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FEATURE

# Depressional wetland classification and ecosystem service predictive models for the Integrative Landscape Modeling partnership

Allison J. Thompson, Scott T. McMurry, and Loren M. Smith

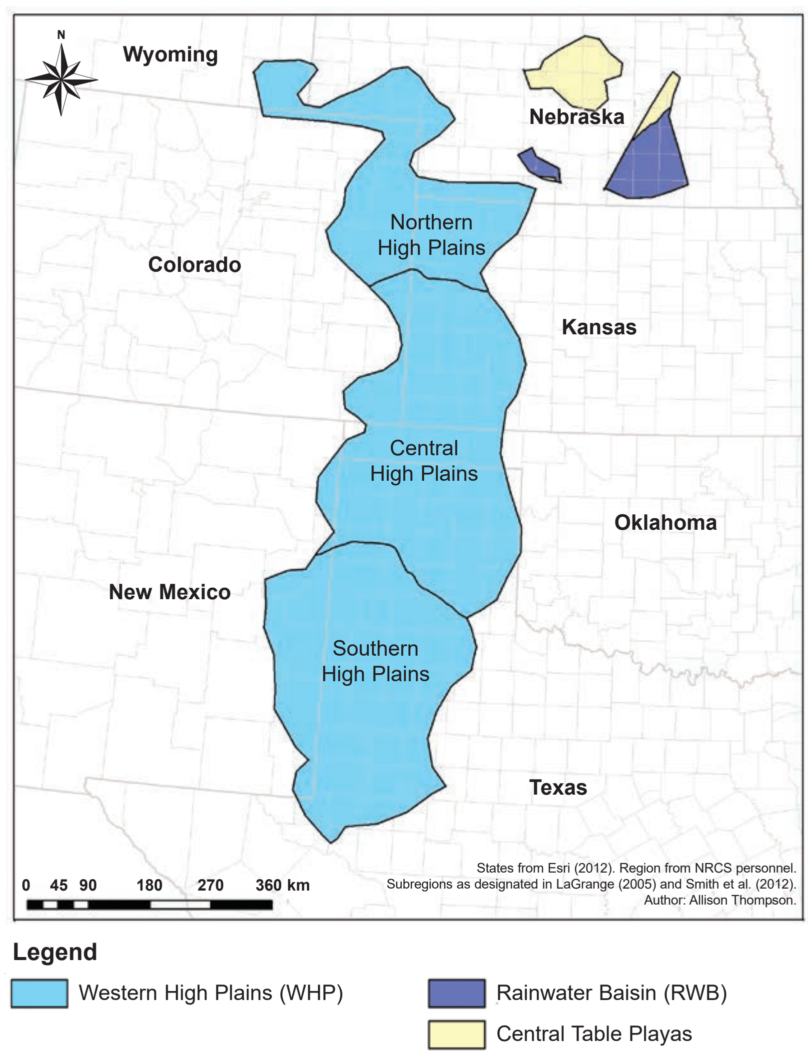
Depressional wetlands in the Great Plains are experiencing watershed alterations that impact the provisioning of important ecosystem services, which are unmonitored on a large scale (Smith et al. 2011). The Natural Resources Conservation Service's (NRCS) Conservation Effects Assessment Project (CEAP) determines the effects of conservation programs on ecosystem service provisioning across the nation (Tomer et al. 2014). CEAP–Wetlands has focused on the effects of conservation programs and land use on ecosystem service provisioning in areas dominated by different wetland types. The High Plains region (HPR) primarily contains playa wetlands. The dominant land uses are native grassland, cropland, and conservation program land (Brinson and Eckels 2011). Predictive models were developed from CEAP–Wetlands data and can be used to estimate services by identifying playa wetlands and measuring variables required to populate the models (Duriancik et al. 2008). We developed a sampling manual that can be used to remotely classify playa function and ecosystem service provisioning. The manual and models can be used to monitor ecosystem services and inform USDA conservation decisions.

Playas are depressional recharge wetlands that exist within western portions of Texas, Oklahoma, and Kansas, western and central Nebraska, the eastern plains of Colorado and New Mexico, and southeast Wyoming (Smith 2003) (figure 1). Playas exist within closed watersheds and receive water only through precipitation and overland flow (Bolen et al. 1989). Playa hydrology is similar among basins, allowing application of CEAP–Wetlands models to estimate service provisioning (Smith et al. 2015). Playas provide ecosystem services including habitat provisioning, floodwater storage, and carbon (C) sequestration (Smith et al. 2011). When cultivation occurs in the watershed of a playa, overland water flow carries sediments causing them to fill a basin, which diminishes wetland function and potentially eliminates the wetland entirely (Tsai et al. 2007). Indeed, playa losses of 60% or greater are due to sedimentation, drainage, and agricultural conversion (Johnson et al. 2012).

Classification of Wetlands and Deepwater Habitats (Cowardin et al. 1979) has been commonly used in the United States to identify wetlands and waterbodies based on abiotic and biotic factors within a hierarchy consisting of

**Figure 1**

The High Plains region as determined by the Conservation Effects Assessment Project Wetlands component (CEAP–Wetlands). Subregions are shown as designated by LaGrange (2005) and Smith et al. (2012).

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**Allison J. Thompson** is a master's research assistant in the Department of Integrative Biology, Oklahoma State University, Stillwater, Oklahoma. **Scott T. McMurry** and **Loren M. Smith** are professors in the Department of Integrative Biology, Oklahoma State University, Stillwater, Oklahoma.

systems, subsystems, classes, subclasses, and modifiers (USGS FGDC 2013). Cowardin et al. (1979) is used within the National Resources Inventory and National Wetlands Inventory (NWI) to monitor gains and losses of wetlands nationwide (USFWS 2011; USDA 2018). Although Cowardin et al. (1979) can describe the presence and habitat characteristics of wetlands, playas often become grouped with functionally different waterbody types within the palustrine class (USDA NRCS 2008).

Hydrogeomorphic (HGM) classification alternatively uses three abiotic characteristics including geomorphic setting, hydrodynamics, and water source to classify wetlands (Brinson 1993). Classes are based on geomorphic setting with hydrodynamics and water sources identified within each class (Smith et al. 1995). Thus, various functions of a wetland can be inferred since geomorphic setting is related to function, which can be used to infer ecosystem service provisioning (Brinson 1993).

We developed a tool that natural resource managers can use to functionally classify playa wetlands and estimate ecosystem service provisioning according to wetland and watershed characteristics. First, we developed an HGM classification key for wetlands in the HPR that can identify playas. Second, we compiled preexisting CEAP–Wetlands predictive ecosystem service models and ranked them by ease of application for a user. Third, we used the HGM key and ecosystem service models to develop an instructional manual for the Integrative Landscape Modeling (ILM) partnership (Mushet and Scherff 2016). Any federal, state, or nongovernment natural resource manager or researcher could use the manual, which is accessible through the CEAP–Wetlands web page (https://www.nrcs.usda.gov/wps/portal/nrcs/detail/national/technical/nra/ceap/na/?cid=nrcs143_014155). This provides a usable guide to estimate ecosystem service provisioning within wetland basins and ultimately within the region. Estimates could indicate how ecosystem services would be affected by different land uses and USDA conservation programs and be used to identify optimal locations for conservation programs on a cost-effective basis.

## THE HIGH PLAINS REGION

The HPR ranges from semiarid in the west to a more mesic climate with annual precipitation amounts ranging from 38 to 63 cm (15 to 25 in), respectively (Smith 2003). Playa hydrology is driven by precipitation events causing variable inundation (Bolen et al. 1989). Most common land use types include native grassland, cropland, and conservation program lands. Conservation programs in this region are the Conservation Reserve Program (CRP), administered by the USDA Farm Service Agency, and the Wetlands Reserve Program (WRP), now carried out as the Wetland Reserve Easement (WRE), which is within the Agricultural Conservation Easement Program, administered by NRCS (Smith et al. 2011). CRP focuses on uplands by taking highly erodible croplands out of production and replacing it with perennial vegetation (USDA NRCS 2014). WRP, now WRE, focuses on restoring wetland condition and conserving wetland basins (USDA FSA 2016). Although CRP does not directly focus on wetland conservation, program activities in the upland have significant impacts on depressional wetlands (Smith et al. 2011).

The HPR is often separated into the Western High Plains (WHP) and the Rainwater Basin (RWB) subregions (Smith 2003) (figure 1). Differences in topography, land use, conservation programs, and climate cause ecosystem services to vary among subregions (Smith 2003). The WHP is often separated into the Northern High Plains (NHP), Central High Plains (CHP), and Southern High Plains (SHP) (figure 1). CRP is the most commonly applied federal conservation program in this subregion (Smith et al. 2011). The RWB receives more rainfall than the WHP, and the landscape consists of gently rolling plains (LaGrange et al. 2011). WRP/WRE is more commonly applied in the RWB.

## A HYDROGEOMORPHIC KEY

We developed an HGM key for the HPR by randomly selecting 200 palustrine wetland polygons from the NWI and building HGM wetland categories (USFWS 2018). Each waterbody was observed individually using Esri ArcMap 10.4 (Esri 2011). We noted geomorphic setting, water source, and likely hydrodynamics based on Esri topographic maps from the US Geological Survey (USGS), Landsat 8 satellite imagery, and the National Hydrography Dataset stream lines (Esri 2017a, 2017b; USGS 2017a). Topography was used to identify geomorphic setting, placing wetlands within closed watersheds into the depressional class and wetlands associated with streambeds into the riverine class (Smith et al. 1995). Using these data, we developed subclasses according to likely formation processes that could be detected, taking note of constructed dikes and excavations. Nine wetlands were not detectable using our data and considered lost since the original mapping date or misclassified due to NWI errors when identifying wetlands via aerial imagery. Replacement polygons were then selected.

Of the 200 palustrine wetlands, 118 (59%) were identified as depressional class with 101 playas and 17 wetlands across three other subclasses (table 1). These other subclasses included the naturally formed "draw," which held water within a drainage not associated with a stream, and the mechanically formed "diked" and "excavated," which held water due to a constructed dike or excavation, respectively. The riverine class contained 81 (40.5%) wetlands within five subclasses. Subclasses included "floodplain," which sat adjacent to the stream; "oxbow," which was a disconnected portion of the stream; and "streambed," where water ponded during low flow, as well as artificially formed "diked" and "excavated." One lacustrine fringe waterbody was identified and associated with a reservoir edge (Smith et al. 1995).

Although depressional types were most common, these were not the only wetlands identified under Cowardin et al. (1979) as palustrine. The HGM types we encountered could not be consistently distinguished using Cowardin labels. Approximately 80% of the identified HGM depressional wetlands shared the same Cowardin label with one or more HGM riverine wetlands. This illustrates the geomorphic differences possible within palustrine waterbody types. Differences in function could be inferred using remotely sensed data, and the HGM rules allowed for rapid labeling by HGM class.

The HGM key that we developed included each HGM class and subclass identified in the 200 sample wetlands, and identifies topography and connectivity to streams or other waterbodies to differentiate between riverine, depressional, and lacustrine fringe classes (figure 2). By design, the classification key is most helpful in identifying a Great Plains palustrine waterbody as playa wetland and use should be limited to the HPR based on the CEAP–Wetlands region designations. Because hydrology is the main driver of wetland function, ecosystem services can be inferred due to current knowledge about the hydrology of these wetland types.

**Table 1**

Hydrogeomorphic classification results for National Wetlands Inventory identified palustrine wetlands sampled in the High Plains region.

| Class and subclass | Palustrine ($n$ = 200) | Percentage |
|---|---|---|
| Depressional | 118 | 59.0% |
| Naturally formed | — | — |
| Pothole/playa | 101 | — |
| Draw | 1 | — |
| Mechanically formed | — | — |
| Diked | 10 | — |
| Excavated | 6 | — |
| Riverine | 81 | 40.5% |
| Naturally formed | — | — |
| Floodplain | 21 | — |
| Oxbow | 2 | — |
| Streambed | 24 | — |
| Mechanically formed | — | — |
| Diked | 30 | — |
| Excavated | 4 | — |
| Lacustrine fringe | 1 | 0.01% |

## PLAYA ECOSYSTEM SERVICE MODELS

Twelve models were developed from CEAP–Wetlands work to predict playa ecosystem services. To rank models based on ease of use, we applied each model and measured variables required to populate the models. Some variables included playa area, dominant land use, and watershed area and were identified using wetland polygons, land use layers, and topographic maps. Models had been developed using data from field projects in the HPR with many models consisting of regression equations requiring explanatory variables that could be measured remotely. To apply models, playa wetlands were randomly selected from the 200 palustrine set and from the Playa Lakes Joint Venture Probable Playa Dataset (PLJV 2010). Models were applied on different playas to incorporate variability among basins. Many of the models were built using data from specific subregions or contained equations specific to three dominant land use types: native grassland or reference wetland, cropland, and conservation programs.

**Figure 2**

Hydrogeomorphic classification key for wetlands in the High Plains region. National Hydrography Database (NHD) stream lines used to designate riverine wetlands. The Italicized text indicates the classes of the hydrogeomorphic classification system as established by Brinson (1993).

**High Plains Region (HPR) Hydrogeomorphic Key**

| | | |
|---|---|---|
| 1 | Wetland is classified as Cowardin palustrine | 2 |
| 1 | Wetland is not classified as palustrine | Stop here (this key is not applicable) |
| 2 | Wetland is detectable via remotely sensed data | 3 |
| 2 | Wetland is not detectable via remotely sensed data | Lost/misclassified |
| 3 | Wetland is associated with a natural, continuous NHD stream or surrounding floodplain | *Riverine (5)* |
| 3 | Wetland is not associated with a natural, continuous NHD stream | 4 |
| 4 | Wetland exists within a closed watershed | *Depressional (9)* |
| 4 | Wetland exists along the edge of a lake or reservoir | *Lacustrine fringe* |
| 5 | Wetland retains water due to landscape alteration (anthropogenic or beaver activity) | 6 |
| 5 | Wetland does not retain water due to landscape alteration | 7 |
| 6 | Wetland is excavated | Riverine excavated |
| 6 | Wetland is diked | Riverine diked |
| 7 | Wetland is situated within current or historic streambed | 8 |
| 7 | Wetland is outside of streambed but within the floodplain | Riverine floodplain |
| 8 | Wetland exists within streambed during low flow | Riverine streambed |
| 8 | Wetland is disconnected and was formed by streamflow at bend | Riverine oxbow |
| 9 | Wetland retains water due to landscape alteration | 10 |
| 9 | Wetland does not retain water due to landscape alteration | 11 |
| 10 | Wetland is excavated | Depressional excavated |
| 10 | Wetland is diked | Depressional diked |
| 11 | Wetland is situated within a drainage | Depressional draw |
| 11 | Wetland is not situated within a drainage | Playa wetland |


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We ran models based on all possible land use types regardless of actual playa location or land use type because the methods for measuring metrics did not change. We identified grassland, cropland, and fallow crop from the CropScape data layer, which is derived from the USDA National Agricultural Statistics Service Cropland Data Layer (Weiguo et al. 2012). Some models require the identification of CRP and WRP/WRE lands, which can be done from our land use identification methods with conservation program spatial data displayed in a geographic information system (GIS). CRP and WRE location data are confidential, and access for non-USDA staff is limited and could restrict the use of these models for some users. Reference wetlands in Nebraska are used in RWB models and can be identified through the Nebraska Game and Parks Commission.

***Abiotic Models.*** The soil organic C (kg $m^{-2}$) model was built for all three land use types within the WHP. Soil characteristics from the Soil Survey Geodatabase (SSURGO) provided by the USGS were required along with the Soil Adjusted Vegetation Index (SAVI) based on vegetative spectral reflectances (Soil Survey Staff n.d.; O'Connell et al. 2016; Zhuoqing et al. 2016b; Thompson et al. 2019). Up to 10 SSURGO values were determined at the playa center or by averaging the values at 10 and 40 m (33 and 131 ft) distances from the southwest playa edge as required by the model. SAVI was determined for the playa basin through the Landsat 8 Level-2 product data downloaded from Earth Explorer (USGS 2017b).

Greenhouse gas flux (g C $ha^{-1}$ $d^{-1}$) was the standardized C equivalent of the estimated change in the sum of greenhouse gasses (carbon dioxide [$CO_2$], methane [$CH_4$], and nitrous oxide [$N_2O$]) (Zhuoqing et al. 2016a; Daniel et al. 2019). This model was built for both the NHP and the RWB. Metrics included Fraction of Photosynthetically Active Radiation (FPAR) and Leaf Area Index (LAI) as determined by the National Aeronautics and Space Administration's Moderate Resolution Imaging Spectroradiometer (MODIS) platform (Thompson et al. 2019). These data were downloaded through the Earth Observation System website at the playa location (Vannan et al. 2009).

The sediment depth (cm) model was developed using data from SHP playas. The regression equation only required percentage cropland in the watershed (McMurry and Smith 2018; Thompson et al. 2019). To apply this model, the watershed was delineated from USGS-based topographic maps by connecting the highest elevation points surrounding the playa basin (Esri 2017a). Land use in the watershed was identified using CropScape data by summing pixels using the Zonal Statistics tool and calculating the percentage cropland.

The floodwater storage ($m^3$) model was also developed from SHP playa data and used one equation to estimate original playa volume by playa area and another to estimate percentage volume loss by sediment depth (Tsai et al. 2007, 2010; Daniel et al. 2014; McMurry and Smith 2018; Thompson et al. 2019). Playa area was measured using the wetland polygon, and sediment depth was determined from the sediment depth model. Total volume lost was calculated by multiplying the original volume by the percentage lost, and floodwater storage was calculated by subtracting this volume lost from the original predicted volume.

The contaminant filtration (%) model was developed from SHP playa data and was applied by selecting mean values according to the vegetative buffer type (Haukos et al. 2016; Thompson et al. 2019). Buffer type was the land use type surrounding greater than 50% of the playa edge. Buffer types could include any noncultivated land use, and CropScape data were used to identify fallow crop and grassland.

The model for contaminant concentration (ppm) in water runoff was developed from SHP playa data. Values were selected based on mean width of any noncrop vegetative buffer (Haukos et al. 2016; Thompson et al. 2019). Land use data were used to determine buffer width in the four cardinal directions (Esri 2011). Four points on the playa edge at 0, 90, 180, and 270 degree angles from the playa centroid were established to correspond with the four cardinal directions, and buffer width was measured orthogonally from the playa edge. These four widths were averaged to calculate mean width.

The model for pesticide residue (μg $kg^{-1}$) in playa soils was developed from WHP and RWB playa data (Belden et al. 2012; Kensinger et al. 2014; Thompson et al. 2019). For the WHP, equations were developed for northern playas in Kansas, Nebraska, Wyoming, and Colorado; and southern playas in Oklahoma, Texas, and New Mexico (Belden et al. 2012). We identified the dominant surrounding land use type within a 500 m (1,640 ft) buffer, visually if obvious, or by summing CropScape land use pixels.

***Biotic Models.*** The model for plant species richness included native wetland plants and native upland plants within WHP playas (O'Connell et al. 2012a, 2012b). Regression equations were specific to the plant group of interest and surrounding land use type (Thompson et al. 2019). The model was applied and metrics included playa area, area of all playas within 1 or 5 km (0.6 or 3 mi), distance to nearest grassland playa, UTM coordinates, and water presence within a playa (1 = present, 0 = absent). Playa area was measured from the playa polygon. Area of all near playas was determined by establishing the required buffer width at either 1 or 5 km, selecting all PLJV probable playa polygons within the buffer (PLJV 2010), and calculating their total area. Distance to nearest grassland playa was identified using nearby playa polygons and identifying CropScape grassland as dominant. Decimal degree coordinates for the playa centroid were converted to UTM coordinates. Water presence was determined as present or absent using the most recent Landsat 8 imagery downloaded from Earth Explorer.

The amphibian species richness model was developed from SHP playa data and used a single regression equation (Venne et al. 2012; Kensinger et al. 2013; Thompson et al. 2019). Metrics included the ratio of watershed area to playa area and the length of hydroperiod in days. Playa area was measured using the playa polygon, and watershed area was determined through delineation as described previously. Once these two values were known, the ratio of watershed to playa area was calculated. Playa hydroperiod is not detectable using most remotely sensed data. For our model application, we used the average number of annual wet days observed by Tsai et al. (2007) as 98. For the purpose of model ranking, we considered hydroperiod measurements to be attainable through field sampling. If field sampling is not feasible, the manual includes average values based on field measurements, which can be substituted to make approximate model estimates.

The model for avian species richness and waterfowl abundance was developed from SHP playa data and consisted of four equations with one for each season (Kensinger et al. 2015; Thompson et al. 2019). Metrics required for model application included playa area (ha), watershed area (ha), tilled index of the watershed (Tsai et al. 2007), water depth, and water presence (present = 1, absent = 0). Playa area was measured from the playa polygon, and watershed area was determined through watershed delineation. The tilled index within the watershed was determined according to Tsai et al. (2007) and is calculated as

$$\text{Tilled Index (TI)} = \frac{\text{Tilled landscape} - \text{Untilled landscape}}{\text{Tilled landscape} + \text{Untilled landscape}}, \quad (1)$$

where tilled landscape is cultivated or conservation program lands and untilled is noncultivated grassland. Water depth, similar to hydroperiod, could not be determined from our available data sets, and an average value of 37 cm (14.5 in) was used from Tsai et al. (2012). For the model ranking, field sampling was considered the method to determine water depth and again, averages were provided for users without access to field measurements. Water presence was detected using Landsat 8 satellite imagery as previously described.

Pollinator models were developed for both the SHP and the RWB with slightly different variables for each. The SHP model estimated hymenoptera abundance and richness from land use, playa area, percentage vegetation cover, vegetation height, and mean monthly precipitation (Luttbeg et al. 2017; Thompson et al. 2019). Land use and area were measured remotely while vegetation metrics were applied using mean values from field measurements. Again, field measurements for vegetation were the considered method in the model ranking. Precipitation values were calculated from nearby west Texas Mesonet weather stations (Schroeder et al. 2005). The RWB model estimated hymenoptera abundance, richness, and diversity from land use; playa area; and percentage forb coverage (Joshi et al. 2018; Thompson et al. 2019). Land use and playa area were determined remotely. Average forb cover values were included by land use, but again, field measurements for this metric were considered in the model ranking.

***Model Ranking.*** Once all models had been applied, they were relatively ranked based on ease of application with 1 being the simplest to apply, and 12 being most difficult. This ranking was based on three characteristics of each model: number of metrics needed to populate the model, method of model application, and data location for required metrics. As the number of required model metrics increased, the relative rank also increased. Methods of model application included selecting mean values (simple) or solving of one or more equations (complex). Acquiring the necessary metrics was simplest when data could be accessed and broadly displayed in ArcMap, ranging to more complex when accessed from a platform such as SSURGO, external modeling, or sampling of field data.

The first three models each required one metric, and the application process was selection of mean values (table 2). The fourth and fifth ranked models each required only one metric, but numerous equations were required for service estimations. Models ranked as sixth through eighth required numerous metrics and equations but also included external data from MODIS, SSURGO, and Landsat 8. The last four ranked models required multiple metrics and equations, but each included at least one feature requiring field measurements to determine a particular value as previously mentioned.

Models that predicted abiotic services were the least complex to apply (relatively ranked 1 to 7), while biotic service models were considered most complex to apply (ranked 8 to 12). While all depressional wetland functions are closely tied to watershed activities (Smith et al. 2008), abiotic ecosystem service metrics are more easily measured using open source remotely sensed data. Metrics such as vegetative buffer width, surrounding land use type, and playa area are detectable from data, which can be easily accessed and broadly displayed in a GIS since these features experience change at a wide temporal scale and can be measured often enough to detect changes. The complexity of applying biotic ecosystem service models is related to the importance of hydroperiod and water presence. In wetlands, biotic services, such as amphibian, pollinator, and avian species presence, are reliant on inundation and hydroperiod (Tsai et al. 2012). Playa hydroperiod is determined by seasonal and intermittent precipitation, which makes measuring hydroperiod using open source data more difficult (Smith 2003).

## HIGH PLAINS REGION SAMPLING MANUAL

The HGM key and ranked ecosystem service models were used to develop a sampling manual for the HPR, titled *Ecosystem Services Estimation for Depressional Wetlands in the High Plains Region: Manual for the Integrated Landscape Modeling Partnership* (Thompson et al. 2019). This manual was developed for the Integrative Landscape Modeling partnership, which seeks to quantify and model ecosystem services provided by wetlands (Mushet and Scherff 2016). Methods detailed within this special feature were used to establish the application instructions of the sampling manual, which can be accessed through the CEAP–Wetlands website (https://www.nrcs.usda.gov/wps/portal/nrcs/detail/national/technical/nra/ceap/na/?cid=nrcs143_014155).

## SUMMARY

The sampling manual can be applied by researchers and natural resource managers to classify depressional wetlands as playas, communicate wetland functions, and estimate ecosystem service provisioning within playa basins in the HPR. Past, present, and future land use conditions could be modeled allowing natural resource managers to

## Table 2

**Conservation Effects Assessment Project wetlands component (CEAP–Wetlands) ecosystem service models for playa wetlands ranked by ease of application from simplest to most complex.**

| Rank | Service | Metric(s) | Model application |
|---|---|---|---|
| 1 | Contaminant filtration (%) | Vegetative buffer type | - Measure 1 metric<br>- Select value(s) |
| 2 | Contaminant concentration (ppm) | Vegetative buffer width | - Measure 1 metric<br>- Select value(s) |
| 3 | Pesticide residue (μg $kg^{-1}$) | Dominant land use | - Measure 1 metric<br>- Select value(s) |
| 4 | Sediment depth (cm) | Percentage crop in buffer | - Measure 1 metric<br>- Apply equation |
| 5 | Floodwater storage ($m^3$) | Playa area | - Measure 1 metric<br>- Apply four equations |
| 6 | Greenhouse gas flux (g C $ha^{-1}$ $d^{-1}$) | Dominant land use<br>MODIS – FPAR<br>MODIS – LAI | - Measure 1 metric<br>- Gather 2 external data features<br>- Apply 1 equation |
| 7 | Soil organic carbon (kg $m^{-2}$) | Dominant land use<br>SSURGO values (up to 10)<br>SAVI | - Measure 1 metric<br>- Gather up to 11 external data features<br>- Apply equation |
| 8 | Plant species richness: Native wetland and native upland | Dominant land use<br>Playa area<br>Area of all near playas<br>UTM coordinates<br>Water presence<br>Distance to near grass playa | - Measure 5 metrics<br>- Gather 1 external data feature<br>- Apply equation |
| 9 | Rainwater basin: Pollinator abundance, richness, and diversity | Dominant land use<br>Playa area<br>Percentage forb coverage | - Measure 2 metrics<br>- Field sampling<br>- Apply equation |
| 10 | Amphibian species richness | Playa area<br>Watershed area<br>Hydroperiod | - Measure 2 metrics<br>- Calculate ratio of metrics<br>- Field sampling<br>- Apply equation |
| 11 | Southern High Plains: Pollinator abundance and species richness | Dominant land use<br>Playa area<br>Percentage vegetation cover<br>Vegetation height<br>Precipitation | - Measure 2 metrics<br>- Field sampling<br>- Gather 1 external data feature<br>- Apply equation |
| 12 | Avian species richness and waterfowl abundance | Playa area<br>Water presence<br>Water depth<br>Tilled index<br>Watershed area | - Measure 3 metrics<br>- Gather 1 external data feature<br>- Field sampling<br>- Apply equation |

estimate the response of ecosystem services. Knowledge of service provisioning over time could communicate the change in ecosystem services and economic values these wetlands provide. Estimates of playa service provisioning within varying land use types could be used to determine the most beneficial location and practices for conservation programs. Playas have experienced losses up to 60% in most of the HPR (Johnson et al. 2012), and application of this sampling manual could assist in improving wetland ecosystem services, which are important to the health of local and global communities.

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