Ecohydrologic Effects of Stream Restoration

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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. Despite the large number of "pond and plug" restoration projects undertaken in the western United States, little research has been conducted to evaluate and quantify the effects of such topographic modification upon hydrology and riparian vegetation in these systems. To predict the changes in hydrologic processes and 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 hydrologic model of a 230 ha mountain meadow along a 3.6 km restored reach of Bear Creek in northeastern California, and used it to simulate the pre- and post-restoration topographic conditions. Next, vegetation 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 predicted as a function of growing season water table depth and range. Last, hydrologic and vegetation models were jointly used to predict the spatial distribution of individual plant species for pre- and post-restoration conditions. Our results document three general hydrologic 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. Vegetation modeling 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. This study supports and quantifies the hypothesis that "pond and plug" type stream restoration projects have the capacity to re-establish hydrologic processes necessary to sustain riparian systems. The methods utilized 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 AND PROBLEM STATEMENT

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 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 stream bank 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.

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 hydrologic sciences (Malakoff 2004) and placed river restoration at the forefront of applied hydrologic sciences (Wohl et al. 2005). Considerable complexity and uncertainty exist in the emerging multidisciplinary science of river restoration (Wohl et al. 2005). 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). Furthermore, 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. Regrettably, few studies documenting the hydrologic and concomitant ecological effects of stream restoration exist.

OBJECTIVES

The purpose of this study is to quantify the ecohydrologic effects of a "pond and plug" type stream restoration. We hypothesize that topographic modification of stream channels and adjacent floodplains, typical of "pond and plug" restoration projects, will result in measurable changes to all surface and subsurface hydrologic processes, and these changes would initiate changes in the distribution of riparian vegetation. To test these hypotheses, 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:

- Develop, calibrate, and validate a hydrologic model of a well-documented "pond and plug" restoration project, and subsequently employ this model to assess the affect of channel and floodplain topographic modification on key hydrologic processes.
- Develop a suite of vegetation models linking water table depth and range to the probability of occurrence of common herbaceous meadow species.
- Use these hydrologic and vegetation models in concert to predict the changes in distribution of herbaceous species due to hydrologic alteration caused by stream restoration.

PROCEDURE

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). 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 m to 2 m of deltaic sands and gravels, and 1 m to 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 hydrologic 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). 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). 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 $m^3 s^{-1}$ to 20.73 $m^3 s^{-1}$ (Figure 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 $m^3 s^{-1}$, 29.6 $m^3 s^{-1}$ and 48.2 $m^3 s^{-1}$, 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"

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 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 (Figures 3 and 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). In addition, 17 ha of new ponds (remnant gully segments and fill sources) exist throughout the meadow. Based upon qualitative observations, these topographic modifications resulted in changes to the distribution of vegetation in the meadow (Figure 5).

Hydrologic Model Development

A numerical hydrologic 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 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} m s^{-1}, with values ranging from 6.3×10^{-6} m s^{-1} to 1.5×10^{-8} m s^{-1}. The arithmetic mean for five slug tests performed in the sand and gravel layer was 4.5×10^{-2} m s^{-1}, with values ranging from 1.5×10^{-2} m s^{-1} to 9.0×10^{-2} m s^{-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} m s^{-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 6).

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 (dominated by *Fraxinus latifolia* and *Crataegus douglasii*), pine forest (dominated by *Pinus jeffreyi*), and grassland (dominant dominated by *Poa pratensis*, *Bromus japonicus*, and *Juncus balticus*, Figure 7). 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). 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 $1.0x10^{-5}$ s^{-1} adopted from the literature (Thompson et al. 2004). Manning's n for channel flow was estimated to be 0.033 s $m^{-1/3}$ 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 s $m^{-1/3}$ was chosen from the literature (Thompson et al. 2004). Each of these values was subsequently altered during model calibration.

Boundary Conditions. 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 8). 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 $2x10^{-2}$ m^3 s^{-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 (Figures 1 and 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 6). 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 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. 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.

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.

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.

Vegetation Model Development

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 (Figure 1). 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 and 400 m distances from the stream edge, as allowed by the width of the meadow. 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.

Model Development. 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 five 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, as simulated with the hydrologic model. 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. 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 Evaluation. 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.

Predictor surfaces. Water table depth surfaces were generated by subtracting water table elevation results (from 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 two-meter 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 hydrologic model of the restored scenario successfully simulates observed conditions (Figures 9 and 10). 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). 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.

Hydrologic Model Application – Incised and Restored Scenario Comparison

Groundwater. Groundwater levels were higher in the restored scenario (Figures 11-13). Restoration had the smallest hydrologic 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. Spatially averaged growing season average water table depths were 0.82 m and 1.86 m for pre- and post-restoration conditions, respectively (Figure 13). 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. 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. 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 14). 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 15). The average channel capacity was 61.7 $m^3\ s^{-1}$ and 5.35 $m^3\ s^{-1}$ 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 $m^3\ s^{-1}$ and 9.7 $m^3\ s^{-1}$. 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 $m^3\ s^{-1}$, 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 14). 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 16). Due to this floodplain storage, flood peak discharges were attenuated in the restored scenario (Figure 17). 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 $m^3\ s^{-1}$ 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 $m^3\ s^{-1}$, 17.25 $m^3\ s^{-1}$ and 20.67 $m^3\ s^{-1}$,

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 15). 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 18). 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 mm d^{-1} occurred on 5/22/05 in the incised scenario, while the peak daily ET rate of 7.0 mm d^{-1} occurred 41 days later on 7/2/05 in the restored scenario. During the 2006 water year, the peak daily ET rate of 5.5 mm d^{-1} occurred on 5/2/06 in the incised scenario, while the peak daily ET rate of 6.9 mm d^{-1} occurred 56 days later on 6/27/06 in the restored scenario. The maximum difference of 3.6 mm d^{-1} occurred on 7/11/06. 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.

Vegetation Model Development and Evaluation

Each species was strongly related to the average water table depth (Table 3). 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 to 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 20, 21 and Table 4). 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 AND CONCLUSION

Hydrologic Effects

This analysis of the Bear Creek Meadow restoration project indicates that plugging the incised channels and construction of a shallow, sinuous, single-thread channel initiated at least three significant hydrologic responses that are likely to have important ecological effects (Table 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.

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 hydrologic 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.

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 hydrologic 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 $m^3 s^{-1}$ vs. 5.35 $m^3 s^{-1}$). 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 $m^3 s^{-1}$) 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)

There is a general perception that stream restoration will improve all hydrologic 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 evapotranspiration 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.

Changes in Vegetation Distribution

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 average growing season water table depth in addition to growing season 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, which 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 13, 20 and 21). 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.

While this work focuses on the hydrologic 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, predicting, 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 ecohydrologic 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. 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 9. Surface locations compare simulated and observed water surface elevations as shown in Figure 10.

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 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

Table 3. 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 (US 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 4. 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.

Species	WIC[1]	Probability of presence Pre-restoration	Probability of presence Post-restoration	Probability of presence 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 (US 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.

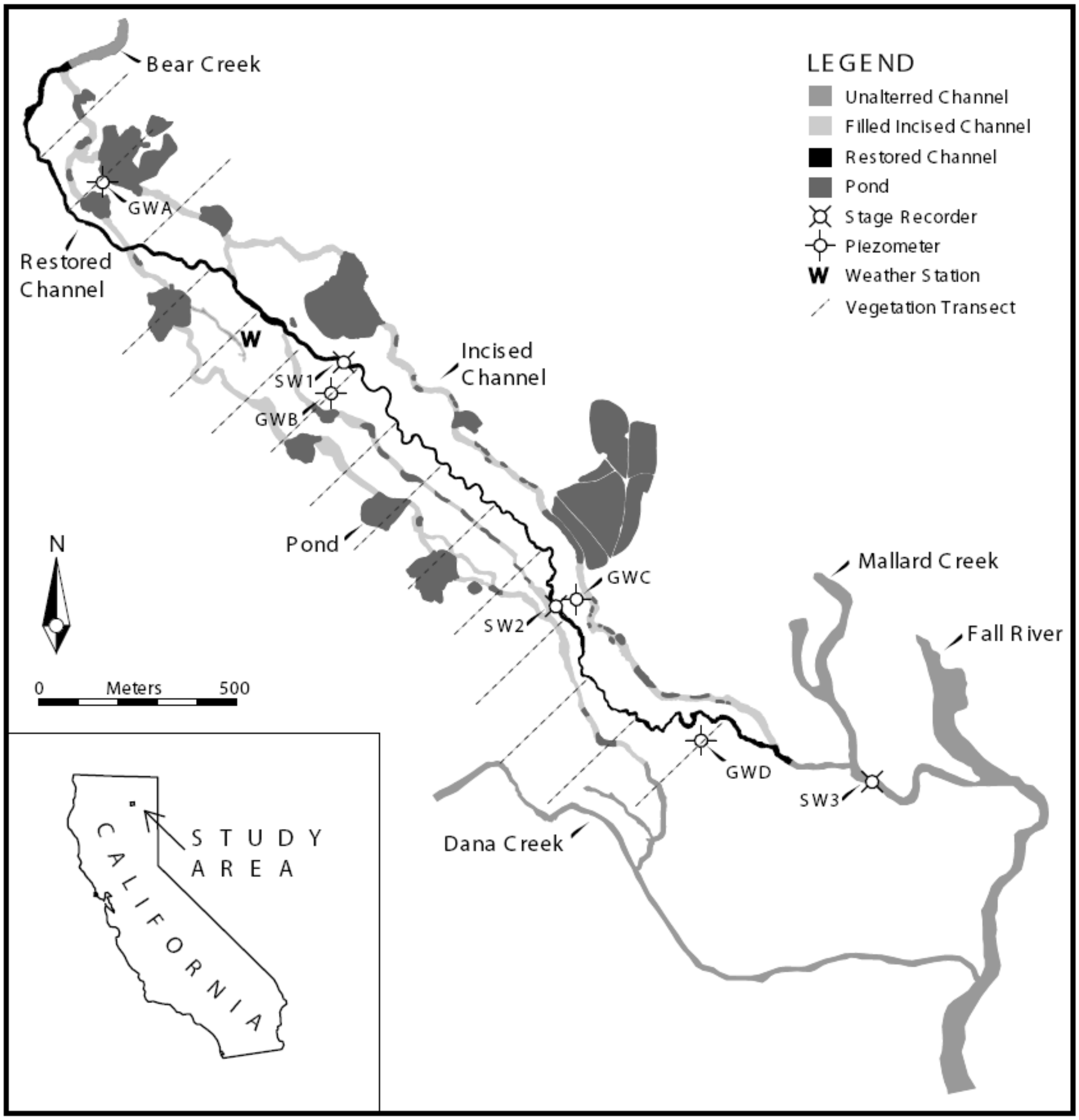


Figure 1. Bear Creek Meadow study area. Portions of the incised channels were filled with alluvium excavated from ponds throughout the meadow. A 3.6 km single thread restored channel reach was created from remnant channel segments and excavated where necessary. Flow direction is from upper left to lower right. Surface and groundwater comparison and vegetation transect locations are also shown.

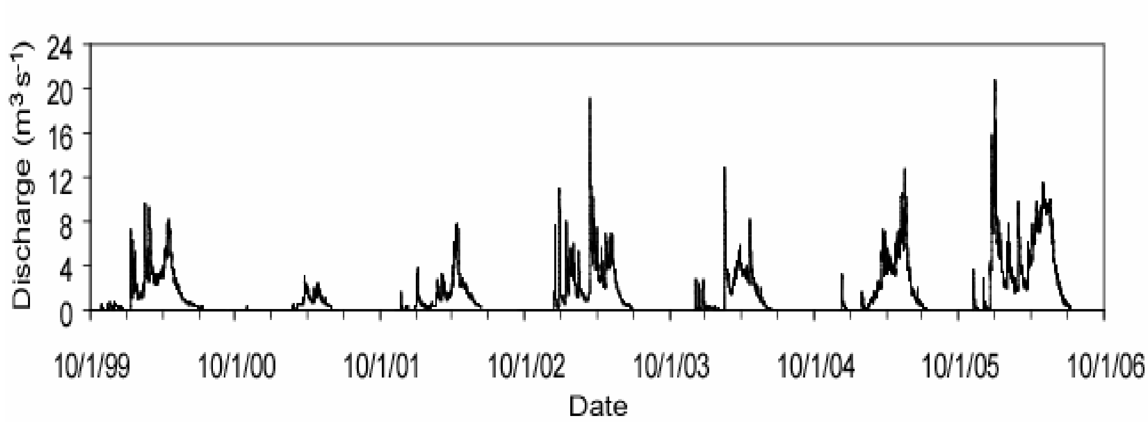
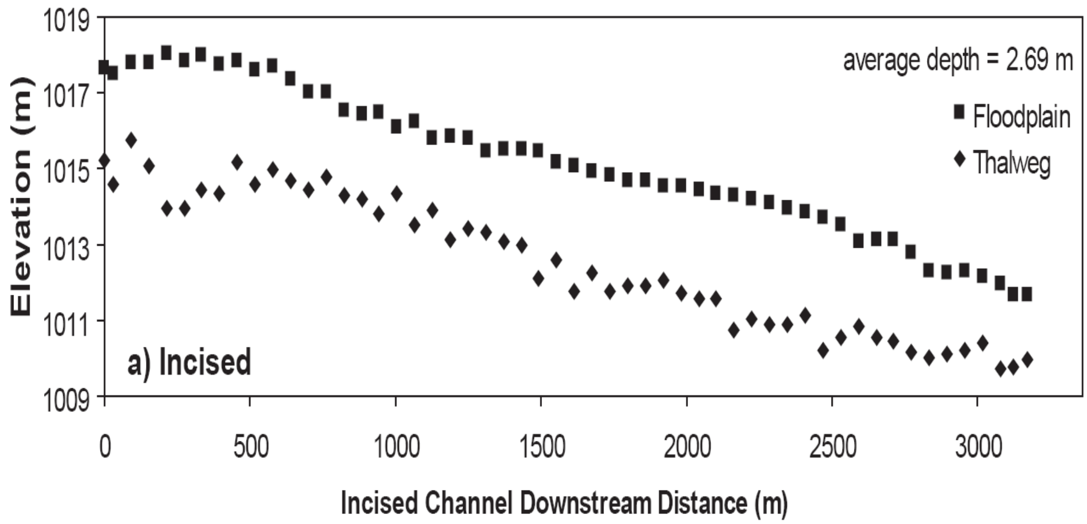


Figure 2. Bear Creek discharge at the upstream extent of the restored reach for the water years of 2000 to 2006. Annual peak discharge ranged from 3.11 $m^3 s^{-1}$ to 20.73 $m^3 s^{-1}$. Stream discharge is intermittent, with flood peaks resulting from rainfall, rain on snow, and spring snowmelt.



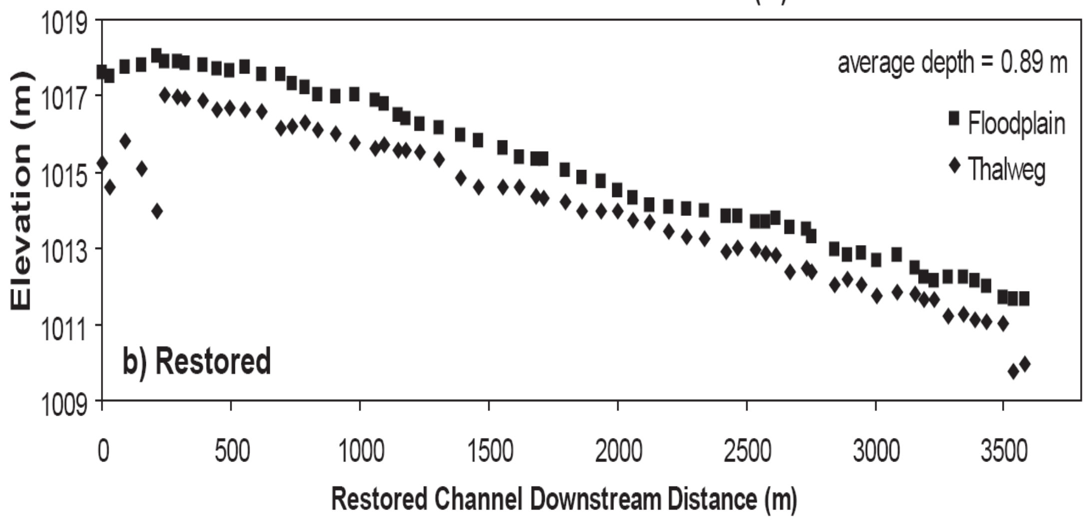


Figure 3. Long profiles of riffle crest thalweg and adjacent floodplain elevations for (a) incised and (b) restored channel geometries. The restored reach begins at restored channel station 800 m and ends at restored channel station 3535 m corresponding to incised channel station 800 m and 3124 m, respectively. The first five and last two points in each of the surveys represent identical locations.

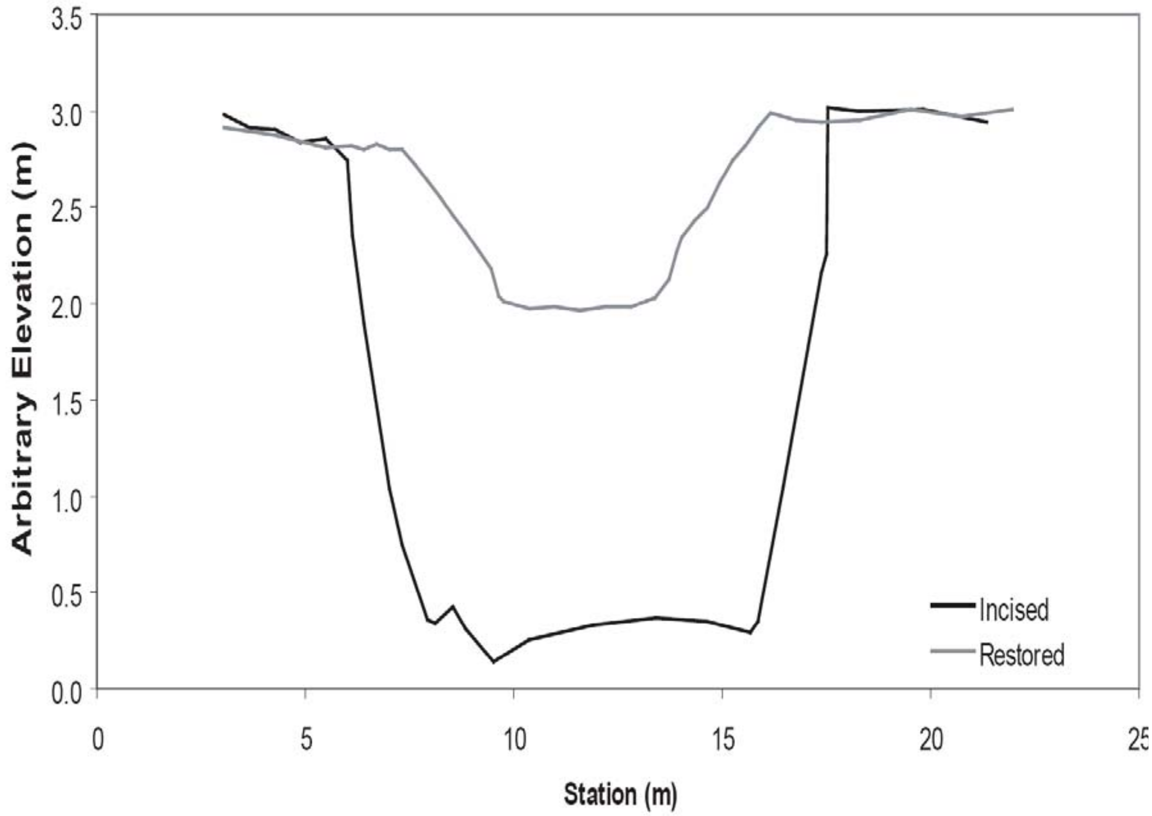


Figure 4. Representative restored and incised cross sections of the Bear Creek channel.

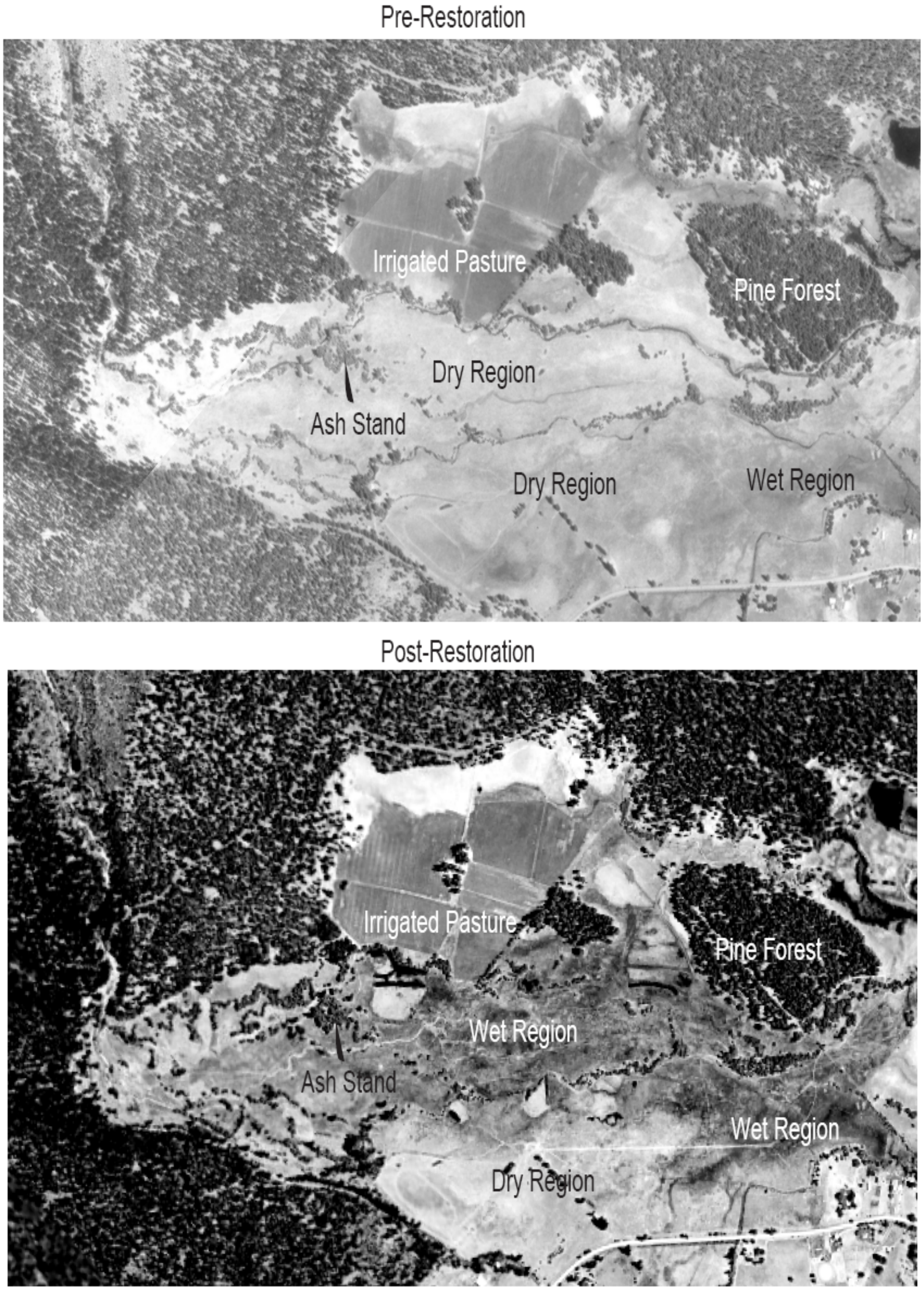


Figure 5. Pre- and post-restoration aerial photographs of the meadow. Qualitative comparisons indicate an increase in mesic and hydric vegetation in the post-restoration photograph. The region immediately below the irrigated pasture and the pine forest experienced the largest degree of hydrologic alteration, and subsequent herbaceous vegetation change. Wet region labels indicate the area occupied by mesic-hydric vegetation communities.

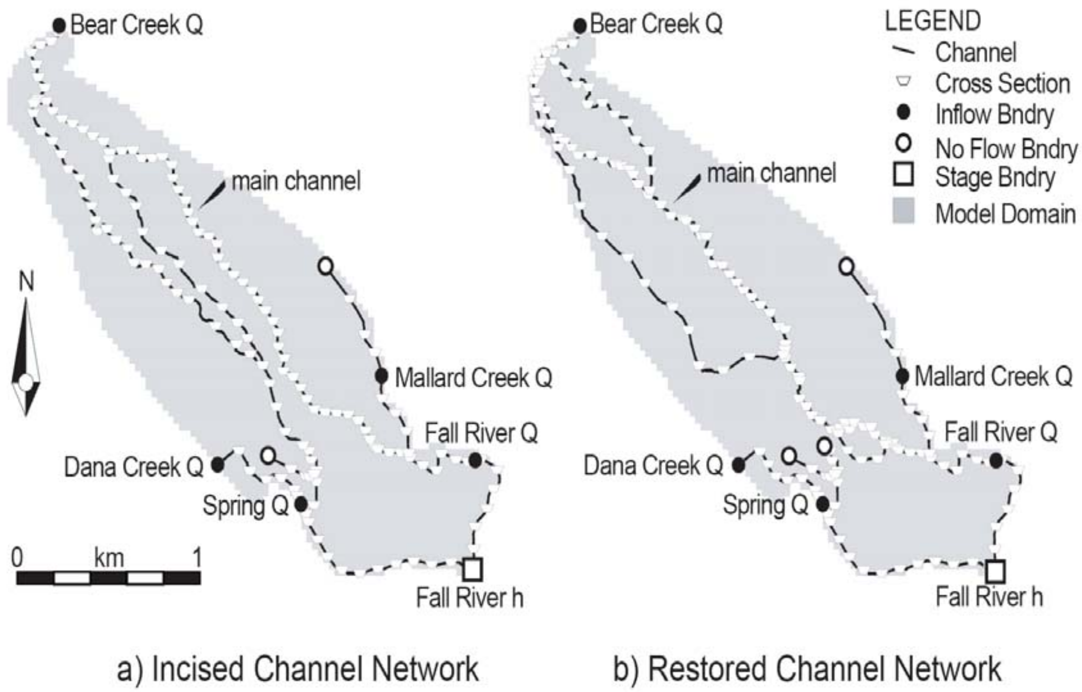


Figure 6. Channel alignment, cross section locations, and surface water boundary condition type and locations for the (a) incised and (b) restored channel flow components of the two models.

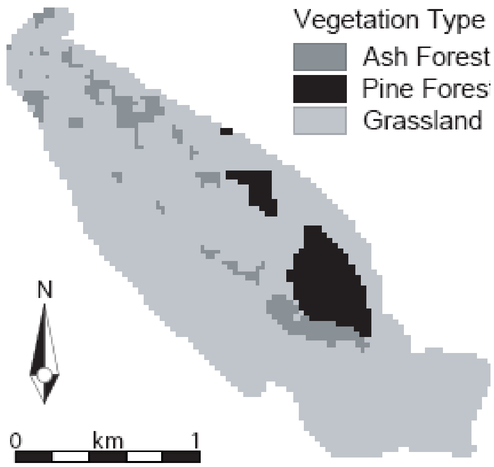


Figure 7. Spatial distribution of the three vegetation types employed in the hydrologic model.

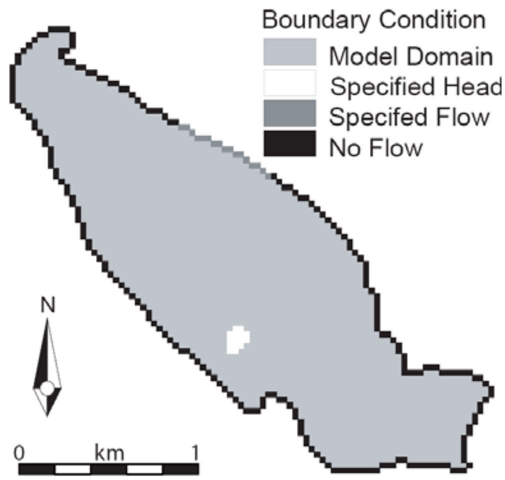


Figure 8. Domain and subsurface boundary conditions for the hydrologic model. Subsurface boundary types include no flow, specified flow and specified head.

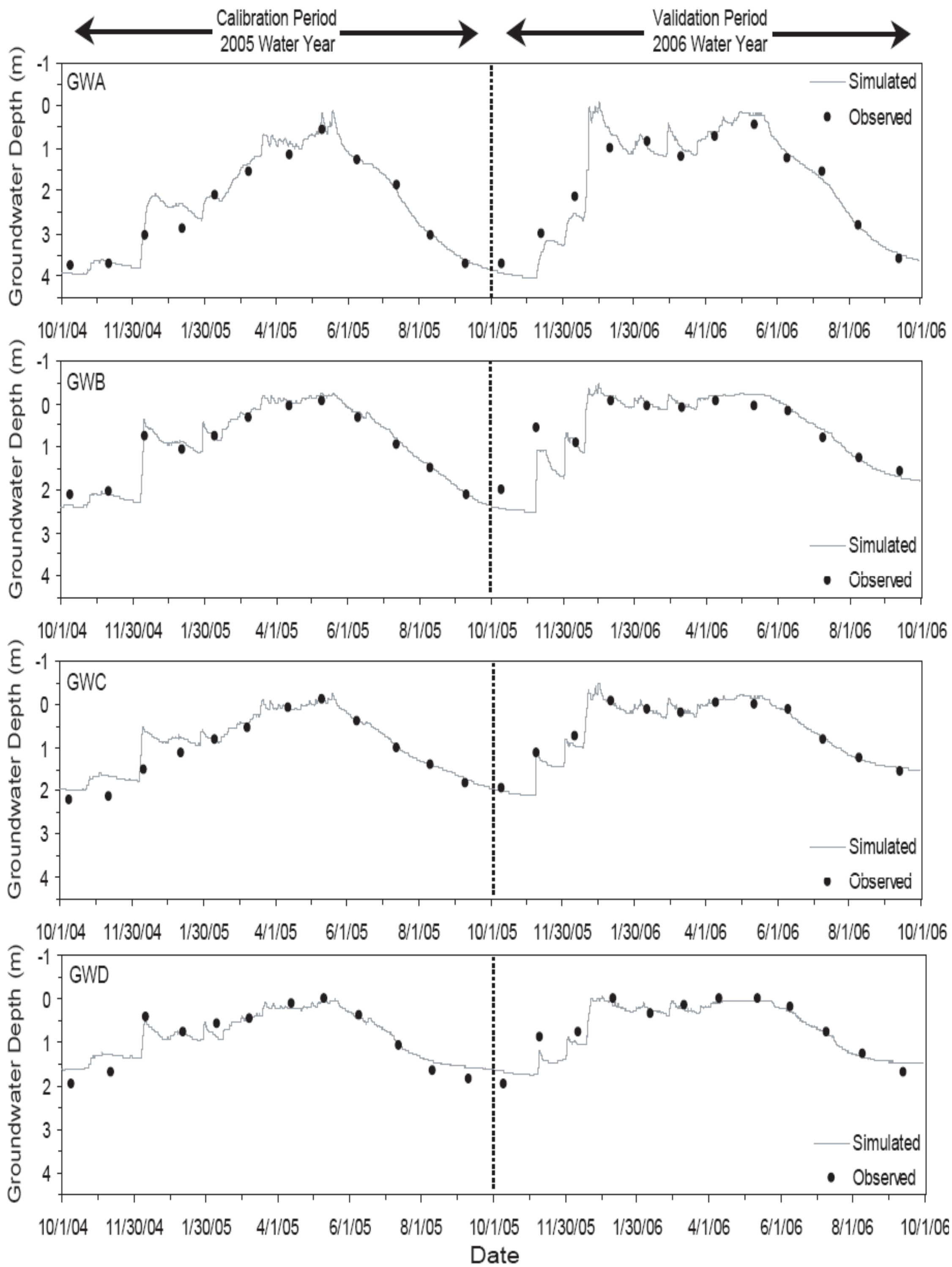


Figure 9. Comparison of simulated and observed groundwater depth at four piezometer locations within the meadow. The 2005 water year (left side) was used for model calibration and the 2006 water year (right side) was used for model validation. Negative groundwater depths indicate surface inundation that is common in the restored meadow. Piezometer locations are shown on Figure 1.

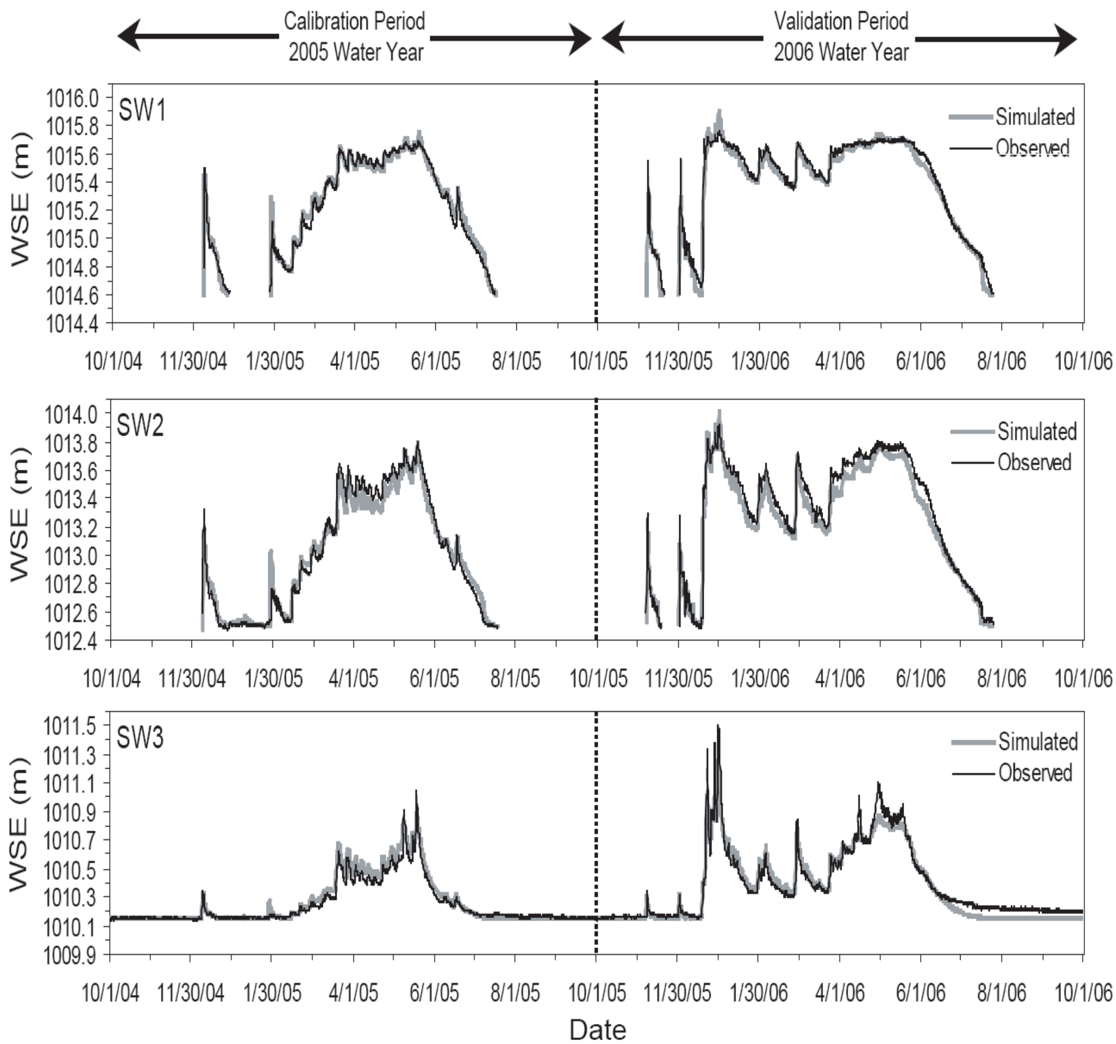


Figure 10. Comparison of simulated and observed water surface elevations (WSE) at three locations along Bear Creek. The 2005 water year (left side) was used for model calibration and the 2006 water year (right side) was used for model validation. At the upper two locations (SW1 and SW2) Bear Creek is 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.

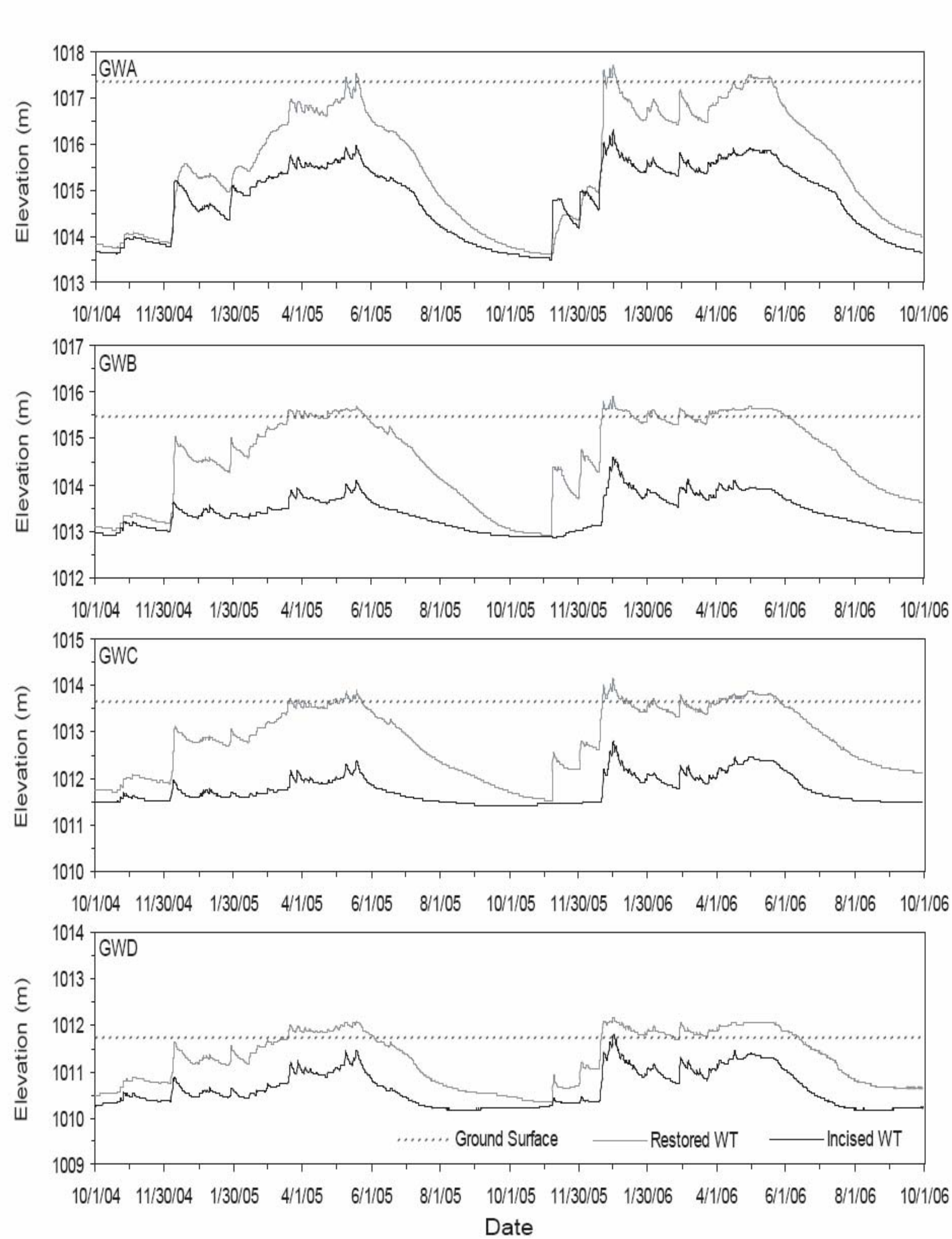


Figure 11. Comparison of water table elevations for the restored and incised scenarios at four locations within the meadow. The largest water table elevation differences are seen in the winter and spring, corresponding to surface flow in Bear Creek. In the restored condition, the elevation of the water table is above the ground surface for extended periods at each location. Comparison locations coincide with the locations of piezometers shown on Figure 1.

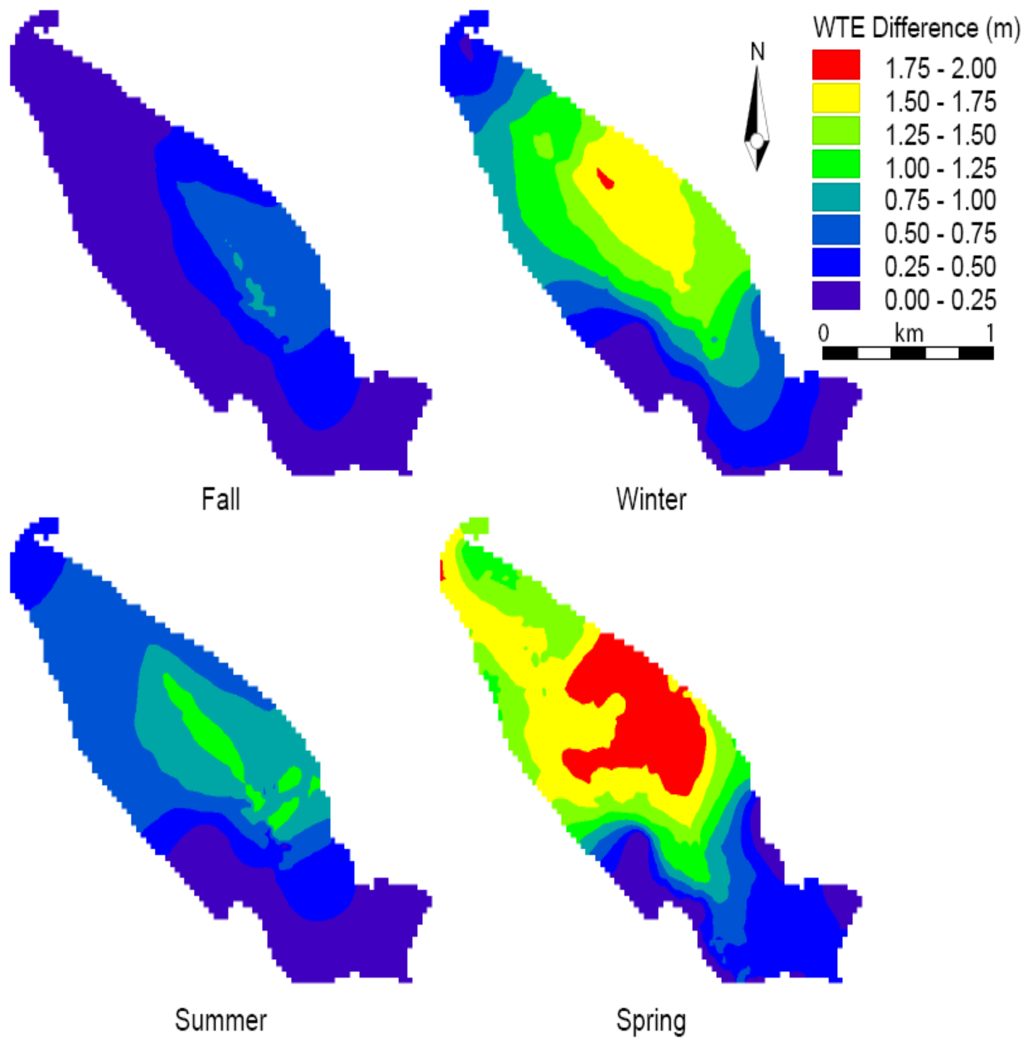


Figure 12. Seasonal water table elevation (WTE) differences between the 2005 water year incised and restored simulations. Clockwise from top left: mid-fall (15 October 2004), mid-winter (14 February 2005), mid-spring (16 May 2005) and mid-summer (15 August 2005). Positive difference indicates the restored 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.

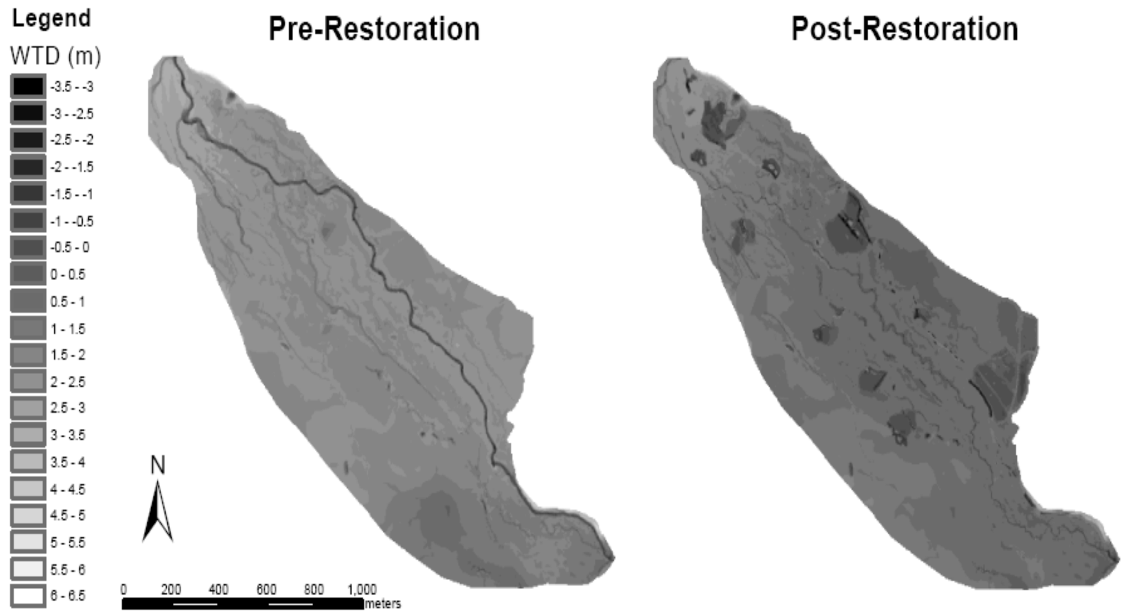


Figure 13. Comparison of growing season average water table depth (WTD) for the pre- and post-restoration hydrologic-topographic scenarios. Spatial water table depth averages are 1.86 m and 0.82 m for the pre- and post-restoration scenarios, respectively. Differences in water table depth result from 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.

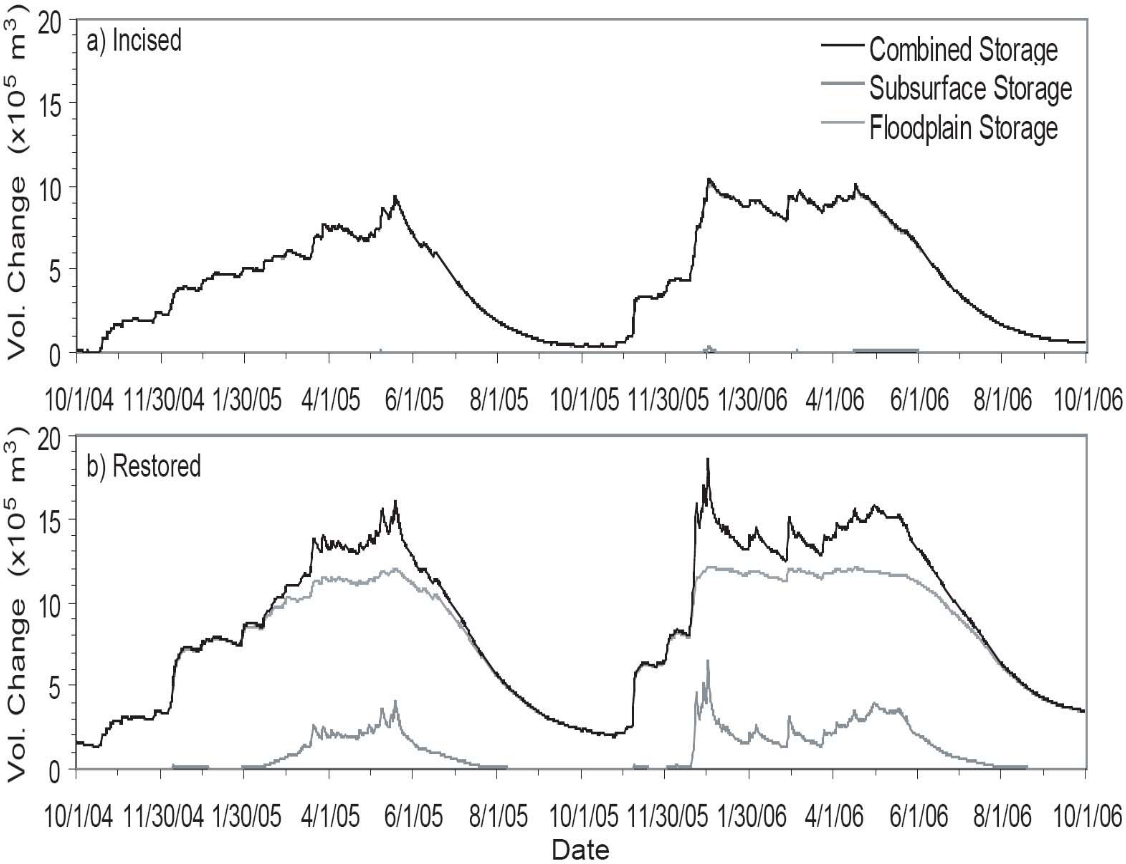


Figure 14. Storage volume change for subsurface storage, floodplain storage and combined (subsurface & floodplain) storage for a) incised and b) restored scenarios. The restored scenario stores a larger volume in each of the three categories, with a maximum combined storage of 10.45×10^5 m^3 and 18.52×10^5 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.

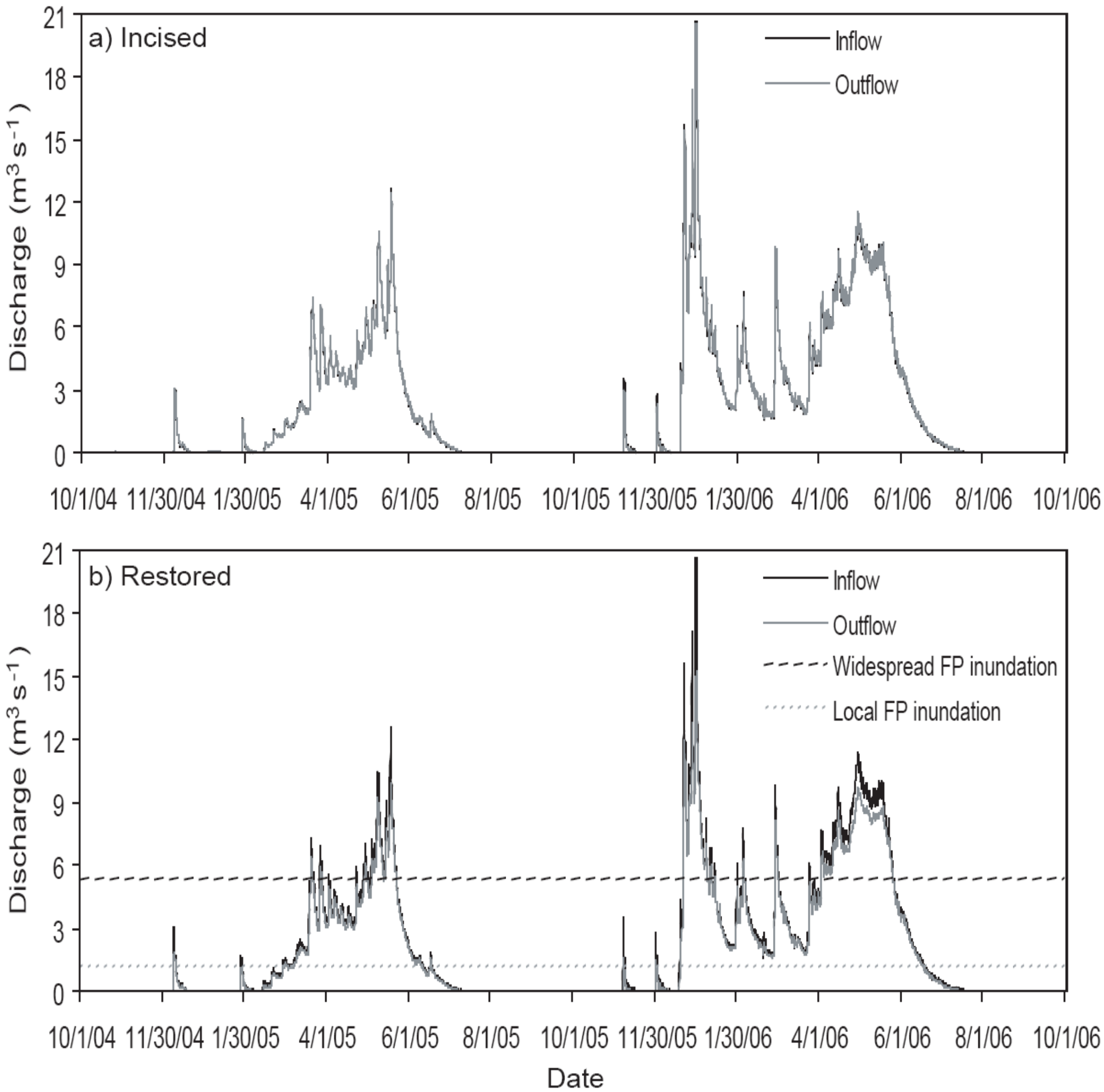


Figure 15. Time series of surface inflow and outflow for the a) incised and b) restored scenarios. Channel-floodplain exchange did not occur in the incised scenario, but occurred frequently and for extended periods in the restored scenario. Incised outflow was nearly identical to inflow, however restored 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 $m^3 s^{-1}$), above which local floodplain inundation occurred. The dashed line corresponds to the average capacity of the restored channel (5.35 $m^3 s^{-1}$) above which widespread floodplain inundation occurred. Minimum bankfull capacity of the incised channel was 28.0 $m^3 s^{-1}$, therefore floodplain inundation did not occur in the incised scenario.

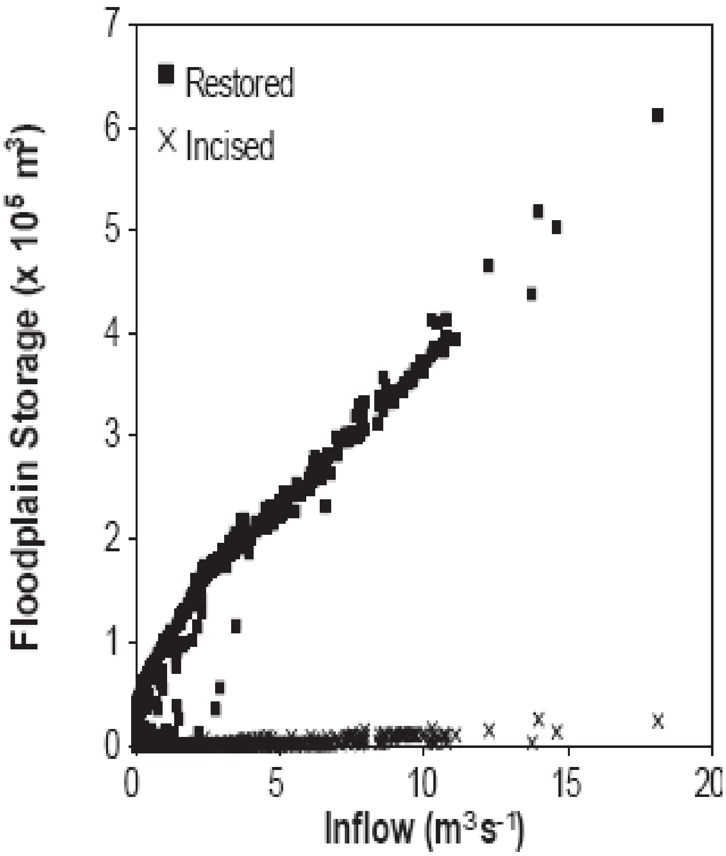


Figure 16. Average daily inflow vs. average daily floodplain storage for the incised and restored scenarios. As inflow increased the volume of water stored on the floodplain increased. A much larger volume of water is stored on the restored floodplain, due to enhanced channel floodplain connectivity resulting from the lower capacity restored channel.

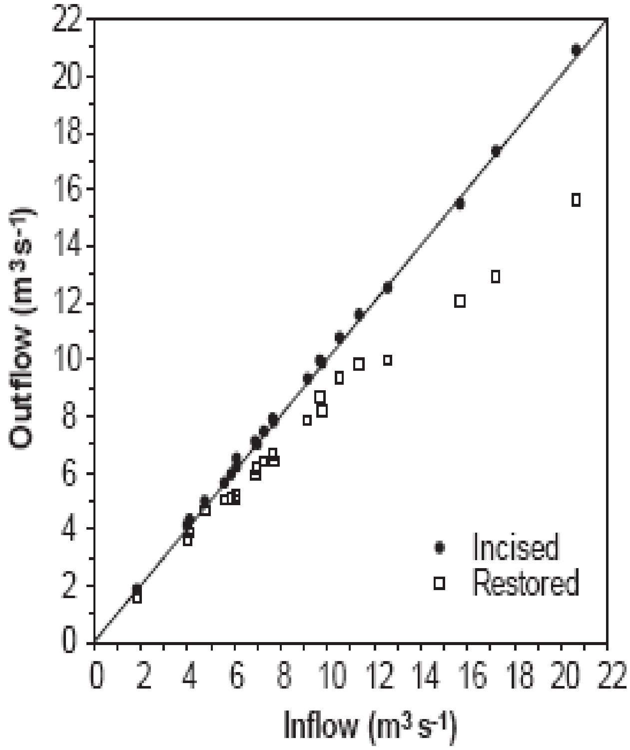


Figure 17. Comparison of flood peak inflow/outflow values for incised and restored conditions. Little change is observed between inflow and outflow values for the incised condition. Flows below ~ 4 $m^3 s^{-1}$ are mostly contained within the restored channel, and only minor reductions are observed due to subsurface recharge. However, for the largest peaks (i.e., >15 $m^3 s^{-1}$) a 25% reduction of the inflow peak is observed.

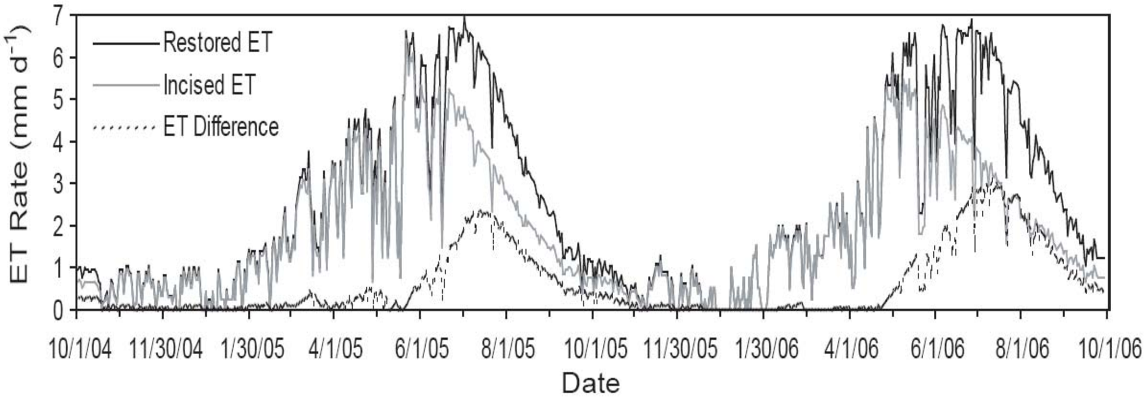


Figure 18. Daily evapotranspiration rates for the restored and incised scenarios. The difference between these two values is also provided. Daily ET rates were similar in both scenarios until mid-April of each year. After this point, daily ET rates declined in the incised scenario, but continued to increase 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 mm d^{-1} occurred on 11 July 2006.

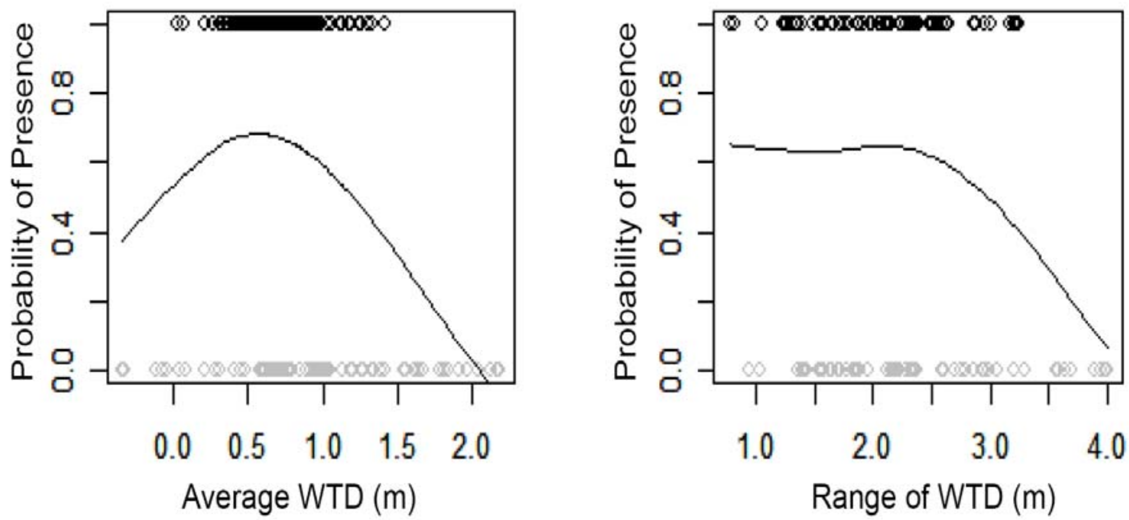


Figure 19. Predictors vs. response curves for *Juncus balticus*, a commonly occurring facultative wetland species. The y-axis represents the probability of presence for *Juncus balticus*, while the x-axis represents the individual predictor variables. Optimum of probability of presence occurs at an average water table depth of ~ 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.

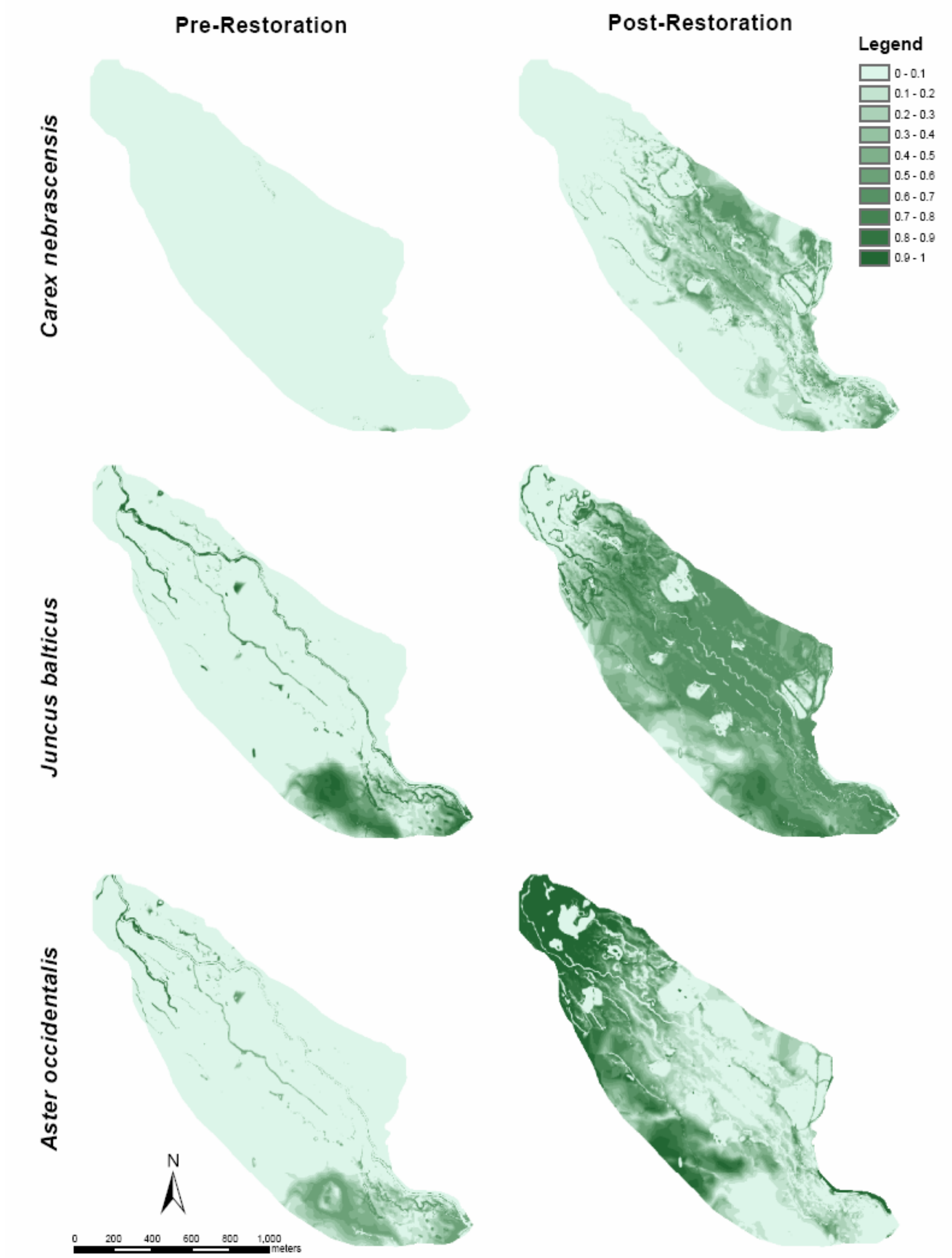


Figure 20. Comparison of pre-restoration and post-restoration probability of presence for three species on the hydric-mesic end of the hydrologic gradient. *Carex nebrascensis, Juncus balticus,* and *Aster occidentalis* belong in the obligate wetland, facultative wetland, and facultative wetland indicator categories, respectively. The meadow average probability of presence for each of these species increased due to stream restoration.

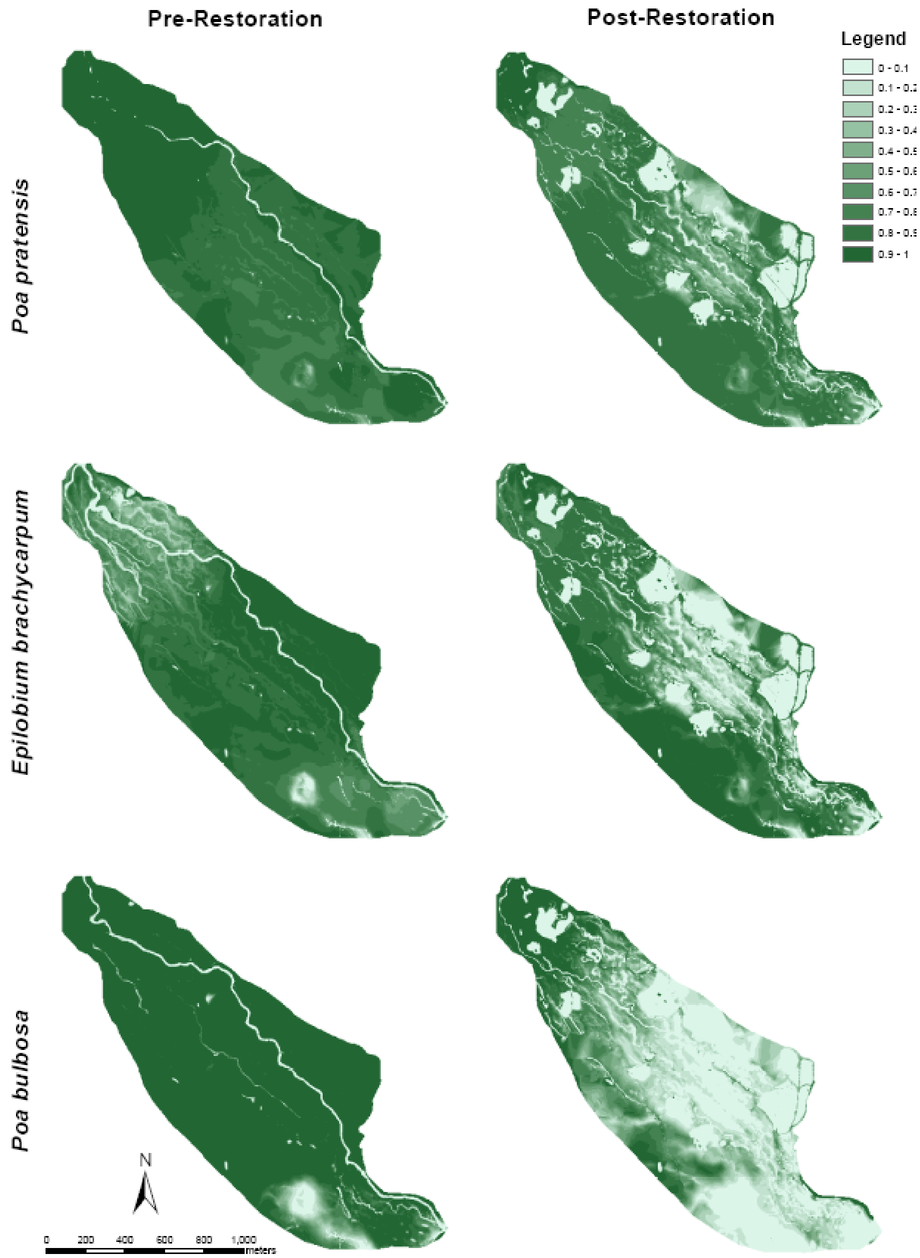


Figure 21. Comparison of pre-restoration and post-restoration probability of presence for three species on the mesic-xeric end of the hydrologic gradient. *Poa pratensis*, *Epilobium brachycarpum,* and *Poa bulbosa* belong in the facultative upland, obligate upland and unassigned (assumed to be obligated upland) wetland indicator categories, respectively. The meadow average probability of presence for each of these species decreased due to stream restoration.