Restoration Notes

Restoration Notes have been a distinguishing feature of *Ecological Restoration* for more than 25 years. This section is geared toward introducing innovative research, tools, technologies, programs, and ideas, as well as providing short-term research results and updates on ongoing efforts. Please direct submissions and inquiries to the editorial staff (ERjournal@sebs.rutgers.edu).

Restoration of Bedrock Vernal Pools for the Threatened Vernal Pool Fairy Shrimp in Central California

Jeff A. Alvarez (corresponding author: The Wildlife Project, P.O. Box 188888, Sacramento, CA 95818, Jeff@thewildlifeproject.com), Colton Rogers (Contra Costa Water District, Concord, CA) and Mary A. Shea (Contra Costa Water District, Concord, CA)

Vernal pool creation and restoration has been an active industry in California for more than 40 years (Black and Zedler 1998, DeWeese 1998). The success of these creation and restoration activities has been less than consistent, with results that may not meet the goals of the restoration activity (Calhoun et al. 2014, Schlatter et al. 2016). For example, creation and restoration projects must mimic surrounding conditions, and more importantly, meet the natural history parameters of the target species for which the activity was designed (Eng et al. 1990, Vanschoenwinkel et al. 2009, Wortley et al. 2013).

Fairy shrimp, which often or exclusively occur in vernal pools, are a taxon for which many species are in steep decline, particularly in California (Eriksen and Belk 1999). Successful restoration of vernal pools is dependent on the target of the restoration (Bakker et al. 2000). We engaged in a restoration project that included the removal of sediment, topsoil and associated vegetation from historic vernal pools that lay within rock outcrops in a vernal pool complex. The goal of our restoration activity was to increase the number of pools and the volume of wetted vernal pool habitat for potential colonization by the federally threatened *Branchinecta lynchi* (vernal pool fairy shrimp). We report herein on the methods and results of those activities.

We surveyed 29 vernal pools in the Kellogg Creek Vernal Pool Complex from 1998 to 2016 to determine the annual presence of *B. lynchi* and other sympatric species. The site was comprised of sandstone bluffs in the upper Kellogg Creek watershed, Contra Costa County, California. These bluffs included various depressions in the rock that were likely formed by weathering (Way 1978) and ranged from 0.1–1.1 m in depth and 0.5–195 m² in surface area. Many pools were completely open and devoid of vegetation; however, approximately 50% of the pools included some level of soil intrusion that supported non-native annual grasses (Figure 1).

Following winter rains and pool inundation—typically late December to early January—surveys at these 29 pools were conducted bimonthly. Pools were considered inundated when they were > 3 cm deep (maximum depth; following USFWS 2017). Each survey included a single biologist conducting visual encounter surveys (i.e., walking or viewing the water column for two minutes per 2 m² of estimated surface area, while an additional biologist took two sweeps with a fine mesh net per 1 m² of approximate surface area. Surveys were conducted twice monthly until fairy shrimp were detected or pools dried. Additionally, physical characteristics of each pool (e.g., depth, surface area, temperature, etc.) were also collected (Table 1). Fairy shrimp samples collected were assessed based on visual estimates of animals captured and resulted in a determination of 1s (≤ 10 animals present); 10s (10–100 animals); 100s (100–1000); or 1000s (≥ 1000 animals present) of shrimp present per pool. Due to a history of presence of *Linderiella occidentalis* (California fairy shrimp), and adjacent populations of *B. longianntenae* (long-horned fairy shrimp) and *B. mackini* (alkali fairy shrimp), voucher specimens were collected each year to confirm the identity of species.

In November 2015 we visually assessed the vernal pool complex and determined that at least 12 sites appeared to have historic vernal pools that were entirely filled by soil and non-native annual grasses (Figure 1). We elected to restore these vernal pools by removing excess soil and allowing the sites to naturally hydrate from seasonal rains. These new pools would then be added to the annual monitoring efforts of the existing 29-pool complex.

Beginning 22 Dec 2015 and continuing until 15 Jan 2016, we hand-excavated each historic (i.e., potential) vernal pool until all soils were removed (Figures 1, 2 and 3). We removed soil and vegetation by hand shoveling and placed it in a wheelbarrow, which was then moved to an off-site

doi:10.3368/er.42.3.169
Ecological Restoration Vol. 42, No. 3, 2024
ISSN 1522-4740 E-ISSN 1543-4079

Figure 1. Initial soils removal in a large bedrock vernal pool (#NP-10), using hand shovel, Contra Costa County, CA, December 2015. *Image credit Jeff Alvarez.*

location where it was deposited away from vernal pools (i.e., no chance of refilling the vernal pool). A fine layer (approximately 0.5–1 cm) of soil was left on the bottom of each pool as a result of using only shovels to remove soil. No pools were inoculated with soils from other pools, and all pools were left to inundate naturally from seasonal rain. Following restoration and inundation, each pool was monitored for vernal pool invertebrates.

Surveys conducted between 2013–2020 indicated that among the 29 original pools, there was a high level of interannual variability in the percentage of rock pools inundated (range: 39–95%). Among those same pools, and during the same time period, the percentage of pools that had *B. lynchi* present ranged from 13–69%, also indicating a high level of interannual variability in fairy shrimp presence.

Our work included restoration of 12 additional pools in 2015, accounting for an increase of approximately 72 m^2 of rock pool habitat. Newly excavated pools ranged from 1–30 m^2 and ranged in depth from 0.1–0.95 m. Following restoration and seasonal rain events, every pool excavated showed signs of inundation (25.4 cm [10 in.] the first year following restoration, with a single pool becoming inundated two years following restoration, and a single pool becoming inundated only after four years following restoration activities). Among the 12 pools that were restored, nine were observed having *B. lynchi* (i.e., 1s to 100s of individuals). Two years following the restoration of pools, an additional pool included *B. lynchi*. A single 2 m^2 restored rock pool (RP-09) was reported to have *L. occidentalis* in 2018, a species not seen in this vernal pool complex for the previous 20 years (Contra Costa Water District, unpublished data). In addition to fairy shrimp, the natural and restored pools included a variety of vernal pool invertebrates (e.g., representatives of orders Ostracada, Copepoda, Cladocera, Hemiptera, Coleoptera and Tricladida) as well as *Pseudacris regilla* (Pacific chorus frog) (Table 1).

We found that careful investigation can reveal cryptic, historic vernal pool depressions that may be obscured by sediments eroded from upland soils surrounding the pool. In our work, some of these potential vernal pools were overgrown with non-native annual grasses and required substantial excavation (Figures 1, 2, and 3.). Unearthing potential vernal pools in this complex significantly

Table 1. Physical characteristics and species observed in 41 pools (original and restored) at the Kellogg Creek Vernal Pool Complex, Contra Costa County, CA, 1998–2015. ORP = Original Rock Pool, RP = Restored Pool. Each physical characteristic is measured in cm and includes a standard deviation (SD). Syntopic species observed is a cumulative list from 1998 to 2016. Fairy shrimp species observed: BL= *Brachinecta lynchi* (vernal pool fairy shrimp), LO = *Linderiella occidentalis* (California fairy shrimp). Other taxa: CH = Chironomidae, CL = Cladocera, CO = Coleoptera, CP = Copepoda, CU = Culicidae, HE = Hemiptera, OS = Ostracoda, PR = *Pseudachris regilla* (Pacific chorus frog), TR = Tricladida.

Pool Number	Average Maximum Length (cm)	Average Maximum Width (cm)	Average Maximum Depth (cm)	Syntopic Species Observed
ORP-1	139 (1.3)	181 (1.6)	8.2 (0.8)	BL, CO, OS, PR
ORP-2	46 (1.4)	40 (1.6)	3.5 (0.1)	BL, CO, OS, PR
ORP-3	88 (1.3)	255 (1.1)	3.4 (0.8)	BL
ORP-4	30 (1.8)	61 (2.4)	3.1 (0.8)	CP, OS
ORP-5	187 (9.8)	58 (9.7)	3.2 (0.1)	CH, CP
ORP-6	105 (1.0)	95 (6.6)	< 3.0 (0.9)	OS
ORP-7	89 (4.2)	41 (5.2)	7.9 (0.9)	BL, CL, CP, HE, OS
ORP-8	210 (4.0)	116 (3.9)	7.6 (2.2)	BL, CP
ORP-9	495 (23.2)	218 (13.5)	23.9 (4.3)	BL, CL, CO, OS, PR, TR
ORP-10	133 (9.6)	79 (4.7)	15.0 (2.5)	BL, CL, CO, LO, OS
ORP-11	99 (9.8)	68 (5.3)	15.1 (6.8)	BL
ORP-12	1809 (120.1)	840 (94.4)	4.9 (0.1)	BL, CL, CO, CP, HE, OS, TR
ORP-13	701 (71.8)	182 (12.9)	24.8 (4.4)	BL, CO, HE, OS, TR
ORP-14	166 (6.7)	222 (4.1)	9.9 (1.2)	BL, CL, OS, OS
ORP-15	114 (2.1)	36 (1.4)	< 3.0 (0.4)	OS
ORP-16	341 (10.7)	398 (12.7)	17.1 (0.7)	BL, CL, CO, HE, OS, PR, TR
ORP-17	192 (1.8)	268 (3.0)	9.6 (1.4)	BL, LO
ORP-18	155 (2.2)	94 (1.9)	8.1 (1.6)	BL, CL, HE
ORP-19	140 (5.3)	81 (5.8)	7.9 (0.9)	BL
ORP-20	143 (1.4)	82 (0.7)	9.4 (2.0)	BL
ORP-21	217 (10.1)	197 (8.5)	4.2 (0.4)	CL, CO OS, TR
ORP-22	101 (1.9)	193 (2.0)	< 3.0 (4.8)	CO, OS
ORP-23	47 (1.3)	202 (1.6)	< 3.0 (3.8)	CO, OS
ORP-24	110 (1.8)	309 (18.8)	11.1 (5.0)	BL, CL, LO, OS
ORP-25	147 (2.5)	59 (2.1)	7.1 (5.8)	BL, OS
ORP-26	149 (8.1)	81 (7.6)	7.5 (4.4)	BL, CH, OS
ORP-27	328 (25.6)	108 (8.8)	7.0 (5.1)	BL, CO, HE. LO
ORP-28	281 (18.6)	59 (3.2)	8.8 (4.7)	BL, CO, HE
ORP-29	96 (1.7)	158 (2.1)	< 3.0 (4.2)	CP, OS
RP-1	202 (9.0)	81 (2.9)	5.9 (1.7)	BL, CH, CL, HE
RP-2	202 (39.1)	214 (37.8)	< 3.0 (0.8)	CP, OS
RP-3	333 (9.6)	65 (1.9)	6.6 (2.2)	BL, CH
RP-4	351 (7.3)	47 (4.2)	6.1 (2.1)	BL
RP-5	185 (6.7)	223 (4.9)	< 3.0 (4.4)	BL, CH
RP-6	225 (9.2)	284 (7.2)	< 3.0 (1.0)	CP, OS
RP-7	148 (1.4)	218 (8.0)	15.8 (3.8)	BL
RP-8	341 (1.8)	150 (1.7)	29.7 (7.8)	BL, CL, OS TR
RP-9	77 (2.9)	244 (5.2)	8.6 (3.2)	BL, CL, CU, HE, LO, OS, TR
RP-10	99 (1.6)	92 (1.7)	34.7 (9.0)	BL, CL, CO, TR, OS
RP-11	105 (1.1)	210 (6.6)	11.0 (6.0)	BL, CO, OS
RP-12	27 (2.7)	47 (5.6)	9.9 (1.9)	BL, CO, TR

Figure 2. Continued hand removal of soils for a large vernal bedrock vernal pool (#NP-10), Contra Costa County, CA, December 2015. *Image credit Jeff Alvarez.*

Figure 3. Six weeks following complete removal of all soils, this (#NP-10) and all other restored vernal pools were inundated by rainfall, and fairy shrimp were noted swimming within the pool, Contra Costa County, CA, January 2016. *Image credit Jeff Alvarez.*

increased vernal pool habitat: the restoration activities increased the surface area of vernal pools in this complex by 34% and a maximum number of pools occupied by fairy shrimp by 11 (55% increase). We speculated that many of the pools were colonized by wind-blown cysts, which is common for *B. lynchi* (Ericksen and Belk 1999), or by surface runoff between pools.

Kistner et al. (1995) suggested that documenting a population's viability was preferred over species presence as a method to determine successful creation or restoration. At this vernal pool complex, we noted a strong tendency toward interannual variability in the number of pools inundated and the number of pools supporting fairy shrimp (Eriksen and Belk 1999). This population has persisted since prior to the initiation of systematic annual surveys in 1998. Additionally, *B. lynchi* has been present annually within up to nine of the restored pools, and 10 of 12 total restored pools since their restoration.

We defined success for the proposed project as 1) an increase in inundated area for rock outcrop vernal pools within the complex, and 2) colonization of one or more pools by *B. lynchi*. Establishing target success criteria can guide or direct the methods and monitoring required to meet those targets (Bakker et al. 2000). Therefore, our project was successful.

In our case, although the amount of human labor to move an estimated 35 m^3 of soil was high (approximately 66 person-hours), follow-up surveys require only a single permitted biologist to check pools at intervals that will increase the likelihood of detection (generally bimonthly, following inundation). We considered the time expended (1.1 hrs per m^2 of surface added) minimal when compared to the number of colonized restored pools ($N = 10$), and continued persistence of the species. The total cost of restoration of 12 pools was approximately $7,200 (in 2015) for both labor and tools.

We made no plan to inoculate pools with soils from occupied sites. Rather, we elected to allow for natural cyst dispersal through wind, intra-complex pool spillover, or from passive recruitment from within the existing soils that remained. We speculated that some cysts were likely banked in much of the upland areas surrounding the vernal pools. Furthermore, disturbing soils during excavation would likely leave some number of cysts within the restored pool depression following excavation. In this effort, 10 pools were colonized by one or more species of fairy shrimp, nine pools within the first wet season. Based on these results, we suggest that inoculation from pools that are within a complex is an unnecessary disturbance to existing sites.

Although the success rate related to the creation of vernal pools has been reported as variable, and even controversial (Sutter and Francisco 1996, Calhoun et al. 2014, Schlatter et al. 2016), we feel that restoration of rock outcrop pool habitat, particularly within a complex of similar habitat, may be valuable. We do acknowledge that monitoring and vigilant management (e.g., continued removal of soil or installation of erosion control measures) may be required to sustain these pools.

We also note that at this site, 14 of the original 29 pools that were monitored had soil encroachment that occupied as much as 50% of the pool area. We suggest a plan be developed for soil excavation in these pools. Based on our successful restoration efforts reported here, soil removal may increase pool hydroperiod and improve conditions for some longer-lived species (Brendonck 1996, Eriksen and Belk 1999, Philippi et al. 2001). Maintaining the greatest pool volume and the largest number of pools in a vernal pool complex will likely enhance habitat over time and should be considered when management decisions are made for maintaining a site similar to ours (Brooks and Hayashi 2002, Dittes et al. 2007).

Acknowledgments

We are grateful to the Contra Costa Water District for supporting this project from conception to implementation and monitoring. Hand excavation of vernal pools was aided by C. Davis, C. Funk, J. Howard, and S. Foster. Assistance with field data collection was aided by C. McClain, R. Lee, E. Pimentel, and M. Wacker. The US Fish and Wildlife Service provided individual (TE-027427) and entity Recovery Permits (TE-797267-20.1), and a Biological Opinion (08ESMF00-2012-F-0646), under which this work was conducted.

References

Bakker, J.P., A.P. Grootjans, M. Hermy and P. Poschlod. 2000. How to define targets for ecological restoration?—Introduction. *Applied Vegetation Science* 3(1):3–6.

Black, C. and P.H. Zedler. 1998. An overview of 15 years of vernal pool restoration and construction activities in San Diego County. Pages 195–205 in C.W. Witham, E.T. Bauder, D. Belk, W.R. Ferren Jr. and R. Ornduff (eds.), *Ecology, Conservation, and Management of Vernal Pool Ecosystems—Proceedings from a 1996 Conference*. Sacramento, CA: California Native Plant Society.

Brendonck, L. 1996. Diapause, quiescence, hatching requirements: What we can learn from large freshwater branchiopods (Crustacea: Branchiopoda: Anostraca, Notostraca, Conchostraca). *Hydrobiologia* 320:85–97.

Brooks, R.T. and M. Hayashi. 2002. Depth-area-volume and hydroperiod relationships of ephemeral (vernal) forest pools in southern New England. *Wetlands* 22:247–255.

Calhoun, A.J.K., J. Arrigoni, R.P. Brooks, M.L. Hunter and S.C. Richter. 2014. Creating successful vernal pools: A literature review and advice for practitioners. *Wetlands* 34:1027–1038.

DeWeese, J.M. 1998. Vernal pool construction monitoring methods and habitat replacement evaluation. Pages 217–223 in C.W. Witham, E.T. Bauder, D. Belk, W.R. Ferren Jr. and R. Ornduff (eds.), *Ecology, Conservation, and Management of Vernal Pool Ecosystems—Proceedings from a 1996 Conference*. Sacramento, CA: California Native Plant Society.

Dittes, J.C., J.L. Guardino and R.A. Radmacher. 2007. A GIS-based vernal wetland acre/density index (VWADI) for classification and conservation of vernal/annual grasslands landscapes in California's Great Central Valley. Pages 125–141 in Schlising, R.A. and D.G. Alexander (eds.), *Vernal Pool Landscapes. Studies from the Herbarium 14*. Chico, CA: California State University.

Eng, L.L., D. Belk and C.H. Eriksen, 1990. Californian Anostraca: distribution, habitat, and status. *Journal of Crustacean Biology* 10:247–277.

Ericksen, C. and D. Belk. 1999. *Fairy Shrimps of California's Puddles, Pools and Playas*. Eureka, CA: Mad River Press.

Kistner, D.H., D.G. Alexander and H.R. Jacobson. 1995. Vernal pool creation and restoration in California: A study plan for evaluating attainable quality and functional performance. Environmental Protection Agency Research Laboratory Washington, D.C., EPA/600/R-95/073.

Philippi, T.E., M.A. Simovich, E.T. Bauder and J.A. Moorad. 2001. Habitat ephemerality and hatching fractions of a diapausing Anostracan (Crustacea: Brachiopoda). *Israel Journal of Zoology* 47(4):387–395.

Schlatter, K.J., A.M. Faist and S.K. Collinge. 2016. Using performance standards to guide vernal pool restoration and adaptive management. *Restoration Ecology* 24:145–152.

Sutter, G. and R. Francisco. 1996. Vernal pool creation in the Sacramento Valley: A review of the issues surrounding its role as a conservation tool. Pages 206–216 in C.W. Witham, E.T. Bauder, D. Belk, W.R. Ferren, Jr. and R. Ornduff (eds.), *Ecology, Conservation, and Management of Vernal Pool Ecosystems—Proceedings from a 1996 Conference*. Sacramento, CA: California Native Plant Society.

United States Fish and Wildlife Service (USFWS). 2017. Survey guidelines for listed large branchiopods. Pacific Southwest Region, Sacramento, CA. 24 pp.

Vanschoenwinkel, B., A. Hulsmans, E. DeRoeck, C. De Vries, M. Seaman and L. Brendonck, 2009. Community structure in

temporary freshwater pools: Disentangling the effects of habitat size and hydroregime. *Freshwater Biology* 54:1487–1500.
Way, D.S. 1978. *Terrain Analysis: A Guide to Site Selection Using Aerial Photograph Interpretation*. Stroudsburg, PA: McGraw-Hill Publishing.
Wortley, L., J. Hero and M. Howes. 2013. Evaluating ecological restoration success: A review of the literature. *Restoration Ecology* 21:537–543.

The Constancy Richness Index: A New Plant Community Metric with Conservation Implications

Craig Annen (corresponding author: 308 North Nine Mound Road, Verona, WI 53593-1036, Annen00@aol.com, Craigannen556@gmail.com) and Erin Green (Integrated Restorations LLC, Verona, WI)

Species richness (S) (McIntosh 1967) is a community metric commonly used in conservation and ecological restoration to inform management decisions and prioritize resources. Also called alpha diversity (S_α) by Whittaker (1965), species richness is defined as the number of taxonomically distinct species at a local scale, typically at a single location that is under consideration for ecological study, management, or conservation. When more than one location is under consideration, gamma diversity (S_Υ, also called pooled species richness) is employed. Gamma diversity is defined as the cumulative species richness among multiple locations across a landscape (Whittaker 1965). A third related concept, beta diversity (S_β, sometimes called proportional species turnover) describes the relationship between alpha and gamma diversity. Beta diversity estimates the degree of overlap in species composition between two locations, yet available metrics for estimating beta diversity are limited to pairwise site comparisons. In restoration ecology we must often consider ecological patterns and management priorities across multiple locations in a given landscape. To address this need, Hui and McGeoch (2014) introduced the concept of zeta diversity (S_ζ), defining it as the compositional heterogeneity among three or more vegetation communities or sampling areas. The zeta diversity parameter algebraically extends the concept of gamma diversity to more than two locations. Zeta diversity is estimated by comparing species overlap among arrays of sampling locations with a matrix of similarity estimates calculated from detailed plant abundance data (e.g., stem density estimated from quadrat samples). While this approach has outstanding merits for detailed scientific investigations with large data sets, application of the zeta matrix-overlap technique to measure zeta diversity has some drawbacks when applied to ecological restoration and management: the method is a computationally intensive, multi-step algorithm that requires detailed abundance data inputs, and the output values that it produces are not immediately intuitive to many restoration ecologists and land managers.

doi:10.3368/er.42.3.174
Ecological Restoration Vol. 42, No. 3, 2024
ISSN 1522-4740 E-ISSN 1543-4079

Plant species are not uniformly distributed across modern landscapes. Ejnæs et al. (2018) introduced the concept of uniquity and developed a metric to quantify uniqueness of species to localized areas across a variety of spatial scales. Their metric calculates species' uniqueness among multiple locations as a probability generated by comparing individual species overlap with computer-generated reference sites constructed by extracting prevalent species from pooled species richness data. However, this metric was designed for use at large spatial scales (regional areas or larger), requires plant community mapping to correct for sampling bias, and its outputs are sensitive to sampling effort and even less ecologically intuitive than the zeta array. Stohlgren et al. (2005) presented a more straightforward and easier to interpret uniquity index based on proportional frequencies of species sampled in nested Whittaker plots, yet this method requires more intensive sampling effort than collecting species presence-absence data.

We developed an alternative metric (the Constancy Richness Index [CRI]) to estimate uniquity among multiple sites at the species richness scale when detailed abundance data are not readily available. The CRI is formulated as: $\alpha_1/(\alpha_2 + \alpha_3 + \ldots \alpha_n)$, where α_1 is the number of species unique to a single location, α_2 is the number of species shared between two locations, α_3 is the number of species shared among three locations, and α_n is the number of species shared among *n* locations, *n* being the total number of locations where species richness data have been collected.

The CRI offers some practical advantages over the zeta array and associated uniquity metrics: it is computationally simple and straightforward, requiring only species presence data (as opposed to abundance data that can be expensive and time-consuming to collect), and it yields results that are clear and easy to conceptualize. It is an index, so it is independent of scale enlargement, and it is not based on any underlying assumptions about plant community structure and assembly. Furthermore, the CRI is calculated from pooled species richness measured from real-world plant communities rather than comparisons with probabilistically contrived reference communities (as in Ejnæs et al. 2018). Previous authors (reviewed in Phillips 1959) used the term segment-frequency to describe this concept, but Phillips (1959) recommended adopting the term species constancy. Values for the CRI range from zero to pooled species richness. When CRI = 1.0, half