Assessing the Hydroecological Effects of Stream Restoration

By

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ABSTRACT

This dissertation consists of three studies undertaken to document and quantify the hydroecological effects of stream restoration. The study was conducted at the Bear Creek Meadow, a particularly well-documented "pond and plug" type stream restoration project, located in northeastern California.

The first study investigates the effects of stream restoration on hydrologic processes. A hydrologic model of the 230 ha meadow was developed, calibrated, validated, and used to simulate the system under pre- and post-restoration topographic conditions. Simulation results document three general hydrologic responses to the meadow restoration effort: 1) increased groundwater levels and volume of subsurface storage; 2) increased frequency and duration of floodplain inundation and decreased magnitude of flood peaks; and 3) decreased annual runoff and duration of baseflow.

The second study explores the relationship between temporally varying water-table elevations and plant community distributions in the restored riparian meadow. Vegetation was sampled throughout the meadow and TWINSPAN was used to classify the vegetation into four community types: *Eleocharis macrostachya / Eleocharis acicularis, Downingia bacigalupii / Psilocarphus brevissimus, Carex nebrascensis / Juncus balticus,* and *Poa pratensis / Bromus japonicus*. The hydrologic model was used to simulate a three-year time series of water-table depth for each plot, and nonmetric multidimensional scaling was utilized to investigate the relationships between community types and hydrologic variables. Community types were distributed along the hydrologic

gradient at reasonably similar positions to those found in previous studies, however the range of water-table depths in this meadow was greater than previously observed, presumably due to the higher temporal resolution of water-table measurements, in addition to the intermittent nature of stream flow in Bear Creek and its substantial control of water-table elevations.

In the final study, the changes in distribution of several commonly occurring herbaceous plant species were investigated. Vegetation models were developed where the probability of occurrence of a particular species was predicted as a function of growing season water-table depth and range. These vegetation models were used in concert with the hydrologic model in order to predict the spatial distribution of individual plant species for pre- and post-restoration topographic-hydrologic conditions. Simulation results indicate an increase in the spatial distribution of obligate wetland, and facultative wetland plant species, as well as a decrease in the distribution of facultative upland and obligate upland plant species. In combination, the results of these studies support and quantify the hypothesis that "pond and plug" type stream restoration projects have the capacity to re-establish hydrologic processes necessary to sustain riparian systems.

CHAPTER 1 - Quantifying the Hydrological Effects of Stream Restoration in a Montane Meadow

ABSTRACT

Stream restoration efforts, particularly within meadow systems, increasingly rely on "pond and plug" type methods in which (a) alluvial materials are excavated from the floodplain, forming ponds; (b) excavated alluvial materials are used to plug incised channels; and (c) smaller dimension channels are restored to the floodplain surface. A commonly stated objective of these efforts is to restore ecologically significant hydrological processes to degraded riparian systems. However, little research has been conducted to evaluate and quantify the restoration of these hydrological processes. Direct comparisons of pre- and post-restoration hydrological observations are often misleading due to inter-annual climatic variability. To overcome this issue and accurately quantify the hydrological effects of restoration, we developed, calibrated and validated a hydrological model of a 230 ha mountain meadow along a 3.6 km restored reach of Bear Creek in northeastern California. We then applied the model to simulate the pre- and post-restoration scenarios by altering the floodplain topography and stream channel networks. Our results document three general hydrological responses to the meadow restoration effort: 1) increased groundwater levels and volume of subsurface storage; 2) increased frequency/duration of floodplain inundation and decreased magnitude of flood peaks; and 3) decreased annual runoff and duration of baseflow. This study supports and quantifies the hypothesis that "pond and plug" type stream restoration projects have the capacity to re-establish hydrological processes necessary to sustain riparian systems. In addition, the results of this study can be used to improve quantitative objectives for "pond and plug" type stream restoration activities in similar settings.

INTRODUCTION

An increased appreciation of the multitude of environmental services that healthy stream systems provide has prompted large investments in restoring degraded watercourses in the United States (U.S. Environmental Protection Agency and U.S. Department of Agriculture 1998) and throughout the world (Moser et al. 1997). An exponential increase in river restoration projects over the last decade (Bernhardt et al. 2005), has made stream restoration one of the most visible elements of hydrological sciences (Malakoff 2004) and placed river restoration at the forefront of applied hydrological sciences (Wohl et al. 2005). An increasingly popular stream restoration strategy is the “pond and plug” method, in which (a) alluvial materials are excavated from the floodplain, forming ponds; (b) excavated alluvial materials are used to plug incised channels; and (c) smaller dimension channels are restored to the floodplain surface. Objectives of “pond and plug” projects typically include: improved aesthetics, improved land productivity, improved aquatic and terrestrial habitats, decreased streambank erosion and downstream sediment delivery, increased water-table elevations, and enhanced baseflow conditions (Benoit and Wilcox 1997, Rosgen 1997). Despite the popularity of this approach, only a small number of projects receive sufficient monitoring and assessment to evaluate their effectiveness and to inform future restoration efforts (Bernhardt et al. 2005), seriously limiting advancement in design and implementation.

The purpose of this study is to quantify the hydrological effects of a “pond and plug” type stream restoration. We hypothesize that topographic modification of channels and floodplains, typical of “pond and plug” restoration projects, will result in measurable changes to all surface and subsurface hydrological processes. Hydrological processes of particular interest are the spatial and temporal changes to groundwater (e.g., water-table elevation, range of water-table elevations, and subsurface storage), surface water (e.g., floodplain inundation frequency, area and duration, flood peak attenuation, baseflow duration, and total annual runoff) and atmospheric exchange (e.g., evapotranspiration). Direct comparisons of pre- and post-restoration hydrological observation data can be instructive, yet also can be misleading due to inter-annual climatic variability, which affects many surface and subsurface hydrological processes. In order to quantify the effects specific to stream restoration, two linked surface water-groundwater numerical models were developed with the MIKE SHE hydrological modeling system using a well-documented "pond and plug" stream restoration project as an example. The two models (incised vs. restored) differ only in the size, shape and alignment of the channels and the presence/absence of ponds on the floodplain surface. Identical boundary conditions are used to simulate the hydrological effects and allow for a direct comparison of the stream restoration's effects on surface and subsurface hydrological processes. The results of this work offer new insight into the impact of this restoration technique on meadow hydrology. In addition, the methods used can guide future efforts to monitor and assess stream restoration efforts.

STUDY AREA

Geology and Hydrology

Bear Creek Meadow (meadow) is a low-gradient alluvial floodplain ~100 km northeast of Redding in northern California, USA (Figure 1.1). The meadow is located at an elevation of ~1010 m, and is situated at the bottom of the ~218 km^2 Bear Creek watershed, immediately upstream of the confluence of Bear Creek with the Fall River, the largest spring-fed river system in California (Grose 1996), and among the largest spring-fed river systems in the United States (Meinzer 1927, Rose et al. 1996).

The meadow is approximately three km long, one km wide, 230 ha in size, and is situated at the northwestern margin of the Fall River Valley. The meadow is bounded on the south and west by the steep slopes of Soldier Mountain, to the north and east by the low-relief basaltic flows of the Medicine Lake Highlands, and to the southeast by the Fall River Valley. The head of the meadow lies at the base of a relatively steep, heavily-forested bedrock reach. The Fall River Valley is underlain by lacustrine deposits consisting of clay, silt and sand. In the meadow, the lacustrine deposits are overlain by 0.5 - 2 m of deltaic sands and gravels, and 1 - 3 m of floodplain silty loam soils (Grose 1996). The meadow vegetation is dominated by grasses, sedges and rushes, in addition to stands of Oregon ash lining inactive stream channels.

The climate of the Fall River Valley is semi-arid, receiving an annual average of 508 mm of precipitation (California Irrigation Management System data for McArthur for water years 1984-2006). Most precipitation in the Fall River Valley occurs as rainfall in late fall-early spring. Higher elevation areas of the Bear Creek watershed, located to the north and west of the meadow, receive considerably more precipitation, which occurs as snow and rain in late fall-early spring.

The hydrological system of the study area is complex, consisting of seasonal or intermittent surface-water inflow from Bear Creek and Dana Creek and perennial spring discharge from the Fall River spring system (Figure 1.1). The latter system is fed by meteoric water, which falls on the Medicine Lake Highlands, perches on low-permeability lacustrine deposits, flows south through fractured basalt and discharges at the downstream end of the meadow (Rose et al. 1996). These springs form the headwaters of the Fall River and several short tributaries (i.e., Mallard Creek and Lower Dana Creek). The local groundwater system is unconfined and down-valley fluxes occur primarily through the deltaic silts, sands and gravels of the shallow subsurface.

Surface-water input to the meadow is supplied primarily by the intermittent Bear Creek and secondarily by the intermittent Dana Creek, which bounds the southwestern edge of the lower meadow (Figure 1.1). Stream discharge results from spring snowmelt, and fall, winter, and spring rain events including episodic rain-on-snow events. In the seven years following the restoration in 1999 that is described below, peak discharge in Bear Creek measured at the head of the meadow ranged from 3.11 - 20.73 m^3s^{-1} (Figure 1.2). Based upon a flow frequency analysis of 15 discontinuous years of annual peak discharge data available, the two-, five- and ten-year recurrence interval discharges are 12.7 m^3s^{-1}, 29.6 m^3s^{-1} and 48.2 m^3s^{-1}, respectively.

Anthropogenic Disturbance, Incision, Widening and Restoration

Prior to restoration, the meadow was channelized and overgrazed (Poore 2003), resulting in degradation of both aquatic and terrestrial ecosystems of the meadow and the Fall River immediately downstream (Spencer and Ksander 2002). After several years of pre-restoration data collection and consultation, the meadow's incised channels were restored in 1999 as a joint venture between California Department of Fish and Game and the private landowner. The restoration design followed the "Natural Channel Design Using a Geomorphic Approach" method developed by David Rosgen (Rosgen 1996, Malakoff 2004). A "priority 1" approach (Rosgen 1997), more commonly referred to as a "pond and plug" strategy was utilized.

Following the usual "pond and plug" method, the incised stream channels were intermittently filled with plugs of locally derived alluvial material. The remaining unfilled incised channel segments were left as ponds, and many were enlarged to provide the fill material necessary to plug portions of the incised channels. When configuring the restored channel, existing remnant channel segments were used when possible, connected by sections of excavated new channel. The restored channel was constructed with reduced width, depth, and cross-sectional area (Figures 1.3 and 1.4, Poore 2003). The restored channel was classified as C4 and E4 types of the Rosgen classification system (Rosgen 1996, Poore 2003). Upon completion, a 3.6 km single thread sinuous channel connected the bedrock controlled upstream reach to the unaltered downstream reach (Figure 1.1). In addition, 17 ha of new ponds (remnant gully segments and fill sources) exist throughout the meadow.

METHODS

Model Development

A numerical hydrological model was developed using the MIKE SHE modeling system (Refsgaard and Storm 1995), which is based upon the Systeme Hydrologique Europeen (SHE) model (Abbott et al. 1986a, b). MIKE SHE is a commercially-available, deterministic, fully-distributed and physically-based modeling system that has been applied to a wide variety of problems where surface water and groundwater are closely linked (for examples see Jayatilaka et al. 1998, Thompson 2004, Sahoo et al. 2006). Using a finite difference methodology, MIKE SHE solves partial differential equations describing the processes of saturated subsurface flow (three-dimensional Boussinesq equation), unsaturated subsurface flow (one-dimensional Richards' equation), channel flow (one-dimensional St. Venant equations), and overland flow (diffusion wave approximation of the two-dimensional St. Venant equations). Channel hydraulics are simulated with the one-dimensional MIKE 11 hydraulic modeling system which is dynamically coupled to the MIKE SHE modeling system. The processes of interception and evapotranspiration are handled with analytical solutions.

Separate MIKE SHE/MIKE 11 models were developed for the pre-project (i.e., incised) and post-project (i.e., restored) scenarios. Initially, a base model of the restored scenario was developed, calibrated and validated. Subsequently the surface topography and channel size and alignments were altered to reflect the incised pre-restoration scenario. The altered surface topography and channel configuration were the only differences between the two models. All other components remained unchanged between the two models. The models were comprised of 2898 30 x 30 m^2 grid squares, representing a total area of 261 ha.

Grose (1996) and three well logs from within the model domain provided the conceptual model of the hydrostratigraphy. The vertical and horizontal extent of the various hydrostratigraphic units were further defined by excavating shallow boreholes with hand augers, excavating test pits with a backhoe, and conducting a three-dimensional survey of the contact of the upper two layers as observed in the restored channel and ponds. Based upon the refined conceptual model, the subsurface component of the model was composed of three layers, with the lower layer a sandy clay, the middle layer a high-permeability alluvial sand and gravel mixture, and the upper layer an alluvial silty-clayey loam.

Slug tests were conducted at three piezometers and analyzed using the Bouwer and Rice (1976) method. The arithmetic mean for six slug tests performed in the upper silty-clayey loam was 9.3×10^{-7} ms^{-1}, with values ranging from 6.3×10^{-6} - 1.5×10^{-8} ms^{-1}. The arithmetic mean for five slug tests performed in the sand and gravel layer was 4.5×10^{-2} ms^{-1}, with values ranging from 1.5×10^{-2} - 9.0×10^{-2} ms^{-1}. These values all lie within values found in the literature for units with similar textural descriptions (Masch and Denny 1966, Adams and Gelhar 1992, Martin and Frind 1998, Woesner et al. 2001, Loheide and Gorelick 2007). No slug tests were conducted in the lower sandy clay unit, instead a value of 1.0×10^{-9} ms^{-1} was taken from the literature (Freeze and Cherry 1979, Martin and Frind 1998). These values for saturated hydraulic conductivity were used as a starting point in the model development, and were subsequently varied during model calibration.

Surface topography was obtained from previous surveys of pre- and post-restoration scenarios. Two digital elevation models (DEMs) were developed, one representing the incised scenario and one representing the restored scenario. The one representing the restored scenario was updated in 2004 with an additional topographic survey. The DEMs were sampled on a 30 m grid to provide surface elevations to the model. Two MIKE 11 models were developed to reflect the altered channel configuration due to restoration. Channel alignments and cross sections were extracted for each MIKE 11 model from the pre- and post-restoration DEMs (Figure 1.5).

Vegetation inputs included the spatial extent of various vegetation types, in addition to leaf area index (LAI) and root depth (RD) of each prescribed vegetation type. Three vegetation types were employed in the model: ash forest (dominant species *Fraxinus latifolia* and *Crataegus douglasii*), pine forest (dominant species *Pinus jeffreyi*) and grassland (dominant species *Poa pratensis*, *Bromus japonicus*, and *Juncus balticus*) (Figure 1.6). The distribution of each vegetation type was determined through a combination of field reconnaissance and aerial photo interpretation. The ash forest was assigned a variable LAI with a maximum of 5 and a constant RD of 1.83 m. The pine forest was assigned a constant LAI of 5 and RD of 3.05 m (Misson et al. 2005). The grassland was assigned a variable LAI with a maximum value of 2.5 (Xu and Baldocchi 2004) and a variable RD with a maximum of 0.45 m (Wu 1985, Weixelman et al. 1996). Unsaturated soil conductivity and moisture retention properties were adopted from Loheide and Gorelick (2007). Meteorological data were collected at 15 minute intervals from a data logging weather station (HOBO weather station, Onset Computer Corporation) deployed within the meadow (Figure 1.1). Reference evapotranspiration was computed using these meteorological data and the FAO Penman-Montieth combination equation (Allen et al. 1998).

Additional input parameters included the leakage coefficient, which governs river-aquifer exchange, and channel and overland flow roughness coefficients (i.e., Manning's n). River-aquifer exchange was simulated using the reduced contact (b) method, with an initial value of 1.0×10^{-5} s^{-1} adopted from the literature (Thompson et al. 2004). Manning's n for channel flow was estimated to be 0.033 $sm^{-1/3}$ based upon values found in the literature for similar channel conditions (Chow 1959, Barnes 1967, Coon 1998). An initial floodplain Manning's roughness value of 0.5 $sm^{-1/3}$ was adopted from the literature (Thompson et al. 2004). Each of these values was subsequently altered during model calibration.

The subsurface domain boundaries consisted of a combination of no-flow and specified-flux subsurface external boundary conditions and one internal specified-head boundary condition (Figure 1.7). Pre- and post-restoration observation data from 28 piezometers arranged along four transects were used to define the subsurface external boundary conditions. No-flow boundaries were on the upper portion of the meadow and along the southwestern border of the meadow. A short specified-flow boundary was along the northeastern border where subsurface irrigation runoff from an irrigated pasture discharges to the meadow. A flux of 2×10^{-2} m^3s^{-1} was applied during the June-September irrigation season, with zero flow applied to the remaining portion of the year. The spring-fed, perennial streams Mallard Creek, Lower Dana Creek and Fall River bound the downstream portion of the model domain (Figures 1.1 and 1.5). While no-flow boundaries were used in the subsurface, these surface channels were linked to the subsurface, essentially acting as specified-head boundaries. The advantage to this approach was that while constant inflow to these surface channels was specified, stream stages were calculated by the model and differed between the incised and restored scenario runs. The specified head internal boundary was used for an area that received subsurface spring discharge. Water levels in this area were not affected by the stream restoration, and a geochemical analysis of groundwater in this area indicated that the groundwater is similar to nearby springs and dissimilar to Bear Creek surface water (Hammersmark unpublished data). The low-permeability lacustrine clay underlying the meadow justified the use of a no-flow boundary along the bottom of the model domain.

The surface domain boundaries for each MIKE 11 model were developed from flow records from Bear Creek inflow, Mallard Creek inflow, Fall River inflow, Dana Creek inflow, Dana spring inflow to Lower Dana Creek and Fall River stage at the downstream extent of the model domain (Figure 1.5). Data logging pressure transducers (Solinst LT 3001 Leveloggers) were installed in spring 2004 to provide stage hydrographs at each location. At the five inflow locations, over a wide range of flow levels, discharge was measured using standard velocity-area methods (Harrelson et al. 1994), with water velocity measurements collected with a flowmeter (Marsh-McBirney Flo-Mate). Flow measurements and corresponding stage levels were used to create rating curves/tables for each inflow location to allow the conversion of the stage hydrographs to discharge hydrographs. Several additional no-flow boundaries were employed at minor channels heads, which did not experience surface inflow but nevertheless played important roles in regulating the elevation of the water-table.

Model Calibration and Validation

Model calibration parameters included hydraulic conductivity, the leakage coefficient, and channel and overland roughness coefficients. Uniform values for each of the parameters were used. The calibration consisted of individual parameter manipulation and subsequent model performance evaluation. Only the post-restoration model was calibrated and validated because water-table and stream flow data of sufficient temporal resolution were not available for the pre-restoration period.

The 2005 water year (i.e., 1 October 2004 - 30 September 2005) was used for model calibration. Values of saturated hydraulic conductivity, the leakage coefficient, and channel roughness were varied during the calibration process, but the best fit was achieved with the initial value estimates, which all fall within reasonable ranges of values found in relevant literature. The value of overland roughness was decreased from 0.5 $sm^{-1/3}$ to 0.1 $sm^{-1/3}$. This final value resulted in improved channel stage agreement and more closely resembles values for floodplains found in the literature (Chow 1959).

The 2006 water year (i.e., 1 October 2005 - 30 September 2006) was used for model validation. Model performance evaluation during both calibration and validation was based upon a combination of graphical assessment and statistical methods. The Nash-Sutcliffe efficiency coefficient (Nash and Sutcliffe 1970, McCuen et al. 2006) was employed to statistically judge the performance of the model simulation as compared to observed data. The Nash-Sutcliffe efficiency coefficient is widely used when evaluating the statistical goodness-of-fit of model simulations, however time-offset bias and bias in magnitude have been observed (McCuen et al. 2006). In addition to the Nash-Sutcliffe efficiency coefficient, the correlation coefficient, and the mean error for each comparison location were calculated and evaluated. Modeled and observed hydraulic heads were compared at 28 shallow piezometers, and modeled and observed stages were compared at two locations on Bear Creek within the meadow and one location on Bear Creek below the meadow.

Model Application

Once model development, calibration and validation were completed, the two models were used to simulate an identical two-year time period (i.e., 1 October 2004 - 30 September 2006). The only differences between the two models were the altered channel configuration (alignment and size), the topography of the meadow surface (ponds vs. no ponds) and the initial water-table elevation. Starting both model simulations with the same potentiometric surface was unrealistic because the incised scenario could not possibly support the same elevated water-table elevations that occur in the restored scenario at the beginning of the water year. To address this issue, both models were first run with initial hydraulic heads determined by interpolating hydraulic head data collected in early October 2004. Each scenario model was then run for the 2005 water year. Water-table elevations from the end of this run were then utilized as initial conditions for the comparison model simulations described below.

RESULTS

Model Calibration and Validation

The hydrological model of the restored scenario successfully simulates observed conditions (Figures 1.8 and 1.9). Nash-Sutcliffe efficiency coefficients are all greater than 0.90, correlation coefficients are all greater than 0.95, and mean error values are all less than ±0.05 m (Table 1.1).

The agreement between modeled and observed hydraulic heads was particularly strong during the winter, spring and summer, when Bear Creek was flowing. The agreement between modeled and observed hydraulic heads was less strong during late fall, prior to the initiation of flow in Bear Creek, and as initial surface flow began to recharge the subsurface.

The agreement between modeled and observed stage was strong throughout the simulation. However, modeled values were variously higher or lower than observed values during many overbank flow events when flows are largely controlled by floodplain topographic features that are below the resolution of the 30 m grid DEM. Furthermore, modeled stage values were lower than observed values during baseflow conditions downstream of the meadow when Bear Creek ceased to flow in the meadow but continued to flow below the meadow due to discharge from spring-fed Mallard Creek.

Model Application – Incised and Restored Scenario Comparison

Groundwater

Groundwater levels were higher in the restored scenario (Figures 1.10 and 1.11). Restoration had the smallest hydrological effect during the summer and fall when Bear Creek ceased to flow and groundwater levels were lowest, and the largest effect during the winter and spring when Bear Creek was flowing and groundwater levels were highest. Winter and spring meadow average groundwater levels were increased by 0.72 m and 1.20 m, respectively, above incised levels. Smaller seasonal differences occurred in summer and fall when restored average groundwater levels for the entire meadow were 0.34 m and 0.06 m higher, respectively. Restoration had the smallest effect in the lower meadow, where inflows from springs maintained relatively stable groundwater levels throughout the year, and the largest effect in the upper and middle meadow where inflows from the springs were absent and groundwater levels were therefore more related to intermittent stream flows. Restoration increased the range of water-table fluctuations throughout the meadow. Groundwater levels were at or above the ground surface at least once during the simulation at 3.8% and 76.7% of the model grid squares in the incised and restored scenarios, respectively.

Maximum groundwater storage and residual groundwater storage was greater in the restored scenario (Figure 1.12). Maximum groundwater storage was 10.11×10^5 m^3 and 12.11×10^5 m^3 for the incised and restored scenarios, respectively. Residual groundwater storage (i.e., the groundwater storage that remained at the end of the 2006 water year) was 5.83×10^3 m^3 and 3.48×10^5 m^3 for the incised and restored scenarios, respectively. Groundwater residence time was greater in the restored scenario. In the incised scenario, the center of mass of the annual groundwater storage occurred on 14 March 2006, while in the restored scenario, the center of mass of the annual groundwater storage occurred 16 days later on 30 March 2006.

Surface Water

Overbank flows were more frequent in the restored scenario (Figure 1.13). The average channel capacity was 61.7 m^3s^{-1} and 5.35 m^3s^{-1} in the incised and restored scenarios, respectively. While average channel capacity values are useful for communication purposes, minimum channel capacity values exert a larger influence upon the frequency and duration of flooding. The capacity of the restored channel varied between 1.2 m^3s^{-1} and 9.7 m^3s^{-1}. In the restored scenario, local floodplain inundation occurred when stream discharge exceeded the minimum channel capacity, and widespread floodplain inundation occurred when discharge surpassed the average channel capacity. The minimum capacity of the incised channel was 28.0 m^3s^{-1}, thus floodplain inundation due to overbank flooding did not occur in the incised scenario. Floodplain inundation also occurred when groundwater levels rose above the ground surface. Annual surface water storage on the floodplain increased in the restored scenario (Figure 1.12). Maximum surface water storage on the floodplain was 0.27×10^5 m^3 and 6.47×10^5 m^3 for the incised and restored scenarios, respectively.

Floodplain storage was positively correlated with surface water inflow to the meadow in the restored scenario (Figure 1.14). Due to this floodplain storage, flood peak discharges were attenuated in the restored scenario (Figure 1.15). Within the restored reach, flood peak stages were increased, but downstream of the reach flood peak stages were reduced. Instantaneous inflow and outflow were essentially equal in the incised scenario, indicating that floodwaters remained within the channel in the incised scenario. Conversely, instantaneous inflow exceeded instantaneous outflow in the restored scenario, indicating that floodwaters flowed overbank onto the floodplain in the restored scenario. The effects of restoration were most apparent when discharge exceeded the 5.35 m^3s^{-1} average channel capacity. Subsequent flood peak reductions ranged from 12.6 -25.0% of the upstream peak value, with the largest reductions of 23.3%, 25.0% and 24.4% for largest magnitude flood peaks of 15.71 m^3s^{-1}, 17.25 m^3s^{-1} and 20.67 m^3s^{-1}, respectively. Most of the overbank water was stored temporarily and returned to the channel at downstream locations, while some of the overbank water infiltrated and/or evapotranspired.

Within the restored reach, baseflow duration was shorter in the restored scenario (Figure 1.13). When compared at the longitudinal midpoint of the meadow, baseflow ceased 16 days earlier in the restored scenario in each of the years simulated. Increased baseflow levels occurred downstream of the restored reach.

Total annual runoff was higher in the incised scenario. During the 2005 water year, total annual runoff was 4.11×10^7 m^3 and 4.05×10^7 m^3 for the incised and restored scenarios, respectively. Therefore, total annual runoff was 6.60×10^5 m^3 (i.e., 1.6%) higher in the incised scenario. During the 2006 water year, total annual runoff was 9.09×10^7 m^3 and

8.99×10^7 m^3 for the incised and restored scenarios, respectively. Therefore, total annual runoff was 9.38×10^5 m^3 (i.e., 1.0%) higher in the incised scenario.

Evapotranspiration

ET was higher in the restored scenario (Figure 1.16). Daily ET rates were very similar in both scenarios until mid-April. After this point, daily ET rates declined in the incised scenario, but continued to increase in the restored scenario. During the 2005 water year, the peak daily ET rate of 6.5 mmd^{-1} occurred on 22 May 2005 in the incised scenario, while the peak daily ET rate of 7.0 mmd^{-1} occurred 41 days later on 2 July 2005 in the restored scenario. During the 2006 water year, the peak daily ET rate of 5.5 mmd^{-1} occurred on 2 May 2006 in the incised scenario, while the peak daily ET rate of 6.9 mmd^{-1} occurred 56 days later on 27 June 2006 in the restored scenario. The maximum difference of 3.6 mmd^{-1} occurred on 11 July 2006. During the 2005 water year, total annual ET was 1.22×10^6 m^3 and 1.52×10^6 m^3 for the incised and restored scenarios, respectively. During the 2006 water year, total annual ET was 9.63×10^5 m^3 and 1.44×10^6 m^3 for the incised and restored scenarios, respectively. Therefore, total annual ET was 25% and 50% greater in the restored scenario for the 2005 and 2006 water years, respectively.

SUMMARY AND DISCUSSION

This analysis of the Bear Creek Meadow restoration project indicates that plugging of the incised channels and construction of a shallow, sinuous, single-thread channel initiated at least three significant hydrological responses that are likely to have important ecological effects (Table 1.2). These include: 1) increased groundwater levels and volume of subsurface storage; 2) increased frequency of floodplain inundation and decreased magnitude of flood peaks; and 3) decreased baseflow and annual runoff.

Increased Groundwater Levels and Volume of Subsurface Storage

Stream channelization and subsequent incision lower water-tables (Choate 1972, Schilling et al. 2004) resulting in altered riparian vegetation patterns and species composition (Jewitt et al. 2004, Loheide and Gorelick 2007). Consequently, a commonly stated objective of many pond and plug type stream restoration projects is to raise groundwater levels in order to improve the health of riparian vegetation (Benoit and Wilcox 1997, Rosgen 1997, Doll et al. 2003, Poore 2003). Based upon simulations, we demonstrate significant increases in groundwater levels and subsurface storage, which occurred largely in response to the raised channel bed. In the incised scenario, the channel bed was well below the meadow surface, acting as a deep linear sink that efficiently drained the subsurface of the meadow. In the restored scenario, the channel bed was raised, the deep linear sink was removed (i.e., plugged), and groundwater levels were raised (e.g., average increase during spring of 1.2 m), in some cases up to and above the meadow surface. Consequently, subsurface storage was consistently greater in the restored scenario.

The increased water-table elevations simulated in this study are consistent with the one-dimensional groundwater modeling simulations of Schilling et al. (2004), and the three-dimensional groundwater modeling simulations of Loheide and Gorelick (2007). However, these previous studies focused on groundwater alone (i.e., floodplain flow was not simulated), in hypothetical situations with perennial stream flow. Conversely, this study simulated actual conditions where substantial overland flow and intermittent stream flow occurred, creating a more complex hydrological response. In addition, the results of this study support the findings of Bradley (2002), who showed that spatial and temporal trends in groundwater levels are closely linked to the stages of adjacent river channels.

Increased Frequency of Floodplain Inundation and Decreased Magnitude of Flood Peaks

The natural flow regime has been identified as the key determinant in the ecology of river and riparian systems (Poff et al. 1997). In addition, multidimensional connectivity (Vannote et al. 1980, Junk et al. 1986, Ward and Stanford 1995, Tockner et al. 2000) and the resulting variable levels of natural disturbance determine successional patterns and habitat heterogeneity in floodplain river systems. Lateral connectivity, in particular is responsible for the transfer of water, sediment, nutrients and organic matter between river channels and their adjacent floodplains (Tockner et al. 1999). In this study, simulations demonstrate a significant increase in the hydrological connectivity of Bear Creek to its floodplain due to stream restoration. The changes in frequency, duration and magnitude of floodplain inundation, along with declines in the magnitude of peak flood flows exiting the meadow appear to all be a response to decreased channel capacity. The average channel capacity of the incised channel was less than 11 times the average capacity of the restored channel (i.e., 61.7 m^3s^{-1} vs. 5.35 m^3s^{-1}). For the two years simulated here, overbank flooding did not occur in the incised scenario. Conversely, overbank flooding was frequent and of long duration in the restored scenario, with 13 widespread flooding events (defined as when flows reached sufficient magnitude to exceed the average channel capacity of 5.35 m^3s^{-1}) for a total duration of 106 days (i.e., 27% of time the stream was flowing) of overbank flooding. This is the most dramatic change in the hydrology of the meadow. These simulation results are consistent with the qualitative observations of local landowners, who recall extremely rare floodplain inundation in the pre-restored condition (i.e., only during 100+ year return interval events), and frequent and long-duration floodplain inundation in the post-restored condition. Increased inundation frequency due to channel restoration is consistent with the findings of Helfield et al. (2007).

Floodwater storage on the floodplain acted to attenuate flood peaks at the base of the meadow. The peak discharge values for the largest events simulated, which lie between two- and five-year return interval flow values, were reduced by up to 25%. Even greater flood-peak reduction is expected for larger flood pulses than those simulated here. However, the magnitude of flood-peak reductions is capped by floodplain accommodation space. Therefore, flood-peak reductions for very large floods are likely to be less dramatic for lower-frequency, higher-magnitude flood flows. Flood peak attenuation coincident with wetland restoration is consistent with the results of other studies where off-channel areas were hypothetically reconnected to adjacent river channels (Hey and Philippi 1995, Hammersmark et al. 2005)

Decreases in Baseflow and Annual Runoff

There is a general perception that stream restoration will improve all hydrological components of a river-riparian system, resulting in improved conditions for all native plant and animal communities. In the meadow restoration simulated here, anticipated improvements in aquatic habitat associated with increases in baseflow did not occur. The decline in channel capacity and the raising of the channel bed decreased the total amount of runoff by 1-2% and shortened the duration of baseflow by two weeks, extending the period of flow disconnection in the meadow.

The decline in baseflow is largely in response to the raised channel bed and the related changes in ET and groundwater flow paths. Increases in ET were responsible for roughly half of the decreases in total annual runoff. In the incised scenario, much of the groundwater flowed laterally across the valley, discharged to the incised channel, and flowed out of the meadow as stream flow. In the restored scenario, groundwater flowed down the valley, in some cases discharging to the meadow surface, and flowed out of the meadow as either shallow groundwater or overland flow. Therefore, some water that flowed out of the meadow as stream flow in the incised scenario instead left the meadow as evapotranspiration or groundwater discharge in the restored scenario.

The increased ET occurred largely in response to both the raised channel bed and the decreased channel capacity and the related increased groundwater levels, increased the frequency of floodplain inundation, and increased surface storage. In the restored scenario, groundwater levels were higher, providing water to the root zone over a greater area and for longer duration. Furthermore, in the restored scenario, surface water – both overbank flows and floodplain ponds – covered a greater area and for longer duration. These results are consistent with the findings of Loehide and Gorelick (2005) who measured ET rates in degraded and pond and plug restored meadows in northern California.

CONCLUSION

Hydrology is the primary driver of the establishment and persistence of wetlands (Mitsch and Gosselink 2000). Natural flow regimes (Poff et al. 1997) and multidimensional connectivity (Ward and Stanford 1995, Stanford et al. 1996) have been identified as key determinants in the ecology of river-riparian systems. Moreover, hydrology is so crucial that a National Research Council report on the management of riparian areas states that "repairing the hydrology of the system is the most important element of riparian restoration" (National Research Council 2002). The restoration of the meadow channel studied here resulted in the restoration of shallow groundwater levels. The project also resulted in the restoration of the natural flow regime and channel-floodplain connectivity, primarily reflected in the increased frequency and duration of floodplain inundation. These changes to the physical attributes of the system are having and will continue to have profound effects upon the ecology of the meadow and riparian forests.

While this work focuses on the hydrological effects of a particular "pond and plug" restoration project, the results should be utilized toward improved goal setting, restoration design and performance monitoring in similar degraded environments. The methods utilized in this study provide an essential tool for monitoring and assessing the performance of restoration efforts. Considerable complexity and uncertainty exist in the emerging multidisciplinary science of river restoration (Wohl et al. 2005). This approach to evaluating the hydrological response of a restored meadow provides an improved understanding of the magnitude of change and the causes of those changes, supplying a learning tool to improve the science of river restoration. Lessons learned in this study should be used in support of similar methods in appropriate environments, and towards setting realistic and quantifiable objectives for similar projects (see Klein et al. 2007 for example).

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Table 1.1. Nash Sutcliffe efficiency coefficient, correlation coefficient and mean error statistics for the two year model simulations at four subsurface and three surface comparison locations. Subsurface locations compare simulated and observed groundwater depths as shown in Figure 1.8. Surface locations compare simulated and observed water surface elevations as shown in Figure 1.9.

Location	Nash-Sutcliffe	Correlation Coefficient	Mean Error (m)
Groundwater comparisons			
GWA	0.95	0.98	-0.01
GWB	0.93	0.98	0.02
GWC	0.90	0.95	-0.05
GWD	0.91	0.97	0.04
Surface water comparisons			
SW1	0.98	0.99	0.01
SW2	0.97	0.99	0.03
SW3	0.93	0.97	0.02

Table 1.2. Hydrological effects and their causes due to pond and plug stream restoration.

Hydrological Effect	Cause
a) raised groundwater levels	raised channel bed no longer acted as a deep line sink
b) increased subsurface storage	raised channel bed no longer acted as a deep line sink
c) increased frequency of floodplain inundation	channel capacity reduced, reconnecting channel and floodplain at lower flow levels
d) decreased magnitude of flood peaks	water transferred from channel to floodplain, and temporarily stored
e) increased surface storage	increased channel-floodplain exchange and increased surface storage in ponds
f) decreased duration of baseflow	raised channel bed no longer drains groundwater after surface water inflow terminates
g) decreased total annual runoff	increased subsurface storage and ET
h) increased evapotranspiration	elevated groundwater levels available to root zone and increased evaporation from ponds

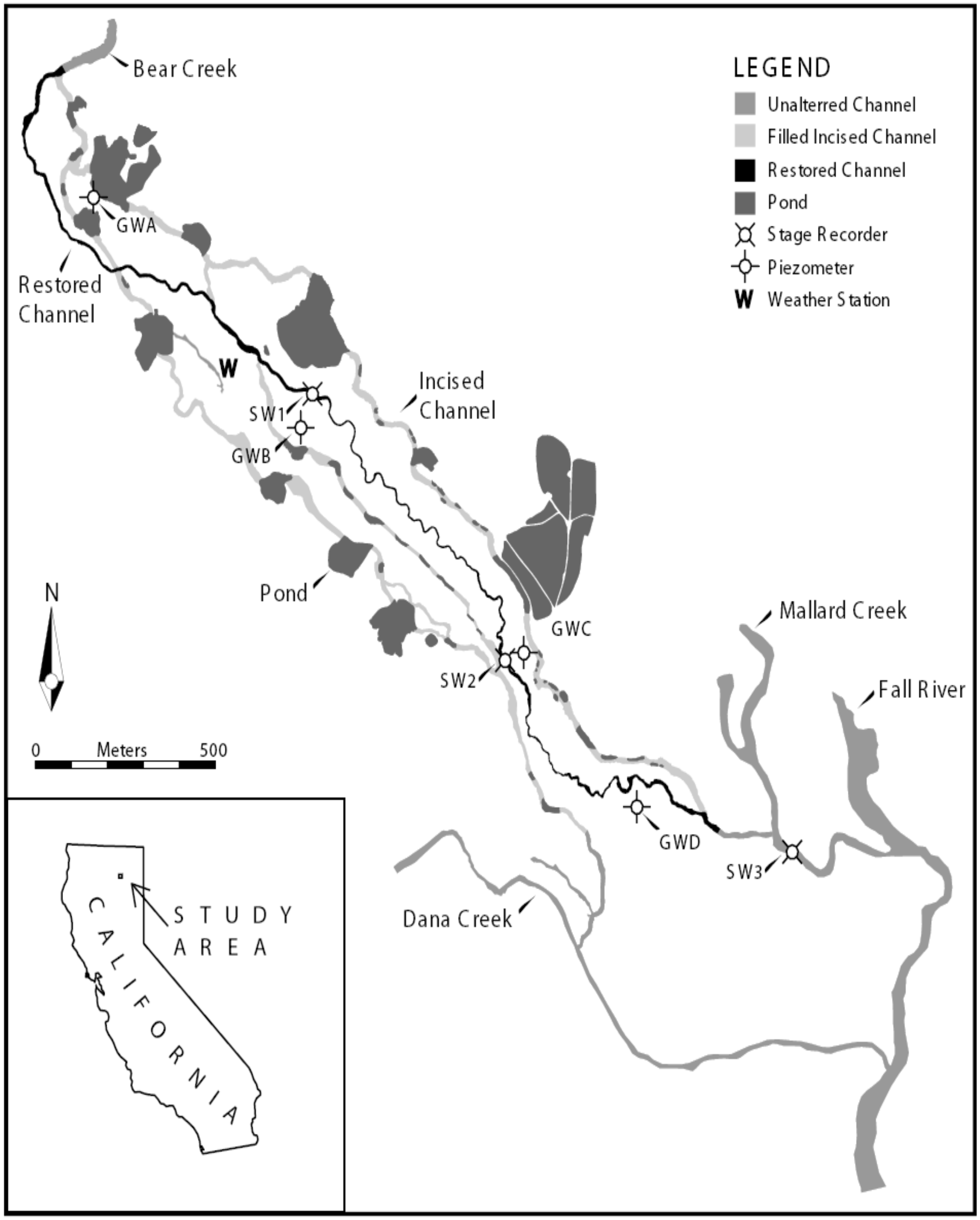

Figure 1.1. Bear Creek Meadow study area. Portions of the incised channels were filled with alluvium excavated from ponds throughout the meadow. A 3.6 km single thread restored channel reach was created from remnant channel segments and excavated where necessary. Flow direction is from upper left to lower right. Surface and groundwater comparison locations are also shown.

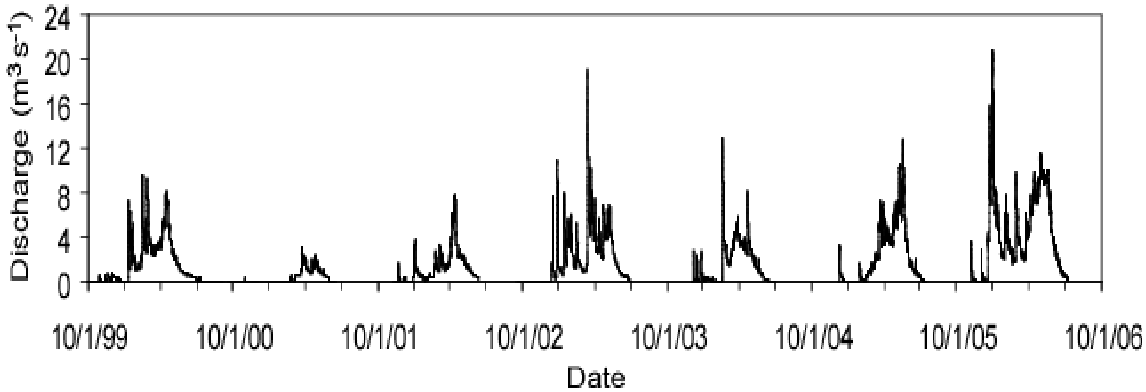

Figure 1.2. Bear Creek discharge at the upstream extent of the restored reach for the water years of 2000 - 2006. Annual peak discharge ranged from 3.11 - 20.73 m^3s^{-1}. Stream discharge is intermittent, with flood peaks resulting from rainfall, rain on snow, and spring snowmelt.

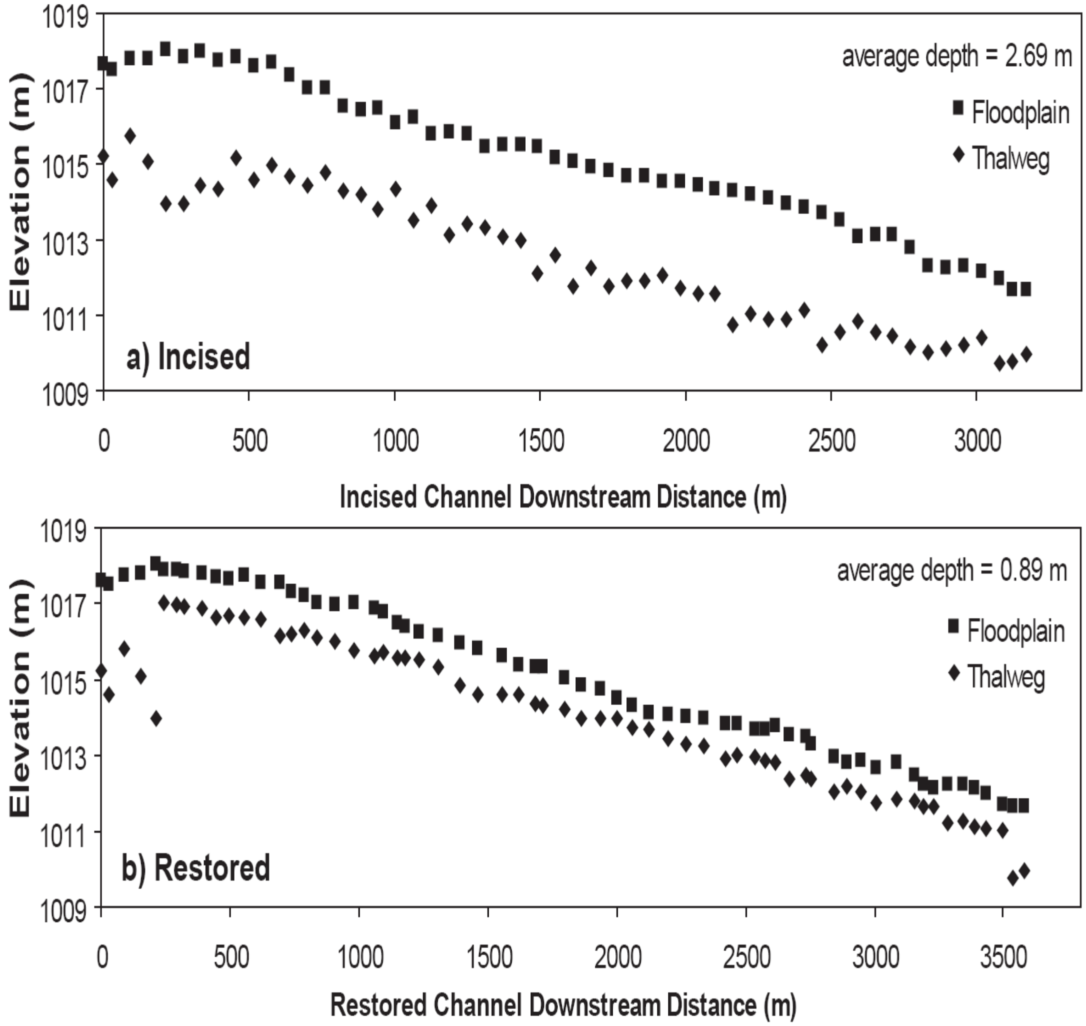

Figure 1.3. Long profiles of riffle crest thalweg and adjacent floodplain elevations for (a) incised and (b) restored channel geometries. The restored reach begins at restored channel station 800 m and ends at restored channel station 3535 m corresponding to incised channel station 800 m and 3124 m, respectively. The first five and last two points in each of the surveys represent identical locations.

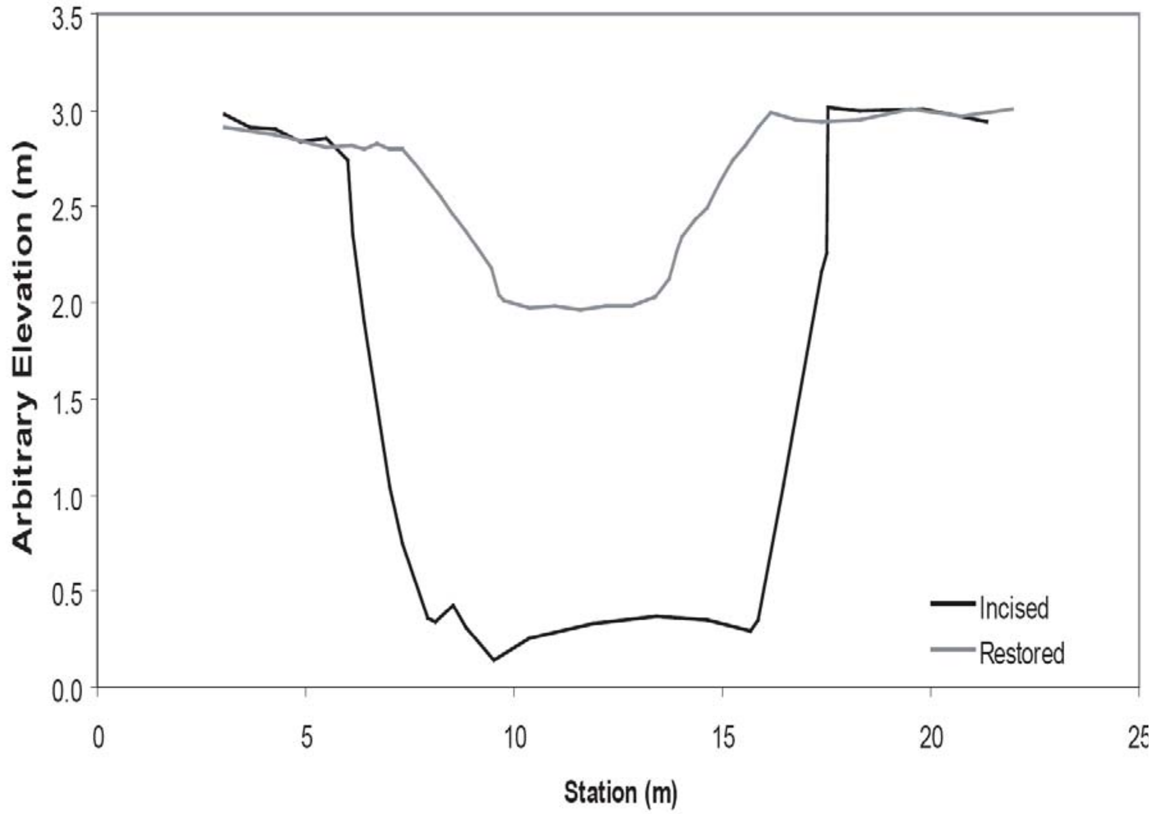

Figure 1.4. Representative restored and incised cross sections of the Bear Creek channel.

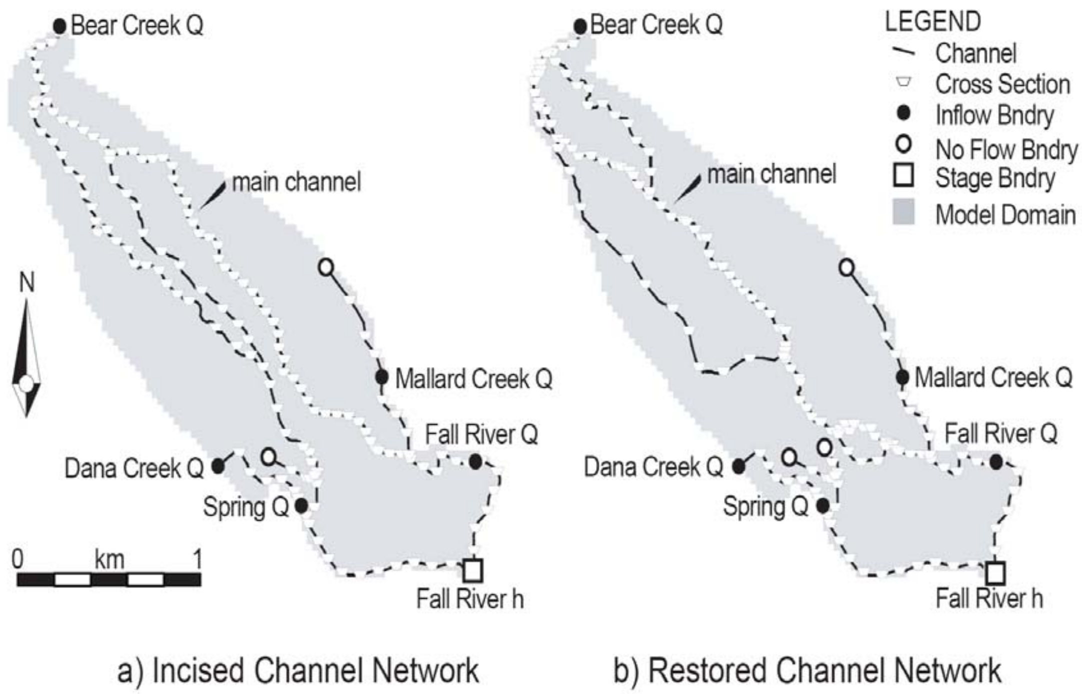

Figure 1.5. Channel alignment, cross section locations, and surface water boundary condition type and locations for the (a) incised and (b) restored channel flow components of the two models.

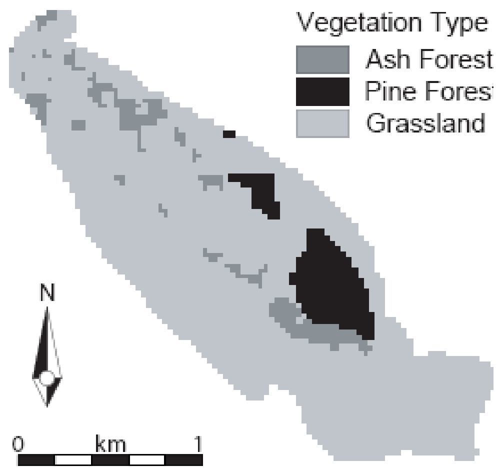

Figure 1.6. Spatial distribution of the three vegetation types employed in the model.

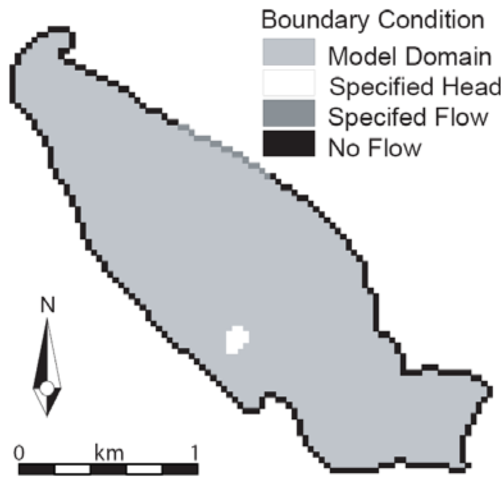

Figure 1.7. Domain and subsurface boundary conditions for the hydrological model. Subsurface boundary types include no flow, specified flow and specified head.

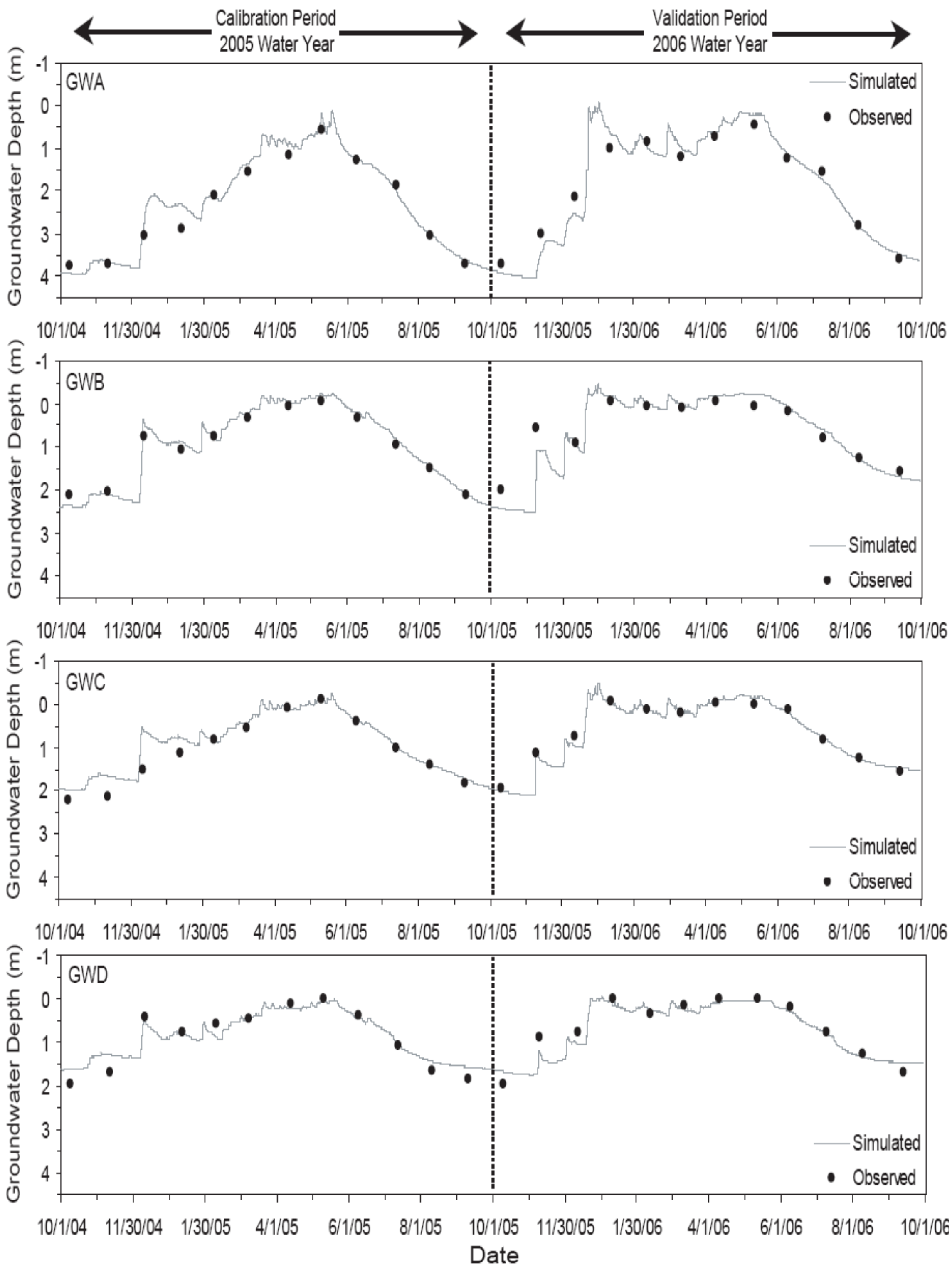

Figure 1.8. Comparison of simulated and observed groundwater depth at four piezometer locations within the meadow. The 2005 water year (left side) was used for model calibration and the 2006 water year (right side) was used for model validation. Negative groundwater depths indicate surface inundation that is common in the restored meadow. Piezometer locations are shown on Figure 1.1.

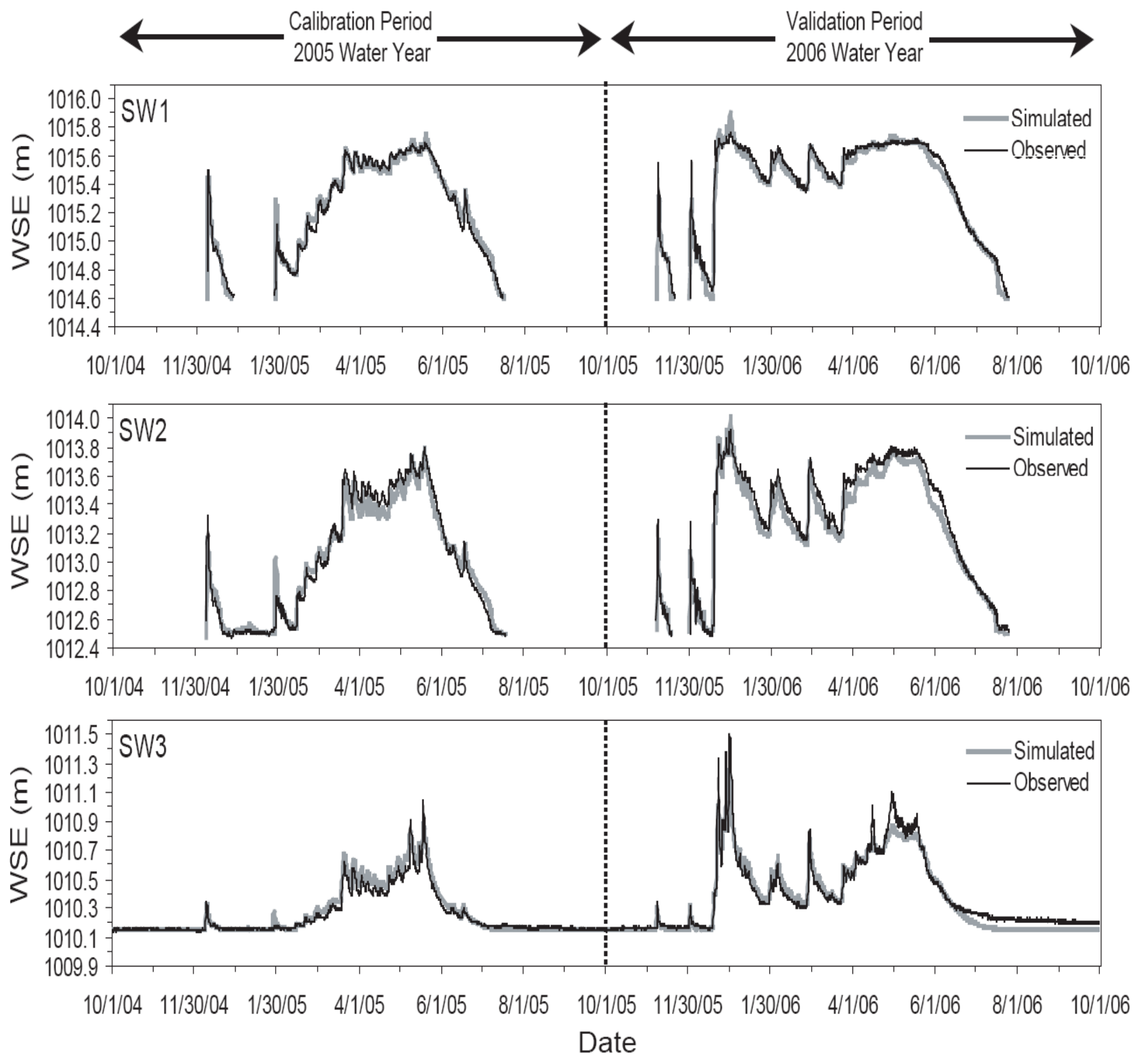

Figure 1.9. Comparison of simulated and observed water surface elevations (WSE) at three locations along Bear Creek. The 2005 water year (left side) was used for model calibration and the 2006 water year (right side) was used for model validation. At the upper two locations (SW1 and SW2) Bear Creek is intermittent, however at the third location (SW3) Bear Creek is perennial due to its confluence with Mallard Creek, a perennial spring channel. Locations of SW1, SW2 and SW3 are shown on Figure 1.1.

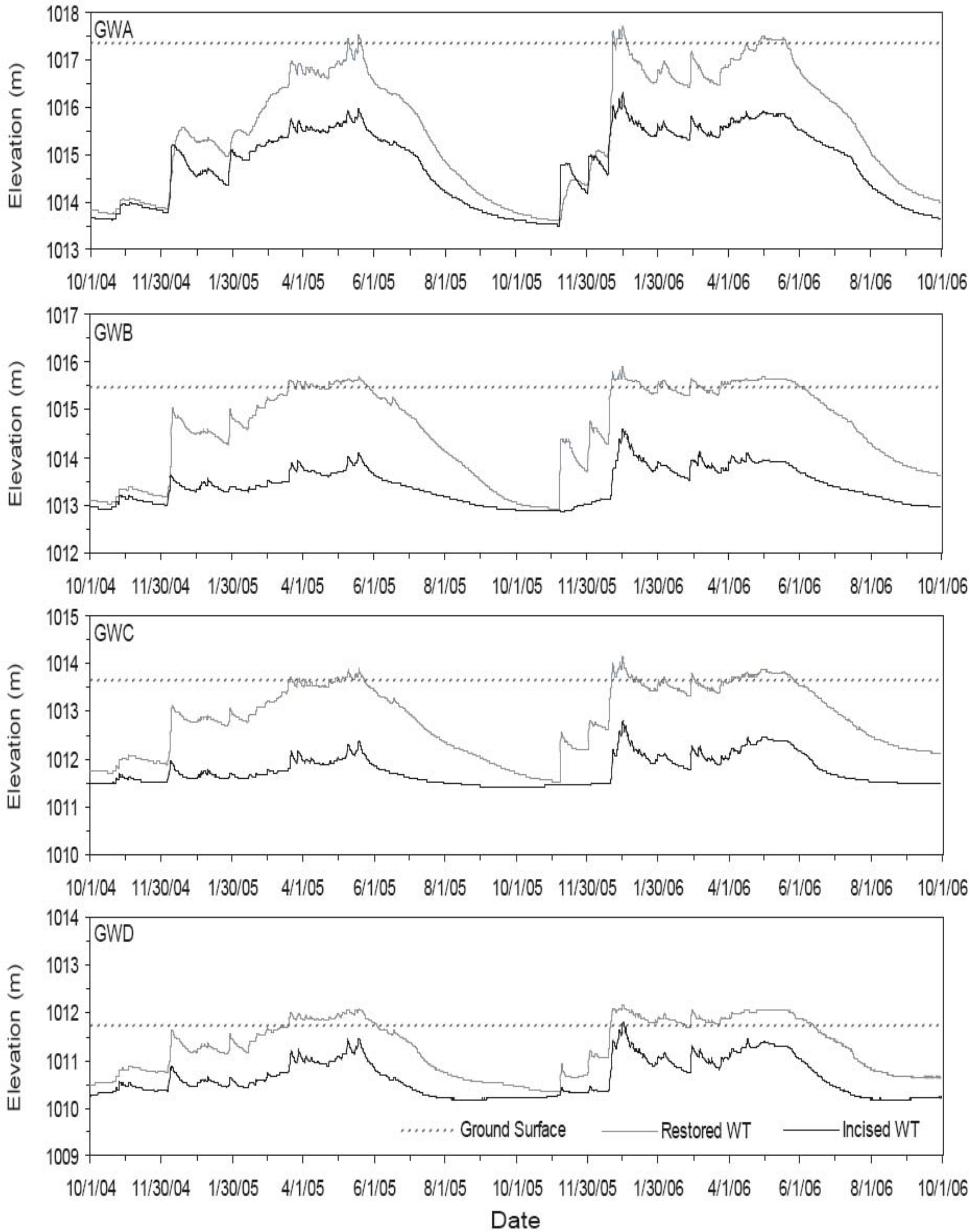

Figure 1.10. Comparison of water-table elevations for the restored and incised scenarios at four locations within the meadow. The largest water-table elevation differences are seen in the winter and spring, corresponding to surface flow in Bear Creek. In the restored condition, the elevation of the water-table is above the ground surface for extended periods at each location. Comparison locations coincide with the locations of piezometers shown on Figure 1.1.

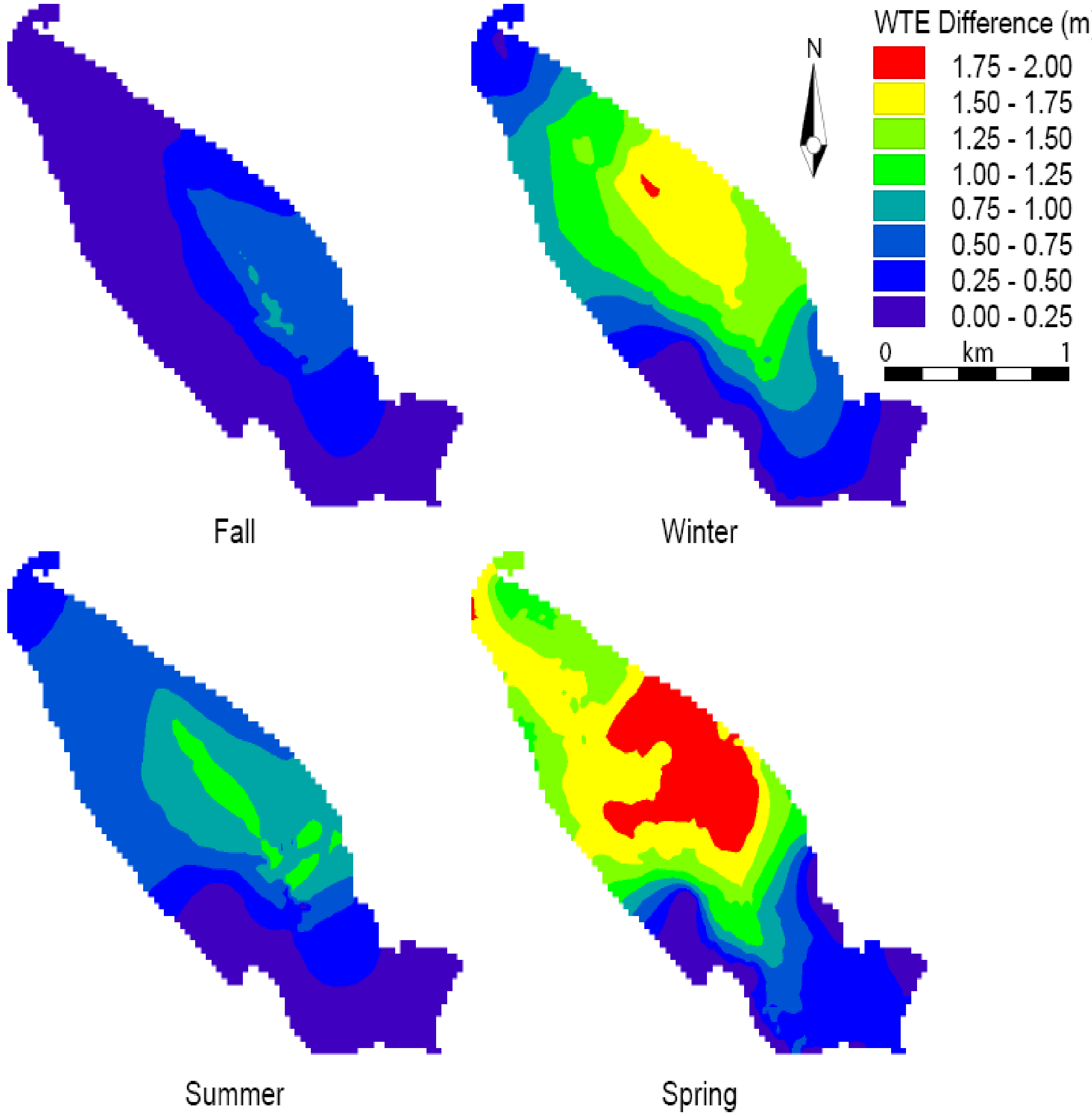

Figure 1.11. Seasonal water-table elevation (WTE) differences between the 2005 water year incised and restored simulations. Clockwise from top left: mid-fall (15 October 2004), mid-winter (14 February 2005), mid-spring (16 May 2005) and mid-summer (15 August 2005). Positive difference indicates the restored water-table is higher than the incised water-table. Spatial patterns in water-table elevation differences are complex due to differing channel alignments, pond locations, subsurface and surface water inputs.

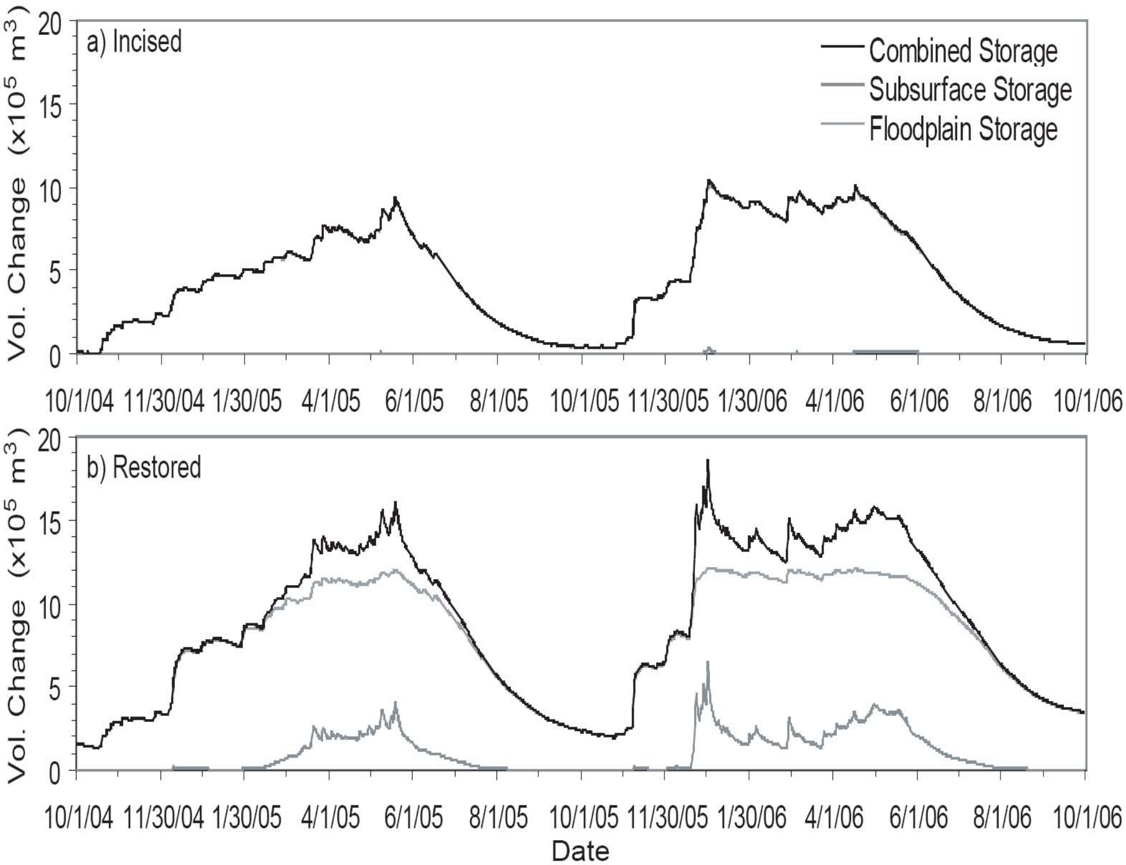

Figure 1.12. Storage volume change for subsurface storage, floodplain storage and combined (subsurface & floodplain) storage for a) incised and b) restored scenarios. The restored scenario stores a larger volume in each of the three categories, with a maximum combined storage of 10.45×10^5 m^3 and 18.52×10^5 m^3 for the incised and restored scenarios, respectively. Due to negligible amounts of water stored on the surface in the incised scenario, the combined storage time series plots nearly on top of the subsurface storage time series. For ease of comparison, stored volume is set equal to 0 m^3 for the beginning of the 2005 water year (i.e., 1 October 2004) in the incised scenario.

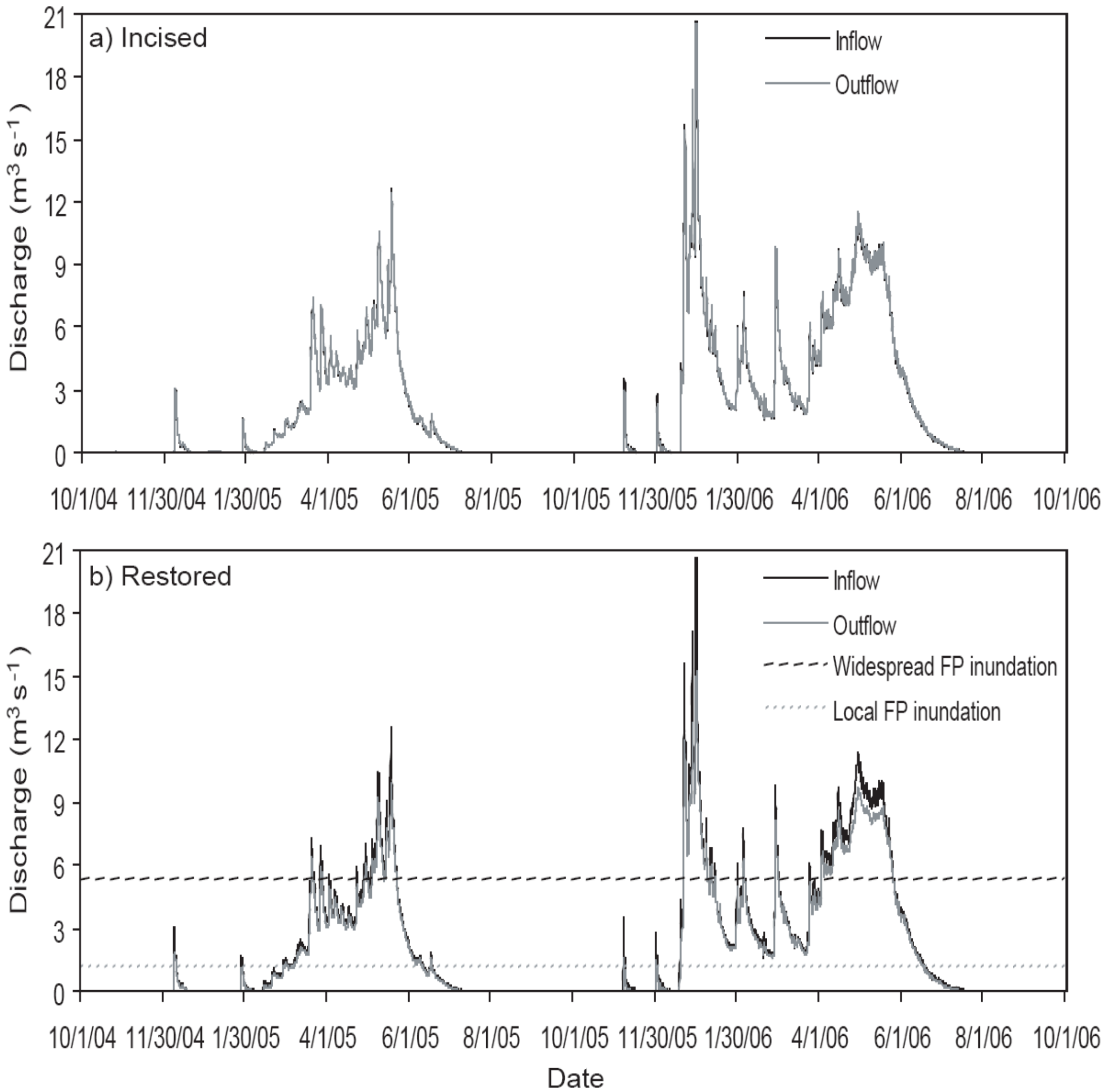

Figure 1.13. Time series of surface inflow and outflow for the a) incised and b) restored scenarios. Channel-floodplain exchange did not occur in the incised scenario, but occurred frequently and for extended periods in the restored scenario. Incised outflow was nearly identical to inflow, however restored outflow was lower than inflow. For the restored scenario, two floodplain inundation thresholds are shown. The dotted line corresponds to the minimum restored channel capacity (1.2 m^3s^{-1}), above which local floodplain inundation occurred. The dashed line corresponds to the average capacity of the restored channel (5.35 m^3s^{-1}) above which widespread floodplain inundation occurred. Minimum bankfull capacity of the incised channel was 28.0 m^3s^{-1}, therefore floodplain inundation did not occur in the incised scenario.

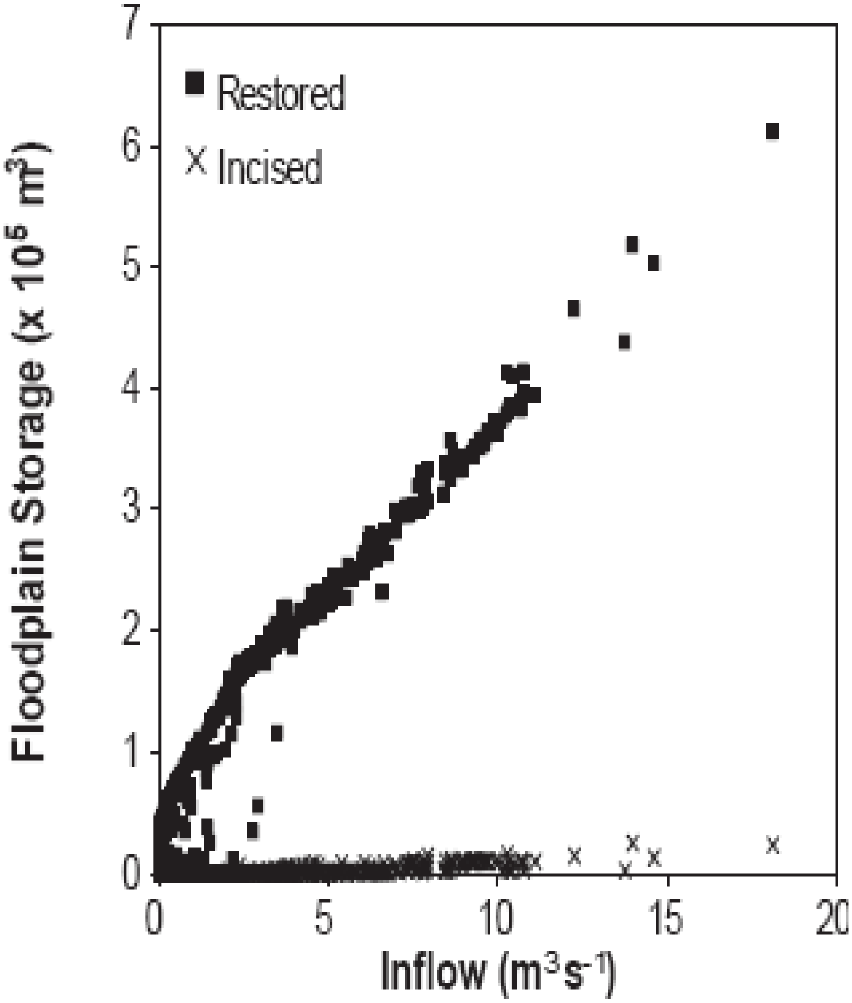

Figure 1.14. Average daily inflow vs. average daily floodplain storage for the incised and restored scenarios. As inflow increased the volume of water stored on the floodplain increased. A much larger volume of water is stored on the restored floodplain, due to enhanced channel floodplain connectivity resulting from the lower capacity restored channel.

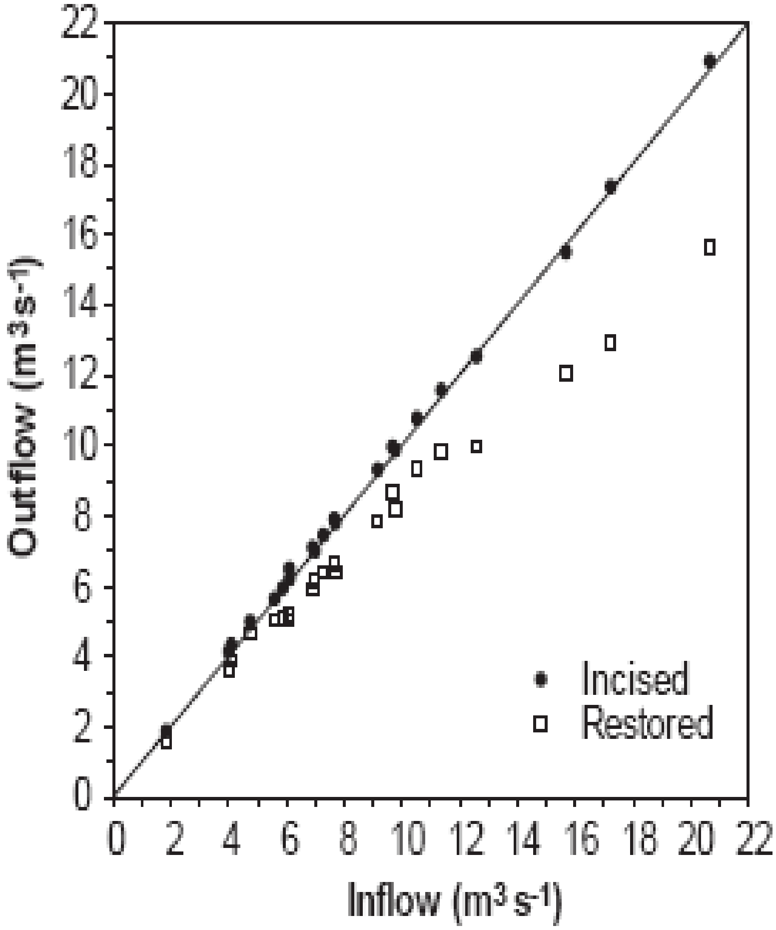

Figure 1.15. Comparison of flood peak inflow/outflow values for incised and restored conditions. Little change is observed between inflow and outflow values for the incised condition. Flows below ~ 4 m^3s^{-1} are mostly contained within the restored channel, and only minor reductions are observed due to subsurface recharge. However, for the largest peaks (i.e., >15 m^3s^{-1}) a 25% reduction of the inflow peak is observed.

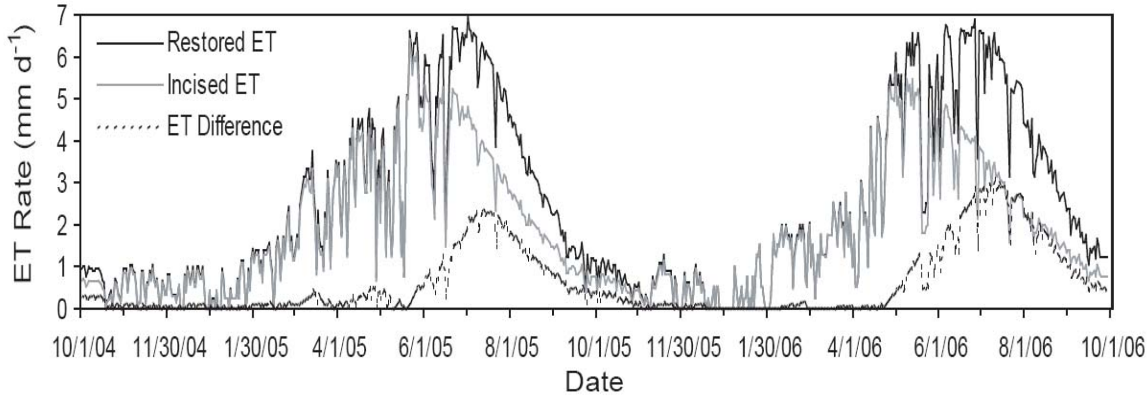

Figure 1.16. Daily evapotranspiration rates for the restored and incised scenarios. The difference between these two values is also provided. Daily ET rates were similar in both scenarios until mid-April of each year. After this point, daily ET rates declined in the incised scenario, but continued to increase in the restored scenario. Peak daily ET rates occurred 41 days and 56 days later for the restored scenario in the 2005 and 2006 water years, respectively. The maximum difference of 3.6 mmd^{-1} occurred on 11 July 2006.

CHAPTER 2 - Vegetation – Water-Table Relationships in a Hydrologically-Restored Riparian Meadow

ABSTRACT

The degraded state of the majority of riparian meadows in the arid west has lead to an increase in management efforts to rehabilitate and restore these ecologically important areas. The distribution of riparian plant communities is primarily driven by hydrologic variables, therefore improved knowledge of vegetation – water-table relationships will increase success of meadow restoration projects. We examined the relationship between temporally varying water-table elevations and plant community distributions in a riparian meadow in northeastern California that recently experienced hydrologic modification by "pond and plug" stream restoration. The objectives of this study were to describe the floristic composition of herbaceous communities found in a recently hydrologically restored riparian meadow and relate them to water-table depth variables. The aerial cover of each species encountered within 128 plots positioned along 15 transects was recorded. A hydrologic model was used to simulate a three-year time series of water-table depth for each plot. TWINSPAN was used to classify the vegetation into four community types (*Eleocharis macrostachya / Eleocharis acicularis, Downingia bacigalupii / Psilocarphus brevissimus, Carex nebrascensis / Juncus balticus,* and *Poa pratensis / Bromus japonicus*) and nonmetric multidimensional scaling was utilized to investigate the relationships between community types and hydrologic variables. Community types were distributed along the hydrologic gradient at reasonably similar positions to those found in previous studies, however *Carex nebrascensis*, a species frequently used as an indicator of shallow water-tables, occurred at greater water-table depths than reported in other studies. The range of water-table depths in this meadow was greater than previously observed, presumably due to the higher temporal resolution of water-table

measurements, in addition to the intermittent nature of stream flow in Bear Creek and its substantial control of water-table elevations. The results of this study can be utilized for improved planning, design, and objective setting in meadow and stream restoration projects in similar Great Basin settings.

INTRODUCTION

In the arid west, riparian areas are ecologically significant and economically important areas that occupy a relatively small percentage of the landscape. Currently, over half of the riparian areas in the Great Basin exist in a poor ecological condition due to both natural and anthropogenic disturbances (Jenson and Platts 1990, Tausch et al. 2004). A common disturbance is lowered water-tables resulting from stream incision (Martin and Chambers 2001, Chambers et al. 2004). While incision has been attributed to geologic factors in many meadow complexes of the Great Basin (Germanoski and Miller 2004), incision also has been attributed to anthropogenic influences including channelization (Emerson 1971) and overgrazing (Kauffman and Krueger 1984, Fleischner 1994, Trimble and Mendel 1995). In an effort to improve the ecological conditions of degraded streams and their adjacent riparian corridors, stream restoration has grown in popularity. An increasingly common technique of raising water-tables in incised meadow environments is the “pond and plug” method, also referred to as meadow re-watering.

Knowledge of plant species and community distributions in relation to water-table elevations is a crucial component in planning and implementing meadow vegetation restoration efforts. Previous studies have investigated vegetation – water-table relationships in pristine or degraded meadows in central Nevada (Chambers et al. 1999, Castelli et al. 2000), eastern Oregon (Stringham et al. 2001, Dwire et al. 2006), the Sierra Nevada (Allen-Diaz 1991, Murrell-Stevenson 2004) and western Montana (Law et al. 2000). However, no studies have investigated vegetation – water-table relationships in hydrologically-restored meadow systems. This is an important distinction, as vegetation – water-table relationships in pristine or degraded meadows are routinely utilized in restoration efforts even though some studies have indicated that plant species may occupy different positions along an altered hydrologic gradient (Leyer 2005).

We examined the relationship between plant community distributions and temporally-varying water-tables in a riparian meadow in northeastern California that was recently hydrologically restored through "pond and plug" stream restoration. The objectives of this study were to: 1) describe the floristic composition of herbaceous plant communities found in a hydrologically-restored riparian meadow and to compare these to other herbaceous plant communities described in the literature; and 2) to relate these herbaceous plant communities to water-table depth variables and to compare these to other vegetation – water-table relationships described in the literature. We hypothesize that similar plant communities will occur at similar locations along the hydrologic gradient, as observed in previous studies of non-restored meadow systems.

STUDY AREA

Bear Creek Meadow (meadow) is a low-gradient alluvial floodplain ~ 100 km northeast of Redding in northern California, USA (Figure 2.1, 41º7'15" N, 121º34'12" W). The meadow is located at an elevation of ~ 1010 m and is situated at the bottom of the ~ 218 km^2 Bear Creek watershed, immediately upstream of the confluence of Bear Creek with the Fall River, the largest spring-fed river system in California (Grose 1996), and among the largest spring-fed river systems in the United States (Meinzer 1927, Rose et al. 1996).

The meadow is approximately 3 km long, 1 km wide, 230 ha in size, and is situated at the northwestern margin of the Fall River Valley near the intersection of the Modoc Plateau and the Cascade Range. The meadow is bounded on the south and west by the steep slopes of Soldier Mountain, to the north and east by the low-relief basaltic flows of the Medicine Lake Highlands, and to the southeast by the Fall River Valley. The head of the meadow lies at the base of a relatively steep, forested bedrock reach. The Fall River Valley is underlain by lacustrine deposits consisting of clay, silt and sand. In the meadow, these deposits are overlain by 0.5-2 m of deltaic sands and gravels, and 1-3 m of floodplain silty loam soils (Grose 1996). Based on a recent soil survey, the dominant soil type is the Matquaw gravelly sandy loam, a mixed, active, mesic Pachic Ultic Haploxeroll (NRCS 2003).

The climate of the Fall River Valley is semi-arid, receiving an annual average of 508 mm of precipitation (California Irrigation Management System data for McArthur for water years 1984-2006). Most precipitation in the Fall River Valley occurs as rainfall in late fall-early spring. Higher elevation areas of the Bear Creek watershed, located to the north and west of the meadow, receive considerably more precipitation that occurs as snow and rain in late fall-early spring.

The hydrological system of the study area consists of intermittent surface-water inflow from Bear Creek and Dana Creek and perennial spring discharge from the Fall River spring system (Figure 2.1). The Fall River spring system is fed by meteoric water, which falls on the Medicine Lake Highlands, perches on low-permeability lacustrine deposits, flows south through fractured basalt, and discharges at the downstream end of the meadow (Rose et al. 1996). These springs form the headwaters of the Fall River and several short tributaries (i.e., Mallard Creek and Lower Dana Creek). The local groundwater system is unconfined, down-valley fluxes occur primarily through the deltaic silts, sands and gravels of the shallow subsurface.

Surface-water input to the meadow is supplied primarily by the intermittent Bear Creek and secondarily by the intermittent Dana Creek, which bounds the southwestern edge of the lower meadow (Figure 2.1). Stream discharge results from spring snowmelt, and fall, winter, and spring rain events including episodic rain-on-snow events. In the 7 years following the 1999 restoration described below, peak annual discharge in Bear Creek measured at the head of the meadow ranged from 3.1-20.7 m^3s^{-1} and the duration of surface flow ranged from 98-229 days (Figure 2.2).

Anthropogenic Disturbance, Incision, Widening and Restoration

Prior to restoration, the meadow was channelized and overgrazed (Poore 2003), resulting in degradation of both aquatic and terrestrial ecosystems of the meadow and the Fall River immediately downstream (Spencer and Ksander 2002). After several years of pre-restoration data collection and consultation, the meadow's incised channels were restored in 1999 as a joint venture between California Department of Fish and Game and the private landowner. The restoration design followed the "Natural Channel Design" method developed by David Rosgen (Rosgen 1996, Malakoff 2004). A "priority 1" approach (Rosgen 1997), more commonly referred to as a "pond and plug" or meadow re-watering strategy, was utilized.

Following the usual "pond and plug" method, the incised stream channels were intermittently filled with plugs of locally derived alluvial material. The remaining unfilled incised channel segments were left as ponds, and many were enlarged to provide the fill material necessary to plug portions of the incised channels. When configuring the restored channel, existing remnant channel segments were used when possible, connected by sections of excavated new channel. The restored channel was constructed with reduced width, depth, and cross-sectional area (Poore 2003). Average channel depth at riffles, was reduced from 2.69 to 0.89 m and average channel capacity was reduced from 61.7 to 5.35 m^3s^{-1} (Hammersmark et al. In press). The restored channel had a meandering riffle-pool morphology, classified as C4 and E4 types in the Rosgen classification system (Rosgen 1996, Poore 2003). Upon completion, a 3.6 km single thread sinuous channel connected the bedrock controlled upstream reach to the unaltered downstream reach (Figure 2.1). In addition, 17 ha of new ponds (remnant gully segments and fill sources) now exist throughout the meadow. Light to moderate seasonal cattle grazing has occurred in the years following restoration activities.

Hydrologic Effects of Restoration

Topographic modification of the stream channels and floodplain surface resulted in substantial changes to the hydrologic regime of the meadow. Based upon simulations from a hydrologic model, Hammersmark et al. (In press) documented these changes, which included: 1) increased groundwater levels and volume of subsurface storage; 2) increased frequency of floodplain inundation and decreased magnitude of flood peaks; 3) increased evapotranspiration; and 4) decreased annual runoff and duration of baseflow. Mean groundwater levels in spring and summer were increased by 1.20 m and 0.34 m, respectively. These were meadow-averaged values, with greater changes occurring near the channels and smaller differences occurring at the distal margins of the meadow. A greater than ten-fold reduction in channel capacity increased the frequency and duration of floodplain inundation. For the two years simulated in the study, overbank flooding did not occur in the pre-restoration scenario, but was frequent and of long duration in the post-restoration scenario, with 13 flooding events creating a cumulative duration of 106 days (i.e., for 27% of time the stream was flowing). Based upon purely qualitative observations, these changes to the hydrologic regime of the meadow resulted in significant changes to the composition and distribution of herbaceous vegetation throughout the meadow (Figure 2.3).

METHODS

Vegetation Sampling

Plant species composition and aerial cover were sampled in 128 2 m x 2 m plots along 15 transects aligned perpendicular to the down valley gradient (Figure 2.1). Along each transect, plots were systematically placed on one side of the channel at 2 m, 5 m, 10 m, 20 m, 40 m, 80 m, 120 m, 160 m, 200 m and 300 m distances from the stream edge, as allowed by the width of the meadow. Data were collected from June 30 to July 20, 2005 when plants were in flower and therefore more easily identified. Percent aerial cover of all vascular plants, bare ground, litter, dung, and wood were ocularly estimated by three observers in 1% increments from 1-5% and then in 5% increments from 5-100% (Daubenmire 1959). Species with <1% cover were recorded as 0.5%, and rare species with only one or two individuals were recorded as 0.1%. The three ocular estimates were then averaged. Nomenclature and native vs. introduced status of each species recorded follows Hickman (1993).

Hydrologic Model Development

A numerical hydrologic model was developed for the study area using the MIKE SHE modeling system (Refsgaard and Storm 1995), which is based upon the Systeme Hydrologique Europeen (SHE) model (Abbott et al. 1986a, b). MIKE SHE is a commercially-available, deterministic, fully-distributed and physically-based modeling system that has been applied to a wide variety of problems where surface water and groundwater are closely linked (for examples see Jayatilaka et al. 1998, Thompson 2004, Sahoo et al. 2006). Using a finite-difference methodology, MIKE SHE solves partial-differential equations describing the processes of saturated subsurface flow (three-dimensional Boussinesq equation), unsaturated subsurface flow (one-dimensional Richards' equation), channel flow (one-dimensional St. Venant equations), and overland flow (diffusion-wave approximation of two-dimensional St. Venant equations). The processes of interception and evapotranspiration are handled with analytical solutions.

The model was comprised of 2898 30 x 30 m^2 grid squares, representing a total area of 261 ha. Surface topography was obtained from a previous topographic survey, and updated in 2004 with an additional topographic survey. Grose (1996) and three well drilling logs from within the model domain provided the conceptual model of the hydrostratigraphy, which was further refined with field investigations. Based upon the refined conceptual model, the subsurface component of the model was composed of three layers, with the lower layer a sandy clay, the middle layer a high-permeability alluvial sand and gravel mixture, and the upper layer an alluvial silty-clayey loam.

Slug tests were conducted at three piezometer locations and analyzed using the Bouwer and Rice method (1976). The arithmetic mean for six slug tests performed in the upper silty-clayey loam was 9.3×10^{-7} ms^{-1}, and the arithmetic mean for five slug tests performed in the sand and gravel layer was 4.5×10^{-2} ms^{-1}. These values lie within ranges found in the literature for units with similar textural descriptions (Masch and Denny 1966, Adams and Gelhar 1992, Martin and Frind 1998, Woesner et al. 2001, Loheide and Gorelick 2007). No slug tests were conducted in the lower sandy clay unit; instead, a value of 1.0×10^{-9} ms^{-1} was taken from the literature (Freeze and Cherry 1979, Martin and Frind 1998). Unsaturated soil hydraulic conductivity and moisture retention properties were adopted from Loheide and Gorelick (2007).

Vegetation inputs included leaf area index, root depth, and spatial extent of various vegetation types. Three vegetation types were employed in the model: ash forest (dominant species *Fraxinus latifolia* and *Crataegus douglasii*), pine forest (dominant species *Pinus jeffreyi*) and grassland (dominant species *Poa pratensis*, *Bromus japonicus*, *Juncus balticus*). The distribution of each vegetation type was determined through a combination of field reconnaissance and aerial photo interpretation. Meteorological data were collected at 15-minute intervals from a data logging weather station (HOBO weather station, Onset Computer Corporation) deployed within the meadow (Figure 2.1). Reference evapotranspiration was computed using these meteorological data and the FAO Penman-Montieth combination equation (Allen et al. 1998).

Additional input parameters included the leakage coefficient, which governs river-aquifer exchange, and channel and overland flow roughness coefficients (i.e., Manning's n). River-aquifer exchange was simulated using the reduced contact (b) method, with a value of $1x10^{-5}$ s^{-1} adopted from the literature (Thompson et al. 2004). Manning's n for channel flow was estimated to be 0.033 $sm^{-1/3}$ based upon values found in the literature for similar channel conditions (Chow 1959, Barnes 1967, Coon 1998). An initial floodplain Manning's roughness value of 0.5 $sm^{-1/3}$ was adopted from the literature (Thompson et al. 2004). Each of these values was subsequently altered during model calibration.

The subsurface domain boundaries consisted of a combination of no-flow and specified-flux subsurface external boundary conditions and one internal specified-head boundary condition. Observation data from 28 piezometers arranged along four transects were used to define the subsurface external boundary conditions. No-flow boundaries were on the upper portion of the meadow and along much of the southwestern border of the meadow. A short specified-flow boundary was along the northeastern border where subsurface irrigation runoff from an irrigated pasture discharges to the meadow. A flux of 2×10^{-2} m^3s^{-1} was applied during the June-September irrigation season, with zero flow applied to the remaining portion of the year. The spring-fed, perennial streams Mallard Creek, Lower Dana Creek and Fall River bound the downstream portion of the model domain (Figure 2.1). The specified head internal boundary was used for an area that received subsurface spring discharge. The low-permeability lacustrine clay underlying the meadow justified the use of a no-flow boundary along the bottom of the model domain.

The surface domain boundaries were developed from flow records from Bear Creek inflow, Mallard Creek inflow, Fall River inflow, Dana Creek inflow, Dana spring inflow to Lower Dana Creek and Fall River stage at the downstream extent of the model domain. Data logging pressure transducers (Solinst LT 3001 Leveloggers) were installed in spring 2004 to provide stage hydrographs at each location. At the five inflow locations, discharge was measured over a wide range of flow levels, using standard velocity-area methods (Harrelson et al. 1994). Water velocity measurements were taken with a flowmeter (Marsh-McBirney Flo-Mate). Flow measurements and corresponding stage levels were used to create rating curves/tables for each inflow location, to allow the conversion of stage hydrographs to discharge hydrographs.

Following calibration and validation, the hydrologic model was used to simulate a 3-year period, the water years of 2004, 2005 and 2006 (i.e., October 1, 2003 through September 30, 2006). Annual precipitation was within normal ranges during the course of this study, with annual precipitation being 510 mm (i.e., 100.2% of average), 529 mm (i.e., 104.1% of average), and 653 mm (i.e., 129.4% of average) for the 2004, 2005 and 2006 water years, respectively.

For each vegetation plot location, a time series of water-table elevation was generated and combined with the ground surface elevation to yield a time series of water-table depth. From each of the three annual time series, average, minimum and range of water-table depth during the growing season were calculated. The growing season was defined as May through August, the period in which aboveground parts of herbaceous plants were observed to be alive on site. The three annual values were then averaged to provide one value for each water-table depth variable at each plot location. In addition, the number of growing-season days the water-table depth was < 0, 30 and 70 cm were calculated for each of the three years. The number growing-season days that the water-table depth was < 0 cm represents the number of days a given plot was flooded, or inundated. The number of growing-season days that the water-table depth was < 30 cm represents the number of days the water-table was within the root zone typical of mesic and hydric herbaceous meadow communities (Manning et al. 1989, Weixelman et al. 1996, Chambers et al. 1999, Castelli et al. 2000). The number of growing-season days that the water-table depth was < 70 cm represents the number of days the water-table was within the root zone typical of xeric herbaceous meadow communities (Weixelman et al. 1996, Chambers et al. 1999, Castelli et al. 2000). The number of growing-season days the water-table depth was < 0, 30, and 70 cm was employed as a proxy for the duration of anoxia in the root zone, as past studies have demonstrated strong correlations between shallow water-table depths and low soil redox potentials (Castelli et al. 2000, Dwire et al. 2006).

Hydrologic Model Calibration and Validation

Hydrologic model calibration parameters included hydraulic conductivity, leakage coefficient, and channel and overland roughness coefficients. Uniform values for each of the parameters were used. The calibration consisted of individual parameter manipulation and subsequent model performance evaluation. The 2005 water year was used for model calibration. The 2006 water year was used for model validation. The hydrologic model performance evaluation during calibration and validation was based upon a combination of graphical assessment and statistical methods. The Nash-Sutcliffe efficiency coefficient was employed to statistically judge the performance of the model simulation as compared to observed data (Nash and Sutcliffe 1970, McCuen et al. 2006). The Nash-Sutcliffe efficiency coefficient is widely used when evaluating the statistical goodness-of-fit of model simulations, though time-offset bias and bias in magnitude have been observed (McCuen et al. 2006). In addition to the Nash-Sutcliffe efficiency coefficient, the correlation coefficient and the mean error for each comparison location were calculated and evaluated. Modeled and observed hydraulic heads were compared at 28 shallow piezometers, and modeled and observed stream stages were compared at two locations on Bear Creek within the meadow and one location on Bear Creek below the meadow.

Data Analyses

Two-way indicator species analysis (TWINSPAN), a polythetic, divisive classification tool, was used to analyze the vegetation data (Hill 1979, McCune and Mefford 1999). Default pseudo-species, percent-cover cutoff values of 0, 2, 5, 10, and 20 were utilized in the classification analysis. Infrequently-observed plant species, which occurred in < 5% of the plots, were excluded from the TWINSPAN analysis (McCune et al. 2002). Indicator-species analysis was used to identify individual species which were both faithful and exclusive to each community (Dufrene and Legendre 1997, McCune and Mefford 1999). Indicator values were tested for statistical significance using a Monte Carlo randomization, with 1000 runs. Only species with a $p<0.001$ were reported.

Nonmetric multidimensional scaling (NMS), an indirect gradient analysis, was utilized to ordinate vegetation data without the influence of environmental variables (Kruskal 1964, Mather 1976, McCune and Mefford 1999). Log-transformed cover values were used to ordinate the plots employing the Sorensen distance measure. Again, infrequently-observed plant species which occurred in < 5% of the plots were excluded from the NMS analysis (McCune et al. 2002). Following the ordination, relationships between the ordination axes and the environmental variables were examined, and TWINSPAN classification groups were overlaid for interpretation purposes. Differences between community means for each variable were tested with analysis of variance and Tukey-Kramer honest significant difference in JMP (SAS Institute 2004).

Each species encountered was assigned to a wetland-indicator category based upon its U. S. Fish and Wildlife Service (1996) wetland-indicator status in the California region. A composite wetland-indicator score for each sample was calculated by weighting the percent cover of each species with index values for each wetland-indicator category as follows: obligate wetland = 1, facultative wetland = 2, facultative = 3, facultative upland = 4, and obligate upland = 5. Unidentified species and those assigned to the NA (no agreement) or NI (no indicator) categories were excluded from this portion of the analysis.

RESULTS

Vegetation Data and Classification

A total of 167 herbaceous, vascular taxa were encountered, 75 of which occurred in ≥ 5% of the plots. Species richness ranged from 3 to 31 species per plot with an average of 17. *Juncus balticus*, *Bromus japonicus*, *Phlox gracilis*, and *Poa pratensis* ssp. *pratensis* were the most frequently encountered species. TWINSPAN classification of the species cover data yielded four community types, named after the two species with the highest total percent cover in each community type (Table 2.1).

In the first community type, the non-native grasses *Poa pratensis* ssp. *pratensis* and *Bromus japonicus* had the highest total percent cover. Additional species contributing a large percent cover in this community type were *Iris missourensis*, *Juncus covellei*, *Aster* *occidentalis*, and *Leymus triticoides*. *Epilobium brachycarpum* and *Pholox gracilis* had low percent cover but high constancy values. The *Poa pratensis / Bromus japonicus* community dominated the upper third of the meadow, even near the stream (i.e., in plots 2-20 m from the stream margin). In the lower two-thirds of the meadow, this community type was limited to locations farther away from the stream (>100 m from the stream margin). This community type shares the dominant species found in the Kentucky Bluegrass community type (Smith 1998), the dry meadow community (Dwire et al. 2006), the mesic meadow community (Castelli et al. 2000), the moist bluegrass community (Stringham et al. 2001), the mesic graminoid ecological types (Weixelman et al. 1996, Weixelman et al. 1999), the *Poa pratensis/Potentilla gracilis* community (Allen-Diaz 1991), the Kentucky bluegrass class (Ratliff 1982), the *Poa pratensis/Potentilla gracilis* plant association (Potter 2005), and the Kentucky bluegrass series (Keeler-Wolf et al. 1998). However, none of the above-mentioned communities include *Bromus japonicus* or *Juncus covillei*, which are commonly found with high cover values in plots of this community type in this study.

In the second community type, *Carex nebrascensis* and *Juncus balticus*, and to a lesser extent *Juncus covillei* and *Plagiobothrys stipitatus* var. *micranthus*, had the highest total percent cover. *Navarettia intertexta* and *Carex athrostachya* had low percent cover but high constancy values. The *Carex nebrascensis* / *Juncus balticus* community type was found near the stream in the lower two-thirds of the meadow. This community type shares dominant species with the *Carex nebrascensis* and *Juncus balticus* community types (Smith 1998), the moist meadow community (Stringham et al. 2001), the Nebraska sedge ecological type (Weixelman et al. 1996), the wet meadow type (Castelli et al. 2000), the Nebraska sedge class (Ratliff 1982), the Nebraska sedge series (Sawyer and Keeler-Wolf 1995) and the *Carex nebrascensis* plant association (Potter 2005).

In the third community type, *Downingia bacigalupii* and *Psilocarphus brevissimus* var. *brevissimus*, *Navaretia leucocephala* ssp. *minima*, and *Mimulus tricolor* had the highest total percent cover. *Plagiobothrys stipitatus* var. *micranthus*, *Polygonum polygaloides* ssp. *confertiflorum* and *Veronica peregrina* ssp. *xalapensis* had low percent cover but high constancy values. The *Downingia bacigalupii* / *Psilocarphus brevissimus* community type was limited to the bottoms and margins of alternate channels and swales, which were intermittently or seasonally inundated. Within this community type, three of the species present were vernal pool indicators and five of the species were vernal pool affiliate species (Keeler-Wolf et al. 1998, Barbour et al. 2005, Barbour et al. 2006). While many of these species are found in wet meadow habitats (Hickman 1993), other examples of this community type in non-vernal pool environments are relatively absent from the published literature. However, this community type does resemble the *Downingia bicornuta* and *Navarretia* community types described by Smith (1998). Despite the differing water sources, this community shares many species in common with those observed in nearby vernal pools (Solomeshch et al. 2007). In addition, vernal pool-like plant communities have been found in "tenajas," described as sites in intermittent streams that exhibit characteristics of vernal pools upon desiccation (Ferren et al. 1995).

In the fourth community type, *Eleocharis macrostachya* and *Eleocharis acicularis*, and to a lesser degree *Juncus nevadensis*, had the highest total percent cover. This community type was limited to depressions on the floodplain inundated in the early growing season. This community type shares dominant species with the *Eleocharis macrostachya* community type (Smith 1998), the *Eleocharis macrostachya* plant association (Potter 2005), the ephemeral-lake class (Ratliff 1982), and the spikerush series (Sawyer and Keeler-Wolf 1995).

Indicator species analysis identified a species or group of species that were significant indicators (diagnostic species) of each community type. *Poa pratensis*, *Bromus japonicus*, and *Epilobium brachycarpum* were identified as significant indicators of the first community type. *Carex nebrascensis* was identified as a significant indicator of the second community type. *Downingia bacigalupii*, *Psilocarphus brevissimus*, *Mimulus tricolor*, *Veronica peregrina*, and *Polygonum polygaloides*, all vernal pool indicators or affiliates, were identified as significant indicators of the third community type. *Eleocharis macrostachya* was the only significant indicator of the fourth community type.

Hydrologic Model Calibration and Validation

Values of saturated hydraulic conductivity, leakage coefficient, and channel roughness were varied during the calibration process, but the best fit was achieved with the initial value estimates, which all fall within reasonable ranges of values found in relevant literature (Chow 1959, Masch and Denny 1966, Barnes 1967, Adams and Gelhar 1992, Coon 1998, Martin and Frind 1998, Woesner et al. 2001, Thompson et al. 2004, Loheide and Gorelick 2007). The value of overland roughness was decreased from 0.5 to 0.1 $sm^{-1/3}$. This final value resulted in improved channel stage agreement and more closely resembles values for floodplains found in the literature (Chow 1959).

The hydrological model successfully simulates observed conditions, with Nash-Sutcliffe efficiency coefficients, calculated for the combined calibration and validation period, all > 0.90, correlation coefficients all > 0.95, and mean error values all < ±0.05 m (Figure 2.4). The agreement between modeled and observed hydraulic heads was particularly strong during the winter, spring, and summer, when Bear Creek was flowing. The agreement between modeled and observed hydraulic heads was less strong during late fall, prior to the initiation of flow in Bear Creek, and as initial surface flow began to recharge the subsurface. For further details on the hydrologic model the reader is referred to Hammersmark et al. (In press).

Changes in Herbaceous Vegetation Distribution

Comparison of aerial photographs taken prior to the restoration (i.e., July 1998) and 6 years after the restoration activities (i.e., July 2005) illustrate the shift in non-woody mesic and hydric vegetation types (Figure 2.3). The distribution of woody vegetation types (e.g., *Fraxinus latifolia, Crataegus douglasii*, and *Pinus jeffreyi*) remain largely unchanged, but the distribution of mesic and hydric herbaceous vegetation types increased in response to the elevated water-table conditions. In the pre-restoration photograph, mesic and hydric herbaceous vegetation is observed only in the lower part of the meadow. In the post-restoration photograph, mesic and hydric vegetation is observed throughout much of the meadow, particularly in the near-stream regions.

Vegetation – Water-Table Relationships

Throughout the three water years simulated, the water-table depth was consistently deepest in the *Poa pratensis / Bromus japonicus* community type, intermediate in the *Carex nebrascensis / Juncus balticus* and *Downingia bacigalupii / Psilocarphus brevissimus* community types, and shallowest in the *Eleocharis macrostachya / Eleocharis acicularis* community type (Figure 5). Surface inundation during the growing season occurred rarely, if at all in the *Poa pratensis / Bromus japonicus* community type, yet was frequent (i.e., occurring in each of the 3 years) and for extended duration (i.e., 25-50 growing season days) in the other three community types. Mean water-table depths for all community types were > 95 cm by the end of the growing season.

Summary hydrologic variables for each community type are presented in Table 2.2. Values represent community type means for the growing season (May – August) of the 3 years simulated. For many variables (e.g., average, minimum, < 0 cm, < 30 cm, < 70 cm), community types were arranged in three significantly different hydrologic groups, with the *Poa pratensis / Bromus japonicus* community type being most xeric, the *Carex nebrascensis / Juncus balticus* and *Downingia bacigalupii / Psilocarphus brevissimus* community types being most mesic, and the *Eleocharis macrostachya / Eleocharis acicularis* community type being most hydric. For every water-table, time-series variable, differences between the *Carex nebrascensis / Juncus balticus* and *Downingia bacigalupii / Psilocarphus brevissimus* communities were not significant.

Wetland Indicator Scores

Wetland indicator scores for each community type reflect the vegetation distribution along the hydrologic gradient (Table 2.2 and Figure 2.7). The *Poa pratensis / Bromus japonicus* community type contained species from each of the categories, but was dominated by species from the mesic-xeric categories with facultative, facultative upland, and upland species covering 32%, 19%, and 7%, respectively. The *Carex nebrascensis / Juncus balticus* and *Downingia bacigalupii / Psilocarphus brevissimus* community types were dominated by species from the hydric-mesic categories. The *Carex nebrascensis / Juncus balticus* community type had obligate wetland and facultative wetland species covering 26% and 35%, respectively, while the *Downingia bacigalupii / Psilocarphus brevissimus* community type had had obligate wetland and facultative wetland species covering 43% and 14%, respectively. The *Eleocharis macrostachya / Eleocharis acicularis* community type was dominated by species from the hydric category, with obligate wetland species and facultative wetland species covering 65% and 16%, respectively.

Nonmetric Multidimensional Scaling Results

In the NMS ordination, a two dimensional solution with a final stress of 18.85 and a final instability of 0.00001 after 86 iterations was obtained (Figure 2.8). The first and second ordination axes captured 62% and 18% (cumulative 80%) of the variance in the vegetation data set. Hydrologic variables were strongly correlated with the first axis, with minimum depth (R^2 = 0.64, p<0.0001), average depth (R^2 = 0.60, p<0.0001), days < 0 cm (R^2 = 0.65, p<0.0001), < 30 cm (R^2 = 0.65, p<0.0001), and < 70 cm (R^2 = 0.61, p<0.0001) being most strongly correlated. The wetland indicator score, a surrogate hydrologic variable computed from vegetation data, also was strongly correlated with axis one (R^2 = 0.77, p<0.0001). Percent cover of bare ground was weakly correlated (R^2 = 0.20, p<0.0001) with the second axis. These results indicate that the depth to water-table and the variables derived from water-table time-series data explain the majority of variation in the vegetation data set.

DISCUSSION

Approach and Assumptions

In this study, we assumed that hydrology, or more specifically, the depth to groundwater, was the primary factor controlling the distribution of herbaceous vegetation communities. This assumption is typically valid for wetland environments, many of which experience both drought and soil saturation with anoxia in the root zone (Mitsch and Gosselink 2000). Indeed, several studies have identified hydrologic variables, typically depth to groundwater, as the primary gradient controlling vegetation distributions in meadow and grassland environments (Allen-Diaz 1991, Castelli et al. 2000, Law et al. 2000, Stringham et al. 2001, Henszey et al. 2004, Dwire et al. 2006). The results of this study further support this assumption, as hydrologic variables were strongly correlated with the primary axis of the NMS ordination gradient (Figure 2.8).

However, hydrologic conditions may simply be surrogates for soil chemical reactions that influence plant productivity, such as redox reactions limiting root oxygen and nutrient availability (Hobson and Dahlgren 2001). Furthermore, other factors including flooding, competition, grazing intensity (past and present), nutrient availability, soil properties (e.g. texture, porosity, amount of organic matter), fire history, and disease are likely to further influence vegetation distributions at the site. For example, *Carex nebrascensis* and *Juncus balticus* dominate heavily-grazed sites with adequate moisture due to their persistent deep rhizomes and extensive fibrous root systems (Weixelman et al. 1996). While the influence of flooding was considered (i.e number of days water-table depth < 0 cm), a number of other elements associated with flooding (e.g., shear stress, sediment delivery, propagule delivery) were not investigated,nor were the additional biotic and abiotic controlling factors mentioned above.

In this study, we also assumed that the herbaceous plant community types were in equilibrium with the restored hydrologic regime. Restoration activities were undertaken in the summer of 1999. Qualitative observation of the meadow's herbaceous vegetation indicated a considerable change following the restoration and consequent hydrologic modification. Six years is surely not a sufficient period for woody species to reach equilibrium with the modified site hydrology, and for this reason they were excluded from this study. In contrast, six years is a sufficient amount of time for annual, biennial, and many perennial herbaceous species to undergo several reproductive attempts and/or complete life cycles. For these short-lived herbaceous species, the equilibrium assumption is considered valid. However the distribution of some of the longer-lived perennial species may still be reaching equilibrium with the restored hydrology, with xeric-mesic species persisting in mesic-hydric locations.

The use of a hydrologic model to simulate water-table levels has both advantages and disadvantages. The simulation results are simply estimates of actual conditions. However, the model utilized in this study was rigorously calibrated and validated, and simulation results closely replicate the temporally and spatially variable water-table. Advantages to utilizing a hydrologic model are numerous, including the ability to simulate water-table conditions: 1) at a large number of vegetation plots, 2) at a high temporal resolution, 3) over long periods of time, and 4) at depths that might otherwise exceed piezometer depths. In this study, we simulated water-table time-series data for each of the 128 vegetation plot locations, greatly exceeding the number of observations in similar studies (Allen-Diaz 1991, Castelli et al. 2000, Stringham et al. 2001, Dwire et al. 2006). Furthermore, we simulated water-table time-series data in 30 minute time steps, while the best temporal resolution in a similar study was obtained from water-table depth measurements taken every 10 days during the growing season (Stringham et al. 2001). To address this coarse temporal resolution issue, others have utilized statistical regression techniques to improve the temporal resolution of water-table dynamics (Henszey et al. 2004). Water-table depths vary on inter-annual and intra-annual scales, and results derived from just one year pose the potential for misleading results. While we simulated three complete water years, other studies have used observation data from only one complete growing season, augmented by incomplete portions of other years (Castelli et al. 2000, Dwire et al. 2006). Ideally, longer periods of water-table data should be used to reduce the effects of inter-annual climactic variation. Other studies have successfully simulated up to 20 years of water-table time-series data for use in the development of water-table – vegetation relationships (Rains et al. 2004). Lastly, similar studies have encountered dry wells during the later portions of the growing season, thereby biasing the water-table – vegetation relationships. Our approach allows for a complete high temporal resolution time series, in many cases demonstrating that water-table depths have a greater variation than previously reported.

Vegetation – Water-Table Relationships

Previous studies have documented various plant community types at distinct positions along the hydrologic gradient in pristine and degraded meadow environments (Allen-Diaz 1991, Chambers et al. 1999, Castelli et al. 2000, Law et al. 2000, Stringham et al. 2001, Dwire et al. 2006). These studies provide valuable information towards understanding the potential influence of changing water-tables on vegetation distributions. However, no previous studies have confirmed such vegetation – water-table relationships in meadow environments where hydrologic processes were restored, thereby justifying their use in restoration planning.

Plant community types were largely distributed along a hydrologic gradient (Table 2.2 and Figures 2.5-2.8). The *Poa pratensis / Bromus japonicus* community type was located at the xeric end of the hydrologic gradient, the *Carex nebrascensis / Juncus balticus* and *Downingia bacigalupii / Psilocarphus brevissimus* community types were located in the middle of the hydrologic gradient, and the *Eleocharis macrostachya / Eleocharis* *acicularis* community type was located at the hydric end of the hydrologic gradient. The *Carex nebrascensis / Juncus balticus* and *Downingia bacigalupii / Psilocarphus brevissimus* community types shared few species in common, with Jaccard's and Sorenson's indices of similarity (calculated with presence/absence data) of only 8 and 15, respectively (Table 2.3). However, water-table depths for the *Carex nebrascensis / Juncus balticus* and *Downingia bacigalupii / Psilocarphus brevissimus* community types did not vary significantly. One possible explanation of the cooccurence of these different community types within this water-table niche relates to the equilibrium assumption of the study. The *Downingia bacigalupii / Psilocarphus brevissimus* community is comprised primarily by annual species, which reflect very early seral conditions (Smith 1998). Perhaps with more time, these sites will be colonized by the rhizomatous, perennial species which dominate the *Carex nebrascensis / Juncus balticus* community. Another possible explanation is that another gradient not investigated exerts a larger degree of control than water-table depth for at least one of these community types. For example, active scour and/or deposition due to flooding may provide exposed substrate that favors the annual species which dominate the *Downingia bacigalupii / Psilocarphus brevissimus* community type. Indeed the *Downingia bacigalupii / Psilocarphus brevissimus* community type had significantly larger amounts of bare ground than the sites occupied by the *Carex nebrascensis / Juncus balticus* community.

Post-restoration vegetation distributions throughout the meadow demonstrate the benefits of hydrologic restoration. Prior to restoration, water-table depths in the meadow were largely conducive to the support of the *Poa pratensis / Bromus japonicus* community type (Hammersmark et al. In Review). The *Poa pratensis / Bromus japonicus* community type was dominated by these two introduced species, commonly found in riparian meadows with high grazing pressure and relatively high water-table depths (Allen-Diaz 1991, Smith 1998, Martin and Chambers 2001). Following restoration, water-table depths decreased and became more conducive to the *Carex nebrascensis / Juncus balticus*, *Downingia bacigalupii / Psilocarphus brevissimus*, and *Eleocharis macrostachya / Eleocharis acicularis* community types. Therefore, hydrologic restoration resulted in widespread shifts from the former to the latter three community types (Figure 2.3), increasing the relative proportion of native species in the meadow and meadow-scale biodiversity.

Keeley and Zedler (1998) define vernal pools as "precipitation-filled seasonal wetlands inundated during a period when temperature is sufficient for plant growth, followed by a brief waterlogged terrestrial stage and culminating in extreme desiccating soil conditions of extended duration." Rains et al. (In Review) quantified this for particular cases where vernal pools on contrasting soils were inundated for ~ 150-200 days and desiccated for much of the remainder of the year. The *Downingia bacigalupii / Psilocarphus brevissimus* community type had a similar hydrologic regime, with inundation for ~ 90-180 days and desiccation for much of the remainder of the year. However, these are not true vernal pools, as the water sources are primarily overbank flooding and regional groundwater flows rather than ponded direct precipitation and local perched groundwater flow as is more typical in vernal pools (Rains et al. 2006, Rains et al. In Review). In addition, this extended duration of inundation is much longer than typically found in vernal pools (Bauder 2000, Barbour et al. 2005, Barbour et al. 2006). Nevertheless, similarities between the plant community types along intermittent streams such as this and vernal pools suggest these habitats occur along a geomorphic-floristic continuum (Talley 2007), and merit further study.

Comparisons to Previous Studies

The plant community types described were similar to others described in previous studies. Regrettably, quantitative comparison of vegetation communities (e.g., Jaccard's or Sorenson's index of similarity) was not possible because data from previous studies were inconsistently and/or incompletely reported. In most cases, only a few of the most common species were reported (Allen-Diaz 1991, Law et al. 2000). In some studies, a more complete species list was reported (i.e., ~ 10 species listed), but the justification for inclusion is generally not stated (Castelli et al. 2000). An exception to these issues is Dwire et al. (2006), in which percent cover values for all species with ≥ 5% cover were reported. Nevertheless, similarity can be inferred from simple comparisons of the most prevalent species. Once identified, vegetation – water-table depth relationships can be compared in these similar plant community types.

The minimum, maximum, and ranges of the water-table depths in the community types in this study generally exceed those reported for similar community types in previous studies (Table 2.3). This is best documented for the *Poa pratensis / Bromus japonicus* and *Carex nebrascensis / Juncus balticus* community types. The *Poa pratensis / Bromus japonicus* community type occurred where the growing-season water-table depth ranged from ~ 10 - 230 cm. Seven similar community types were described in five previous studies, where the combined growing-season water-table depth ranged from ~ 0 - 150 cm. Similarly, the *Carex nebrascensis / Juncus balticus* community type occurred where the growing-season water-table depth ranged from ~ -20 cm (i.e., 20 cm above the ground surface) to 140 cm. Eight similar community types were described in seven previous studies, where the combined growing-season water-table depth ranged from ~ 0 - 100 cm. Though comparative data are limited, the same pattern may be true for the *Downingia bacigalupii / Psilocarphus brevissimus* and *Eleocharis macrostachya / Eleocharis acicularis* community types.

The higher minimum, maximum, and ranges of the water-table depths in the community types in this study could be a reflection of the greater temporal resolution of the hydrologic modeling approach. Discreet measurements are unlikely to reflect transient conditions, particularly transient higher water conditions associated with flood events. The higher maximum and ranges of the water-table depths in the community types could be a reflection of the fact that similar studies have encountered dry wells during the later portions of the growing season, thereby biasing the water-table – vegetation relationships. The higher maximum and ranges of the water-table depths in the community types also could indicate that access to water is more critical in the early growing season when individuals are growing, flowering, and fruiting than in the late growing season when individuals have set seed and are preparing to senesce.

In this regard, perennial vs. intermittent stream flow is a crucial hydrologic aspect that should be reported when vegetation – water-table relationships are reported. In many riparian systems, stream flow controls water-tables in the adjacent riparian area (Dwire et al. 2006, Hammersmark et al. In press). Therefore, water-tables adjacent to intermittent streams can have larger ranges than water-tables adjacent to perennial streams. This, in turn, can strongly affect mean growing season water-table depths.

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Table 2.1. Common species with mean cover ≥ 1% of the four community types.

Community / Common Species	Mean Cover	Cover Range	Constancy	Native/ Introduced	Duration[1]	WIC[2]
Poa pratensis / Bromus japonicus						
Poa pratensis L. ssp. *pratensis**	19.5	0 - 80	93	I	P	F
Bromus japonicus Murr*	13.2	0 - 65	96	I	A	FU
Iris missouriensis Nutt.	6.5	0 - 58	48	N	P	FW
Juncus covillei Piper	5.0	0 - 36	46	N	P	FW
Aster occidentalis (Nutt.) Torrey & A. Gray	5.0	0 - 38	38	N	P	F
Leymus triticoides (Buckley) Pilger	3.7	0 - 43	48	N	P	F
Juncus balticus Willd.	2.2	0 - 17	64	N	P	FW
Potentilla gracilis Hook var. *gracilis*	2.2	0 - 38	45	N	P	FW
Epilobium brachycarpum C. Presl*	1.5	0 - 11	80	N	A	OU
Elymus trachycaulus (Link) Shinn.	1.2	0 - 25	29	N	P	F
Phlox gracilis E. Greene	1.2	0 - 15	80	N	A	FU
Achillea millefolium L.	1.0	0 - 9	41	N	P	FU
Carex nebrascensis / Juncus balticus						
Carex nebrascensis Dewey*	16.6	0 - 81	59	N	P	OW
Juncus balticus Willd.	12.6	0 - 58	80	N	P	FW
Juncus covillei Piper	6.5	0 - 70	43	N	P	FW
Plagiobothrys stipitatus var. *micranthus* (Piper) I.M. Johnston	4.9	0 - 50	70	N	A	OW
Potentilla gracilis Hook var. *gracilis*	2.4	0 - 37	41	N	P	FW
Iris missouriensis Nutt.	2.3	0 - 35	25	N	P	FW
Carex athrostachya Olney	1.9	0 - 13	61	N	P	FW
Alopecurus pratensis L.	1.4	0 - 13	41	I	P	FW
Leucanthemum vulgare Lam.	1.4	0 - 35	29	I	P	NI
Penstemon rydbergii Nelson var. *oreocharis* (E. Greene) N. Holmgren	1.2	0 - 16	48	N	P	F
Elymus trachycaulus (Link) Shinn.	1.2	0 - 30	27	N	P	F
Bromus japonicus Murr	1.2	0 - 25	41	I	A	FU
Poa pratensis L. ssp. *pratensis*	1.0	0 - 11	41	I	P	F
Navarettia intertexta (Benth.) Hook.	1.0	0 - 18	68	N	A	FW
Juncus nevadensis S. Watson	1.0	0 - 20	36	N	P	FW
Downingia bacigalupii / Psilocarphus brevissimus						
Downingia bacigalupii Weiler*	15.7	0 - 60	86	N	A	OW
Psilocarphus brevissimus var. *brevissimus**	9.1	0.5 - 55	100	N	A	OW
Navaretia leucocephala ssp. *minima* (Nutt.) Day	5.9	0 - 24	71	N	A	FW
Mimulus tricolor Lindley*	4.6	0 - 14	86	N	A	OW
Plagiobothrys stipitatus var. *micranthus* (Piper) I.M. Johnston	3.9	0.5 - 9	100	N	A	OW
Juncus nevadensis S. Watson	2.6	0 - 14	57	N	P	FW
Madia elegans Lindley	1.6	0 - 10	29	N	A	OU
Eremocarpus steigerus (Hook.) Benth.	1.5	0 - 5	71	N	A	OU
Rumex crispus L.	1.2	0 - 6	57	I	P	FW
Polygonum polygaloides Meissner ssp. *confertiflorum* (Piper) J. Hickman*	1.0	0.5 - 2	100	N	A	OW
Veronica peregrina ssp. *xalapensis* (Kunth) Pennell*	1.0	0 - 3	86	N	A	OW
Eleocharis macrostachya Britton	1.0	0 - 3	43	N	P	OW
Eleocharis macrostachya / Eleocharis acicularis						
Eleocharis macrostachya Britton*	38.9	4 - 70	100	N	P	OW
Eleocharis acicularis (L.) Roemer & Schultes	10.8	0 - 50	67	N	AP	OW
Juncus nevadensis S. Watson	5.1	0 - 40	56	N	P	FW
Juncus balticus Willd.	3.0	0 - 21	33	N	P	FW
Carex athrostachya Olney	2.2	0 - 5	78	N	P	FW
Carex nebrascensis Dewey	1.7	0 - 11	22	N	P	OW
Iris missouriensis Nutt.	1.6	0 - 11	22	N	P	FW

Notes follow on next page.

Table 2.1 Notes:
* - indicator species of the community type
1 - Duration: A = annual and P = perennial
2 - Wetland Indicator Category (WIC, U.S. Fish and Wildlife Service 1996): OW-obligate wetland, FW-facultative wetland, F-facultative, FU-facultative upland, OU-obligate upland.

Table 2.2. Species richness, wetland indicator score and simulated growing-season water-table depths (WTD) for the four community types. Values are reported as means ± standard deviations. Superscript letters indicate significant differences (α= 0.05).

Community	*Eleocharis macrostachya / Eleocharis acicularis*	*Downingia bacigalupii / Psilocarphus brevissimus*	*Carex nebrascensis / Juncus balticus*	*Poa pratensis / Bromus japonicus*
n	7	7	47	67
Species Richness	8.3 ± 5.1^{a}	16.7 ± 5.0^{b}	18.0 ± 7.2^{b}	18.9 ± 5.2^{b}
Vegetation Cover (%)	82.7 ± 24.4^{a}	56.1 ± 26.4^{a}	76.7 ± 18.2^{a}	82.5 ± 19.2^{a}
Bare Ground Cover (%)	16.4 ± 18.1^{abc}	42.4 ± 28.2^{a}	12.0 ± 18.7^{b}	4.5 ± 8.9^{c}
Litter Cover (%)	9.3 ± 6.5^{ab}	5.9 ± 6.2^{b}	17.8 ± 13.9^{ab}	20.1 ± 10.7^{b}
Wetland Indicator Score	1.19 ± 0.22^{a}	1.47 ± 0.46^{ab}	1.92 ± 0.57^{b}	3.15 ± 0.41^{c}
WTD average (cm)	18.4 ± 28.0^{c}	58.5 ± 19.8^{bc}	60.3 ± 12.6^{b}	119.4 ± 44.4^{a}
WTD minimum (cm)	-66.2 ± 30.7^{c}	-33.4 ± 29.2^{b}	-22.1 ± 15.7^{b}	12.1 ± 24.2^{a}
WTD maximum (cm)	94.8 ± 24.2^{b}	154.2 ± 11.1^{b}	137.4 ± 25.2^{b}	231.2 ± 74.1^{a}
WTD range (cm)	161.0 ± 20.0^{b}	187.6 ± 39.9^{ab}	159.5 ± 30.5^{b}	219.1 ± 66.6^{a}
Days WT < 70 cm	91.3 ± 20.5^{c}	65.4 ± 8.8^{b}	65.6 ± 7.5^{b}	41.6 ± 18.3^{a}
Days WT < 30 cm	65.4 ± 16.1^{c}	46.8 ± 18.0^{b}	42.4 ± 10.2^{b}	22.3 ± 11.4^{a}
Days WT < 0 cm	49.7 ± 17.2^{c}	33.7 ± 18.3^{b}	24.9 ± 8.4^{b}	9.8 ± 7.1^{a}

Table 2.3. Growing-season, water-table depths for the four community types identified in this study and similar community types found in the literature.

Community Name	**Growing Season WTD (cm)**	**Source**
Poa pratensis / Bromus japonicus	10 - 230	this study
Poa pratensis / Potentilla gracilis	26 - 62	Allen-Diaz, 1991
Moist meadow	~ 0 - 50	Dwire et al., 2006
Dry meadow	~20 - 85	Dwire et al., 2006
Mesic meadow (Corral Canyon)	90 - 150	Castelli et al., 2000
Moist bluegrass	~35 - 120	Stringham et al., 2001
Dry bluegrass	~80 - 140	Stringham et al., 2001
Mesic graminoid	midseason soil saturation at 55-100	Weixelman et al., 1996
Carex nebrascensis / Juncus balticus	~ -20 - 140	this study
Carex nebrascensis ecological type	0 - 20	Chambers et al., 1999
Wet meadow	0 - 30	Castelli et al., 2000
Wet meadow	0 - 30	Chambers et al. 2004a
Deschampsia caespitosa / Carex nebrascensis	6.4 - 93.8	Allen-Diaz, 1991
Moist meadow	~20 - 100	Stringham et al., 2001
Carex nebrascensis ecological type	season long soil saturation at 50	Weixelman et al., 1996
Carex nebrascensis community type	33 [1]	Smith, 1998
Juncus balticus community type	66 [1]	Smith, 1998
Downingia bacigalupii/Psilocarphus brevissimus	-34 - 154	this study
Downingia bicornuta community type	-33 [1]	Smith, 1998
Navarretia community type	33 [1]	Smith, 1998
Eleocharis macrostachya / Eleocharis acicularis	-66 - 95	this study
Eleocharis macrostachya community type	0 [1]	Smith, 1998

Notes:
1 - Values represent single water-table depth measurements, which do not reflect the temporal variation present during the growing season.

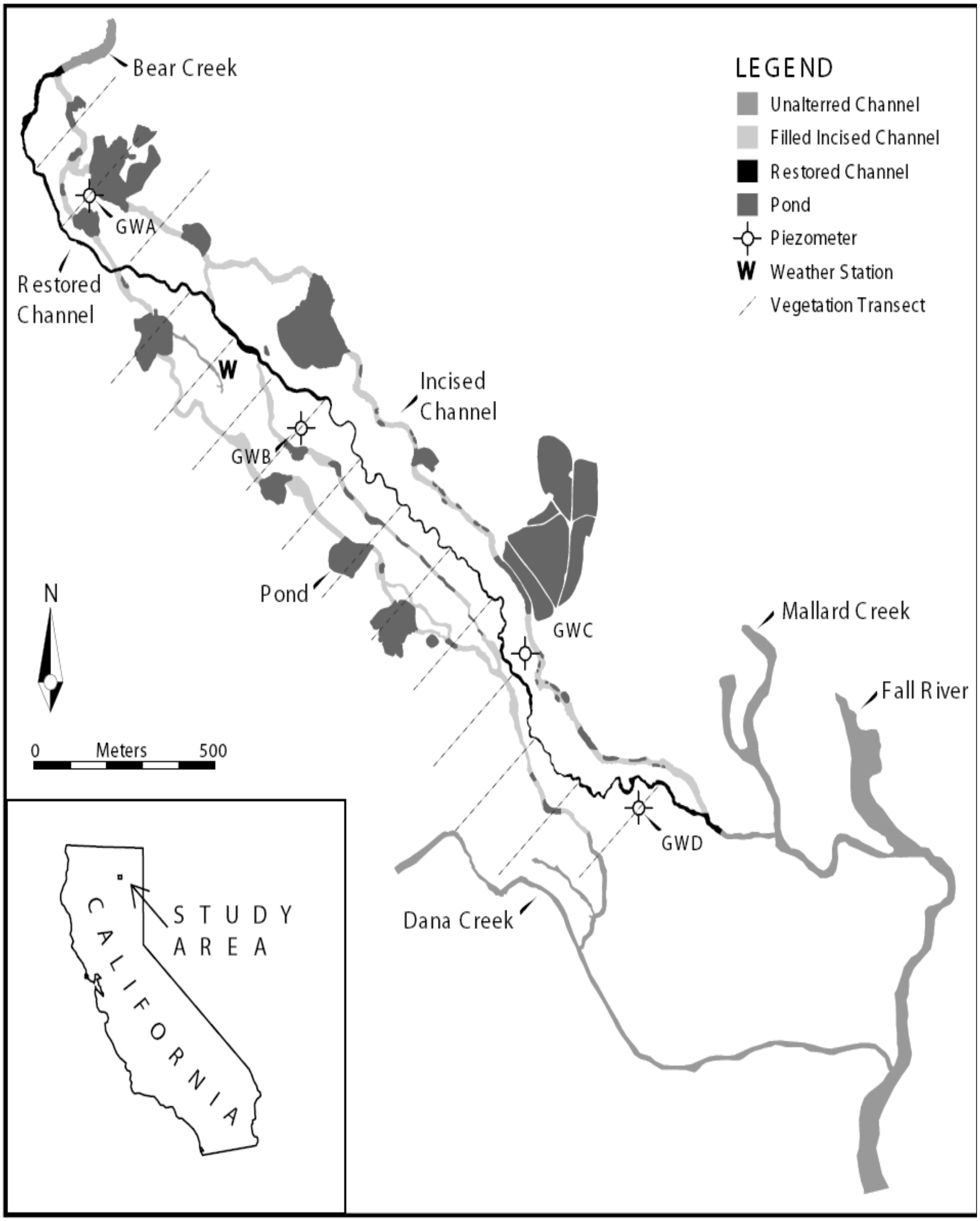

Figure 2.1. Bear Creek Meadow study area. Portions of the incised channels were filled with alluvium excavated from ponds throughout the meadow. A 3.6 km single thread restored channel reach was created from remnant channel segments and excavated where necessary. Flow direction is from upper left to lower right.

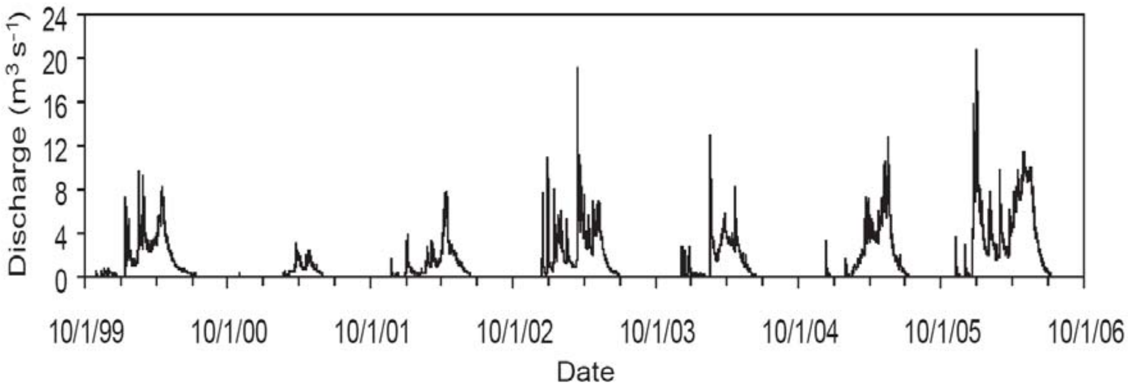

Figure 2.2. Bear Creek discharge at the upstream extent of the restored reach for the water years of 2000 - 2006. Stream discharge is intermittent, with flood peaks resulting from rainfall, rain on snow, and spring snowmelt. Annual peak discharge ranged from 3.1 - 20.7 m^3s^{-1} and the annual duration of surface flow ranged from 98 - 229 days.

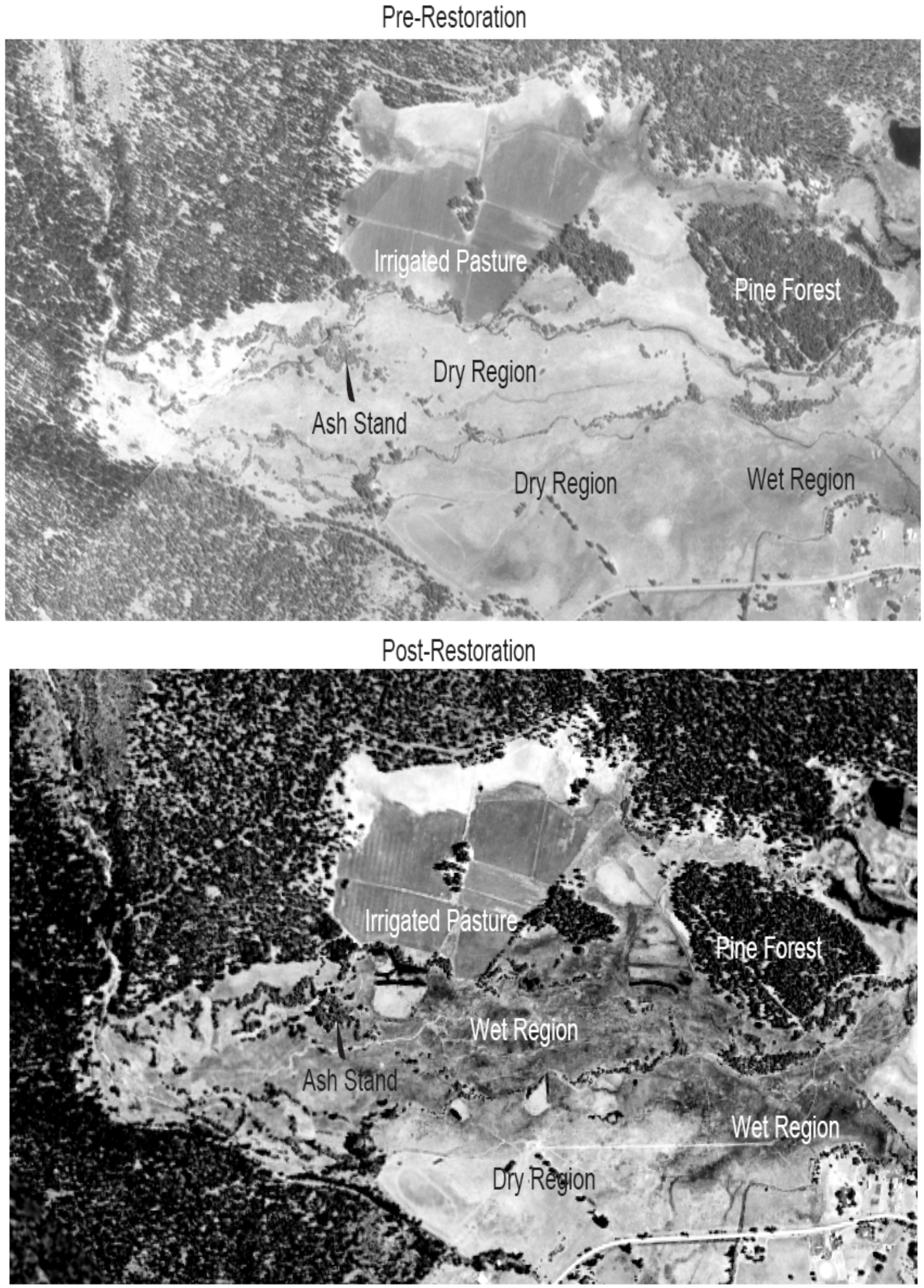

Figure 2.3. Pre- and post-restoration aerial photographs of the meadow. Qualitative comparisons indicate an increase in mesic and hydric vegetation in the post-restoration photograph. The region immediately below the irrigated pasture and the pine forest experienced the largest degree of hydrologic alteration, and subsequent herbaceous vegetation change. Wet region labels indicate the area occupied by mesic-hydric vegetation communities.

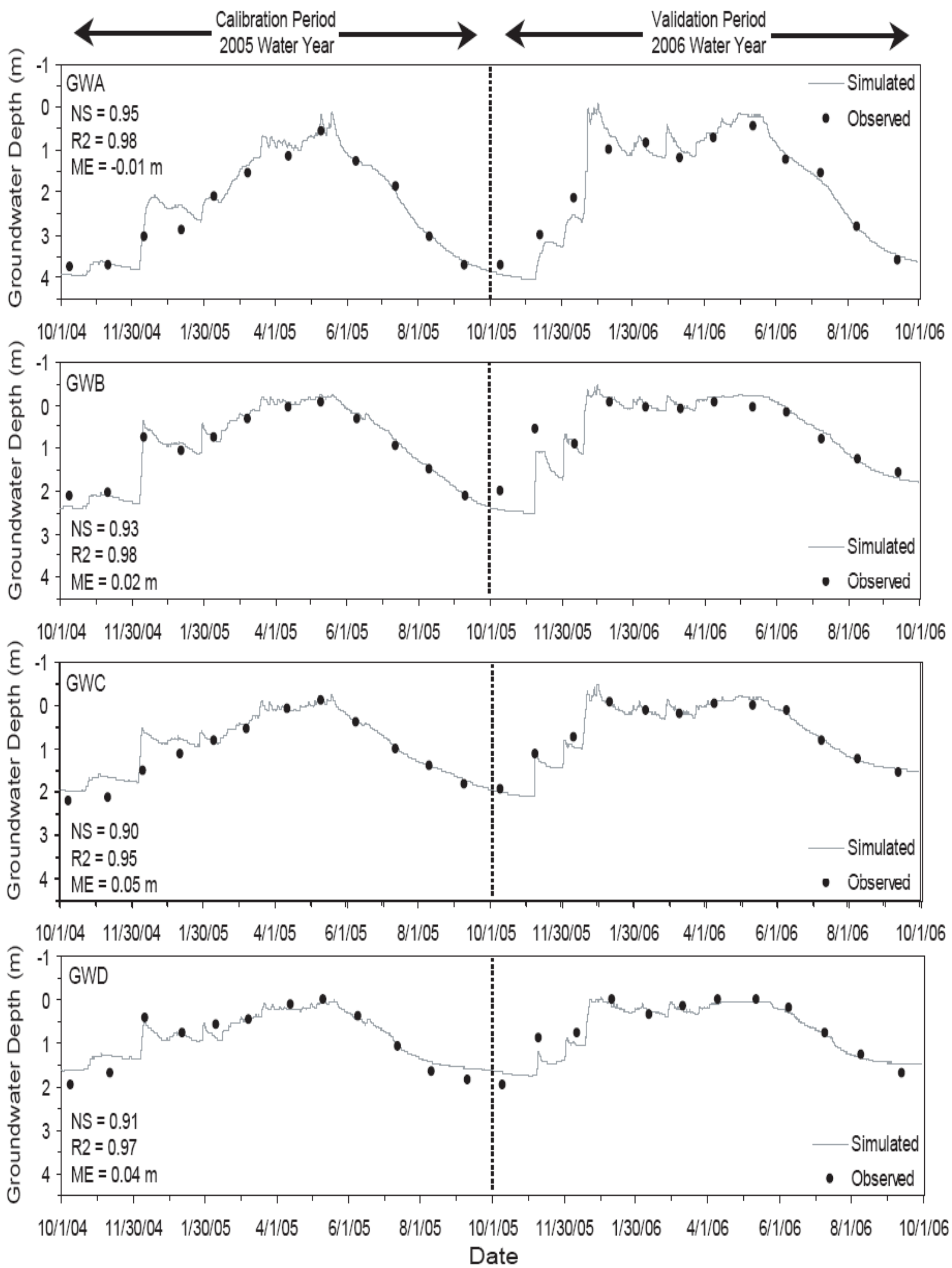

Figure 2.4. Comparisons of simulated and observed groundwater depths at four piezometers within the meadow. The 2005 water year (left side) was used for model calibration and the 2006 water year (right side) was used for model validation. Nash-Sutcliffe efficiency coefficients (NS), correlation coefficients (R2) and mean error values (ME) are provided for each location. Negative groundwater depths indicate surface inundation that is common in the restored meadow. Piezometer locations are shown on Figure 2.1.

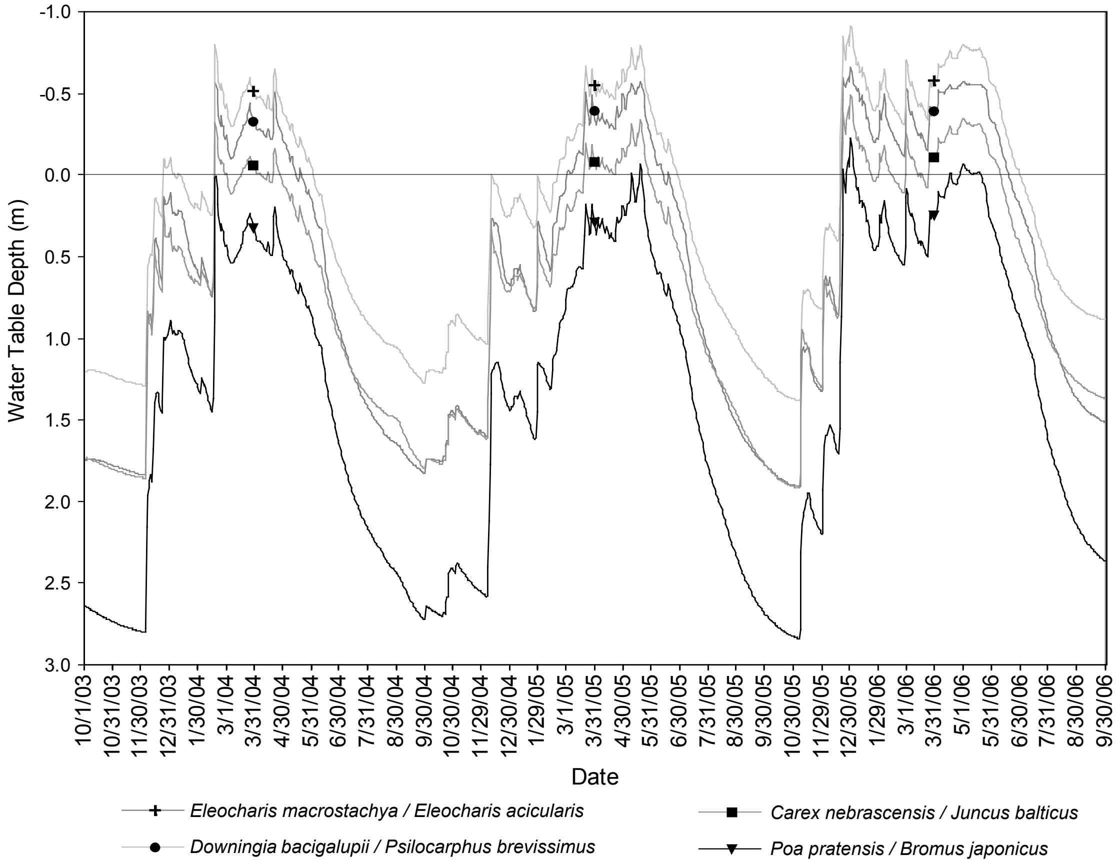

Figure 2.5. Mean time series of water-table depths for the four community types for water years 2004 - 2006. Negative water-table depths indicate inundation.

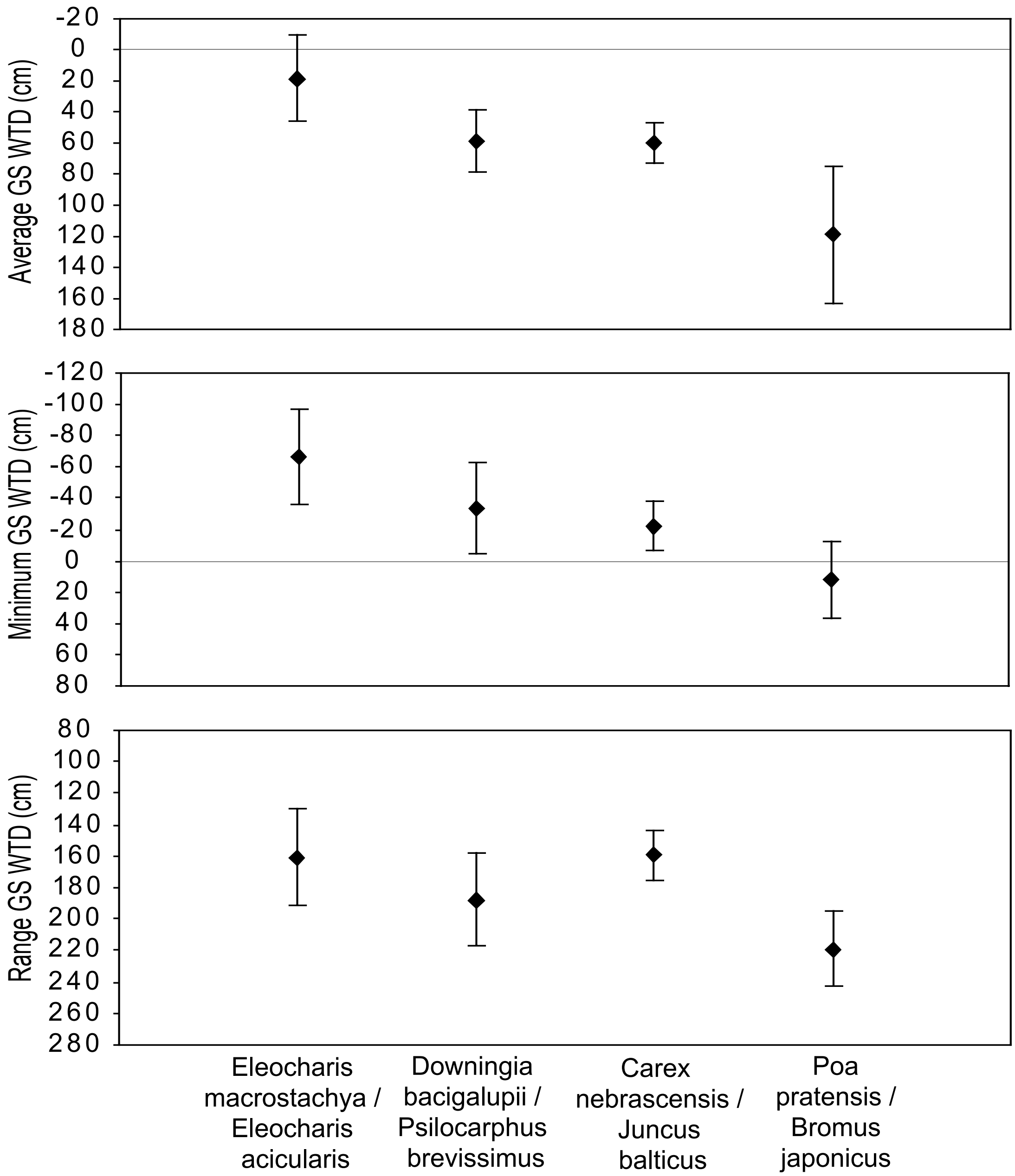

Figure 2.6. Mean (±standard deviation) growing-season, water-table depths (GS WTD) for the four community types.

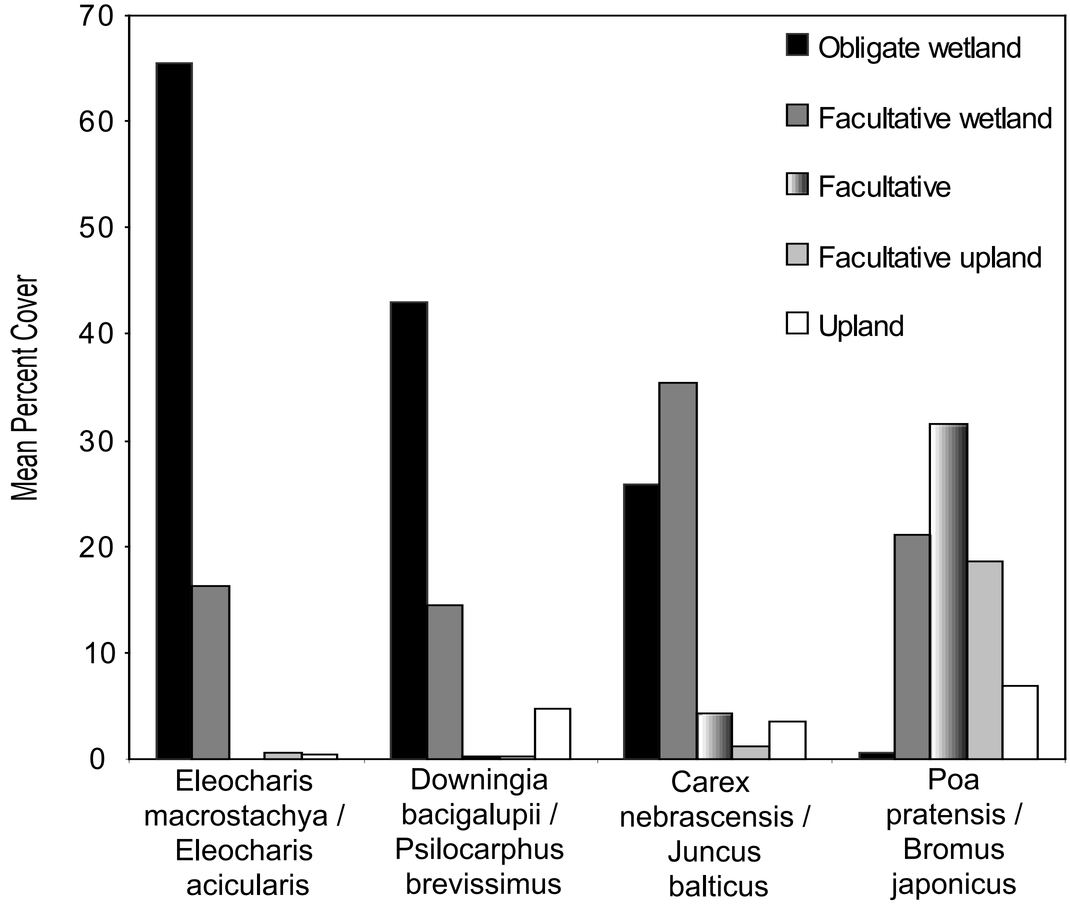

Figure 2.7. Mean percent cover of plant species in each wetland indicator category within the four community types.

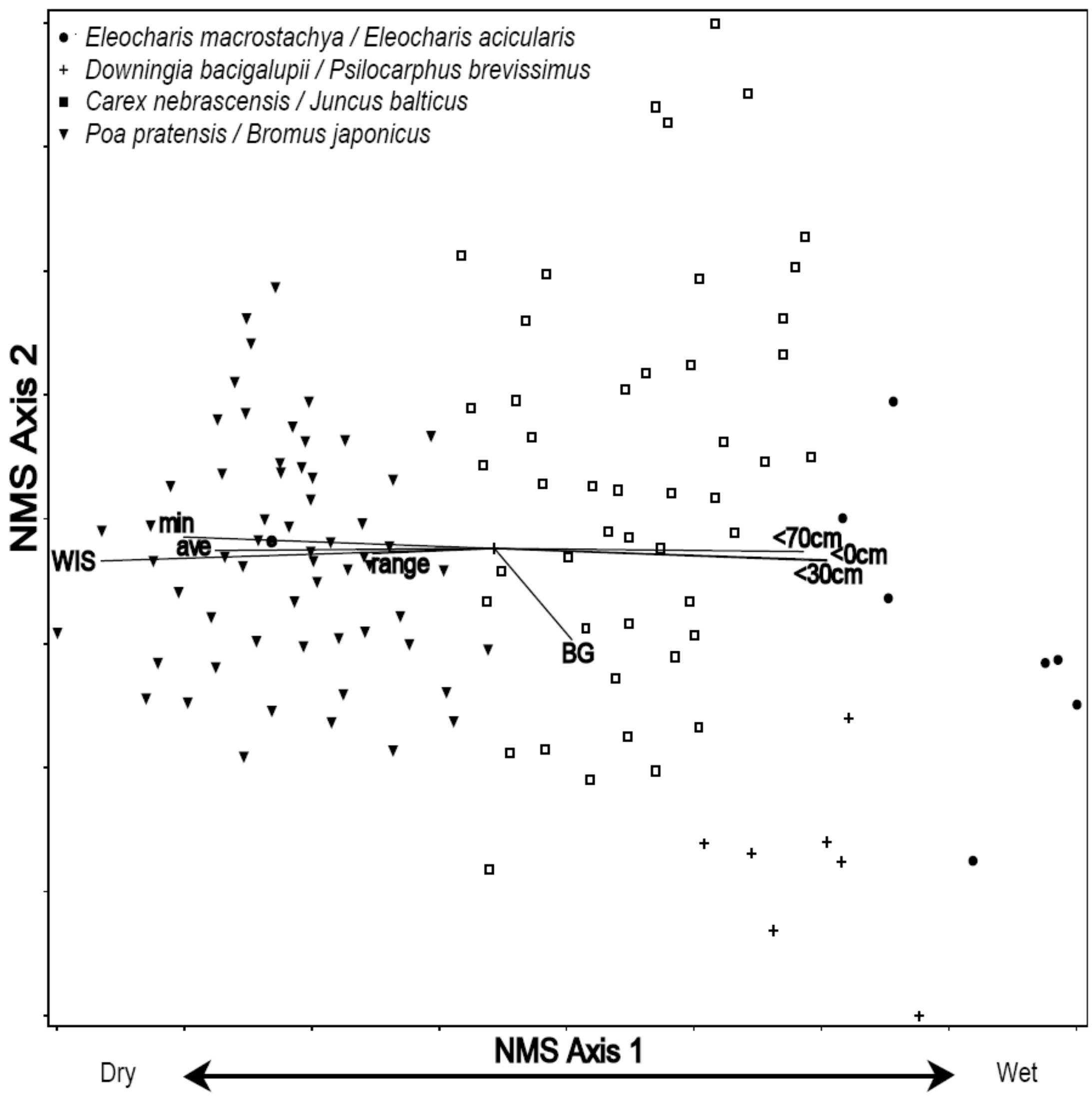

Figure 2.8. Two-dimensional nonmetric multidimensional scaling ordination biplot displaying the vegetation samples, their community types, and the correlation of environmental variables with the two axes. Community types are shown with different symbol types, and correlation strength with the various environmental variables are represented by the scale of the arrow. Axis 1 is highly correlated with many of the hydrologic variables. Min, ave and range refer to minimum, average and range of growing-season, water-table depths; <0cm, <30cm and <70cm refer to the number of days the water-table was < 0 cm, 30 cm and 70 cm from the ground surface; and WIS and BG refer to the wetland indicator score and percent cover of bare ground, respectively.

CHAPTER 3 - Simulated Effects of Stream Restoration on Herbaceous Vegetation Distribution

ABSTRACT
Meadow restoration efforts are increasingly commonplace typically involving the restoration of stream channels to re-establish hydrologic conditions necessary for wetland plant species. Particularly common are "pond and plug" type stream restoration projects, in which (a) alluvial materials are excavated from the floodplain, forming ponds; (b) excavated alluvial materials are used to plug incised channels; and (c) channels are restored to the floodplain surface. Despite the large number of "pond and plug" restoration projects undertaken in the western United States, the ability to predict effects of topographic modification upon hydrology and riparian vegetation in these systems is limited. To predict the changes in the distribution of commonly found meadow riparian plant species a hydrologic model and a suite of individual vegetation species models were used in concert. First we developed, calibrated and validated a MIKE SHE hydrological model of a 230 ha mountain meadow along a 3.6 km restored reach of Bear Creek in northeastern California. Next, vegetation presence/absence data from 170 plot locations distributed throughout the study area were combined with simulated water-table depth time series to develop species distribution models for individual plant species. In each vegetation model, the probability of occurrence is predicted as a function of growing season water-table depth and range. The hydrologic model was used to simulate water-table depth time series for the pre- and post-restoration topographic conditions, and these results were used to predict the distribution of several plant species. Hydrologic modeling results indicated significant changes to shallow groundwater levels throughout the study area, extending well beyond the near stream region. Combined modeling results indicated an increase in the spatial distribution of obligate wetland, and facultative

wetland plant species, as well as a decrease in the distribution of facultative upland and obligate upland plant species. The methods utilized in this study could be used to improve realistic objective setting in similar projects in similar environments, in addition to providing a quantitative, science-based approach to guide riparian restoration and active re-vegetation efforts.

INTRODUCTION

In the arid west, riparian areas are ecologically significant and economically important areas that occupy a relatively small percentage of the landscape. Currently, over half of the riparian areas in the Great Basin exist in a poor ecological condition due to both natural and anthropogenic disturbances (Jenson and Platts 1990, Tausch et al. 2004). A common disturbance in these systems is the lowering of water-tables due to stream channel incision (Martin and Chambers 2001, Chambers et al. 2004). While incision has been attributed to geologic factors in many meadow complexes of the Great Basin (Germanoski and Miller 2004), incision also has been attributed to anthropogenic influences including channelization (Emerson 1971) and overgrazing (Kauffman and Krueger 1984, Fleischner 1994, Trimble and Mendel 1995).

Stream channelization and subsequent incision lower water-tables (Choate 1972, Schilling et al. 2004) resulting in altered riparian vegetation patterns and species composition (Jewitt et al. 2004, Loheide and Gorelick 2007). In an effort to improve the ecological condition of degraded streams and their adjacent riparian corridors, stream restoration has grown in popularity. An increasingly common technique for raising water-tables in incised, degraded meadow environments is the “pond and plug” method, also referred to as meadow re-watering. A frequently stated objective of many of these projects is to raise groundwater levels in order to improve the health of riparian vegetation (Benoit and Wilcox 1997, Rosgen 1997, Doll et al. 2003, Poore 2003).

Previous studies have demonstrated the relationship between herbaceous riparian vegetation and water-table depth observations in pristine, degraded and/or restored meadows in northeastern California (Hammersmark el al. In prep.), central Nevada (Chambers et al. 1999, Castelli et al. 2000), eastern Oregon (Stringham et al. 2001, Dwire et al. 2006), the Sierra Nevada (Allen-Diaz 1991, Loheide and Gorelick 2007) and western Montana (Law et al. 2000). Despite the body of literature relating riparian vegetation to water-table depth, considerable uncertainty regarding the effects of system modification accompanies most stream restoration efforts (Wohl et al. 2005). Palmer and Bernardt (2006) suggest that efforts to evaluate the ecological effectiveness of floodplain reconnection and channel reconfiguration restoration projects should be given top research priority. In an effort to address the uncertainty surrounding the ecological effectiveness of “pond and plug” stream restoration efforts, we developed a quantitative, science-based tool to predict the changes in herbaceous vegetation distribution due to stream restoration. Towards this end, our objectives were to: 1) develop, calibrate, and validate a hydrologic model of a well-documented “pond and plug” restoration project; 2) develop a suite of vegetation models linking water-table depth to the probability of occurrence of common herbaceous meadow species; and 3) use these models in concert to predict the changes in distribution of individual species due to hydrologic alteration caused by stream restoration.

STUDY AREA

Bear Creek Meadow (meadow) is a low-gradient alluvial floodplain ~100 km northeast of Redding in northern California, USA (Figure 3.1). The meadow is located at an elevation of ~1010 m, and is situated at the bottom of the ~218 km^2 Bear Creek watershed, immediately upstream of the confluence of Bear Creek with the Fall River, the largest spring-fed river system in California (Grose 1996), and among the largest spring-fed river systems in the United States (Meinzer 1927, Rose et al. 1996).

The meadow is approximately 3 km long, 1 km wide, 230 ha in size, and is situated at the northwestern margin of the Fall River Valley near the intersection of the Modoc Plateau and the Cascade Range. The head of the meadow lies at the base of a relatively-steep, forested bedrock reach. The Fall River Valley is underlain by lacustrine deposits consisting of clay, silt and sand. In the meadow, these lacustrine deposits are overlain by 0.5-2 m of deltaic sands and gravels, and 1-3 m of floodplain silty loam soils (Grose 1996). Based on the survey, the dominant soil type is the Matquaw gravelly sandy loam, a mixed, active, mesic Pachic Ultic Haploxeroll (NRCS 2003).

The climate of the Fall River Valley is semi-arid, receiving an annual average of 508 mm (± 243 mm standard deviation) of precipitation (California Irrigation Management System data for McArthur for water years 1984-2006). Most precipitation in the Fall

River Valley occurs as rainfall in late fall-early spring. Higher elevation areas of the Bear Creek watershed, located to the north and west of the meadow, receive considerably more precipitation as snow and rain in late fall-early spring.

The hydrological system of the study area consists of intermittent surface-water inflow from Bear Creek and Dana Creek and perennial spring discharge from the Fall River spring system (Figure 3.1). The Fall River spring system is fed by meteoric water, which falls on the Medicine Lake Highlands, perches on low-permeability lacustrine deposits, flows south through fractured basalt and discharges at the downstream end of the meadow (Rose et al. 1996). These springs form the headwaters of the Fall River and several short perennial tributaries (i.e., Mallard Creek and Lower Dana Creek). The local groundwater system is unconfined and down-valley fluxes occur primarily through the deltaic silts, sands and gravels of the shallow subsurface.

Surface-water input to the meadow is supplied primarily by intermittent Bear Creek and secondarily by intermittent Dana Creek, which bounds the southwestern edge of the lower meadow (Figure 3.1). Stream discharge results from spring snowmelt, and fall, winter, and spring rain events, including episodic rain-on-snow events. In the 7 years following the restoration in 1999 that is described below, peak discharge in Bear Creek measured at the head of the meadow ranged from 3.1-20.7 m^3s^{-1}. In these 7 years, the duration of surface flow for each water year ranged from 98-229 days.

Anthropogenic Disturbance, Incision, Widening, and Restoration

Prior to restoration, the meadow was channelized and overgrazed (Poore 2003), resulting in degradation of both aquatic and terrestrial ecosystems of the meadow and the Fall River immediately downstream (Spencer and Ksander 2002). After several years of pre-restoration data collection and consultation, the meadow's incised channels were restored in 1999 as a joint venture between the California Department of Fish and Game and a private landowner. The restoration design followed the "Natural Channel Design" method developed by David Rosgen (Rosgen 1996, Malakoff 2004). A "priority 1" approach (Rosgen 1997), more commonly referred to as a "pond and plug" or meadow re-watering strategy was utilized.

Following the usual "pond and plug" method, the incised stream channels were intermittently filled with plugs of locally derived alluvial material. The remaining unfilled incised channel segments were left as ponds, and many were enlarged to provide the fill material necessary to plug portions of the incised channels. When configuring the restored channel, existing remnant channel segments were used when possible, connected by sections of excavated new channel. The restored channel was constructed with reduced width, depth, and cross-sectional area (Poore 2003). Average channel depth at riffles, was reduced from 2.69 m to 0.89 m and average channel capacity was reduced from 61.7 m^3s^{-1} to 5.35 m^3s^{-1} (Hammersmark et al. In press). The restored channel has a meandering riffle-pool morphology, classified as C4 and E4 types of the Rosgen classification system (Rosgen 1996, Poore 2003). Upon completion, a 3.6 km single thread sinuous channel connected the bedrock controlled upstream reach to the unaltered downstream reach (Figure 3.1). In addition, 17 ha of new ponds (remnant gully segments and fill sources) exist throughout the meadow.

Hydrologic Effects of Restoration

Topographic modification of the stream channels and floodplain surface resulted in substantial changes to the surface water and groundwater regimes of the meadow, including: 1) increased groundwater levels and volume of subsurface storage; 2) increased frequency of floodplain inundation and decreased magnitude of flood peaks; 3) decreased baseflow and annual runoff; and 4) increased evapotranspiration (Hammersmark et al. In press). Specifically, spring and summer meadow average groundwater levels were increased by 1.20 m and 0.34 m, respectively, above pre-restoration levels. These are meadow-averaged values, with greater changes occurring near the channels and smaller differences occurring at the distal margins of the meadow. In addition, a greater than ten fold reduction in the channel capacity increased the frequency and duration of floodplain inundation. For the two water years simulated in the study, overbank flooding did not occur in the pre-restoration scenario, however, overbank flooding was frequent and of long duration in the restored scenario, with 13 widespread flooding events (defined as when flows reached sufficient magnitude to exceed the average channel capacity of 5.35 m^3s^{-1}) for a total duration of 106 days (i.e., 27% of the time the stream was flowing) of overbank flooding. Based upon qualitative observations these hydrologic changes resulted in changes to the distribution of vegetation in the meadow (Figure 3.2).

METHODS

Hydrological Model Development

A numerical hydrological model was developed for the study area using the MIKE SHE modeling system (Refsgaard and Storm 1995), which is based upon the Systeme Hydrologique Europeen (SHE) model (Abbott et al. 1986a, b). MIKE SHE is a commercially available, deterministic, fully-distributed and physically-based modeling system that has been applied to a wide variety of problems where surface water and groundwater are closely linked (for examples see Jayatilaka et al. 1998, Thompson 2004, Sahoo et al. 2006, Hammersmark et al. In press). Using a finite difference methodology, MIKE SHE solves partial differential equations describing the processes of saturated subsurface flow (three-dimensional Boussinesq equation), unsaturated subsurface flow (one-dimensional Richards' equation), channel flow (one-dimensional St. Venant equations), and overland flow (diffusion wave approximation of the two-dimensional St. Venant equations). Channel hydraulics are simulated with the one-dimensional MIKE 11 hydraulic modeling system which is dynamically coupled to the MIKE SHE modeling system. The processes of interception and evapotranspiration are handled with analytical solutions.

Separate MIKE SHE/MIKE 11 models were developed for the pre-restoration (i.e., incised) and post-restoration (i.e., restored) scenarios. Initially, a base model of the restored scenario was developed, calibrated and validated. Subsequently, the surface topography and channel size and alignments were altered to reflect the incised pre-restoration scenario. Altered surface topography and channel configuration were the only differences between the two models. All other components remained unchanged between the two models. The models were comprised of 2898 30 x 30 m^2 grid squares, representing a total area of 261 ha.

Surface topography was obtained from previous surveys of pre- and post-restoration scenarios. Two digital elevation models (DEMs) were developed, one representing the incised scenario and one representing the restored scenario. The DEM representing the restored scenario was updated in 2004 with an additional topographic survey. The DEMs were sampled on a 30 m grid to provide surface elevations to the model. Two MIKE 11 models were developed to reflect the altered channel configuration due to restoration. Channel alignments and cross sections were extracted for each MIKE 11 model from the pre- and post-restoration DEMs.

Grose (1996), coupled with three well logs from within the model domain, provided the conceptual model of the hydrostratigraphy, which was further refined with field investigations. Based upon the refined conceptual model, the subsurface component of the model was composed of three layers, with the lower layer a sandy clay, the middle layer a high-permeability alluvial sand and gravel mixture, and the upper layer an alluvial silty-clayey loam. Hydraulic conductivity was computed for the upper two layers by conducting slug tests at three piezometers and analyzing the resulting data using the Bouwer and Rice method (1976). Mean hydraulic conductivity for six slug tests performed in the upper silty-clayey loam was 9.3×10^{-7} ms^{-1}, while mean hydraulic conductivity for five slug tests performed in the middle sand and gravel layer was 4.5×10^{-2} ms^{-1}. These values are consistent with those found in the literature for units with similar textural descriptions (Masch and Denny 1966, Adams and Gelhar 1992, Martin and Frind 1998, Woesner et al. 2001, Loheide and Gorelick 2007). No slug tests were conducted in the lower sandy clay unit, instead a hydraulic conductivity of 1.0×10^{-9} ms^{-1} was taken from the literature (Freeze and Cherry 1979, Martin and Frind 1998). Unsaturated hydraulic conductivity and moisture retention properties were adopted from Loheide and Gorelick (2007).

Vegetation inputs included the spatial extent of various vegetation types, in addition to leaf area index and root depth of each prescribed vegetation type. Three vegetation types were employed in the model: ash forest (dominated by *Fraxinus latifolia* and *Crataegus douglasii*), pine forest (dominated by *Pinus jeffreyi*), and grassland (dominated by *Poa pratensis*, *Bromus japonicus*, and *Juncus balticus*). The distribution of each vegetation type was determined through a combination of field reconnaissance and aerial photo interpretation. Meteorological data were collected at 15-minute intervals from a data logging weather station (HOBO weather station, Onset Computer Corporation) deployed within the meadow (Figure 3.1). Reference evapotranspiration was computed using these meteorological data and the FAO Penman-Montieth combination equation (Allen et al. 1998).

Additional input parameters included the leakage coefficient, which governs river-aquifer exchange, and channel and overland flow roughness coefficients (i.e., Manning's n). River-aquifer exchange was simulated with the reduced contact (b) method, with a value of 1.0×10^{-5} s^{-1} adopted from the literature (Thompson et al. 2004). Manning's n for channel flow was estimated to be 0.033 sm $^{-1/3}$, based upon values found in the literature for similar channel conditions (Chow 1959, Barnes 1967, Coon 1998). An initial floodplain Manning's roughness value of 0.5 sm $^{-1/3}$ was chosen from the literature (Thompson et al. 2004). Each of these values was subsequently altered during model calibration.

The subsurface domain boundaries consisted of a combination of no-flow and specified-flux subsurface external boundary conditions and one internal specified-head boundary condition. Observation data from 28 piezometers arranged along four transects were used to define the subsurface external boundary conditions. No-flow boundaries were on the upper portion of the meadow and along much of the southwestern border of the meadow. A short specified-flow boundary was along the northeastern border where subsurface irrigation runoff from an irrigated pasture discharges to the meadow. A flux of $2x10^{-2}$ m^3s^{-1} was applied during the June-September irrigation season, with zero flow applied to the remaining part of the year. The spring-fed, perennial streams Mallard Creek, Lower Dana Creek, and Fall River bound the downstream portion of the model domain (Figure 3.1). The specified head internal boundary was used for an area located in the southeastern portion of the meadow that received subsurface spring discharge. The low-permeability lacustrine clay underlying the meadow justified the use of a no-flow boundary along the bottom of the model domain.

The surface domain boundaries for the MIKE 11 models were developed from flow records of Bear Creek inflow, Mallard Creek inflow, Fall River inflow, Dana Creek inflow, Dana spring inflow to Lower Dana Creek and Fall River stage at the downstream extent of the model domain. Data logging pressure transducers (Solinst LT 3001 Leveloggers) were installed to provide stage hydrographs at each location. At the five inflow locations, over a wide range of flow levels, discharge was measured by standard velocity-area methods (Harrelson et al. 1994), water velocity measurements being collected with a flowmeter (Marsh-McBirney Flo-Mate). Flow measurements and corresponding stage levels were used to create rating curves/tables for each inflow location to allow the conversion of the stage hydrographs to discharge hydrographs.

Hydrologic Model Calibration and Validation

The hydrologic model was calibrated with 2005 water year data and validated with 2006 water year data. Hydrologic model calibration parameters included hydraulic conductivity, leakage coefficient, and channel and overland roughness coefficients. Uniform values for each of the parameters were used. The calibration consisted of individual parameter manipulation and subsequent model performance evaluation. Values of saturated hydraulic conductivity, leakage coefficient, and channel roughness were varied during the calibration process, but the best fit was achieved with the initial value estimates, which all fall within reasonable ranges of values found in relevant literature (Chow 1959, Masch and Denny 1966, Barnes 1967, Adams and Gelhar 1992, Coon 1998, Martin and Frind 1998, Woesner et al. 2001, Thompson et al. 2004, Loheide and Gorelick 2007). The value of overland roughness was decreased from 0.5 $sm^{-1/3}$ to 0.1 $sm^{-1/3}$, resulting in improved channel stage agreement and more closely resembles values for floodplains found in the literature (Chow 1959).

The hydrologic model performance evaluation during calibration and validation was based upon a combination of graphical assessment and statistical methods. The Nash-Sutcliffe efficiency coefficient was employed to statistically judge the performance of the model simulation as compared to observed data (Nash and Sutcliffe 1970, McCuen et al. 2006). The Nash-Sutcliffe efficiency coefficient is widely used when evaluating the statistical goodness-of-fit of model simulations, though time-offset bias and bias in magnitude have been observed (McCuen et al. 2006). In addition to the Nash-Sutcliffe efficiency coefficient, the correlation coefficient and the mean error for each comparison location were calculated and evaluated. Modeled and observed hydraulic heads were compared at 28 shallow piezometers, and modeled and observed stream stages were compared at two locations on Bear Creek within the meadow and one location on Bear Creek below the meadow.

Following calibration and validation (discussed below), each hydrologic model was used to simulate a 3-year period, the water years of 2004, 2005 and 2006 (i.e., 1 October 2003 through 30 September 2006). The annual precipitation of the 3 water years simulated ranged from average to above average. Annual precipitation was 510 mm (i.e., 100.2% of average), 529 mm (i.e., 104.1% of average), and 653 mm (i.e., 129.4% of average) for the 2004, 2005 and 2006 water years, respectively.

For each vegetation plot location (discussed below), a time series of water-table elevation was generated and combined with the ground surface elevation to yield a water-table depth time series. From each of the three annual time series, average and range of water-table depth during the growing season were calculated. The growing season was defined as May through August, the period in which the above ground parts of herbaceous plants were observed to be actively growing on site. The three annual values were then averaged to provide one value for each water-table depth variable at each plot location.

Vegetation Sampling

Plant species composition and aerial cover were sampled in 2 x 2 m plots placed along 15 transects aligned perpendicular to the down valley gradient. Along each transect, plots were systematically placed at 2 m, 5 m, 10 m, 20 m, 40 m, 80 m, 120 m, 160 m, 200 m, 300 m, 400 m and 500 m distances from the stream edge, as allowed by the width of the meadow, resulting in a total of 170 plots. Vegetation data were collected from 30 June to 20 July 2005 when plants were in flower and therefore more easily identified. Percent aerial cover of all vascular plants was ocularly estimated by three observers in 1% classes from 1-5% and then in 5% classes from 5-100% (Daubenmire 1959). In addition, rare species with only one or two individuals were recorded as 0.1% and species with less that 1% cover were recorded as 0.5%. The three ocular estimates were then averaged. Nomenclature follows Hickman (1993). Each species encountered was assigned to a wetland indicator category based upon its U. S. Fish and Wildlife Service (1996) wetland indicator status in the California region.

Vegetation Model Development and Evaluation

Preference models were developed for 11 herbaceous, vascular taxa to investigate the effect of hydrologic alteration due to stream restoration on species distributions. Species were chosen based upon two criteria: frequency of presence in the sample and wetland indicator category membership. Only species with ≥ 30 occurrences were considered. From this subset of the herbaceous species present, 2-3 species were chosen from each of the 5 wetland indicator categories. For each of the resulting 11 species, preference models were developed with logistic generalized additive modeling.

Generalize additive modeling is a semi-parametric regression technique that utilizes non-parametric smoothing functions (e.g loess or spine smoothers) when relating predictor and response variables (Hastie and Tibshirani 1990). Thus, generalized additive models (GAMs) can accommodate for non-linear and complex response shapes. In each GAM, the probability of occurrence for a given species is determined as a function of one or more environmental variables. These environmental variables included average growing season water-table depth and range of the growing season water-table depth. Models were first developed using average growing season water-table depth alone as a predictor, and subsequently developed using average and range of the growing season water-table depth. Water-table range was included as a predictor variable when deviance was significantly reduced as judged with a χ^2 statistic at the 5% level.

Prior to GAM development, the data set was transformed and screened. First, species abundance data were converted to presence-absence data. In many cases the number of absent observations greatly outnumbered the number of present observations. A large number of absent data points beyond the range of suitable habitat can negatively influence the shape of the response surface. Therefore each data set was screened to reduce the large number of occurrences of species absence along a particular gradient, such that the data set was limited to all data (presence and absence) within the range of occurrence, in addition to 10 absence observations on each end of the occurrence envelope. In each GAM, a quasibinomial error term and a logit link function were used due to the nature of the presence-absence data set. A third order spline smoothing function was used to relate response and predictor variables. GAMs were developed using GRASP (Generalized Regression Analysis and Spatial Prediction), a suite of tools within R (Lehmann et al. 2002, R-Development Core Team 2004).

Model performance was quantitatively assessed using the area under the curve (AUC) statistic (Fielding and Bell 1997). AUC is a threshold independent metric of a model's goodness-of-fit (Fielding and Bell 1997). AUC values scale from 0.5 (indicating a completely random model) to 1 (prefect agreement of predicted and observed). Generally speaking, a value above 0.9 indicates an outstanding model, 0.8-0.9 excellent, 0.7-0.8 acceptable, and 0.6-0.7 poor (Hosmer and Lemeshow 2000). A five-fold cross-validation technique was employed for model performance assessment. Each species' predictor-response data set was randomly divided into 5 groups, 4 of which were used for model training and the remaining group used for performance evaluation. Individual AUC values for each of the 5 permutations of the partitioned data sets were calculated and averaged to provide the cross-validation AUC.

Water-table depth surfaces were generated by subtracting water-table elevation surfaces (as predicted by the hydrologic model) from the surveyed DEMs. Growing season average, minimum and maximum water-table depth surfaces were generated for both pre- and post-restoration hydrologic-topographic scenarios. These surfaces were sampled on a 2 m grid to provide a raster data set for species occurrence predictions. GAM predictions were analyzed and visualized with ArcMap 9.2 (ESRI Inc.).

RESULTS

Hydrologic Model Calibration and Validation

The hydrological model successfully simulates observed conditions, with Nash-Sutcliffe efficiency coefficients, calculated for the combined calibration and validation period, all greater than 0.90, correlation coefficients all greater than 0.95, and mean error values all less than ±0.05 m (Figure 3.3). The agreement between modeled and observed hydraulic heads was particularly strong during the growing season. The agreement between modeled and observed hydraulic heads was less strong during late fall, prior to the initiation of flow in Bear Creek, and as initial surface flow began to recharge the subsurface. For further details on the hydrologic model, see Hammersmark et al. (In press).

Hydrologic Model Scenario Comparison

Groundwater depths were shallower in the restored scenario (Figure 3.4). Spatially-averaged, growing-season, water-table depths were 0.82 m and 1.86 m for pre- and post-restoration conditions, respectively. Thus, average water-table depths were reduced by 1.04 m due to stream restoration. Differences in water-table depth result from topographic (i.e., channel plugging and pond excavation) and hydrologic (i.e., increased water-table elevation) alterations. Larger differences were observed in the near-channel areas as compared to the distal margins of the meadow. Restoration had the smallest effect in the lower meadow, where inflows from springs maintained relatively stable groundwater levels throughout the year, and the largest effect in the upper and middle meadow, where inflows from the springs were absent and groundwater levels were therefore more related to intermittent stream flows. Restoration increased the range of water-table fluctuations throughout the meadow. Spatially-averaged, growing-season, water-table ranges were 0.97 m and 1.89 m for pre- and post-restoration conditions, respectively. Again, larger differences were observed in the near-channel areas as compared to the distal margins of the meadow. See Hammersmark et el. (In press), for more details on the hydrologic effects of stream restoration in the meadow.

Vegetation Model Development and Evaluation

Each species was strongly related to the average water-table depth (Table 3.1). The explained deviance for models using average water-table depth alone varied widely (19%-46%) and accounted on average for 32% of the total deviance. For all but one species, *Bromus japonicus,* the explained deviance increased significantly when water-table range was included in the model. The explained deviance for models using average and range of water-table depth together varied widely (28%-47%) and accounted on average for 38% of the total deviance. Both the level of significance, and the increase in explained deviance by adding range as a predictor variable were smallest for species at the xeric end of the hydrologic gradient (i.e., *Poa bulbosa, Epilobium brachycarpum*, and

Poa pratensis). Cross-validation AUC values for the final models ranged from 0.78-0.91, with an average value of 0.85, indicating strong model fits.

Changes in Species Distribution

Combined hydrologic and species-prediction model results indicate a change in the distribution of suitable habitat for all species investigated due to hydrologic and topographic modification of the meadow (Figures 3.6, 3.7 and Table 3.2). The average probabilities of occurrence increased for species occurring at the hydric end of the hydrologic gradient, belonging to obligate-wetland and facultative-wetland indicator classes (i.e., *Carex athrostachya, Carex nebrascensis, Eleocharis macrostachya, Epilobium densiflorum* and *Juncus balticus*). *Juncus balticus* had the largest increase in average probability of occurrence, changing from 0.11 to 0.47. The average probabilities of occurrence decreased for species occurring at the xeric end of the hydrologic gradient, belonging to obligate-upland and facultative-upland indicator classes (i.e., *Bromus japonicus, Epilobium brachycarpum, Poa bulbosa and Poa pratensis*). *Poa bulbosa* had the largest decrease in probability of occurrence, dropping from 0.91 to 0.34. Species located in the middle of the hydrologic gradient, assigned to the facultative indicator class, experienced varying results, with *Aster occidentalis* increasing slightly (0.10) and *Leymus triticoides* declining slightly (-0.10).

DISCUSSION

Despite recent advances in the science of stream restoration, considerable uncertainty still exists when attempting to predict the outcome of altering fluvial components of riparian ecosystems (Wohl et al. 2005). The methodology presented in this study provides a practical, quantifiable, and science-based method to predict changes in herbaceous vegetation distribution due to hydrologic alteration, a product of topographic modification of stream channels and adjacent floodplain areas. This approach utilizes standard techniques in hydrologic modeling, vegetation ecology, and statistical modeling, requiring no more than a typical desktop computing system. While the hydrologic and statistical modeling techniques can be data-intensive, the required data are readily obtainable. This method could be used prior to channel modification to screen potential restoration alternatives, when specific vegetation types are required, or once a restoration design is chosen to guide the most successful location of specific species plantings. Current industry standards for vegetative restoration rely upon reference locations to guide vegetation-planting efforts based upon communities found on similar geomorphic surfaces (e.g., stream bank, floodplain, terrace, etc.). In the meadow, this would likely have led to the failure of re-vegetation efforts in many areas, because the depth to groundwater varies along the length of the restored reach, and moving laterally away from the channel, resulting in different species assemblages.

Previous studies have modeled vegetation as a function of surface or groundwater in riparian ecosystems (Franz and Bazzaz 1977, Auble et al. 1994, Toner and Keddy 1997, Springer et al. 1999, Primack 2000, Rains et al. 2004, Leyer 2005, Loheide and Gorelick 2007). A subset of these studies have employed hydrologic or hydraulic models to predict shifts in vegetation due to altered hydrology (Auble et al. 1994, Springer et al. 1999, Rains et al. 2004, Loheide and Gorelick 2007). Some of these studies have utilized water-table depth as the controlling environmental variable (Springer et al. 1999, Rains et al. 2004, Leyer 2005, Loheide and Gorelick 2007), while the others have utilized inundation duration as the controlling environmental variable (Franz and Bazzaz 1977, Auble et al. 1994, Primack 2000). The current study builds upon these past efforts; however, new approaches have been added to both the hydrologic and vegetation modeling components of the study. The hydrologic model used in this study incorporates all relevant aspects of the hydrologic cycle, including channel and floodplain flow, in addition to unsaturated and saturated groundwater flow, allowing for dynamic simulation of the spatially and temporally variable water-table. The species-specific vegetation models were developed with a GAM method, utilizing temporal averages of growing-season, water-table depth in addition to water-table range. The inclusion of water-table range as a predictor variable produced statistical models with stronger fits and improved ability to accurately predict species presence. Indeed, previous research has illustrated the importance of this range gradient in the determination of herbaceous meadow vegetation (Allen-Diaz 1991, Leyer 2005). In addition, studies investigating dampened water level fluctuation due to river regulation have shown that reduced water level ranges result in a greater separation of xeric and hydric vegetation classes, contrasting the continuum of species distribution found along unregulated rivers (Auble et al. 1994, Merritt and Cooper 2000).

This study assumes that the depth to groundwater is the dominant environmental gradient controlling the distribution of herbaceous vegetation in meadow systems. This assumption is typically valid for wetland environments, many of which experience both drought and soil saturation and the consequent anoxia in the root zone (Mitsch and

Gosselink 2000). Indeed, several studies have identified hydrologic variables, typically depth to groundwater, as the primary gradient controlling vegetation distributions in meadow and grassland environments (Allen-Diaz 1991, Castelli et al. 2000, Law et al. 2000, Stringham et al. 2001, Henszey et al. 2004, Dwire et al. 2006, Hammersmark et al. In Review). However, hydrologic conditions may simply be surrogates for soil chemical reactions that influence plant productivity, such as redox reactions limiting root oxygen and nutrient availability (Hobson and Dahlgren 2001). A number of factors beyond the accessibility of shallow groundwater control the distribution of vegetation in riparian environments: competition, disease, seed banks, and herbivory. These factors act in combination with abiotic gradients (e.g., depth to groundwater) to limit species distributions to a realized niche which is a subset of their fundamental niche (Guisan and Zimmermann 2000, Austin 2002). For this reason, vegetation-distribution models developed from field data are generally limited to the area where the training data were collected. In addition, abiotic controls such as soil texture and degree of compaction, flooding, nutrient availability and fire, may further influence vegetation distributions.

Static distribution models, such as the models developed in this study, assume equilibrium or at least pseudo-equilibrium. While the woody species present in the meadow have surely not reached an equilibrium state with the altered hydrology, herbaceous species likely have. Hammersmark (In Review) investigated the water-table – vegetation relationships of this restored meadow, and found that vegetation communities in this restored meadow occur at similar locations along the hydrologic gradient as vegetation communities in other meadows that were considered to be in equilibrium. However, it is possible that herbaceous species are still approaching equilibrium with the altered hydrology. One alternative to the static distribution approach taken is a state and transition modeling approach, which assigns transitional probabilities between any number of states that reflect plant successional and disturbance pathways. Such methods require substantial parameterization which in turn requires intensive knowledge of the species involved, and thus have more limited application to spatially explicit prediction (Guisan and Zimmermann 2000).

The general results of this study are largely predictable without the use of sophisticated hydrologic and statistical models. One would expect that raising water-tables would lead to an increase in vegetation adapted to living in mesic and hydric environments, and a decrease in the prevalence of upland species. However, the degree of these changes would remain uncertain, as these changes are dependent upon the degree of hydrologic and topographic modification, which are temporally and spatially variable. As expected, the linked hydrologic-vegetation models predict a quantifiable increase in obligate wetland and facultative wetland species (i.e., *Carex athrostachya, Carex nebrascensis, Eleocharis macrostachya, Epilobium densiflorum,* and *Juncus balticus*), and a quantifiable decrease in facultative upland and obligate upland species (i.e., *Bromus japonicus, Epilobium brachycarpum, Poa bulbosa,* and *Poa pratensis*). Furthermore, the approach presented provides a spatially explicit and quantifiable method that allows for improved objective setting, restoration design screening, and active re-vegetation in similar projects in similar environments.

Both water-table depth and species-prediction maps suggest the importance of micro-topography to the development of a riparian vegetation mosaic in floodplain environments (Figures 3.4, 3.6 and 3.7). The 2 m grid utilized in this study captures many relict and alternate stream channels and depressions, which, due to their lower ground-surface elevations, provide access to shallower groundwater. This access to shallower groundwater makes these environments more conducive to hydric and mesic species, and less conducive to more xeric upland species. If the spatial scale of prediction were increased, then the influence of these areas would likely not be seen.

Lastly, the results of this study highlight the potential impact of hydrologic and subsequent vegetation changes due to stream restoration on geomorphic processes, specifically bank erosion and channel widening. Common goals of similar restoration efforts include decreased streambank erosion and downstream sediment delivery (Benoit and Wilcox 1997, Rosgen 1997). Indeed, this objective was the primary motivation for the restoration of this reach of Bear Creek (Poore 2003). While reconnecting stream channels to the adjacent floodplains is intended, among other things, to dissipate energy and encourage floodplain sedimentation, the subsequent raised water-table, and consequent shifts in vegetation likely play a role in bank stability and erosion. Obligate wetland and facultative wetland vegetation communities have higher root density and mass as compared to upland community types (Manning et al. 1989), and the compressive strength of stream banks increases with root density (Kleinfelder et al. 1992). Vegetation communities dominated by *Carex nebrascensis* and *Juncus balticus* have lower erosion rates than communities dominated by *Poa pratensis* (Dunaway et al.

1994). Likewise banks lined with wet meadow plant communities have less susceptibility to bank erosion than banks with xeric scrub and grasses (Micheli and Kirchner 2002). The predicted increases in *Juncus* and *Carex* species likely translate to increased bank stability and decreased downstream sediment delivery in the restored Bear Creek Meadow.

CONCLUSION

Hydrology is the primary driver of the establishment and persistence of wetlands (Mitsch and Gosselink 2000). Natural flow regimes (Poff et al. 1997) and multidimensional connectivity (Ward and Stanford 1995, Stanford et al. 1996) have been identified as key determinants in the ecology of river-riparian systems. Moreover, hydrology is so crucial that a National Research Council report on the management of riparian areas states that "repairing the hydrology of the system is the most important element of riparian restoration" (National Research Council 2002). The restoration of the meadow channel studied here resulted in the restoration of shallow groundwater levels and consequent changes to herbaceous meadow vegetation. Specifically the likelihood of occurrence of obligate wetland and facultative wetland species increased while the likelihood of obligate upland and facultative upland species decreased. While this work focuses on the hydro-ecological effects of a particular "pond and plug" restoration project, the results and methodology could be utilized toward improved goal setting and restoration design in similar degraded environments. The methods utilized provide a practical tool for the assessment of designs in the planning phase of restoration efforts. Considerable complexity and uncertainty exist in the emerging multidisciplinary science of river restoration (Wohl et al. 2005). This approach to predicting vegetation changes in meadow environments provides an improved understanding of the magnitude of change and the causes of those changes, supplying a learning tool to improve the science of river restoration. While the coupled modeling framework presented herein was applied to assess the ecological effects of “pond and plug” stream restoration, the approach could also be employed to predict the outcome of reservoir operations, water extraction (e.g. surface and/or groundwater) or channel modification on the distribution of riparian vegetation.

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Table 3.1. Summary of wetland indicator category, regression model analysis and cross-validation AUC results for herbaceous species studied.

Species	WIC[1]	n	Total deviance	Explained deviance[3] Average (%)	Explained deviance[3] Average & Range (%)	AUC[4]
Aster occidentalis (Nutt.) Torrey & A. Gray	F	30	158.4	24.2**	28.1*	0.81
Bromus japonicus Murr	FU	107	224.2	46.1**	NS	0.86
Carex athrostachya Olney	FW	48	202.4	31.7**	46.6**	0.90
Carex nebrascensis Dewey	OW	34	170.1	33.1**	41.3**	0.86
Eleocharis macrostachya Britton	OW	30	131.0	31.4**	43.4**	0.88
Epilobium brachycarpum C. Presl	OU	76	233.8	37.4**	39.5*	0.86
Epilobium densiflorum (Lindley) P. Hoch & Raven	OW	42	190.1	26.9**	34.0**	0.83
Juncus balticus Willd.	FW	98	231.7	19.4**	25.0**	0.78
Leymus triticoides (Buckley) Pilger	F	39	183.1	19.1**	32.0**	0.85
Poa bulbosa L.	OU[2]	50	206.0	45.7**	49.4*	0.91
Poa pratensis L. ssp. *pratensis*	FU	100	230.3	34.0**	36.7*	0.82

1 – Wetland indicator category designation (U.S. Fish and Wildlife Service 1996). OW - obligate wetland, FW - facultative wetland, F - facultative, FU - facultative upland, OU - obligate upland.
2 – *Poa bulbosa* L. is not assigned to a wetland indicator category and is assumed to be an obligate upland species in this study.
3 – **$p<0.0001$, *$p<0.05$.
4 – Average AUC of five training-evaluation data-set combinations.

Table 3.2. Comparison of meadow-averaged probability of species presence for pre- and post-restoration scenarios. Average probability of occurrence increased for species assigned to obligate wetland and facultative wetland categories, and decreased for species assigned to the facultative upland and obligate upland categories.

		Probability of presence		
Species	WIC[1]	Pre-restoration	Post-restoration	Change
Aster occidentalis (Nutt.) Torrey & A. Gray	F	0.076	0.170	0.095
Bromus japonicus Murr	FU	0.835	0.697	-0.138
Carex athrostachya Olney	FW	0.002	0.274	0.272
Carex nebrascensis Dewey	OW	0.001	0.150	0.149
Eleocharis macrostachya Britton	OW	0.008	0.157	0.149
Epilobium brachycarpum C. Presl	OU	0.770	0.347	-0.423
Epilobium densiflorum (Lindley) P. Hoch & Raven	OW	0.004	0.279	0.275
Juncus balticus Willd.	FW	0.111	0.465	0.354
Leymus triticoides (Buckley) Pilger	F	0.204	0.100	-0.104
Poa bulbosa L.	OU[2]	0.913	0.335	-0.578
Poa pratensis L. ssp. *pratensis*	FU	0.878	0.643	-0.235

1 – Wetland indicator category designation (U.S. Fish and Wildlife Service 1996). OW - obligate wetland, FW - facultative wetland, F - facultative, FU - facultative upland, OU - obligate upland.
2 – *Poa bulbosa* L. is not assigned to a wetland indicator category and is assumed to be an obligate upland species in this study.

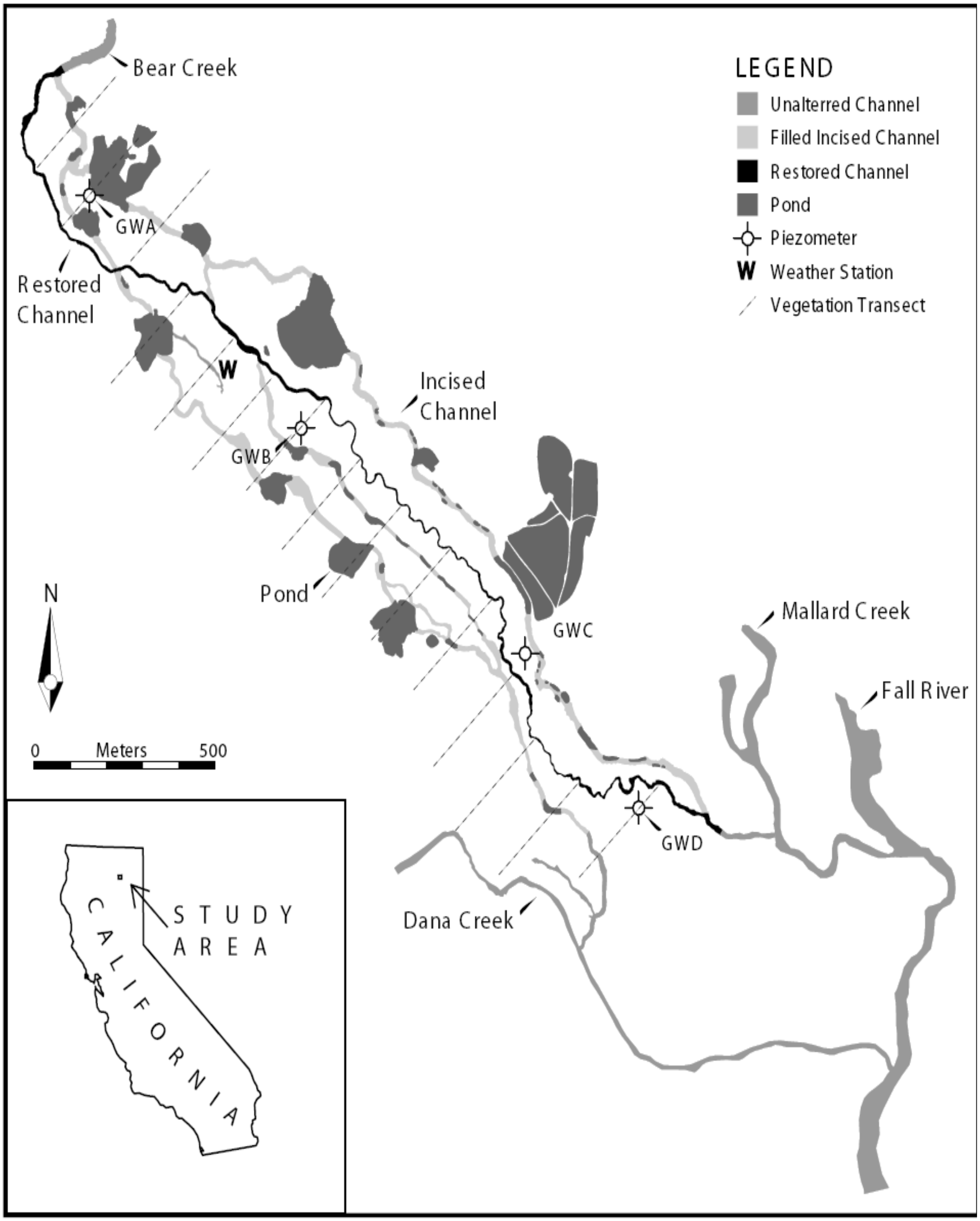

Figure 3.1. Bear Creek Meadow study area. Portions of the incised channels were filled with alluvium excavated from ponds throughout the meadow. A 3.6 km single thread restored channel reach was created from remnant channel segments and excavated where necessary. Flow direction is from upper left to lower right.

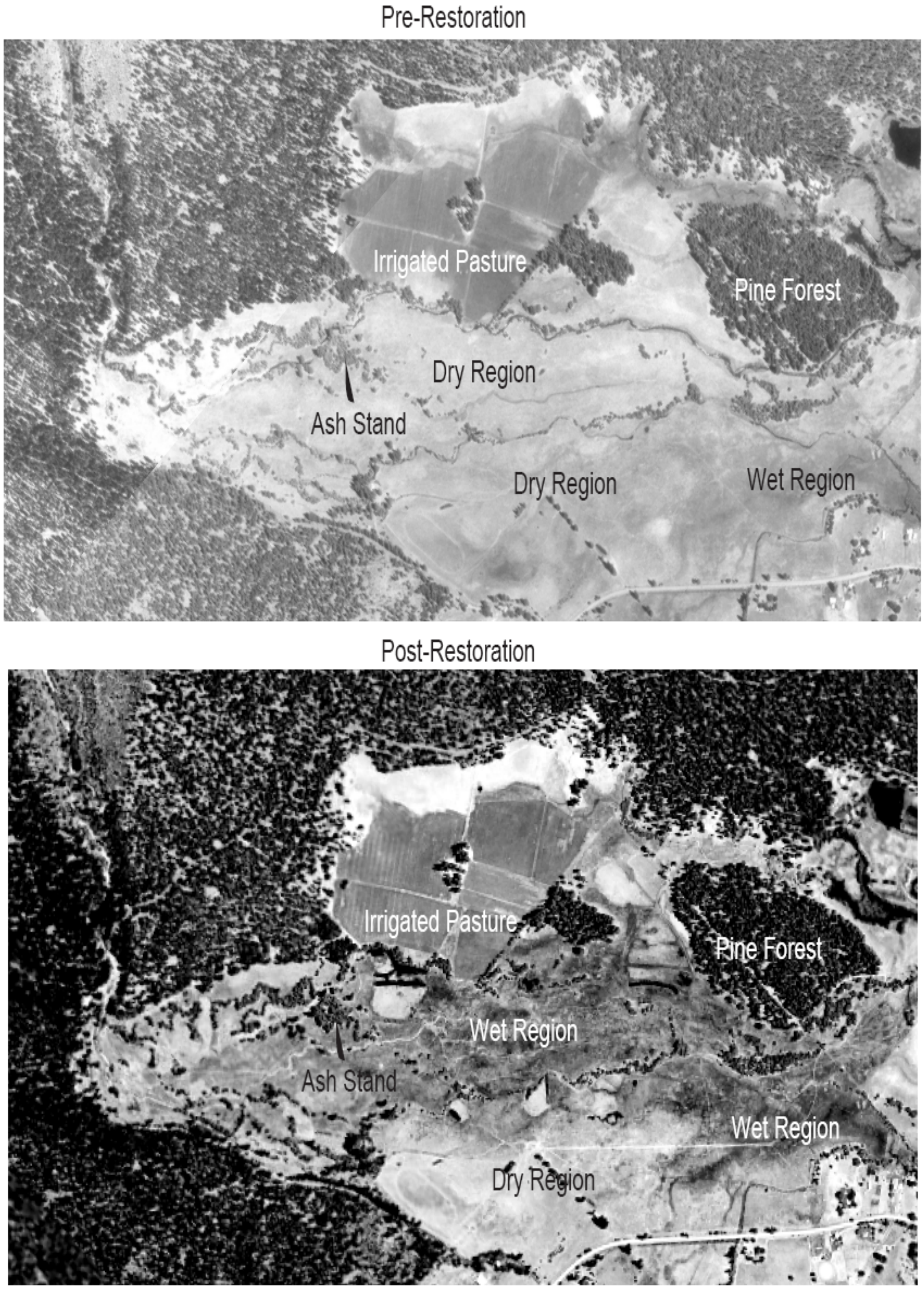

Figure 3.2. Pre- and post-restoration aerial photographs of the meadow. Qualitative comparisons indicate an increase in mesic and hydric vegetation in the post-restoration photograph. The region immediately below the irrigated pasture and the pine forest experienced the largest degree of hydrologic alteration, and subsequent herbaceous vegetation change. Wet region labels indicate the area occupied by mesic-hydric vegetation communities.

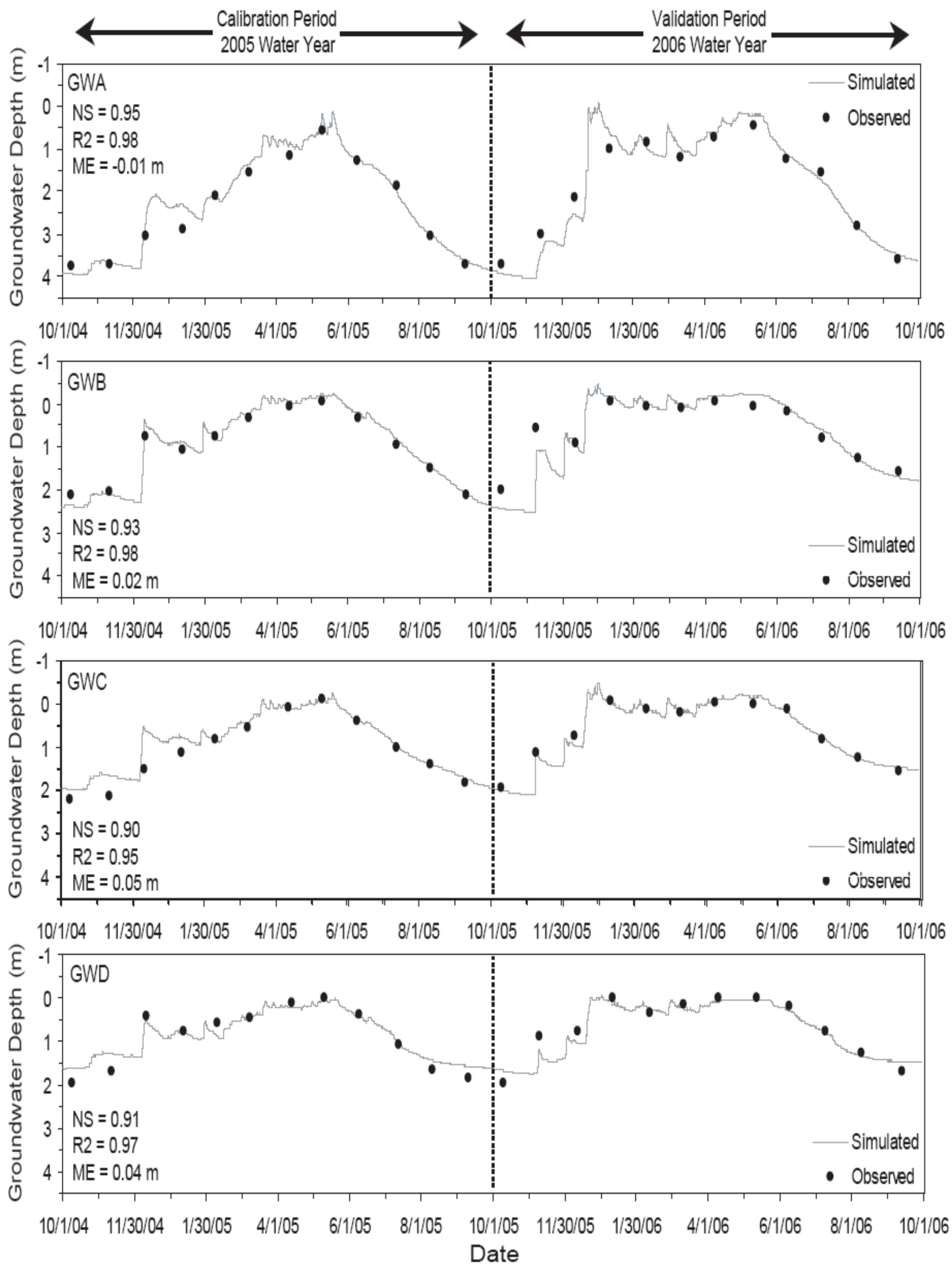

Figure 3.3. Comparisons of simulated and observed groundwater depths at four piezometers within the meadow. The 2005 water year (left side) was used for model calibration and the 2006 water year (right side) was used for model validation. Nash-Sutcliffe efficiency coefficients (NS), correlation coefficients (R2) and mean error values (ME) are provided for each location. Negative groundwater depths indicate surface inundation that is common in the restored meadow. Piezometer locations are shown on Figure 3.1.

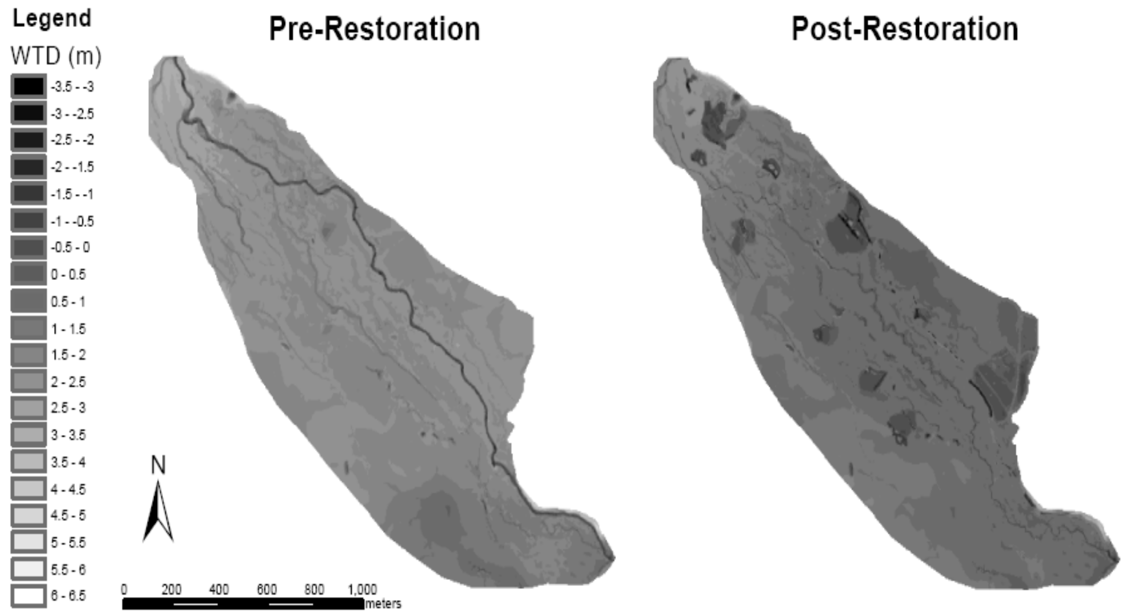

Figure 3.4. Comparison of growing season average water-table depth (WTD) for the pre- and post-restoration hydrologic-topographic scenarios. Spatial water-table depth averages are 1.86 m and 0.82 m for the pre- and post-restoration scenarios, respectively. Differences in water-table depth result from topographic (i.e., channel plugging and pond excavation) and hydrologic (i.e., increased water-table elevation) alterations. In the pre-restoration case, shallow groundwater is limited to the bottom of the incised channels, whereas in the post-restoration case, shallow groundwater occurred throughout much of the study area, with negative values (indicating the ground surface is inundated) occurring in most of the pond areas.

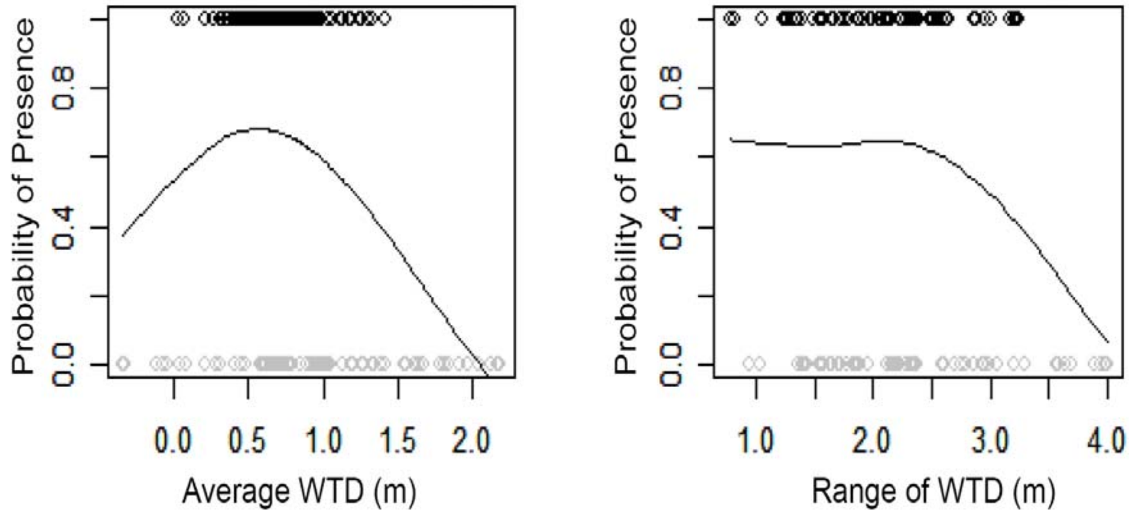

Figure 3.5. Predictor vs. response curves for *Juncus balticus*, a commonly occurring facultative wetland species. The y-axis represents the probability of presence for *Juncus balticus*, while the x-axis represents the individual predictor variables. Optimum of probability of presence occurs at ~ 0.55 m, while probability of presence decreases for water-table depth ranges of > 2.3 m. Black circles along top of plots indicate species presence and gray circles along the bottom of plots indicate species absence.

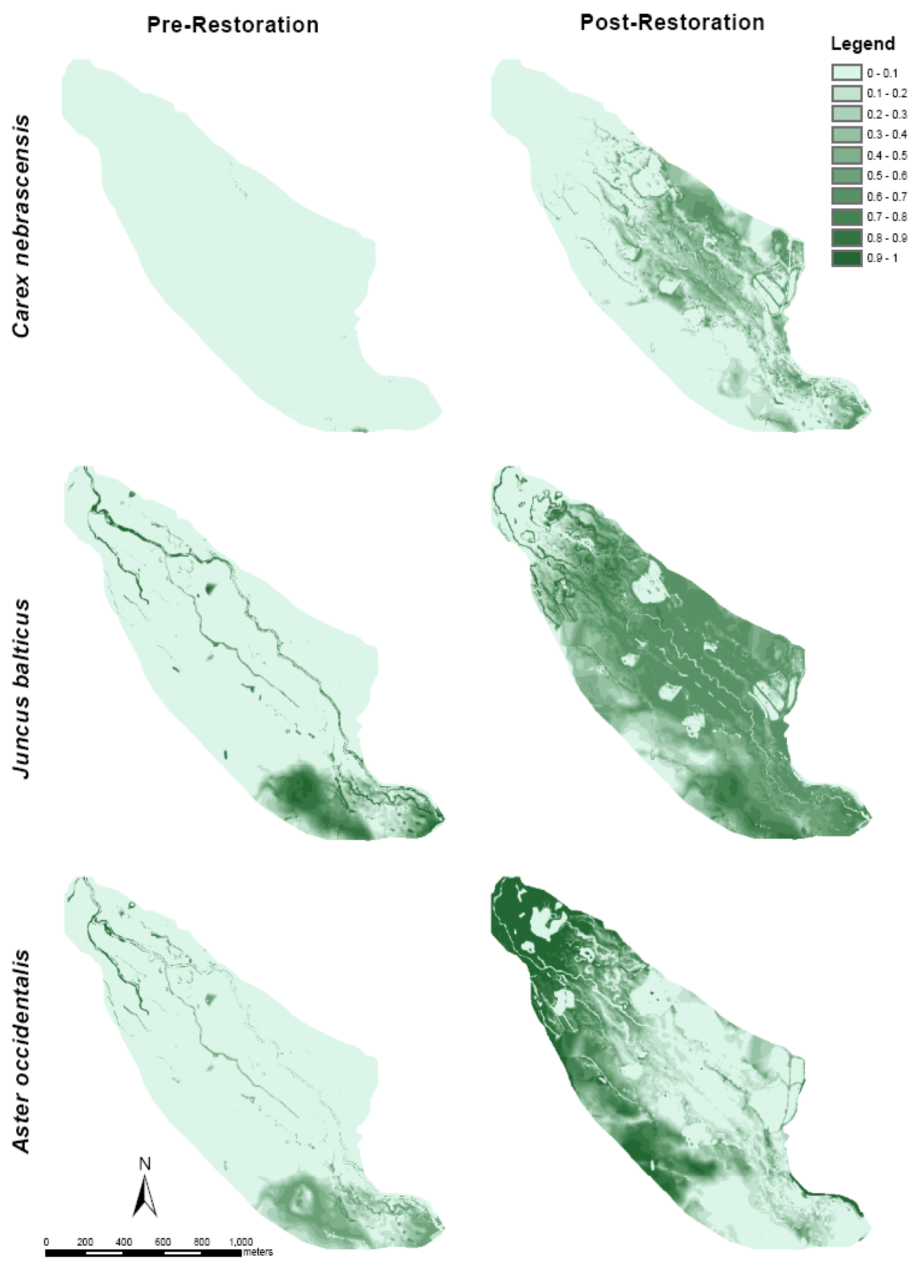

Figure 3.6. Comparison of pre-restoration and post-restoration probability of presence for three species on the hydric-mesic end of the hydrologic gradient. *Carex nebrascensis, Juncus balticus,* and *Aster occidentalis* belong in the obligate wetland, facultative wetland, and facultative wetland indicator categories, respectively. The meadow average probability of presence for each of these species increased due to stream restoration.

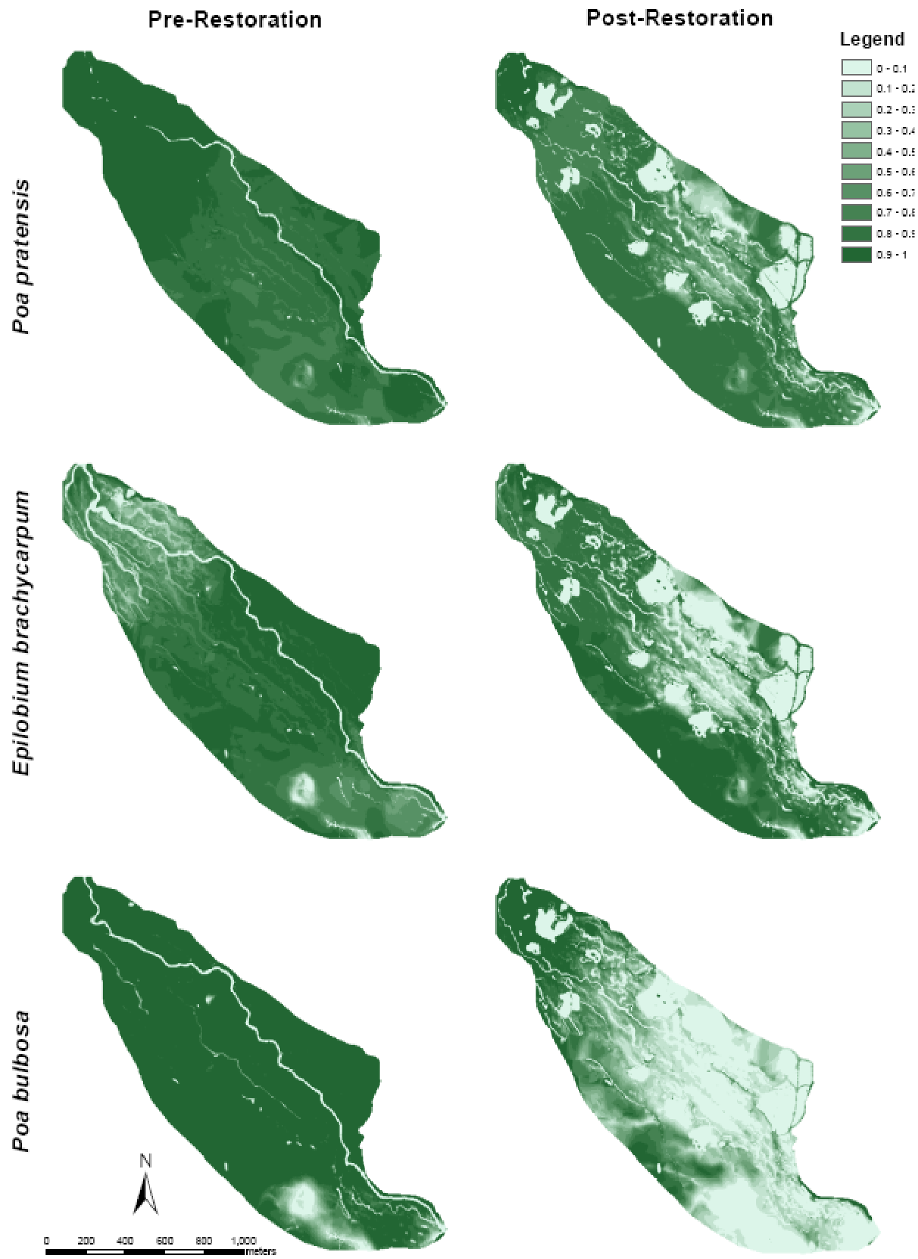

Figure 3.7. Comparison of pre-restoration and post-restoration probability of presence for three species on the mesic-xeric end of the hydrologic gradient. *Poa pratensis*, *Epilobium brachycarpum,* and *Poa bulbosa* belong in the facultative upland, obligate upland and unassigned (assumed to be obligated upland) wetland indicator categories, respectively. The meadow average probability of presence for each of these species decreased due to stream restoration.

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