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Research article

A new approach for hydrologic performance standards in wetland mitigation

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ABSTRACT

Wetland restoration performed as a requirement of compensatory mitigation does not always replace lost acreage or functions. Most new projects are required to identify performance standards to evaluate restoration outcomes. Current performance standards are primarily related to vegetation with little to no evaluation of wetland hydrologic regimes. Because of the agreement in the scientific literature about the role of hydrology in creating and maintaining wetland structure and function, hydrologic performance standards may be an ecologically meaningful way to evaluate restoration outcomes. This research tests the use of water level data from project specific reference sites to evaluate restored water levels for three distinct wetland types across the United States. We analyzed existing datasets from past and ongoing wetland mitigation projects to identify the number of years it took water levels in restored wetlands to match reference sites, and to test whether similar water levels between restored and reference sites leads to increased vegetation success. Wetland types differed in the number of years it took for water levels to match reference sites. Vernal pools in California took nine years to match reference sites, fens and wet meadows in Colorado took four years, and forested wetlands in the southeastern US were hydrologically similar to reference sites the first year following restoration. Plant species cover in all three restored wetland types was related to the water level similarity to reference sites. Native cover was higher when water levels were more similar to reference sites, and was lower in areas where water levels were different. Exotic species cover showed the opposite relationship in fens and wet meadows, where hydrologic similarity led to low cover of exotic species. Along with the general agreement of the importance of hydrology for wetland form and function, this research shows that hydrologic performance standards may also lead to increased vegetation success in some wetland types.

1. Introduction

Defining success in ecological restoration has been difficult for many ecosystem types (Suding, 2011). For wetland permits authorized under Section 404 of the U.S. Clean Water Act (CWA), the Army Corps of Engineers (ACE) may require compensatory mitigation to restore wetlands and offset unavoidable losses to existing wetlands. These mitigation projects are required to specify quantitative performance standards to evaluate restoration success, which have been shown to improve restoration outcomes (Fennessy et al., 2007; National Research Council, 2001; Schlatter et al., 2016). However, even when meeting defined standards, mitigation wetlands can have simplified vegetation (Gutrich et al., 2009), soils with higher bulk density and lower organic matter content (Fennessy et al., 2004b), and lower rates of carbon and nitrogen cycling than suitable reference areas (Hossler et al., 2011). Current performance standards are primarily based on vegetation with little to no evaluation of wetland hydrologic processes (Environmental Law Institute, 2004).

Wetland hydrologic regimes, including the depth, duration, and seasonality of surface and groundwater, vary among wetland types and are a primary control over wetland form and function (Cook and Hauer, 2007; Sieben et al., 2010). In contrast to this known variation among wetland types, the CWA defines any wetland as having saturation within 30 cm of the ground surface for 12.5% of the frost-free growing season (Environmental Laboratory, 1987). All wetland restoration projects under the CWA must meet this hydrologic criterion. However, this simple requirement does not account for important differences in water table depth and dynamics within and among wetland types (Johnson et al., 2012). Additionally, hydrologic regimes of mitigation wetlands that meet the ACE minimum requirement can substantially differ from reference wetlands (Fennessy et al., 2007).

Ecological performance standards are required for many CWA permitted wetland mitigation projects (USACE and USEPA, 2008). Performance standards are designed to indicate whether a project has met its pre-determined goals. Existing standards often focus on plant species richness and cover (Environmental Law Institute, 2004), but vegetation

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and habitat structure may not develop within the typical mitigation monitoring timeframe of five years (Dee and Ahn, 2012). Wetland water levels can develop more rapidly following restoration in some wetland types (Schimelpfenig et al., 2014), although it may take multiple years in others (Black and Zedler, 1998). Identifying the time required for water levels to match reference wetlands can inform required monitoring timelines following restoration.

An approach to create ecologically meaningful hydrologic performance standards would provide a robust tool for the analysis of wetland mitigation programs (Gardner et al., 2009). Hydrologic performance standards are most effective when they are developed using site specific success criteria (Schlatter et al., 2016). Because hydrologic regimes create the template for wetland biotic composition, structure, and function (Gowing, 2005; Silvertown et al., 1999), restored wetlands with a hydrologic regime similar to reference sites may be more likely to develop vegetation similar to reference sites.

In this paper, we present and evaluate a new approach for creating quantitative hydrologic performance standards using water level data collected concurrently from wetland reference and mitigation sites. We test this approach in three wetland types spanning the continental US: California vernal pools, Colorado Rocky Mountain fens and wet meadows, and forested mineral soil wetlands in Virginia and North Carolina. We specifically ask: 1) do wetland types differ in the time required for restored water levels to match water levels in reference sites; and 2) does similarity in water levels between restored and reference sites lead to successful vegetation development in restored wetlands? We use these analyses to evaluate whether one approach for creating hydrologic performance standards can be used to evaluate widely differing wetland types.

2. Materials and methods

2.1. Study sites

Existing hydrology and vegetation datasets were obtained for 17 wetland mitigation projects conducted over the last 20 years for three different wetland types (Table 1). We obtained datasets from individual researchers and the Regulatory In-lieu Fee and Bank Information Tracking Systems (RIBITS) database managed by the USACE (https://ribits.usace.army.mil/) (Fig. 1). Datasets included depth to water table for restored and reference wetlands and annual vegetation data. Construction and planting methods were similar among projects for each wetland type and each project had reference wetlands monitored concurrently with the restored wetlands. Naturally occurring wetlands adjacent to, and representing the same wetland type as, the proposed project area served as reference wetlands.

2.1.1. Fens and wet meadows

Fen and wet meadow data were obtained from the Telluride Ski and Golf Company (Telski) where 11 wetlands were restored in Mountain Village, Colorado. More than 30 ha of wetlands had been filled during the 1980s and 1990s without a USACE permit. Following a legal settlement with the United States Environmental Protection Agency (EPA), Telski restored about half of the impacted wetland area from 1998 to 2003. The two dominant wetland types restored were groundwater driven wet meadows with mineral soils and fens with organic soils. Fens and wet meadows were restored by removing fill to recreate historic grade, disabling artificial drainage features, and planting native shrub and herbaceous species as seedlings or rooted cuttings (Cooper et al., 2017). All wetlands were supported by groundwater discharge from both snow melt and late summer precipitation. Vegetation was predominantly species of the genus *Salix* in the overstory and sedges in the genus *Carex* in the understory.

2.1.2. Vernal pools

Vernal pool data were obtained from two restoration and creation projects in San Diego and six projects in Sacramento, both in California, USA. These two areas have distinct substrate, landforms and climate and represent the range of environmental conditions in the region. Restoration reestablished pools that had been filled and levelled over time, and many areas retained undisturbed pools that were used as reference sites. Suitable restoration or creation sites had a shallow, low-permeability clay-rich aquitard. Pools were created by excavating surface sediments to reach the aquitard, with the final pool geometry designed to simulate reference pools. Restoration and creation projects ranged from 4 to 30 ha in size and from 40 to 1379 restored pools. Vernal pools rely on precipitation during the spring months and are characterized by annual grasses and forbs with little to no woody vegetation.

Table 1
Mitigation project information obtained and used in these analyses, including wetland type, site name, location, year restoration took place, the number of wetlands restored, the total hectares restored, the number of reference wetlands and the number of reference monitoring wells, the number of years of data available, the monitoring frequency water levels were recorded, and the type of vegetation data collected.

Wetland Type	Site	Location	Year Restored	# of wetlands restored	Total Hectares Restored	# of reference wetlands (wells)	Record Length (years)	Hydrologic Observation Frequency	Vegetation data type
Fens	Telluride Ski and Golf	Telluride, CO	1999–2001	2	1.2	2 (5)	12	Weekly/daily	Cover
Wet Meadows	Telluride Ski and Golf	Telluride, CO	1999–2003	9	10.9	6 (11)	12	Weekly	Cover
Vernal Pools	Dennery	San Diego	2008	40	4	12 (1)	7	biweekly	Cover
	SR125	San Diego	2006	103	4.9	20 (1)	5	weekly	Cover
	Aitken	Sacramento	2003	200	4.2	15 (1)	7	weekly	Cover
	Locust Road	Sacramento	2008	108	30.4	58 (1)	5	weekly	Cover
	Meridean	Sacramento	2012	553	14.6	11 (1)	3	weekly	Cover
	Toad Hill	Sacramento	2010	1379	17.4	30 (1)	5	weekly	Cover
	Van Vleck	Sacramento	2009	248	7.9	50 (1)	6	biweekly	Cover
	Vincent	Sacramento	2005	224	5.1	11 (1)	10	weekly	Cover
Non-Riverine Forested Mineral Flats	ABC	North Carolina	2000	1	75	1 (4)	2	daily	Tree Counts
	Dover	Virginia	2009	1	71.2	1 (1)	6	weekly	Tree Counts
	Edge Farm	Virginia	2006	1	199.1	1 (1)	8	daily	Tree Counts
	Hall	Virginia	2000	1	12.5	1 (2)	6	daily	Tree Counts
	Roquist	North Carolina	2007	1	15.0	1 (5)	4	daily	Tree Counts
	Sliver Moon	North Carolina	2011	1	5.7	1 (1)	5	daily	Tree Counts
	Stephens	Virginia	2003	1	57.5	1 (1)	7	daily	Tree Counts
	Su	Virginia	2000	1	22.3	1 (6)	13	daily	Tree Counts

Fig. 1. Study sites across the United States with aerial images from select projects including restored vernal pools in California (A), fens and wet meadows in Colorado (B), and forested wetlands in the mid-Atlantic (C). **A** Restored and monitored vernal pools are identified with the orange point and adjacent reference pools are identified by the orange points with a black hatch mark. **B** Restored fens and wet meadows are outlined in blue, with the adjacent reference wetland shown with blue hatching. **C** Restored forested wetlands outlined in green, with the adjacent reference well shown with the green point. Aerial images show representative projects from each wetland type. Images sourced from Esri 2018. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

2.1.3. Non-riverine forested wetlands

Forested wetland data were from eight restoration projects within the historic extent of the Great Dismal Swamp in Virginia and North Carolina (Table 1). Forested wetlands in the eastern US have been drained for more than a hundred years, with drainage canals installed to lower the local water table to create suitable conditions for agriculture. Restoration of forested wetlands in this area involved converting cropland to forested wetlands by filling or blocking drainage ditches and planting native wetland trees. Forested wetland restoration projects ranged from 6 to 200 ha. Forested wetlands rely on precipitation to create high water tables during the fall, winter, and spring months. Vegetation in forested wetlands is dominated by a variety of tree species in the overstory, including *Acer rubrum, Liquidambar styraciflua, Nyssa sylvatica, Taxodium distichum*, and various species of *Quercus*.

2.2. Hydrologic monitoring data

Groundwater monitoring wells in the fens and wet meadows were made of slotted schedule 40 PVC installed to a depth of approximately 1 m (Cooper et al., 2017). Water levels in wells were measured weekly from May through October from one year prior to restoration (1997–2000) through the final approval of each wetland by the USACE and EPA (2001–2007), and again in 2013 and 2016.

Vernal pool ponded water depth was manually measured using staff gauges in the center of each monitoring pool and recorded every 3–7 days for the period when pools were filled, typically between January and April. Many projects identified a subset of restored and reference pools for monitoring, as it was not feasible nor required by their mitigation permit conditions to monitor all pools.

Groundwater monitoring wells in restored and reference forested wetlands included slotted PVC installed to a depth of at least 50 cm, measured manually or using recording pressure transducers or capacitance rods. Groundwater monitoring wells were measured daily if automated loggers were used and weekly if done manually. Water level monitoring typically occurred from March through June every other year following restoration through final approval by the USACE.

2.3. Vegetation monitoring data

Vegetation monitoring methods differed among wetland types in accordance with regional permit conditions. In fens and wet meadows, 3×5 m plots were established adjacent to monitoring wells in reference and restored sites where plant cover by species was visually estimated to the nearest percent in 1999, 2000, 2001, and 2013. Vernal pool plant species absolute cover was monitored annually. Although monitoring methods were consistent within each vernal pool project, vegetation plot size varied between projects. Some projects used the

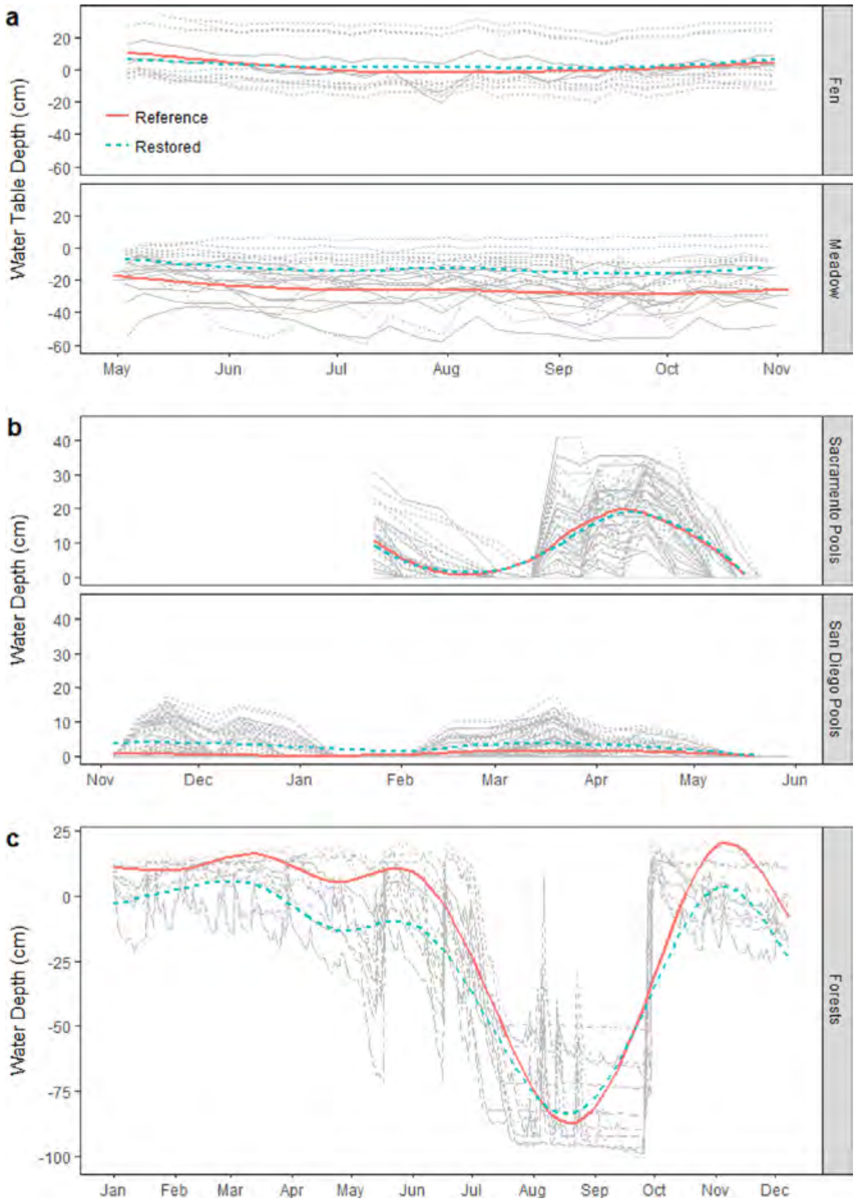

Fig. 2. Hydrographs for restored (dashed) and reference (solid) wetlands: (a) fens and wet meadows in 2001, (b) vernal pools in 2009, and (c) non-riverine forested wetlands in 2010. Measured water level at each site is shown in grey, with average water depth shown in blue for restored and red for reference sites. **A** Restored fens and wet meadows had a wider range of water table depths than reference sites. Reference fens had consistently higher water tables than reference meadows, though some restored meadows were wetter than restored fens. **B** Vernal pools in Sacramento have higher ponding depths than vernal pools in San Diego. Vernal pools filled between November and February, dried by June, and remained dry the rest of the year (Note compressed x-axis). Restored pools in Sacramento had similar hydroperiods than reference pools, though restored pools in San Diego had higher ponded water depths than reference pools. **C** Water table depths in restored and reference forested wetlands were near the ground surface in the fall and spring, and over 75 cm below ground during the summer. Restored forested wetlands had lower water tables than reference forested wetlands during the fall and spring. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

entire vernal pool as a single vegetation plot, while others established multiple plots within each pool. Forested wetland vegetation plots were 10 × 10 m. Living tree stems were identified to species and counted within each plot on an annual basis.

2.4. Statistical analysis

2.4.1. Years to hydrologic similarity

The time required for restored water levels in each wetland type to match reference sites was defined as the number of years until no statistical difference existed between the restored and reference site average annual water table depth. Average annual water table depth for restored wetlands was compared to reference sites each year following restoration using a hierarchical Bayesian model. Annual mean water depths were created by averaging water depths from each well over the annual monitoring period for each wetland type. The statistical model was based on a linear deterministic equation commonly used in Bayesian analyses (Eqn (1) to estimate the effect size of an experimental treatment (Hobbs and Hooten, 2015).

$$g(\boldsymbol{\beta}, x_{ijt}) = \beta_{oj} + \beta_{1t} x_{ijt} \tag{1}$$

All parameters were evaluated for the i^{th} reading in the j^{th} wetland type in the t^{th} year. Restored plots are given an x value of 1 and reference sites are given the x value of 0. Using this design, the β_0 coefficient identifies the reference water level mean for wetland *j*, and the β_1 coefficient identifies the difference in water levels between reference and restored wetlands. The β coefficients were distributed independently for each wetland type and year using a normal distribution with vague priors. The number of years required for restored water levels to match reference sites for each wetland type was defined as the estimated time between restoration and when the 95% credible intervals of the β_1 coefficient include zero.

2.4.2. Vegetation response to hydrologic similarity

We estimated the vegetation response to hydrologic similarity between restored and reference wetlands for each wetland type using a separate Bayesian model. Vegetation metrics included the absolute cover of all native and exotic species in fens, wet meadows, and vernal pools, and the number of tree seedlings and tree species richness in forested wetlands. We used a linear model (Eqn (1) where the β_0 coefficient estimated the cover of each vegetation metric for plot *i* when the hydrologic conditions in the restored wetland perfectly matches the reference site (intercept term), and the β_1 coefficient is the slope of the regression line relating mean weekly hydrologic similarity in monitoring well *i* from its reference site to changes in vegetation in the restored site.

Each well's mean weekly hydrologic similarity to the reference site was calculated by averaging the weekly water table depth in each reference site and subtracting it from the restored water levels for the same week. This value was then averaged across all weeks in each year to calculate a mean weekly hydrologic similarity between the restored and reference sites for each year. Vegetation metrics from each plot were paired with the hydrologic conditions in the adjacent well. Only two of the forested mitigation projects used paired wells and vegetation plots and were included in this analysis. Absolute percent cover in fens, wet meadows, and vernal pools was modelled using a normal distribution, while tree seedling counts and species richness in forested sites were modelled with a Poisson distribution.

All parameters were estimated in JAGS using the R package rjags, with convergence evaluated using the Gelman statistic. Posterior predictive checks were evaluated by using the squared error from the observed data and a simulated data set. Bayesian p-values were obtained from the squared error for standard deviation, mean, and the discrepancy between the observed and simulated datasets.

3. Results

Each wetland type had a distinct hydrologic regime (Fig. 2). Fens and wet meadows had consistent water table depths from May through November, vernal pools were ponded from December to May, and forested wetlands had water tables near the ground surface from October through June (Fig. S1). Water levels in reference fens averaged −2.0 cm (±1.1) compared to −6.8 cm (±0.9) for restored fens (Fig. 2a). Reference wet meadows had an annual water table average of −26.6 cm (±0.4), compared to −26.4 cm (±0.2) for restored wet meadows (Table 2). The mean summer water table across restored fens ranged from −20.1 to +35.7 cm and −108.2 to +7.9 cm for restored wet meadows.

Vernal pools filled with water during rain events in December through April and were dry by May in most years (Fig. S2). Ponding depth was much lower in San Diego than Sacramento pools (Fig. 2b). The two San Diego reference sites had average water depths of 1.8 cm (±0.2) and 4.5 cm (±0.3) during the spring, while in Sacramento the reference pools ranged from 7.3 (±0.2) to 12.3 cm (±0.3). Average restored water depth for San Diego pools was similar to reference sites and averaged 4.4 cm (±0.1) and 5.2 cm (±0.1). Restored Sacramento pools were also similar to their reference pools with average water depths from 6.7 (±0.1) to 15.3 cm (±0.3).

The seasonal water table variation in forested wetlands was similar between restored and reference sites (Fig. S3), with water levels near the ground surface from October through June, and much lower from July through September (Fig. 2). Restored forested wetlands had lower water table depths in the fall, winter, and spring than reference areas. March through June water levels averaged across all monitoring years were similar in restored and reference forested wetlands and ranged from −25.7 cm (±0.4) to −14.1 cm (±0.5) in reference sites and −30.0 cm (±0.5) to −7.3 cm (±0.2) in restored sites (Table 2).

3.1. Years to hydrologic similarity

The number of years required for water levels in restored wetlands to match reference wetlands differed among wetland types (Fig. 3). Restored fens and wet meadows had higher groundwater levels than reference sites following restoration (Pr ($\beta_1 > 0$) = 0.96), though no difference between restored and reference water levels existed after four years (Pr ($\beta_1 > 0$) = 0.90; Fig. 3a). Restored vernal pools had higher water levels than reference pools for six years following restoration (Pr ($\beta_1 > 0$) = 1.00), lower water levels in years seven and eight (Pr ($\beta_1 > 0$) = 0.02) and were not statistically different nine years after restoration (Pr ($\beta_1 > 0$) = 0.69; Fig. 3b). Restored and reference forested wetlands had similar water levels the first year following restoration (Pr ($\beta_1 > 0$) = 0.50; Fig. 3c). The 95% credible intervals included zero during the entire monitoring period except year six when restored forested wetlands had higher water levels than reference sites (Pr ($\beta_1 > 0$) = 0.99).

3.2. Hydrologic similarity and successful vegetation establishment

Hydrologic similarity between restored and reference sites influenced the percent canopy cover of native and exotic plant species in fens and wet meadows, native plant species cover in vernal pools, and tree species richness and seedling density in forested wetlands (Fig. 4; Table 3). Native species cover in fens and wet meadows was highest when restored water levels matched reference sites (β_0 = 105.91), and native cover decreased as restored water levels became less similar to reference sites (β_1 = −0.87, Pr ($\beta_1 < 0$) = 0.98). In contrast, exotic species cover was low in restored fens and wet meadows when water levels were more like reference sites (β_0 = −0.07) and increased as restored water levels were less like reference sites (β_1 = 0.66, Pr ($\beta_1 > 0$) = 1.00).

Table 2
Average water level (±one standard error) for each wetland type and project site. Data are averaged over all wells and all monitoring years. Fen and wet meadow data are from May through October. Vernal Pools data are from January through April. Forested wetland data are from March through June.

Wetland Type	Site	Reference	Restored
Fens	Telski	−2.0 (±1.1)	−6.8 (±0.9)
Wet Meadows	Telski	−26.6 (±0.4)	−26.4 (±0.2)
Vernal Pools - San Diego	Dennery	1.8 (±0.2)	5.2 (±0.1)
	SR125	4.5 (±0.3)	4.4 (±0.1)
Vernal Pools - Sacramento	Aitken	12.4 (±0.3)	14.5 (±0.2)
	Locust Road	11.0 (±0.2)	15.3 (±0.3)
	Meridean	9.0 (±0.5)	9.7 (±0.2)
	Toad Hill	7.3 (±0.2)	6.7 (±0.1)
	Van Vleck	9.0 (±0.2)	15.0 (±0.2)
	Vincent	11.5 (±0.3)	10.6 (±0.1)
Non-Riverine Forested Mineral Flats	ABC	−18.5 (±0.4)	−7.3 (±0.2)
	Dover	−24.4 (±3.2)	−30.0 (±0.5)
	Edge Farm	−21.0 (±0.8)	−20.7 (±0.1)
	Hall	−25.7 (±0.4)	−23.0 (±0.3)
	Roquist	−14.1 (±0.5)	−23.2 (±0.3)
	Sliver Moon	−17.0 (±0.5)	−22.2 (±0.2)
	Stephens	−24.9 (±1.3)	−23.2 (±0.2)
	Su	−25.7 (±0.4)	−29.7 (±0.1)

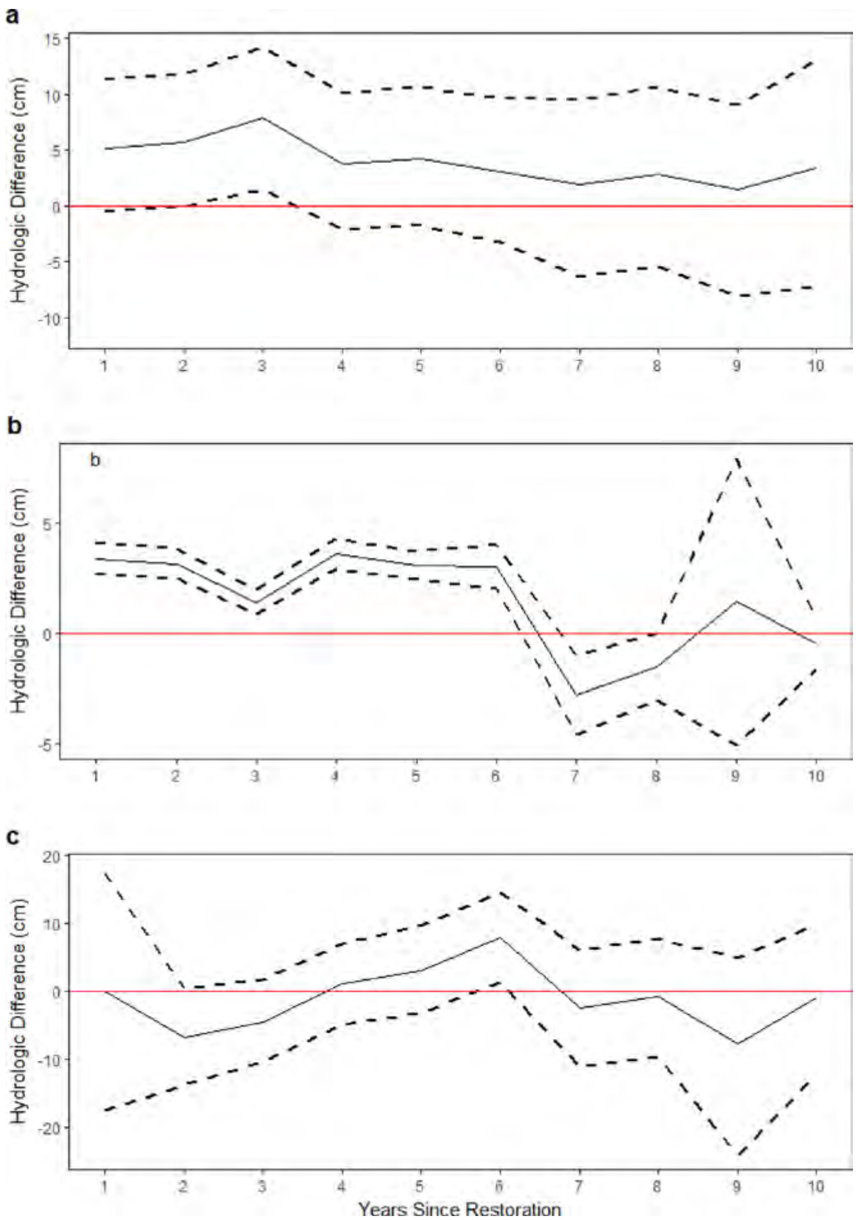

Fig. 3. The estimated hydrologic difference between restored and reference wetlands for each year following restoration for (a) fens and wet meadows, (b) vernal pools, and (c) forested wetlands. Average hydrologic difference between restored and reference wetlands (solid line) and 95% credible intervals of the estimate (dashed lines) indicate the restored wetland hydrologic difference from reference water tables through time following restoration. **A** Water tables in restored fens and wet meadows are significantly higher than reference sites for the first three years following restoration, and have no credible difference from reference sites after four years following restoration. **B** Restored vernal pools have greater inundation than reference pools for the first six years, but no credible difference between restored and reference pools after 8 years. **C** Forested wetlands are hydrologically similar to reference wetlands from the first year following restoration.

For three of the eight vernal pool projects, native species cover was highest when water levels in restored pools were similar to reference pools (Pr ($\beta_1 < 0$) = 1.00). Native species cover in the other five vernal pool restoration projects, all from the Sacramento region, was not significantly related to hydrologic similarity to reference pools. Hydrologic similarity between restored and reference pools did not influence exotic species cover for any vernal pools project (Table 3).

The number of tree seedlings and tree species richness in forested wetlands was related to the hydrologic difference between restored and reference sites for one of the two forested projects and unrelated in the other project. The highest tree seedling counts ($\beta_0 = 19.91$) and tree species richness ($\beta_0 = 5.05$) at the Roquist site occurred in areas where water levels matched the reference site, and declined where water levels were less similar to the reference site (tree seedling counts: $\beta_1 = -0.82$, Pr ($\beta_1 < 0$) = 0.99; tree species richness: $\beta_1 = -0.14$, Pr ($\beta_1 < 0$) = 0.99). Tree seedling counts and tree species richness were not significantly related to hydrologic similarity to the reference at the Edge Farm site.

4. Discussion

Hydrologic comparisons between restored wetlands and project-specific reference sites provide an ecologically meaningful approach for evaluating wetland restoration outcomes. Restored wetlands for all three wetland types became hydrologically similar to reference sites within the 10 years of monitoring. Additionally, hydrologic similarity between mitigation and reference sites was correlated with higher cover of native plant species and lower cover of exotic species in fens and wet meadows, higher native species cover in three vernal pool projects, and higher tree seedling counts and tree species richness in one of the two forested wetland projects.

The time required for water levels in restored wetlands to match their reference sites differed among the three studied wetland types. Water levels in restored fens and wet meadows in Colorado took four years to match reference sites, a similar time for herbaceous plants in these wetlands to reach maximum shoot density (Cooper et al., 2017). Abundant groundwater flow through these wetlands and the dominance of herbaceous and shrub species likely limit future changes to water levels due to increased evapotranspiration as vegetation matures.

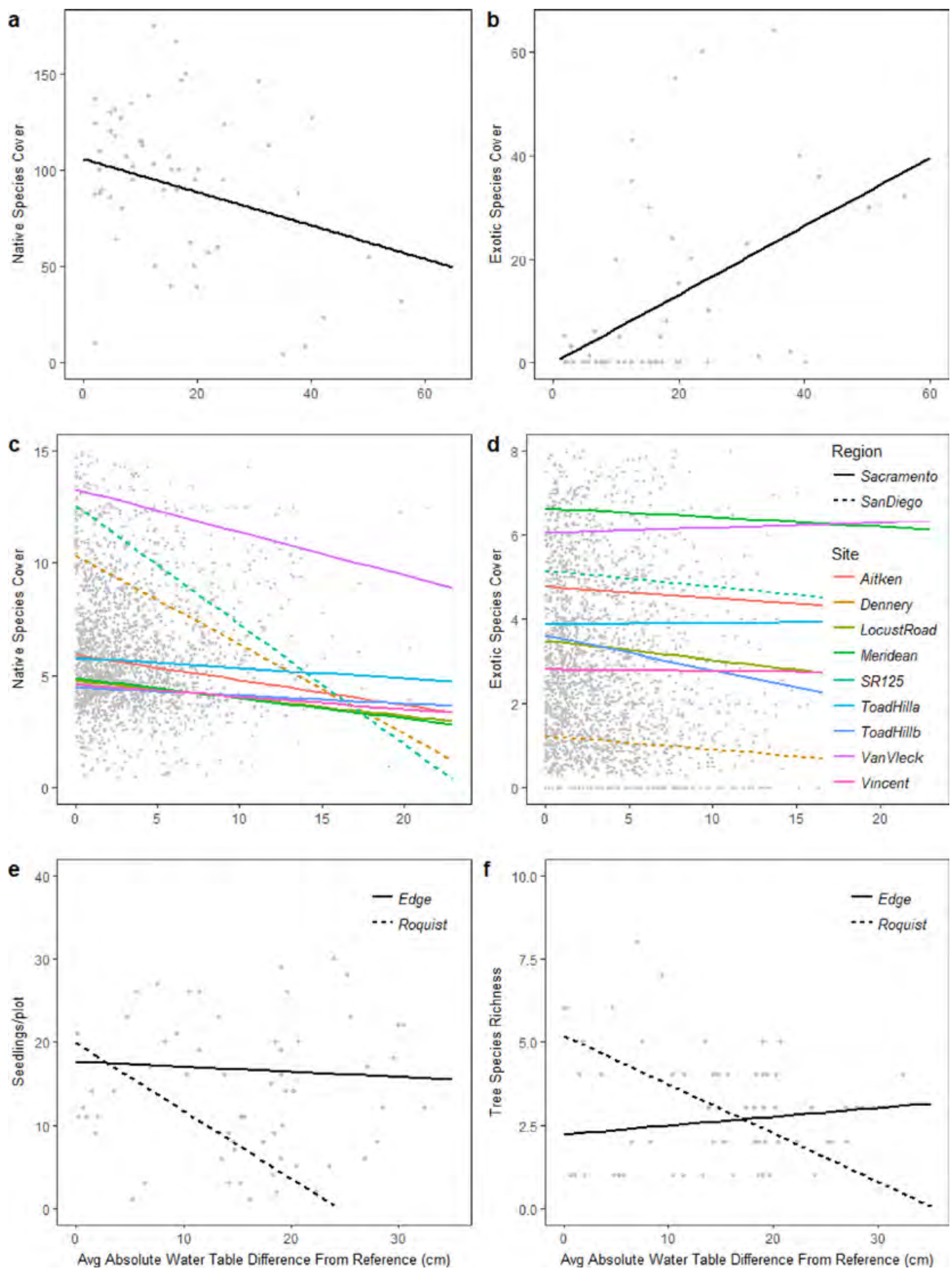

Fig. 4. Average weekly absolute water table difference between restored and reference wetlands and associated vegetation metrics for fens and wet meadows in Colorado (a, b), vernal pools in California (c, d) from the San Diego region (dashed lines) and Sacramento (solid lines), and mid-Atlantic forested wetlands (e, f). Trendlines for each site are created from the median estimates of β_0 and β_1 from the Bayesian models. Restored fen and wet meadows had higher native species cover (a) and lower exotic species cover (b) when water tables were similar to reference sites. As restored hydrologic conditions in fens and wet meadows became less similar to reference sites, native species cover decreased (a) and exotic species cover increased (b). Native species cover in three of the eight vernal pool projects was greatest when inundation depths in restored pools were similar to reference pools (c), and decreased as restored vernal pool hydrologic conditions became less similar to reference pools. Vernal pool exotic species cover (d) was independent of the hydrologic similarity between the restored and reference pools in all eight projects. Forest seedling counts (e) and tree species richness (f) was highest when water table levels were similar to reference sites and decreased as water tables differed from reference sites for one of the two forested sites. Seedling counts and tree species richness was not related to hydrologic similarity in the other forested site.

However, the extent of the peat soil degradation from the original filling or draining remains unknown, though can result in long term hydrologic changes due to the reduction in hydraulic conductivity in degraded peat soils (van Seters and Price, 2002). Any change in hydraulic conductivity can influence water holding capacity and limit colonization of the restored area by peat forming species.

Restored vernal pools in California had greater inundation depths than reference pools for their first six years and became hydrologically similar to reference pools after eight years. This finding is similar to other research that showed a hydrologic development period of 9 years

Table 3
Bayesian estimates for the relationship between vegetation metrics and hydrologic similarity between restored and reference wetlands. Vegetation metrics include native and exotic species cover in fens, wet meadows, and vernal pools, and tree seedlings/plot and tree species richness in forested wetlands. The Bayesian median estimate for each β coefficient and the 95% credible interval for the β_1 coefficient is provided, along with the probability that β_1 is less than zero. B_0 is an estimate of the intercept in the model, indicating the vegetation value when restored water levels match reference water levels. β_1 estimates the slope of the regression, indicating the rate of change in each vegetation metric when water levels in restored sites are different than reference sites. β_1 was expected to be negative for native species cover, tree seedling density and tree species richness, and positive for exotic species cover.

Wetland Type	Metric	Site	β_0	β_1	β_1 CI	Pr ($\beta_1 < 0$)
Fens & Wet Meadows	Native Cover	Telluride	105.91	−0.87	−1.63, −0.04	0.98
	Exotic Cover	Telluride	−0.07	0.66	0.34, 0.99	0.00
Vernal Pools	Native Cover	Dennery	10.34	−0.40	−0.57, −0.22	1.00
		SR125	12.55	−0.52	−0.78, −0.29	1.00
		Aitken	5.90	−0.11	−0.27, 0.05	0.92
		Locust Road	4.79	−0.08	−0.28, 0.14	0.76
		Meridean	4.88	−0.09	−0.34, 0.17	0.77
		Toad Hill	5.80	−0.05	−0.25, 0.17	0.68
		Van Vleck	13.29	−0.19	−0.30, −0.07	1.00
		Vincent	4.64	−0.06	−0.24, 0.14	0.72
	Exotic Cover	Dennery	1.24	−0.04	−0.15, 0.08	0.77
		SR125	5.15	−0.04	−0.19, 0.08	0.78
		Aitken	4.78	−0.03	−0.14, 0.08	0.74
		Locust Road	3.50	−0.05	−0.20, 0.06	0.82
		Meridean	6.66	−0.03	−0.17, 0.12	0.68
		Toad Hill	3.88	0.00	−0.11, 0.16	0.52
		Van Vleck	6.07	0.01	−0.07, 0.12	0.43
		Vincent	2.82	−0.01	−0.11, 0.13	0.55
Non-Riverine Forested Mineral Flats	Tree Seedlings	Edge Farm	17.70	−0.06	−0.38, 0.26	0.65
		Roquist	19.91	−0.82	−1.45, −0.19	0.99
	Tree Species Richness	Edge Farm	2.18	0.03	−0.03, 0.08	0.16
		Roquist	5.05	−0.14	−0.24, −0.03	0.99

for newly created vernal pools (Black and Zedler, 1998; Collinge et al., 2013). Newly restored and created pools often have soils that are overly compacted during construction, making a five-year monitoring period insufficient to detect the long-term development of restored hydrologic conditions.

Native species cover in both San Diego vernal pool projects was higher in restored pools with similar water depths to reference pools, although this was not true for all projects from Sacramento. Previous research has shown sensitivity of plant communities to water availability is greater in drier areas (Cleland et al., 2013). Hydrologic similarity to reference vernal pools may thus be more important for native species colonization in the more arid region of San Diego than in Sacramento. There was no relationship between exotic species cover and hydrologic similarity to reference sites in any vernal pool project, highlighting a significant challenge to restoration and conservation of California vernal pools (Gerhardt and Collinge, 2003).

Restored forested wetlands had similar water levels to reference sites within one year following restoration. However, long term hydrologic processes in restored forested wetlands are poorly understood. An increase in evapotranspiration rates as trees grow and leaf area increases may alter groundwater levels (Bruland and Richardson, 2005), although this remains untested in restored forested wetlands. Tree density decreases as restored forests mature, and although the surviving trees have greater leaf area, the decrease in tree density may balance this water use. Water level changes in response to the harvesting of trees in forested wetlands can be minimal (Sun et al., 2001), and the relationship between forest evapotranspiration and wetland water levels may depend more on climate than forest structure (Lu et al., 2009).

Most forested wetland restoration projects with available data did not have vegetation data from the same location as groundwater monitoring wells, making the response of the number of tree seedlings and richness to hydrologic conditions unclear. The number of tree seedlings and tree species richness was highest when restored water levels were similar to reference sites for only one project. Natural tree seedling recruitment is often limited by hydrologic conditions (Johnson, 2000). Although the planting of mature seedlings is meant to overcome the hydrologic limitations of germination and recruitment (Young et al., 2005), long-term forest community regeneration may be limited if restored hydrology does not permit native tree colonization (Battaglia et al., 2002).

Although restored wetlands had higher native species cover when water levels were more similar to reference areas, more research is warranted on whether hydrologic similarity leads to similarity in vegetation community composition in restored wetlands. Vegetation composition following restoration can be notoriously variable (Laughlin et al., 2017), and having a different plant community composition in a restored site than its reference should not mean restoration was a failure. Restored species composition of a site can be the results of legacy site impacts (Flinn and Vellend, 2005), restoration methods (Kiehl et al., 2010), and initial year climatic conditions (Stuble et al., 2017). Caution is warranted in evaluating restoration solely by vegetation, as is the case in many restorations, as many factors can affect vegetation composition through time.

4.1. Creating hydrologic performance standards

Monitoring that indicates whether a project will meet its success criteria within the required monitoring time frame can provide opportunities for adaptive management. A recent analysis of mitigation sites for several wetland types found an increase in the cover of invasive species after the required five year monitoring period had ended (Van den Bosch and Matthews, 2017). Rather than relying only on vegetation performance standards, hydrologic performance standards specific to each project using water level data from local reference sites are thus needed to ensure long term ecological success in mitigation wetlands. The current hydrologic requirement used by the ACE to evaluate mitigation wetlands in the US is an insufficient indicator to determine whether the correct hydrologic regime for most wetland community types has been restored (Johnson et al., 2012) or whether water table depths differ from those proposed in the mitigation plan (Petru et al., 2014). Hydrologic comparisons to local reference wetlands provide a more accurate reflection of the appropriate water levels for a proposed wetland type.

Identifying and using reference sites that encompass the natural range of hydrologic variability for a proposed wetland type provides an excellent basis for designing restoration projects and developing standards to verify restoration success (White and Walker, 1997). However, the availability of, and access to, appropriate reference sites can be a challenge. Reference sites may have unknown disturbance histories (Moorhead, 2013) and because of the wide temporal and spatial variability in wetlands, and significant urban development in many regions, it can be hard to find reference sites that are hydrologically representative of the site to be restored (White and Walker, 1997). Although project-specific reference sites provide a more accurate characterization of local wetlands and are ideal for creating hydrologic performance standards, the use of monitoring data from regional reference sites to characterize wetlands at a regional scale should be evaluated further (Fennessy et al., 2004a,b; Steyer et al., 2003).

5. Conclusions

Wetland types differ in their hydrologic regimes and the time required for hydrologic conditions in restored wetlands to match reference sites. A single hydrologic requirement for all wetland mitigation sites is therefore inadequate in evaluating whether the hydrologic regime for the proposed wetland type has been obtained. In contrast, project specific reference sites provide a meaningful approach for evaluating mitigation outcomes. Native species cover and tree seedling richness in many sites was higher when restored water tables were similar to reference areas. Hydrologic similarity to reference sites did not lead to vegetation success in all cases, however, highlighting the continued need for vegetation performance standards and invasive species management. Hydrologic performance standards should be created using concurrently collected monitoring data from nearby reference sites to account for water level variations due to climate variance within and between years, water level variations across the landscape, and hydrologic differences among wetland types. Our analyses suggest a post restoration analysis timeline of at least 10 years is needed for some wetlands types to quantify long-term hydrologic conditions. Analyzing mitigation outcomes using project specific reference sites and concurrent hydrologic monitoring provides an effective way to gauge long-term restoration success across wetland types.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2018.11.001.

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Supplementary Information

While the data were used to create hydrologic performance standards for each wetland type, it can also be helpful to evaluate and understand the hydrologic regime of each wetland type throughout the year. Example hydrographs are provided below for each wetland type included in the paper.

Fens and Wet Meadows in Telluride, CO

Figure S1. Wetland water table depth from the ground surface from May through October in Fens (top panels) and Wet Meadows (bottom panels) for both reference (left panels) and restored (right panels) sites.

California Vernal Pools

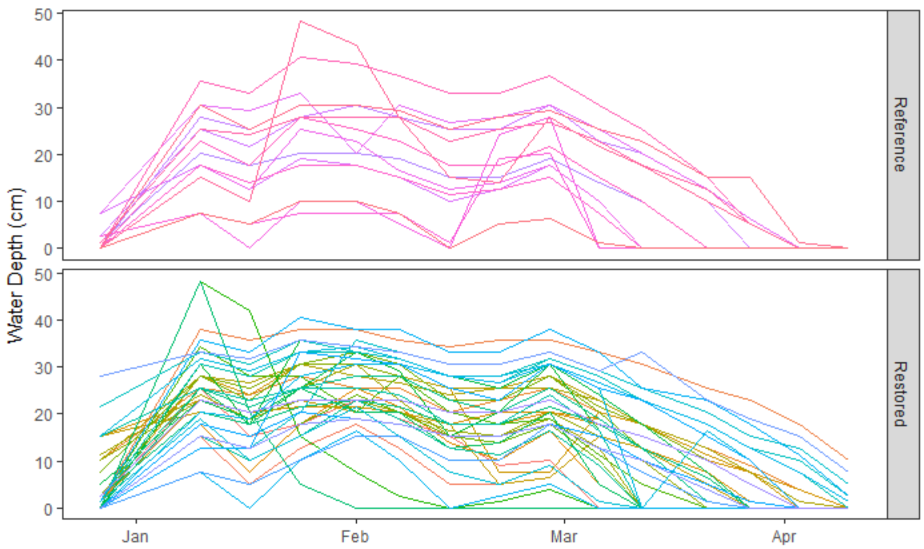

Figure S2. Vernal pool inundation depths above the ground surface from January through May for both reference (top panel) and restored (bottom panel) sites.

Southeastern Forested Mineral Flats

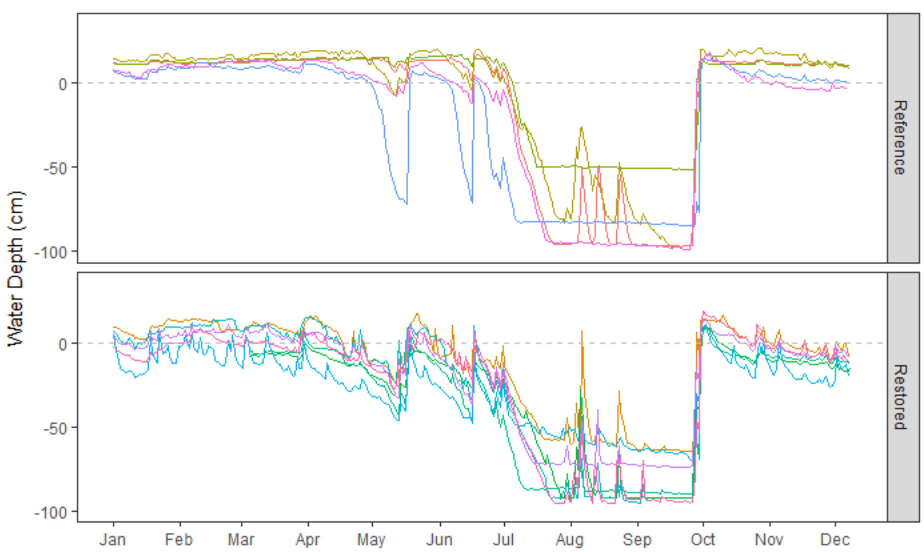

Figure S3. Forested wetland water table depth from the ground surface from January through December in reference (top panel) and restored (bottom panel) sites.