other uses (woodland, wildlife, buildings, etc.)	______________	______________

B5. Of the acres in your farming operation in **2009**, how much falls into each of the following slope classes?

Flat (0-2% slope)	__________ acres
Gently rolling (2-6% slope)?	__________ acres
Hilly (greater than 6%)?	__________ acres

B6. "HEL," or highly erodible land, is land that has an erodibility index of 8 or more as designated by the Natural Resources Conservation Service (NRCS).
Is any of the land that you operate classified as HEL?
☐ Yes ☐ No ☐ Don't know

B7. Is any of the land you operate certified with the Farm Service Agency (FSA) in order to qualify for government payments under Federal support programs?
☐ Yes ☐ No

IF YES, is there any acreage in your conservation plan that is NOT ELIGIBLE for payment under NRCS programs for conservation practices? For example, areas that do not have the required cropping history cannot receive cost-share support for grass waterways.

☐ Yes, some land is NOT eligible ☐ No ☐ Don't know

IF YES, how many acres are NOT eligible for NRCS programs?
___________ acres.

B8. How would you characterize the dominant soil texture on your farm? (mark one)

☐ Clay ☐ Clay loam ☐ Silty loam
☐ Loam ☐ Sandy loam ☐ Sandy

B9. Is there a river or stream (permanent or intermittent) that borders or runs through the largest property that you operate?

☐ Yes ☐ No

IF NO, how far is closest stream or river? ________feet **or** ______ miles

B10. Did you manage livestock on your farm in either 2005 or 2009?

☐ Yes ☐ No

If YES, what was the MAXIMUM number of animals, regardless of ownership, managed by you in during these years?

	Animals in 2009	**Animals in 2005**
Dairy cattle and calves (both dry and in milk)	________________	________________
Beef cattle and calves	________________	________________
Hogs and pigs	________________	________________
Poultry including layers, broilers and turkeys	________________	________________
Horses	________________	________________

B11. What was your gross revenue ($ from farm sales) in 2009?

Under $25,000	☐
$25,000 - $49,999	☐
$50,000 - $99,999	☐
$100,000 - $249,999	☐
$250,000 - $499,999	☐
$500,000 - $999,999	☐
$1,000,000 & over	☐

B12. What percent of your household income was earned FROM FARMING or ranching in 2009?

☐ Low (0-10%)
☐ Moderate (11-50%)
☐ High (51% or more)

B13. Indicate what interactions you have with staff from your county's Soil and Water Conservation District and/or Natural Resources Conservation Service (NRCS) office. (check all that apply)

- ☐ Never
- ☐ Newsletters or bulletins
- ☐ Meetings or seminars
- ☐ Infrequent personal contact (one time per year)
- ☐ Occasional personal contact (several times per year).
- ☐ Frequent personal contact (every month or more)

Section C: Awareness of Conservation Cost-share Programs

In this section we ask you about several programs that provide support for conservation practices.

	A	B	C	D
Program	Have you **heard about** this program?	If YES to A, have you ever **applied** to this program?	If YES to B, have you **received cost-share** payment from this program?	If YES to C, indicate the period(s) during which you **applied** for cost-share from this program.
Environmental Quality Incentive Program (**EQIP**)	☐ Yes ➡ ☐ No	☐ Yes ➡ ☐ No	☐ Yes ➡ ☐ No	☐ Before 2002 ☐ 2003-2005 ☐ 2006–2009
Conservation Reserve Program (**CRP**)	☐ Yes ➡ ☐ No	☐ Yes ➡ ☐ No	☐ Yes ➡ ☐ No	☐ Before 2002 ☐ 2003-2005 ☐ 2006–2009
Conservation Reserve Enhancement Program (**CREP**)	☐ Yes ➡ ☐ No	☐ Yes ➡ ☐ No	☐ Yes ➡ ☐ No	☐ Before 2002 ☐ 2003-2005 ☐ 2006–2009
Conservation Security Program (**CSP**)	☐ Yes ➡ ☐ No	☐ Yes ➡ ☐ No	☐ Yes ➡ ☐ No	☐ Before 2002 ☐ 2003-2005 ☐ 2006–2009
Great Miami River Watershed Water Quality Trading Program (**WQTP**) managed by the Miami Conservancy District	☐ Yes ➡ ☐ No	☐ Yes ➡ ☐ No	☐ Yes ➡ ☐ No	

Section D: Conservation Practices and Cost-share Funding
In this section we ask about several conservation practices that you may have used at some time.

D1. Please complete BOTH columns A and B.
If you have never used a practice, check NO in both columns.

	A	B
Conservation Practice	Did you use this practice in 2009? If so, indicate the extent of the practice.	Have you CHANGED the extent of your use of this practice SINCE 2005?
Conservation Tillage (No-Till or Reduced Tillage so that 30% or more of the soil surface is covered with crop residue after planting)	☐ No ☐ Yes _____ acres	☐ No ☐ Increased by _____acres ☐ Decreased by _____acres
Grass Waterways	☐ No ☐ Yes _____ acres	☐ No ☐ Increased by _____acres ☐ Decreased by _____acres
Filter Strips along streams	☐ No ☐ Yes _____ Length (ft) _____ Avg. Width	☐ No ☐ Increased by _____feet ☐ Decreased by _____feet
Manure Lagoon, Storage Facility, or Livestock Wastewater Collection System	☐ No ☐ Yes _____ gallons	☐ No ☐ Increased by _____gallons ☐ Decreased by _____gallons
Cover Crops planted after row crop harvests	☐ No ☐ Yes _____ acres	☐ No ☐ Increased by _____acres ☐ Decreased by _____acres
Hayfield or Grassland Established as a conservation practice	☐ No ☐ Yes _____ acres	☐ No ☐ Increased by _____acres ☐ Decreased by _____acres
Grid Sampling and Reduced Fertilizer Application	☐ No ☐ Yes _____ acres	☐ No ☐ Increased by _____acres ☐ Decreased by _____acres

We also are interested in whether you received cost-share support for the practices in question D. We list some of the sources that may have provided financial assistance.

- **EQIP:** Environmental Quality Incentive Program
- **CRP:** Conservation Reserve Program
- **CREP:** Conservation Reserve Enhancement Program
- **CSP:** Conservation Security Program
- **WQTP:** Great Miami River Watershed Water Quality Trading Program

D2. Complete only the rows for practices that you indicated you have used in QUESTION D.

	A	B	C	D
Conservation Practice	Have you ever RECEIVED cost-share support to implement this practice?	IF YES TO A, indicate the name(s) of the programs. (*Check all that apply*).	IF YES TO A, during which period(s) did you ENROLL in a cost share program? (*Check all that apply*).	IF YES TO A, indicate the dollar amount **or** cost-share percentage of your MOST RECENT agreement.
Conservation Tillage	☐ Yes ☐ No	☐ EQIP ☐ CRP ☐ CREP ☐ CSP ☐ WQTP ☐ Other	☐ 2002 or before ☐ 2003 – 2005 ☐ 2006 - present	$_____ **or** ______% ☐ Don't know
Grass Waterways	☐ Yes ☐ No	☐ EQIP ☐ CRP ☐ CREP ☐ CSP ☐ WQTP ☐ Other	☐ 2002 or before ☐ 2003 – 2005 ☐ 2006 - present	$_____ **or** _____% ☐ Don't know
Filter Strips along streams	☐ Yes ☐ No	☐ EQIP ☐ CRP ☐ CREP ☐ CSP ☐ WQTP ☐ Other	☐ 2002 or before ☐ 2003 – 2005 ☐ 2006 - present	$_____ **or** _____% ☐ Don't know
Manure Lagoon, Storage Facility or Livestock Wastewater Collection System	☐ Yes ☐ No	☐ EQIP ☐ CRP ☐ CREP ☐ CSP ☐ WQTP ☐ Other	☐ 2002 or before ☐ 2003 – 2005 ☐ 2006 - present	$_____ **or** _____% ☐ Don't know
Cover Crops planted after row crop harvests	☐ Yes ☐ No	☐ EQIP ☐ CRP ☐ CREP ☐ CSP ☐ WQTP ☐ Other	☐ 2002 or before ☐ 2003 – 2005 ☐ 2006 - present	$_____ **or** _____% ☐ Don't know
Conversion of cropland to hay or grassland	☐ Yes ☐ No	☐ EQIP ☐ CRP ☐ CREP ☐ CSP ☐ WQTP ☐ Other	☐ 2002 or before ☐ 2003 – 2005 ☐ 2006 - present	$_____ **or** _____% ☐ Don't know
Grid Sampling and Reduced Fertilizer Application	☐ Yes ☐ No	☐ EQIP ☐ CRP ☐ CREP ☐ CSP ☐ WQTP ☐ Other	☐ 2002 or before ☐ 2003 – 2005 ☐ 2006 - present	$____________ **or** _________% ☐ Don't know

Section E: Views on Environmental Issues

What is your immediate reaction to the following statements?	Strongly Agree	Agree	Neutral	Disagree	Strongly Disagree
E1. No one has the right to tell farmers what practices to use on their land.	☐	☐	☐	☐	☐
E2. Farmers have a responsibility to society to reduce the causes of water pollution that originate on their farms.	☐	☐	☐	☐	☐
E3. Water pollution is a major problem in Ohio.	☐	☐	☐	☐	☐
E4. When compared with conventional tillage, CONSERVATION TILLAGE tends to make yields vary more from one year to the next	☐	☐	☐	☐	☐
E5. It would be acceptable for a city to pay a farmer to reduce water pollution instead of reducing pollution directly at its waste water treatment plant.	☐	☐	☐	☐	☐
E6. I often try new methods on my farm, before most of other farmers in my region.	☐	☐	☐	☐	☐
E7. Applying for cost-share tends to be a time consuming process.	☐	☐	☐	☐	☐
E8. The design specifications and implementation required by cost-share programs are quite restrictive.	☐	☐	☐	☐	☐
E9.Federal cost-share programs require contracts with long-term commitments, making them less attractive.	☐	☐	☐	☐	☐

THE FOLLOWING QUESTIONS ARE OPTIONAL:
If you have any comments, you may add them here.

If you are willing, please provide a contact name and phone number in case there is a need to clarify some of your survey responses. Please note that this information would be given to the researchers at The Ohio State University and Texas A&M University, meaning that your responses to the survey would no longer be anonymous.

Thank you for your cooperation.
Please return the survey in the enclosed postage paid return envelope to
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The Ohio State University
Department of Agricultural, Environmental, and Development Economics
2120 Fyffe Rd.
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