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USE OF INDIVIDUALISTIC STREAMFLOW-VEGETATION RELATIONS ALONG THE FREMONT RIVER, UTAH, USA TO ASSESS IMPACTS OF FLOW ALTERATION ON WETLAND AND RIPARIAN AREAS

Gregor T. Auble, Michael L. Scott, and Jonathan M. Friedman
United States Geological Survey
Fort Collins Science Center
2150 Centre Avenue, Building C
Fort Collins, Colorado, USA 80526-8118

Abstract: We analyzed the transverse pattern of vegetation along a reach of the Fremont River in Capitol Reef National Park, Utah, USA using models that support both delineation of wetland extent and projection of the changes in wetland area resulting from upstream hydrologic alteration. We linked stage-discharge relations developed by a hydraulic model to a flow-duration curve derived from the flow history in order to calculate the inundation duration of 361 plots (0.5 × 2 m). Logistic regression was used to relate plant species occurrence in plots to inundation duration. A weighted average of the wetland indicator values of species was used to characterize plots as Aquatic, Wetland, Transitional, or Upland. Finally, we assessed how alterations in the flow duration curve would change the relative widths of these four zones. The wetland indicator values of species and the wetland prevalence index scores of plots were strongly correlated with inundation duration. Our results support the concept that plants classified as wetland species typically occur on sites inundated at least two weeks every two years. The portion of the riparian zone along the high-gradient study reach of the Fremont River that satisfied the vegetation criterion for a regulatory wetland was narrow (2 m wide). Both the unvegetated Aquatic zone (7.8 m) and the Transitional zone (8 m) were substantially wider. The Transitional zone included the maxima of several species and was, therefore, not merely a combination of elements of the Wetland and Upland zones. Multiplicative increases or decreases in streamflow regime produced a wetter, or drier, bottomland vegetation, respectively. Systematic reductions in flow variability reduced the width of both the Wetland and Transitional zones and increased the width of the Upland zone. Our approach is widely applicable to inform water management decisions involving changes in flow regime.

Key Words: instream flow, hydraulic modeling, downstream impact assessment, wetland delineation, moisture gradient, riparian vegetation plant communities

INTRODUCTION

Wetland regulation is conducted by multiple organizations operating under a complex set of mandates, laws, and court decisions. The domain of these activities has two primary dimensions: the extent of area delineated as wetland and the set of actions being considered. Both of these dimensions have been problematic for riverine wetlands in arid and semi-arid landscapes of the western United States.

Development of operational wetland delineation procedures has been a major area of technical activity over the last 20 years (Wentworth et al. 1988, NRC 1995, Tiner 1999, Wakeley 2002). These procedures are generally based on a concept of wetland as land that is wet enough to produce a distinctive plant community tolerant of saturated soils. Ecological riparian definitions, on the other hand, are based on a concept of land with communities distinctive from the surrounding landscape because of proximity to a watercourse (Lowe 1964). In arid and semi-arid landscapes, flood durations are typically short, and the area subject to anoxia is small. On the other hand, the importance of water subsidy from the river is magnified by the scarcity of water on the uplands. Therefore, rivers in dry regions typically support distinctive plant communities that are much broader than the set of species tolerant of saturated soils (Hughes 1994, Friedman and Auble 2000). This has created a tension between a desire to manage the response of the riparian system as a whole because of its importance as a landscape feature and the application of delineation protocols based on the importance of saturated and anoxic soils (NRC 1995, 2002). In spite of these difficulties, wetland indicator status (Reed 1988, USDA-NRCS 2003) has been shown to be an accurate reflection of the relative positions of plant species along the transverse gradient in western riparian systems (Auble et al. 1994, Stromberg et al. 1997).

Consideration of wetland impacts has tended to focus on either on-site hydrologic modifications such as levee construction, drainage, or placement of fill material in the wetland or on direct alteration of the vegetation such as clearing. These are important sources of wetland impacts in western riparian systems. However, the dominant factor changing many western riparian systems in arid lands is upstream water management that alters streamflow regimes (Brinson et al. 1981, Ohmart et al. 1988). Actions related to this factor tend to be considered in different legal and institutional structures from those dealing with eastern wetlands. The Federal Energy Regulatory Commission, the Bureau of Reclamation, the Army Corps of Engineers, and State appropriative-doctrine water law have primary roles, while the Environmental Protection Agency and Clean Water Act are less dominant. The relations between water management and wetland impacts are also less direct than the case of eliminating a marsh by clearing and filling. The downstream effect of water management on riparian communities is an off-site impact that must be weighed alongside competing social values such as recreation, power generation, and flood control. In order for riparian impacts to be considered along with these other issues, we need methods that quantitatively relate proposed changes in the flow regime to wetland response.

The central role of hydrologic regime, or hydroperiod, in structuring wetland communities has been recognized in formal wetlands classifications from Wells (1928) to Brinson (1993). Transverse (cross-valley) patterns in species distributions related to hydrologic gradients are also distinctive characteristics of most riparian ecosystems (Johnson and Lowe 1985, Malanson 1993, Bendix 1994, Patten 1998). Physical measurements of the hydrologic gradient both directly as inundation duration or flooding frequency and indirectly as relative elevation have been widely used to explain spatial patterns of riparian vegetation (Bedinger 1979, Leitman et al. 1983, Hupp and Osterkamp 1985, Smith 1996, Chapin et al. 2002). Stream discharge is the common currency of river management decisions and appropriative water law. Thus, relating stream discharge to site conditions in a riverine bottomland facilitates consideration of wetland and riparian impacts in water management decisions. For example, Johnson et al. (1999) used a water surface profile model to quantify changes in the hydroperiods of riverine wetlands from a proposed diversion of the Greybull River in Wyoming and suggested that this approach could meet the more general need for tools to evaluate effects of streamflow alteration on riverine wetlands.

Combining explicit hydraulic models with gradient relations of vegetation extends the domain of analysis by connecting stream discharge to vegetation response. Franz and Bazzaz (1977) used distributions of species along a gradient of inundation frequency to project vegetation changes associated with different reservoir levels. Auble et al. (1994, 1997) used a hydraulic model, in combination with flow duration curves, to quantify the distribution of herbaceous communities on an inundation-duration gradient and to predict the direction of shifts in vegetation community extent and location resulting from flow alterations. Similarly, Primack (2000) used inundation duration curves correlated to stream discharge, in combination with community positions on an inundation gradient, to examine potential effects of altered discharge associated with climate change projections.

Our goal is to develop practical methods for projecting the likely responses of streamflow-dependent vegetation to potential changes in flow regime. Here, we analyze the transverse pattern of vegetation along a reach of the Fremont River in Capitol Reef National Park, Utah, using procedures that support both delineation of wetland extent and projection of the changes in wetland area that might result from upstream hydrologic alteration. We fit the probability of occurrence of individual plant species to a moisture gradient of inundation duration. The resulting species distributions describe the transverse pattern of vegetation directly as a function of flow regime. We then calculate a weighted average of the wetland indicator values of individual species (Reed 1988, USDA-NRCS 2003) to score plots by a wetland prevalence index similar to the indices used to evaluate the wetland character of vegetation in wetland delineation procedures (Michener 1983, Wentworth et al. 1988, USDA-SCS 1994, NRC 1995, Tiner 1999). Finally, we illustrate how these relations can be used to estimate the indirect impacts of upstream flow alteration.

STUDY AREA

The Fremont River watershed, occupies an area of 5,025 km^2 east of the high plateaus of central Utah (U.S. Geological Survey Hydrologic Accounting Unit #14070003, Figure 1). The river flows from 3,300 m along the eastern flanks of the Fish Lake Hightop Plateau south through Rabbit Valley, a small agricultural area, and then eastward, through Capitol Reef National Park, to the town of Hanksville, where it joins Muddy Creek to form the Dirty Devil River at an elevation of 1,300 m. The climate of the Fremont River watershed ranges from semiarid at the highest elevations to arid at middle and lower elevations. Average annual precipitation ranges from less than 15 cm in low-elevation desert areas to over 100 cm on the high plateaus that define the upper Fremont river valley.

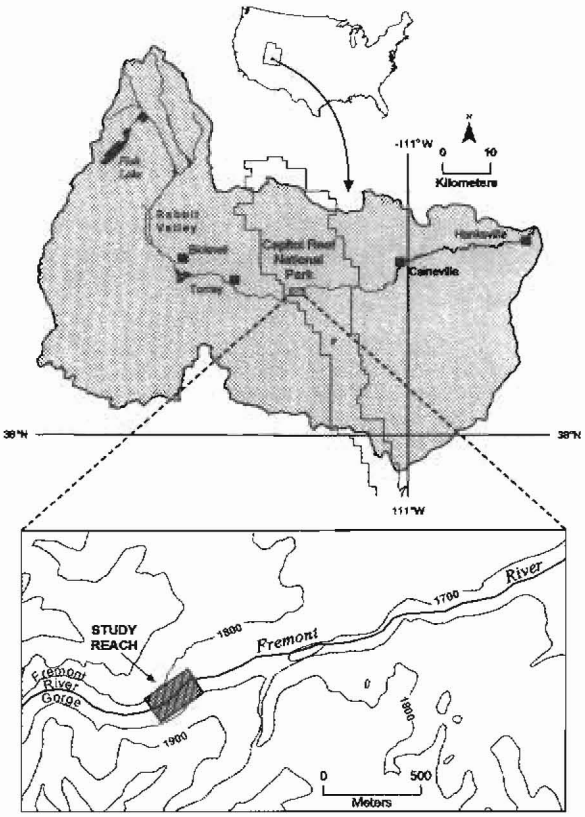

Figure 1. Location of the study site along the Fremont River in Capitol Reef National Park, Utah, USA.

The study area was a 265-m reach of the Fremont River in Capitol Reef National Park, just above the downstream end of the Fremont River Gorge (Figure 1). The study reach was selected by the National Park Service to provide a reasonable and physically accessible representation of the relation between streamflow and vegetation in the Fremont River Gorge while minimizing confounding influences of grazing and historic land-management activities. This reach consisted of a high-gradient (0.019) gravel and cobble-bedded channel (approximately 4 m wide) with narrow alluvial surfaces constrained by colluvium and rock falls from steep canyon walls. The distribution of these narrow, relatively level surfaces, and the associated presence of a contiguous set of relatively homogeneous vegetation plots, was highly variable among cross sections. In addition to variations in width of relatively level surfaces, on some cross sections, they occurred on only one side of the channel, and on some cross sections, they were absent altogether. Downstream from the study area, the Fremont River is typically dominated by Fremont cottonwood, *Populus deltoides* Bartr. ex Marsh. ssp. *wislizeni* (S. Wats.) Eckenwalder (Everitt 1995). Within the study area, however, the woody riparian vegetation was an Alder-Birch association, typical of narrow canyon portions of the Fremont River Gorge, where the dominant woody species include *Alnus incana* (L.) Moench ssp. *tenuifolia* (Nutt.) Breitung, *Betula occidentalis* Hook., *Salix exigua* Nutt., *Acer negundo* L., and *Fraxinus anomala* Torr. Ex S. Wats. (Romme et al. 1993). Vegetation on lower slopes within the canyon and adjacent to the riparian zone corresponded with the Dry and Mesic Canyon Bottom Shrub associations of Romme et al. (1993), including the shrubs *Fraxinus anomala*, *Rhus trilobata* Nutt. var. *trilobata*, *Artemisia bigelovii* Gray, and *Gutierrezia sarothrae* (Pursh) Britt. & Rusby, the herb *Artemeisia ludoviciana* Nutt., and the grass *Achnatherum hymenoides* (Roemer & J.A. Schultes) Barkworth.

METHODS

Hydrology

Water-surface elevations were related to stream discharge using a hydraulic model developed for the study site by the Water Rights Branch of the National Park Service's Water Resources Division (L. Hammack and B. Gillies, unpublished data). Channel geometry for the HEC-RAS (HEC 1998) step-backwater hydraulic model was defined by 23 cross sections surveyed in 1996. Eight of these cross sections were located at hydraulic controls and the additional 15 cross sections were distributed throughout the reach to ensure an adequate sample of the vegetation. Roughness coefficients for cross sections were estimated following Arcement and Schneider (1989), and the model was validated with observed water-surface elevations at discharges of 0.85, 2.55, 7.65, and 21.52 m^3/s. The difference in predicted water-surface elevation from 0.85 to 21.52 m^3/s averaged 1.08 m across 15 validation cross sections. Root mean square errors (the square root of the average squared difference between observed and predicted water-surface elevations) were 0.07 m at 0.85 m^3/s (n=12); 0.13 m at 2.55 m^3/s (n=5); 0.08 m at 7.65 m^3/s (n=12); and 0.07 m at 21.52 m^3/s (n=9). The hydraulic model was then run to predict water-surface elevation at 21 discharges ranging from 0.28 to 28.3 m^3/s. The resulting stage-discharge pairs at each cross section were used to construct rating curves of the form:

$$Q = \alpha(S - S_0)^\beta$$

where, Q is discharge, S is water-surface elevation, S_0 is water-surface elevation at zero discharge, and α and β are parameters specific to each cross section.

We sampled vegetation in 0.5 × 2 m plots, with an extension to 0.5 × 4 m for those woody species capable of achieving 10 cm diameter at breast height (dbh). Plots were oriented with the long dimension

parallel to the river and spaced along cross sections at 1-m intervals on channel, active flood plain, and low terrace surfaces, and at 3-m intervals on higher terraces. A topographic survey of each cross section included points at the streamward edge of each sampling plot and at topographic breaks. Interpolation between the surveyed points was used to define the elevation of each edge of a plot, and these two elevations were averaged to produce a single elevation value for each plot. Plot orientation and size were selected to minimize within plot variation in elevation and hydraulics while providing an adequate sample of vegetation present. However, heterogeneity due to bank slope and intraplot features was substantial in some cases and likely contributes to variation in observed plant distributions. We used our rating curves to determine plot-specific inundating discharges, defined as the minimum discharge required to inundate a plot of a given elevation on a specific cross section.

The National Park Service estimated a mean daily flow record for the Fremont River at the study site by relating flows at a local, temporary gage to the flows at the USGS Caineville gage (09330230), 23 km downstream on the Fremont River, below the confluence with Pleasant Creek. The linear regression was based on concurrent discharges measured from March–November of 1995 and 1996 and then applied to a modified version of the daily flow record at the Caineville gage for water years 1968–1996. The modification consisted of truncating the June–September Caineville flows to a maximum of 3.4 m^3/s to correct for the greater influence of flows produced by summer thunderstorms below Pleasant Creek. This modification was based on observed magnitudes of the summer storm peaks at the gages in 1995 and 1996. It was applied in all years except 1983–1984 when high flows after June 1 were due to late snowmelt and extended-duration summer rainfall.

Inundation durations were calculated for each sampled plot by applying the plot-specific inundating discharge to the nominal flow-duration curve representing the 1968–1996 estimated daily flow record. We discarded data from plots with inundating discharge greater than 250 m^3/s. Seven plots located on large boulders were also excluded as being unable to support rooted vegetation under any flow scenario. This resulted in 361 sampled plots. Using our flow record of 19 years, plots inundated only one day during the highest flow of the entire record had an inundation duration of $10^{-4.025}$; plots with greater inundating discharges were assigned an inundation duration of $10^{-4.03}$.

Vegetation

Species presence was recorded in each plot on July 12–15, 1996. Nomenclature and wetland indicator assignments for species follow USDA-NRCS (2003). Presence was based on individuals rooted in the plot. Three types of relations were used to fit the presence-absence distributions of individual species to a gradient of inundation duration. Gaussian logit curves (ter Braak and Looman 1995) were used for the majority of species. Logistic regression (Hosmer and Lemeshow 1989, SAS 1999) was used to fit the equation

$$p = \frac{c \exp[-.5(x - u)^2/t^2]}{[1 + c \exp(-.5(x - u)^2/t^2)]}$$

where p = probability of occurrence of a species; c, u, and t are estimated parameters; exp (a) = e^a; and x = $\log_{10}$ (inundation duration). The parameter u represents the environmental optimum, or value of inundation duration that produces the maximum probability of occurrence. Tolerance, t, is a measure of ecological amplitude corresponding to the standard deviation of the Gaussian distribution. The maximum probability of occurrence, p_{max}, is related to the parameter c by $p_{max} = c/(1 + c)$. Gaussian logit regression results are reported in terms of the readily interpretable p_{max}, u, and t. A likelihood ratio test statistic, distributed as χ^2 with 2 df, is reported for the null hypothesis that all the explanatory variables have regression coefficients of 0. The max-rescaled R^2 of Nagelkerke (1991) is also reported for each logistic regression.

A linear logistic, or sigmoid, curve was used for species, including the Water pseudo-species that had most of their occurrences at one extreme of the environmental gradient (always wet or always dry). Logistic regression was used to fit species presence-absence data using the equation:

$$p = [\exp(b_0 + b_1 x)]/[1 + \exp(b_0 + b_1 x)]$$

where p = probability of occurrence of a species; b_0 and b_1 are regression parameters; and x = $\log_{10}$ (inundation duration). The parameters b_0 and b_1, the likelihood ratio test χ^2 (with 1 df), and the max-rescaled R^2 are reported for these regressions. Because these regression coefficients are less directly interpretable than those derived for the Gaussian logit form, the predicted probabilities of occurrence are also reported for the dry (duration of inundation = $10^{-4.03}$) and wet (inundation duration = 1) ends of the gradient, and for an inundation duration of 10^{-2}.

In two cases, all occurrences were at the dry extreme of the inundation duration gradient. These distributions were represented as a step function with the probability of occurrence at the dry extreme equal to the observed proportion of such plots containing the species, and with the probability of occurrence elsewhere equal to zero.

Simulation

The combination of a hydraulic model defining the discharge to inundate each plot and a flow-duration curve representing a particular hydrologic regime produced an inundation duration for each plot. The species distribution curves estimated a probability of occurrence of each modeled species as a function of the inundation duration. In order to approximate the vegetation criterion used in wetland delineation procedures, we used weighted averaging to calculate a predicted wetland prevalence index (WPI) for each plot:

$$\mathrm{WPI} = \left(\sum_{sp=1}^{n} p_{sp} \cdot WI_{sp}\right) \Big/ \sum_{sp=1}^{n} p_{sp}$$

where WPI = wetland prevalence index for a plot; p_{sp} = probability of occurrence for species *sp*, based on the inundation duration of the plot and the species distribution curve of the species; WI_{sp} = wetland indicator value for species *sp* from the regional lists of plant species that occur in wetlands (Reed 1988, USDA-NRCS 2003); and n = number of species for which a species distribution curve is defined. Wetland indicators for individual plants of Obligate (OBL), Facultative Wetland (FACW), Facultative (FAC), Facultative Upland (FACU), and Upland (UPL) were assigned numeric values of 1 to 5, respectively (Tiner 1999). Plots with values of the predicted wetland prevalence index less than or equal to 2.5 are likely wetland by most delineation procedures; plots with values between 2.5 and 3.5 are inconclusive, which we refer to as transitional; and plots with values greater than 3.5 are likely to be uplands (Tiner 1999). We used these values to define Wetland, Transitional, and Upland zones. We used a value of 50% or greater for the probability of occurrence of unvegetated Water psuedo-species to separate Aquatic from Wetland but excluded the Water psuedo-species from the wetland prevalence index calculation. In addition to the predicted wetland prevalence index, we also calculated an observed wetland plot score, which was the average of the wetland indicator values for the species occurring in a plot.

Aggregate values for the study area were calculated by weighting each plot by the width of the cross section that it represents to account for variable plot spacing, and then averaging across the 15 vegetation cross sections, assuming that they are all equally representative. The resulting variables (individual species occurrence, wetland prevalence index, inundation duration, and inundating discharge) have units of width (m) associated with a unit length of river.

To assess the effects of flow alteration, we explored changes in zone width resulting from changes in the flow-duration curve. In the absence of a specific proposed flow alteration, such as a new diversion or reservoir, we examined combinations of two simple, systematic alterations of the flow regime. The focus is not on any class of common flow alteration but, rather, on separating dimensions of flow magnitude and flow variability. The first was a flow magnitude multiplier where each flow in the nominal flow record was multiplied by a parameter to increase or decrease all flows by a fixed proportion. We used flow magnitude multipliers ranging from 0.1 to 2.0, where a magnitude multiplier of 1.0 is equivalent to the unmodified, historical flow record. The second flow alteration was a deviation multiplier that multiplied all the deviations from mean flow in a given flow sequence by a fixed fraction. This dampened flow variability, reducing the standard deviation of flows by the deviation multiplier, without changing the total flow volume. We used deviation multipliers of 0.25, 0.5, 0.75, and 1.0. The two alterations, flow magnitude multiplier and variability dampening, were combined to produce a factorial set of flow alterations.

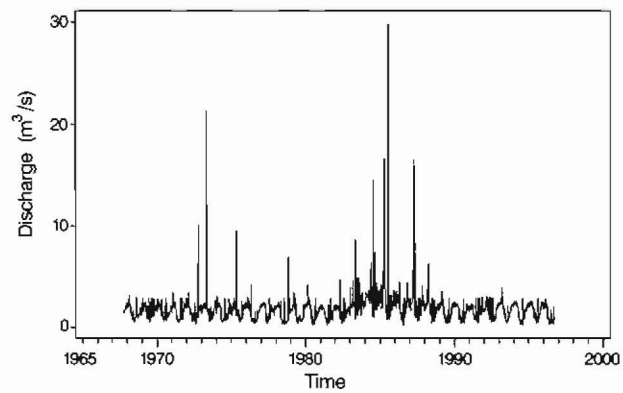

Figure 2. The nominal flow regime (reconstructed record of mean daily flow for the Fremont River at the study site, 1968–1997).

RESULTS

Current Pattern

Mean daily discharge in the nominal flow regime was 1.76 m³/s, with a standard deviation of daily flows of 1.18 m³/s (Figure 2). Daily flows ranged from 0.2 m³/s to 29.7 m³/s. Species distributions were defined for 41 of the 69 species encountered in the sampled plots and the pseudo-species unvegetated Water (Tables 1 and 2). Twenty-six species had too few occurrences (generally less than four total occurrences) to establish a clear distribution. In two other cases, the species could not be identified sufficiently to assign a wetland indicator. These species were excluded from all analyses and represented 6% of the total species occurrences. Gaussian logit relations were used for 28 species (Table 1), logistic sigmoid relations increasing

Table 1. Gaussian logit species distributions. Model parameters u and t are in units of $\log_{10}$ of inundation duration expressed as fraction of time inundated, and p_{max} is probability of occurrence. R^2 is max-rescaled Rsquare (Nagelkerke 1991). Wetland indicators are from USDA-NRCS (2003).

Species	Wetland Indicator	Model Parameters: u $\log_{10}$ (ID)	Model Parameters: t $\log_{10}$ (ID)	Model Parameters: p_{max}	Model Statistics: Likelihood Ratio X^2 (df = 2)	Model Statistics: Likelihood Ratio $Pr > X^2$	Model Statistics: R^2
Acer negundo L.	FACW	−2.23	0.86	0.24	29.4	<.0001	0.18
Agrostis stolonifera L.	FACW	−1.73	0.68	0.77	116.8	<.0001	0.44
Alnus incana (L.) Moench ssp. *tenuifolia* (Nutt.) Breitung	FACW	−1.68	0.69	0.50	60.4	<.0001	0.33
Asclepias speciosa Torr.	FACW	−2.02	0.78	0.43	55.3	<.0001	0.27
Betula occidentalis Hook.	FACW	−2.05	0.88	0.11	11.2	0.0037	0.12
Bromus inermis Leyss.	UPL	−3.01	0.23	0.29	36.8	<.0001	0.40
Bromus tectorum L.	UPL	−3.78	0.14	0.31	39.7	<.0001	0.28
Clematis ligusticifolia Nntt.	FACU	−2.92	1.14	0.17	20.8	<.0001	0.11
Cornus sericea L. ssp. *sericea*	FACW	−1.80	0.64	0.30	38.0	<.0001	0.33
Eleocharis palustris (L.) Roemer & J.A. Schultes	OBL	−1.04	0.33	0.90	54.0	<.0001	0.47
Elymus canadensis L.	FACU	−2.19	0.83	0.27	34.4	<.0001	0.21
Elymus repens (L.) Gould	FACU	−3.25	0.45	0.38	56.3	<.0001	0.25
Epilobium ciliatum Raf.	FAC	−1.20	0.45	0.35	15.0	0.0006	0.29
Equisetum arvense L.	FAC+	−1.64	0.73	0.28	28.4	<.0001	0.23
Equisetum hyemale L.	FACW	−2.13	0.58	0.98	228.5	<.0001	0.60
Helianthus nuttallii Torr. & Gray	FACW	−2.37	0.62	0.32	56.9	<.0001	0.34
Juncus balticus Willd.	FACW	−1.79	0.70	0.55	73.7	<.0001	0.36
Maianthemum stellatum (L.) Link	FAC	−2.74	0.52	0.29	52.1	<.0001	0.31
Mentha arvensis L.	FACW	−1.13	0.37	0.71	32.0	<.0001	0.40
Muhlenbergia asperifolia (Nees & Meyen ex Trin.) Parodi	FACW+	−3.25	0.43	0.17	23.0	<.0001	0.18
Poa compressa L.	FACU	−2.29	0.72	0.54	91.6	<.0001	0.37
Ranunculus cymbalaria Pursh	OBL	−0.90	0.29	0.70	25.1	<.0001	0.42
Salix exigua Nutt.	OBL	−2.07	0.79	0.81	105.8	<.0001	0.34
Solidago canadensis L.	FACU	−2.12	0.69	0.54	92.0	<.0001	0.40
Stuckenia filiformis (Pers) Boerner ssp. *occidentalis* (J.W. Robbins) Haynes, D.H. Les. & M. Kral	OBL	−1.05	0.69	0.14	14.5	<.0001	0.20
Symphyotrichum lanceolatum (Willd.) Nesom ssp. *hesperium* (Gray) Nesom var. *hesperium*	OBL	−1.60	0.68	0.11	10.2	0.0062	0.20
Thermopsis montana Nutt.	UPL	−2.32	0.92	0.11	12.0	0.0025	0.11
Veronica americana Schwein. ex Benth. or *V. anagallis-aquatica* L.	OBL	−0.37	0.12	0.33	20.8	<.0001	0.48

to the dry extreme of the gradient were used for 11 Upland and Facultative Upland species, step functions at the dry end of the gradient were used for two Upland species, and a logistic sigmoid curve increasing to the wet end of the gradient was used for the pseudo-species unvegetated Water (Table 2). All logistic regressions were significant at levels less than 0.01. Maximum probabilities of occurrence, p_{max}, ranged from 0.11 (*Thermopsis montana*) to 0.98 (*Equisituem hyemale*) for the Gaussian distributed species. For species with sigmoid distributions, maximum predicted probabilities on the gradient ranged from 0.03 (*Artemisia tridentata*) to 0.34 (*Rhus trilobata* var. *tribolata*). Representative distributions are depicted in Figure 3.

Positions of species on the gradient of inundation duration, as represented by the value of the environmental optimum parameter, u, for Gaussian distributed species, were strongly correlated with the wetland indicator value (Pearson $r^2 = 0.66$, $p < 0.0001$). Plot scores for wetland indicator prevalence index (Figure 4) were correlated with inundation duration for both the observed plot presence-absence data (Pearson $r^2 = 0.45$, $p < 0.0001$) and for predicted plot composition using the fitted species distributions (Pearson $r^2 = 0.90$, $p < 0.0001$).

The flow duration curve for the nominal hydrologic regime (Figure 2) had a maximum value of 29.7 m^3/s, which inundated 54% of the study area width (where lateral bound of study area is defined by an inundating discharge of 250 m^3/s). Application of the nominal flow duration curve to individual species distributions and subsequent calculation and aggregation of wetland prevalence index values for plots produced a cnmulative distribution of valley width by predicted wetland

Table 2. Linear logit and fixed proportion species distributions. Predicted probabilities of occurrence are given for the upland and aquatic extremes and for an inundation duration of 1 percent. R^2 is max-rescaled Rsquare (Nagelkerke 1991). Wetland indicators are from USDA-NRCS (2003).

Species	Wetland Indicator	Probability of Occurrence			Model Parameters		Model statistics		
		Upland	ID = 0.01	Aquatic	b_0	b_1	Likelihood Ratio X^2 (df = 1)	Likelihood Ratio $Pr > X^2$	R^2
Achnatherum hymenoides (Roemer & J.A. Schultes) Barkworth	UPL	0.31	<.01	<.01	−18.15	−4.31	84.8	<.0001	0.38
Artemisia ludoviciana Nutt.	FACU	0.25	0.05	<.01	−4.67	−0.89	41.9	<.0001	0.19
Artemisia tridentata Nutt.	UPL	0.03	<.01	<.01	−17.26	−3.45	8.1	0.0045	0.16
Elymus elymoides (Raf.) Swezey ssp. *elymoides*	UPL	0.14	<.01	<.01	−25.00	−5.74	36.2	<.0001	0.28
Elymus trachycaulus (Link) Gould ex Shinners	UPL	0.14	<.01	<.01	−13.53	−2.90	32.5	<.0001	0.23
Ericameria nauseosa (Pallas ex Pursh) Nesom & Baird	UPL	0.19	0.01	<.01	−8.66	−1.78	40.4	<.0001	0.23
Fraxinus anomala Torr. Ex S. Wats.	UPL	0.09	<.01	<.01	−25.63	−5.77	22.6	<.0001	0.23
Rhus trilobata Nutt. var. *trilobata*	UPL	0.34	0.02	<.01	−7.58	−1.72	77.4	<.0001	0.32
Rosa woodsii Lindl.	FAC−	0.12	0.04	0.01	−4.28	−0.57	13.4	0.0002	0.08
Sporobolus cryptandrus (Torr.) Gray	FACU−	0.05	<.01	<.01	−15.01	−3.01	12.4	0.0004	0.17
Thelypodium integrifolium (Nutt.) Endl. ex Walp. ssp. *integrifolium*	UPL	0.22	0.01	<.01	−8.76	−1.85	47.8	<.0001	0.25
WATER (unvegetated)	—	<.01	<.01	0.87	1.93	8.48	326.3	<.0001	0.86
Astragalus sp.	UPL	0.08	—	—	—	—	—	—	—
Atriplex canescens (Pursh) Nutt.	UPL	0.06	—	—	—	—	—	—	—

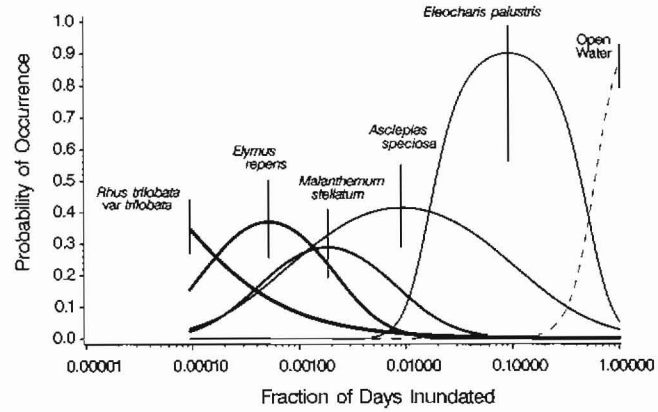

Figure 3. Fit distributions of representative species along the Fremont River in Capitol Reef National Park, Utah. Vertical line represents a 95% confidence interval for probability of occurrence at the optimum value of inundation duration.

prevalence index (Figure 5). The zone meeting the wetland vegetation criterion constituted a small fraction of the riparian community (Figure 5). Both the Wetland and Transitional zones occupied a linear portion of the relation between bottomland width and inundating discharge where inundating discharge increased gradually with width. Most of the Aquatic plots were inundated by very low discharges, including many plots that were under water at zero flow because of backwater effects. Although most of the Upland zone occurred on steeper sections of bank where the inundating discharge of plots was increasing rapidly, these plots were generally above the stage of the maximum discharge in the nominal hydrologic regime. Area-weighted average elevations of the zones were 0.1 m above the water stage at zero discharge for the

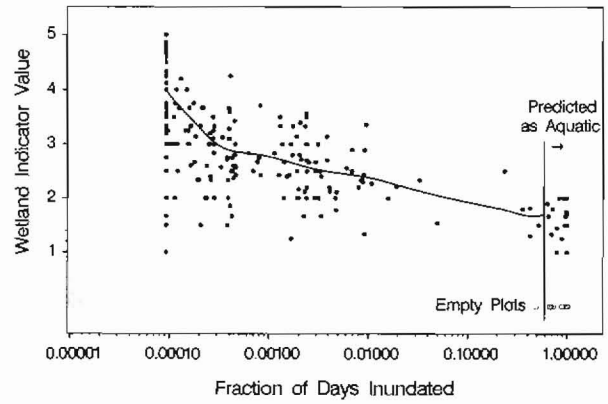

Figure 4. Predicted wetland prevalence index and observed wetland scores for plots on gradient of inundation duration. Circles are observed plots scores based on presence-absence. Solid line is predicted plot wetland prevalence index score derived from predicted species composition at different inundating discharges. Open circles indicate open water plots with no vegetation.

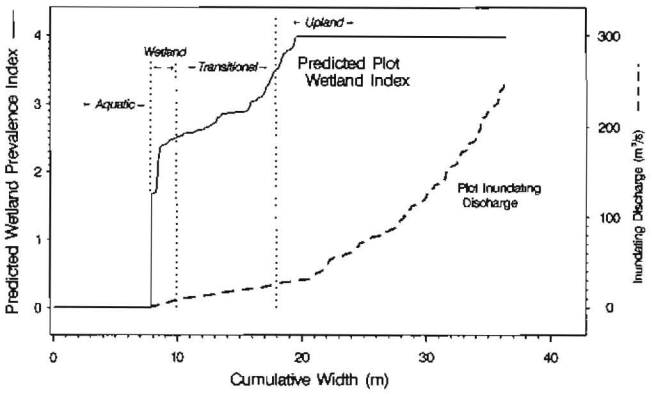

Figure 5. Distribution of predicted plot wetland prevalence index and inundating discharge along aggregated bottomland width. Cumulative width represents the total aggregated width at or below a given value of predicted wetland prevalence index or inundating discharge. Plots predicted as Aquatic are assigned a wetland prevalence index value of 0.

unvegetated Aquatic zone, 0.7 m for the Wetland zone, 1.3 m for the Transitional zone, and 2.8 m for the Upland zone within the study area as defined above.

Projected Responses to Altered Streamflow

Across the range of flows examined, from 0.1 to 2 times the nominal magnitude, increasing discharge generally increased the widths of the Aquatic, Wetland, and Transitional zones, while decreasing the width of the xeric Upland zone (Figure 6). The distribution of bottomland width among the Upland, Transitional, and Wetland zones was heavily dependent on flow variability. Systematic reductions in flow variability reduced the widths of the moderately and infrequently inundated Wetland and Transitional zones and increased the width of the rarely or never inundated Upland zone.

Whereas the broad patterns were generally consistent throughout the range of flow alterations explored here, details depended on the non-linear combinations of flow volume and variability with the distributions of plot inundating discharges and species occurrence on the inundation gradient. For example, in the range of flow magnitude multipliers from 1.6 to 2.0, a deviation multiplier of 0.75 resulted in greater widths of the Transitional zone than did the nominal deviation multiplier of 1.0. When the nominal deviation multiplier is decreased from 1.0 to 0.75, the Transitional zone gains more width from the Upland zone than it loses to the Wetland zone.

DISCUSSION

The transverse patterning of vegetation is strong along the Fremont River, as it is on the margins of

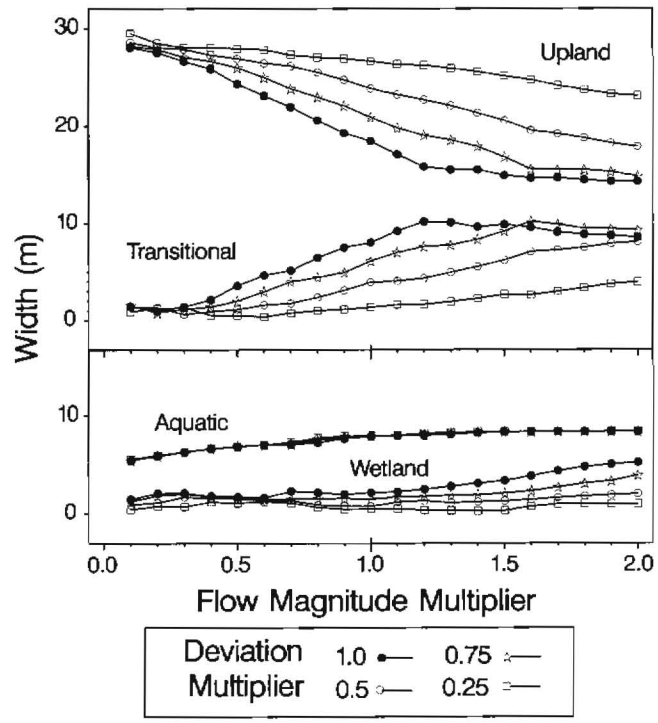

Figure 6. Aggregate width of zones predicted by flow scenarios. The four lines for each zone represent different values of the flow deviation multiplier. The combination of a flow magnitude multiplier = 1 and deviation multiplier = 1 corresponds to the unmodified, nominal flow regime. Zones are based on predicted plot wetland prevalence index values as described in text.

many rivers in the western United States (Auble et al. 1994, 1997, Castelli et al. 2000, Merritt and Cooper 2000). Moderate intensity of presence-absence sampling defined highly significant Gaussian or sigmoid distributions on the transverse gradient of inundation duration for 39 species, representing over 90% of the total species occurrences in plots. The positions of species on the inundation gradient were highly correlated with their wetland indicator values, providing some confidence for both the assignment of indicator values and our physical model of the moisture gradient.

Weighting wetland indicator values of species at a plot by predicted probabilities of occurrence from fit distributions on the inundation gradient produced a predicted wetland prevalence index that approximated the procedures used in wetland delineation. The predicted wetland prevalence index increased monotonically, but non-linearly, with decreasing inundation duration (Figure 4). Observed plot wetland scores show a similar pattern but with considerably more scatter. Sources of the wide variation in observed plot wetland scores include (a) microsite variation within plots; (b) plot size, which produces a restricted set of species occurring on an individual plot; and (c) the lack of a

probability of occurrence weighting in the observed plot scores (each species on a plot receives equal weight). Furthermore, the much wider dispersion of plot scores on the upland end of the gradient suggests that intermediate (i.e., FACW, FAC, and FACU) species may be physiologically more able to occur in small numbers on the dry side of their optimum than on the wet side.

NRC (1995) recommended a hydrology criterion for wetland delineation of at least two growing season weeks of soil saturation to within 0.3 m of the surface at least every other year. Simply interpreting this as an overall annual inundation duration of 0.019 (14 of 730 days) corresponds to a predicted wetland prevalence index of 2.25 from our model of the Fremont River vegetation (Figure 4). NRC (1995) recommended that prevalence index scores of less than 2.0 be interpreted as a high probability that the site is a wetland, scores between 2.0 and 2.5 as a good probability the site is a wetland, and scores between 2.5 and 3.5 as being equivocal. Thus, there is excellent correspondence between our measure of the wetland hydrology criterion (inundation duration) and our modeled wetland prevalence index (vegetation criterion).

Our analysis of flow magnitude and deviation multipliers illustrates the capability of using individual species distributions on an inundation duration gradient, coupled with a hydraulic model, to estimate the direction and magnitude of vegetation changes with flow alteration. As expected, increases in streamflow produced a wetter bottomland vegetation. The area of Upland vegetation decreased and the sum of Aquatic, Wetland, and Transitional zones increased. The rates of these changes with respect to discharge and the distribution of area among the Aquatic, Wetland, and Transitional zones were more complex. These depend on the shapes of the individual species distributions (e.g., tolerance parameter for Gaussian-distributed species), the hydraulic geometry of the bottomland, and flow variability.

The importance of flow variability in promoting riparian vegetation can be seen by comparing curves representing multiplicative reductions in flow deviations (Figure 6). For a given average flow, the distribution of area among the Upland, Transitional, and Wetland zones was heavily dependent on the magnitude of flow deviations from the mean flow. Lower flow variability produced more Upland and less of the intermittently inundated Transitional and Wetland zones. The extreme case of constant daily flow would result in no predicted width of the Wetland or Transitional zones, as all plots would be either always or never inundated. Similar effects of dampening flow variability have been predicted for the Black Canyon of the Gunnison, Colorado (Auble et al. 1994) and observed along the Green River, Colorado, downstream from Flaming Gorge Reservoir (Merritt and Cooper 2000).

There was an interaction between flow variability and flow magnitude (Figure 6). For example, the effect of flow deviation on the amount of Wetland and Transitional area differed at different mean flows. This is because the additional amount of bottomland area inundated by an increment of discharge varies with the starting discharge.

The approach we illustrate for a reach of the Fremont River meets the basic requirements of a procedure estimating the effects of flow alteration on downstream wetland and riparian vegetation—it quantifies their extent and projects how their composition and extent would change under alternative flow regimes. The basic components of this approach consist of alternative flow regimes, a set of stage-discharge relations, and species parameters that represent the location of vegetation along a physical gradient. We used the physical gradient of inundation duration, summarized the flow regime as a flow duration curve, and calibrated individual species locations using a portion of the historical flow regime at the site. This general approach is widely applicable to water management decisions involving changes in flow regime. The strength of the approach is that it connects hydrologic alteration to a comprehensive prediction of vegetation change across a bottomland, through a series of relatively straightforward and standard relations.

The simplicity that produces practicality can ignore important detail. The limitations of this approach need to be considered carefully to avoid inappropriate application. Limitations center around the assumption of an equilibrium between vegetation and flow regime, reliance on an invariant hydraulic geometry of the bottomland, use of a single physical gradient of inundation duration, and issues of calibration and validation. Prentice and Solomon (1991) distinguished static versus dynamic vegetation models in the context of predicting vegetation response to global change. Our use of a static model, expressing vegetation distribution as a function of innndation duration calculated from a long-term flow duration curve, suppresses temporal dynamics. In fact, the plant composition of the bottomland changes on scales of days to decades in response to seasonal variables and the sequence of streamflows. At best, the type of equilibrium vegetation produced by this approach represents a point in a cluster of possible compositions associated with a given long-term flow-duration curve. At worst, collapsing a flow regime into a flow-duration curve could ignore critical sequence dependencies or timing components of flow relative to species life history requirements (Scott et al. 1996, Auble and Scott 1998).

We assumed that our hydraulic model relating streamflow to inundation was invariant. This is a reasonable assumption in many cases, especially in constrained channels such as our study reach of the Fremont River. However, if the stage-discharge relation is likely to change under an altered flow regime (Williams and Wolman 1984, Shafroth et al. 2002), this change should be considered in assessment of vegetation response.

We represented the influence of the river using the single physical gradient of inundation duration. In fact, there are a number of strongly correlated physical variables influencing plant composition in the riparian zone. Other studies have shown a strong correlation between transverse vegetation pattern and flood frequency, recurrence interval, shear stress, unit stream power, and height above a reference water-surface elevation (Wharton et al. 1982, Hupp and Osterkamp 1985, Smith 1996, Bendix 1999, Friedman and Auble 1999, Chapin 2002). We used inundation duration for simplicity, assuming it was an effective surrogate for other correlated variables. This assumption is not always adequate. In particular, inundation duration lacks resolution in representing ground-water mediated influences beyond the zone of inundation and poorly represents the disturbance effects of extreme floods.

Direct quantitative validation of model predictions would require comparing vegetation associated with multiple long-term hydrologic sequences on otherwise replicate reaches. A suitable set of different long-term hydrologic regimes and concomitant vegetation data is not available for our study area on the Fremont River and is rare for any river. Use of a single physical gradient limits space for time substitutions using multiple reaches or rivers. Variation in longitudinal variables such as temperature, evapotranspiration, precipitation, and the importance of side-slope hydrologic and disturbance processes may alter the position of species on the inundation duration gradient. These unrepresented variables will limit the extent to which calibration or predictions from one reach are applicable to another river or reach.

Qualitative confidence in the model is supported by the significance of individual distributional fits, the correspondence between species position and their wetland indicator status, known mechanisms by which inundation duration influences plant distributions, and the general directions of model predictions (e.g., less streamflow means more xeric plant composition and movement of hydric plants to lower positions in the bottomland). Nonetheless, the limitations on calibration and validation mean that this type of approach is best employed as a first-order indication of the direction and extent of vegetation change associated with streamflow rather than a precise or accurate prediction of specific composition at any point in time.

CONCLUSIONS

The combination of (a) flow-duration curves summarizing hydrologic regimes, (b) hydraulic models relating stream discharge to water surface elevation, and (c) distributions of individual species along a gradient of inundation duration can be used to characterize the transverse pattern of riparian vegetation and estimate changes associated with streamflow alteration. This approach has important limitations but is widely applicable to inform water management decisions about likely changes in downstream wetland and riparian vegetation dependent on streamflow.

There has been substantial uncertainty about the extent to which wetland management and regulatory programs are appropriate for riparian areas (NRC 1995, NRC 2002). The emphasis on riparian management is strongest in the western United States, where the vegetation associated with water courses is often strikingly different from that of the background arid uplands and where streamflows are managed under appropriative doctrine State water law. Our results confirm that the riparian zone with vegetation dependent on streamflow can be much larger than the zone that might fall under most conceptual and operational definitions of wetlands. The portion of the riparian zone along our study reach of the Fremont River that satisfies the vegetation criterion for a regulatory wetland is narrow, with a width of approximately 2 m compared to a Transitional zone width of 8 m and an unvegetated Aquatic zone width of 7.8 m. The transitional zone is substantially wider and includes many species characteristic of neither the Wetland zone nor the Upland zone (Table 1).

Much of the identifiable riparian zone along the Fremont River would not meet the vegetation criterion for a regulatory wetland. However, there is a definite and measurable zone that would meet the vegetation criterion. Furthermore, the cumulative area of this zone can be substantial when integrated over the length of river that might be affected by a change in flow regime. For example, 20 km of the Fremont River would have slightly over 4.1 ha of associated jurisdictional wetland. Furthermore, the area with flow-dependent plant composition (sum of Aquatic, Wetland, and Transitional zones) that would be strongly influenced by flow alteration is 35.8 ha for a similar 20-km reach.

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