

## Research Paper

## Quantifying the effects of projected urban growth on connectivity among wetlands in the Great Plains (USA)

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## ABSTRACT

Urban wetlands often have prolonged hydroperiods relative to non-urban ones, so they may play an outsized, positive role for wildlife. Ecological studies of urban wetlands have typically focused on large metropolitan areas, but non-traditional urbanizing areas such as the towns of the Great Plains of North America are projected to experience land-use and climate changes that will alter connectivity among the freshwater wetlands along a continental-scale migratory wildlife corridor. We used seven graph theory metrics to quantify connectivity among 89,798 of these wetlands under landscape-change forecasts from two models built for three climate-change and development scenarios, projected to the year 2050. We compared outcomes from models that differed in focal variable (impervious surface or developed land use). Overall, models with impervious surface projections resulted in the most wetlands affected, whereas models featuring developed land use projections resulted in the largest spatial distribution of effect. There were differences in how many and which wetlands were forecast to become urbanized by model and scenario, resulting in different wetland network topologies and differences in the connectivity roles of individual wetlands. A consensus network was therefore developed based on the wetlands that were projected to increase in impervious surface and exist within developed land use by 2050. These 126 wetlands can be prioritized for urban ecological studies or management because they are highly likely to be affected regardless of model or scenario. Lastly, our study highlights the utility of considering a range of developmental futures when planning urban wetland management in non-traditional urbanizing areas.

## 1. Introduction

Urban ecosystem studies have traditionally focused on large metropolitan centers. However, urbanization can produce a variety of environmental changes that have ripple effects along the urban-rural continuum (McDonnell & Pickett, 1990). Although popular, the concept of the urban-rural continuum itself is subject to changes both social and political (Lichter & Ziliak, 2017) as well as ecological (Geneletti, La Rosa, Spyra, & Cortinovis, 2017; Hansen et al., 2005). Moreover, urban periphery studies may assist future planning and sustainability efforts by managers (Geneletti et al., 2017; La Rosa, Geneletti, Spyra, & Albert, 2017). Therefore, since the effects of urbanization can occur across a wide range of habitats and at several levels simultaneously, stakeholders and managers are increasingly interested in studies of “rural urbanization.” Studies of rural urbanization can inform regional conservation practices, ecosystem service management, and environmental hazard response (Cutter, Ash, & Emrich, 2016; Oliver & Thomas, 2014; Vias, 2012; Walker, de Beurs, & Henebry, 2015). Rural urbanization ecological studies will be especially important for those regions facing

increasing demographic pressures and climate change-induced restrictions to large-scale management actions. The Great Plains, an approximately 1.4–1.9 million km<sup>2</sup> expanse of arid and semi-arid prairies, grasslands, and steppes of the North American continental interior (Samson & Knopf, 1994), represents just such an area ripe for non-traditional, rural urban ecological studies.

Although the Great Plains are known derisively as “flyover country,” this region is one of the world’s leading agricultural production areas, with several historic periods of population shifts of settlement abandonment as well as urban expansion (Kotkin, 2012; Wishart, 2004). The Great Plains has experienced a recent surge in economic and population growth, attributed to energy resource development, increased agricultural production, and favorable economic conditions (Kotkin, 2012; Scott, 2017; U.S. Census Bureau, 2013, 2014). However, these regional trends may be variable at local scales, with some areas of the Great Plains experiencing population declines (Parton, Gutmann, & Ojima, 2007; Wishart, 2004). Despite these potentially contrasting socio-economic influences, recent effects (Kotkin, 2012; Scott, 2017; U.S. Census Bureau, 2013, 2014) have resulted in

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increased amounts of impervious surface coverage and expansion of developed land usage both within and surrounding regional urban centers in the Great Plains. As a consequence of these local landscape changes, a yet-unknown number of regional wetlands are also subject to increases in surrounding impervious surface and expansion of developed land usage.

Just as we do not normally view the Great Plains as urban, neither do we typically think of it as being wet. However, the Great Plains contains millions of freshwater wetlands that form an ecological network of wildlife habitat, the Central Flyway for migratory birds (Smith, Pederson, & Kiminski, 1989; Tiner, 2003). This continental-scale migratory wildlife corridor links the glacially formed prairie potholes of the northern Great Plains to the aeolian wetlands of the central and southern Great Plains, known as playas (Bolen, Smith, & Schramm, 1989; Guthery & Bryant, 1982; McIntyre et al., 2014). The playas of the Great Plains of the United States are similar to other networks of ephemeral wetlands in an otherwise arid landscape that are vulnerable to effects of land-use and climate changes, such as the sabkhas of north Africa (Briere, 2000) and playas of inland Australia (Bourne & Twidale, 2010). Projected climatic and development patterns for the Great Plains will further alter the landscape in terms of its composition, configuration, and connectivity among the 89,798 playas of the region (Fig. 1). Investigating future patterns of rural urbanization and associated impacts on playa wetlands can inform management actions to more sustainably mitigate climatic and development effects within the Great Plains.

Playas are the primary source of aboveground freshwater for the central and southern Great Plains, making them regional biodiversity hotspots through provision of critical resources for aquatic and amphibious wildlife (Bolen et al., 1989; Hall et al., 2004; Haukos & Smith, 1994; Hernandez, Reece, & McIntyre, 2006; Ramesh, Griffis-Kyle, Perry, & Farmer, 2012; Tsai, Venne, Smith, McMurry, & Haukos, 2012), and the primary sources of groundwater recharge for the Ogallala Aquifer (Gurdak & Roe, 2010; Smith, 2003). Playas have been modified for stormwater management and recreation in urban areas (Collins et al., 2014; Haukos & Smith, 1994; Heintzman, Anderson, Carr, & McIntyre, 2015); these alterations are expected to increase with expanding urbanization projected for the Great Plains region. Playas are fed from precipitation runoff, making them sensitive to weather and to human land-use decisions that alter watershed structure (Smith, 2003; Venne, Tsai, Cox, Smith, & McMurray, 2012).

Great Plains wetlands are being altered by human land use, including urbanization, and by climate change. However, these drivers do not always impair function: urban playas often have prolonged hydroperiods (up to 1312 days; Collins et al., 2014) relative to non-urban ones (up to 453 days; Tsai, Venne, McMurry, & Smith, 2007; Venne

et al., 2012) as a result of anthropogenic inputs of water and basin modifications for long-term water retention (Collins et al., 2014; Ganesan et al., 2016; Uden et al., 2015; VanLandeghem, Meyer, Cox, Sharma, & Patino, 2012). Additionally, playas within developed areas differ from non-urbanized playas in terms of water chemistry and their microbial community (Durham, Porter, Webb, & Thomas, 2016; Heintzman et al., 2015; Moorhead, Davis, & Wolf, 1998; Starr, Heintzman, Mulligan, Barbato, & McIntyre, 2016; Warren, Jeter, Kimbrough, & Zak, 2004), yet during drought, playas in developed areas may be the only ones containing water (Collins et al., 2014), so their reliability may allow them to play an unexpectedly positive role in terms of supporting landscape connectivity (Ruiz et al., 2014). Although the biological productivity of playas is driven by their natural wet-dry cycling (Haukos & Smith, 1994), the prolonged hydroperiod of urban playas may facilitate the development or dispersal of some species (Collins et al., 2014; Venne et al., 2012). Thus, urban playas are habitats that entail regional ecological trade-offs for managers as to whether to prioritize more natural ecosystem functionality or to prioritize prolonged aquatic habitat persistence across the landscape, the latter of which may be especially important during times of drought (Collins et al., 2014) and for migratory species.

The importance of playas as migratory stopover sites is well-recognized (McIntyre et al., 2014; Smith, 2003; Tiner, 2003), but the ecological network of playas is dynamic and subject to continued anthropogenic alterations. Interannual differences in precipitation and land use generate different topologies of wet playa occurrence from year to year (Ruiz et al., 2014). The consequences of these different topologies for wildlife migrating through the network have rarely been considered. Given regional climate change projections and increased likelihood of drought in the Great Plains, urban playas with their prolonged hydroperiods may be of increasing importance to migratory route connectivity. Despite the importance of playas for wildlife and humans, limited knowledge exists on current and expected rates of playa urbanization (i.e., the transformation of playas that do not exist in an urban context to those that do, and/or playas that are expected to experience increased ecological alterations as a result of anthropogenic pressures associated with urban development), which is needed for regional planners and private stakeholders to mitigate land-use and climatic changes on the Great Plains, especially with respect to landscape connectivity for wildlife.

For our study we documented projected increases in playa urbanization and subsequent changes to wetland connectivity using data developed by the U. S. Environmental Protection Agency's Integrated Climate and Land-Use Scenarios (ICLUS; <https://www.epa.gov/iclus>) and the U. S. Geological Survey's FOREcasting SCEnarios of land-use change model (FORE-SCE; <http://landcarbon.org/categories/land-use/>

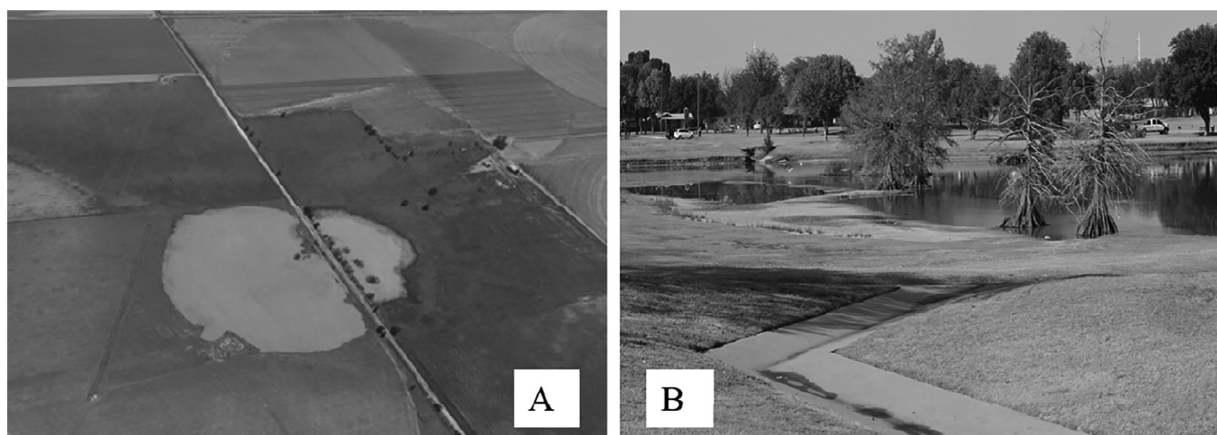


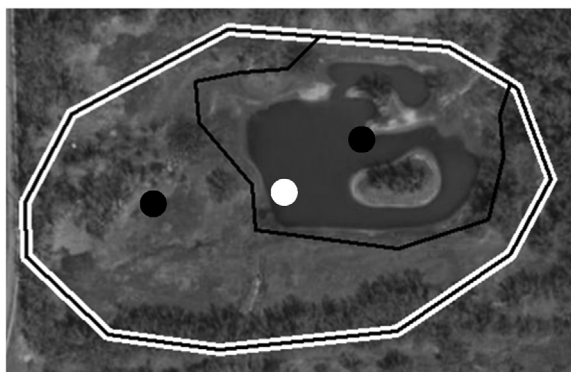
Fig. 1. Examples of playas of the Great Plains of the United States. (A) Playa with direct impervious surface alterations but not occurring within developed land use. (B) Playa within developed land use but with only limited surrounding impervious surface. Photographs courtesy of the authors.

download/) (Bierwagen et al., 2010; Sohl, Sayler, Drummond, & Loveland, 2007; U.S. EPA, 2010). These models generate climate and land-use projections and have been used to quantify projected urban growth (Bierwagen et al., 2010; Caldwell, Sun, McNulty, Cohen, & Moore Myers, 2012; Georgescu, 2015; Georgescu, Morefield, Bierwagen, & Weaver, 2014; Mondal, Butler, Kittredge, & Moser, 2013; Reinmann, Hutrya, Trlica, & Olofsson, 2016; Sohl et al., 2012; Sohl, Wimberly, Radeloff, Theobald, & Sleeter, 2016). From these models we derived projected trajectories of impervious surface and developed land use. These trajectories were then used to examine potential alterations within the ecological network of prairie wetlands via a graph theory approach quantifying topological characteristics of the Great Plains playa network (Bunn, Urban, & Keitt, 2000; Calabrese & Fagan, 2004; Minor & Urban, 2007, 2008; Urban & Keitt, 2001). Overall, we predicted that ICLUS models based on impervious surface coverage (as a continuous variable) would result in the largest number of urban playas and thus correspond with higher connectivity values (both in terms of overall network structure and importance of individual wetlands) than those that used FORE-SCE models (with its static, majority-rule classification scheme). We were guided to this prediction based on the observation that impervious surface development (e.g. roads) can occur at any point along an urban-rural gradient. Because landscape connectivity underpins species persistence, managing for connectivity is important in landscape and urban planning (Fahrig, 2003; Taylor, Fahrig, Henein, & Merriam, 1993; Trakhtenbrot, Nathan, Perry, & Richardson, 2005), so our overall objectives were to quantify connectivity among urban playas under a range of potential futures, and to identify any wetlands that were consistently identified across models and climate scenarios as being affected in future as good candidates for connectivity conservation.

## 2. Methods

### 2.1. Data

Our focal region was based on the digital playa wetland features contained within the Maps of Probable Playas (MPP) database developed by the Playa Lakes Joint Venture (<http://pljv.org/for-habitat-partners/maps-and-data/maps-of-probable-playas/>, accessed May 2016). This database mapped 89,798 playa wetlands across ~68 million ha of the U. S. portion of the Great Plains in Colorado, Kansas, Nebraska, New Mexico, Oklahoma, and Texas. Within the MPP, an individual playa basin may have multiple units as a result of the presence of multiple within-basin sub-features (Fig. 2). Some of these features are natural (e.g. presence of open water in one portion of a basin and presence of woody wetland vegetation in another portion of the same



**Fig. 2.** Example of MPP classifications of playa wetlands consisting of two within-basin sub-features (black outlines, with centroids of each sub-feature symbolized by black circles). Alternatively, a playa basin may be depicted by the simple perimeter of the basin (white outline, with centroid of entire basin symbolized by a white circle).

basin resulted in two wetland classification types within the same unique basin ID in the MPP), and others are due to human activity (e.g. a road bisecting a playa). Each sub-feature within a basin was treated as a separate analysis unit because each sub-feature has unique hydrological properties (e.g. hydroperiod). This approach also better reflects real-world conditions and more accurately reflects habitat availability constraints on wildlife.

We used ICLUS and FORE-SCE models to determine the extent, type, and rate of playa urbanization across the Great Plains playa region. ICLUS' primary focus is on changes in impervious surface coverage and population growth changes, whereas FORE-SCE's is on changes in land use. Impervious surface and developed land use are both associated with urban growth but are not synonymous with each other: for example, a rural highway has impervious surface but may not be associated with developed land use, and a developed area can have relatively little impervious surface (e.g. due to the presence of parks and vacant lots). Impervious surface and developed land use adjacent to or near playas have been shown to affect playa hydrology, water quality, and biota through changes in water flow, water chemistry, and other proximal drivers (Collins et al., 2014; Heintzman et al., 2015; Starr et al., 2016). Although the effects of impervious surface are not necessarily the same as those of developed land use (impervious surface may increase runoff, for example, whereas some forms of land use may disrupt runoff and increase water infiltration), these two features are defining traits of urbanized landscapes (Alberti, 2005; Arnold & Gibbons, 1996). Using both ICLUS and FORE-SCE thus allowed us to perform a more comprehensive estimate of playa urbanization trajectories than examining either dataset alone.

Both ICLUS and FORE-SCE models were developed from Intergovernmental Panel on Climate Change (IPCC) Special Report on Emission Scenarios (SRES) storylines for the continental United States and were developed during the early part of the current decade. We therefore used commensurate pairings between the ICLUS climate scenarios A1, A2, and B1 and the FORE-SCE climate scenarios A1B, A2, and B1, respectively, to represent a range of possible future conditions. These pairings are broadly consistent across both models (Sohl et al., 2012) and can be adapted to more recent IPCC Representative Concentration Pathways (RCP) climate scenarios (Moss et al., 2010; van Vuuren et al., 2011). Although the RCP scenarios are not exactly equivalent to the older SRES scenarios used in ICLUS and FORE-SCE, there are similarities with respect to emissions, concentrations, and temperatures (U. S. Global Change Research Program, 2014). For a detailed conversion between IPCC climate scenarios, see Rogelj, Meinshausen, and Knutti (2012). To develop comprehensive estimates of future urban playa development, we compared projected changes in impervious surface coverage from ICLUS and developed land use from FORE-SCE under all three of the climate-change scenarios (Table 1). Our analysis used a baseline year of 2020 and projected to 2050 to analyze future playa development trajectories; we used these years due to the decadal timeframe of ICLUS projections and because 2050 represents the latest available projection of FORE-SCE.

**Table 1**

Description of ICLUS v 1.3.2 and FORE-SCE climate models and their approximate IPCC SRES and IPCC RCP equivalents. "N/A" = not applicable because no equivalent value.

IPCC RCP	IPCC SRES	ICLUS (SRES Storylines)	FORE-SCE (SRES Storylines)
8.5	A1f	A2	A2
6.0	A1B	A1	A1B
4.5	B1	B1	B1
2.6	N/A	N/A	N/A



## 2.2. Data processing and general workflow

For each of the three ICLUS impervious surface raster datasets (under climate scenarios A1, A2, and B1), data were resampled in ArcGIS 10.3 (Redlands, CA) to match the cell sizes in the three FORE-SCE land-use change raster datasets (A1B, A2, B1). The data from ICLUS were only available at a 1 km resolution whereas data from FORE-SCE were only available at 250 m; we thus had data with two different grain sizes that we resampled to align the resolutions for spatial analysis in ArcGIS. We resampled the coarser-grained ICLUS data to match the finer-grained FORE-SCE data (dividing larger cells to match the size of the smaller ones) rather than the reverse (merging smaller cells to match the size of the larger ones). To do so, we divided each original  $1000\text{ m} \times 1000\text{ m}$  ICLUS cell to create four identical  $250\text{ m} \times 250\text{ m}$  cells that covered the same extent as the original ICLUS cell. Although rescaling from smaller to larger may be applicable to some studies, our raster resampling method allowed for a more detailed depiction of urban development of playa wetlands via expansion of both impervious surface and urban land use. Our rationale for our choice (resampling ICLUS to match FORE-SCE, i.e., resampling larger cells to create smaller ones) was based on preserving data integrity that can be lost when merging smaller cells to create larger ones (see [Turner, O'Neill, Gardner, & Milne, 1989](#) for more information). This resampling did not introduce false precision into the analysis, because values from the rasters were extracted to wetland centroids.

Both raster layers and the MPP layer were spatially projected to

UTM Zone 13 N. ICLUS and FORE-SCE data were then clipped to match the extent of the MPP. The MPP native polygon data were then converted to point data, and centroid coordinates of each playa were determined for subsequent connectivity analyses. The resulting layer thus contained impervious surface and land-use data for each of the 89,798 playa features in the MPP for each of the three ICLUS and three FORE-SCE datasets. This layer was then queried using the Select by Attributes Tool to determine individual playas trajectories in impervious surface coverage and projected land use ([Fig. 3](#)). Designation of a playa as urban was thus based on the single cell value (land-use type or percent impervious surface) at a cell centroid for each of the ICLUS and FORE-SCE models.

Because our objectives were to investigate the role that urban expansion will have on playa wetlands and thence on their contribution to connectivity, we quantified connectivity only among those playas expected to be impacted by current or future urban growth. By not including the ~99% of playas that do not exist within an urban context, we were able to focus on our targeted habitat type. Non-urban playas do play important roles in connectivity ([Albanese & Haukos, 2016](#)); however, they are dry far more often than they are wet ([Johnson, Rice, Haukos, & Thorpe, 2011](#)). Because playas surrounded by urban land use have longer hydroperiods than do playas with other forms of land cover in their watersheds ([Collins et al., 2014](#)), urban playas represent actual potential habitat for aquatic and wetland-associated species within an arid region. As such, urban playas have been shown to play an outsized role in overall connectivity through the playa network, relative to non-

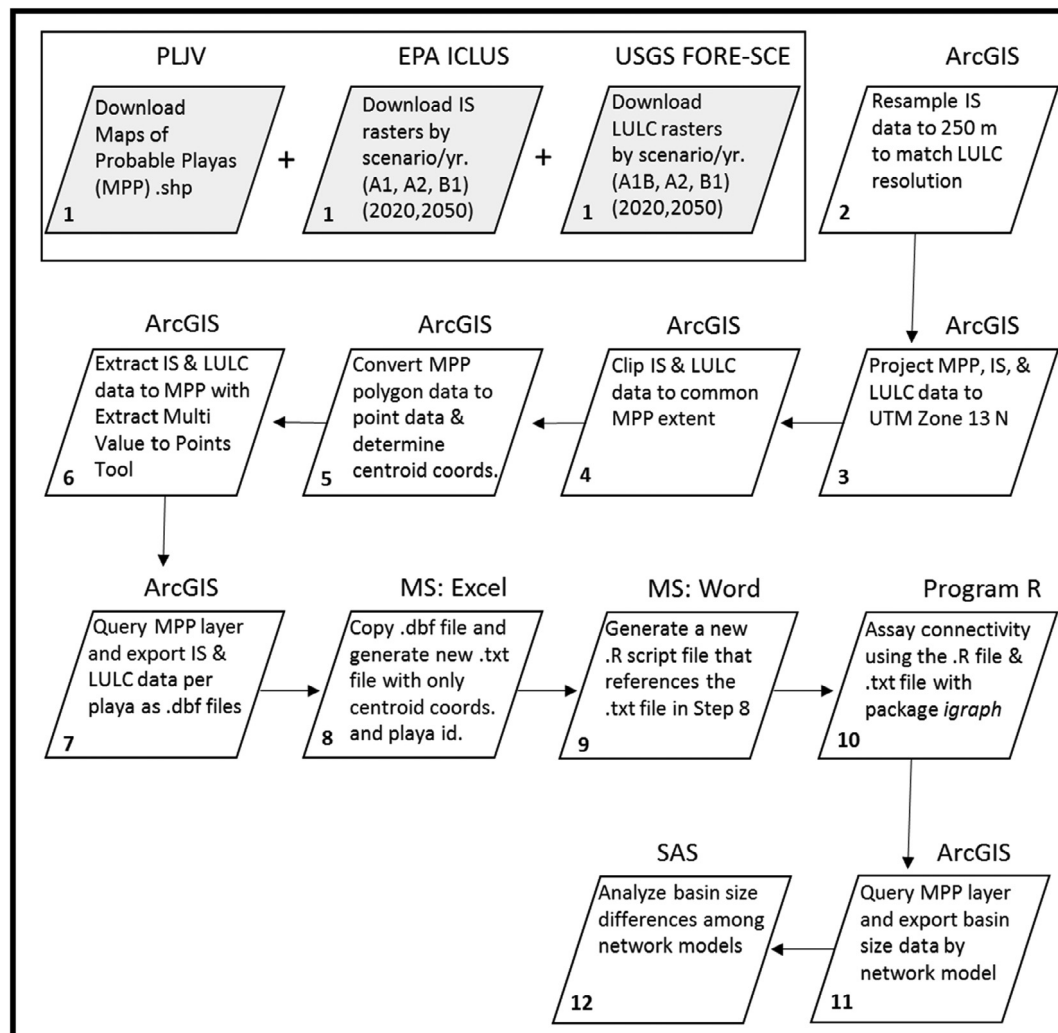
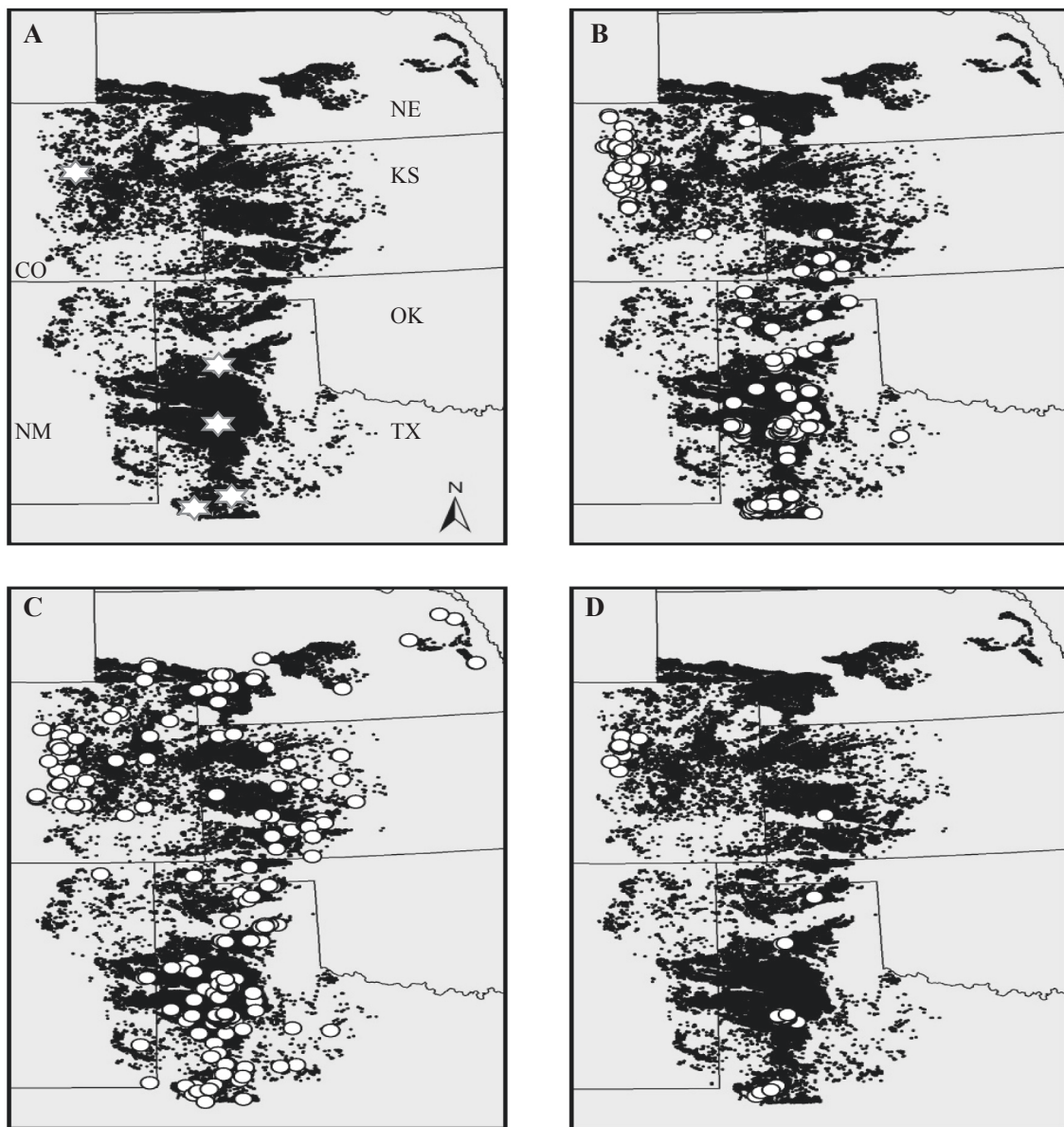


Fig. 3. Schematic of data processing and general workflow.



**Fig. 4.** Extent and distribution of Great Plains playa features (black dots). (A) Large metropolitan centers (white stars) of the region in 2020; from north to south: Denver, Amarillo, Lubbock, Midland, Odessa. (B) Distribution of urban playas in 2050 (white circles) as identified by the ICLUS Model Network. (C) Distribution of urban playas in 2050 (white circles) as identified by the FORE-SCE Model Network. (D) Distribution of urban playas in 2050 (white circles) as identified by the Consensus Model Network.

urban ones (Collins et al., 2014; Ruiz et al., 2014). Therefore, we identified and examined only those playas projected to experience urban development in terms of (1) any increase in impervious surface coverage under each climate scenario (i.e., playa centroids within raster cells projected to increase in impervious surface) (ICLUS); (2) those projected to experience an increase in developed land use under each climate scenario (i.e., playa centroids projected to occur within a raster cell that exists within or transitions to developed land use) (FORE-SCE); and (3) a consensus model that identified playas projected to experience an increase in both impervious surface and developed land use (i.e., playas centroids in cells that increase in impervious surface and occur within developed land use) (both ICLUS and FORE-SCE). Comparing projections from both models in this way provided the opportunity for identification of playas affected, regardless of model, for future landscape-planning activities across a range of potential development pathways. This resulted in the ICLUS Model Network, which identified all playas in the MPP that are expected to experience

an increase in surrounding impervious surface coverage among each of the ICLUS climate scenarios by 2050; the FORE-SCE Model Network, which depicted all playas in the MPP that are expected to exist within urban land use among each FORE-SCE climate scenario by 2050; and the Consensus Model Network, which depicted those playas projected to experience an increase in impervious surface coverage and exist within classified developed land use by 2050 among all ICLUS and FORE-SCE climate scenarios. Since each of these model networks were built from the MPP (which natively contained basin size information), we were able to examine whether significant differences were projected among networks on the basis of playa size using the statistics tool in ArcGIS 10.3, followed by analysis of variance (ANOVA) in SAS 9.4 (Cary, NC). Significant ANOVA models were then followed with a Fisher's Least Significant Difference test of means.

**Table 2**

Summary of differences in landscape contexts for urban playa features within the ICLUS Model Network, the FORE-SCE Model Network, and the Consensus Model Network. A1\*, A2\*, and B1\* are naming conventions to delineate the combined climate models for the Consensus Model Network. The proportion of playas affected by a given scenario and year are the numbers of playas present for a given scenario and year divided by the total number of playas predicted by 2050 to be affected.

		ICLUS Model Network (795 playa features)					
Scenario		A1		A2		B1	
Year		2020	2050	2020	2050	2020	2050
Average % of Surrounding Impervious Surface Among Playa Features In A Network (Ranges)		5.21 (0.01–41.71)	7.20 (0.35–51.85)	4.90 (0.01–37.48)	7.31 (0.36–51.85)	4.97 (0.01–37.48)	6.59 (0.35–44.37)
% of Playas in Developed Land Use (Number of playas)		16.1 (128)	24.0 (191)	16.1 (128)	21.9 (174)	14.7 (117)	18.6 (148)
		FORE-SCE Model Network (420 playa features)					
Scenario		A1B		A2		B1	
Year		2020	2050	2020	2050	2020	2050
Average % of Surrounding Impervious Surface Among Playa Features In A Network (Ranges)		8.84 (0.00–51.85)	9.97 (0.00–48.04)	8.53 (0.00–41.71)	9.73 (0.00–51.85)	8.81 (0.00–41.71)	10.12 (0.00–51.85)
% of Playas in Developed Land Use (Number of playas)		86.9 (365)	100.0 (420)	89.8 (377)	100.0 (420)	83.3 (350)	100.0 (420)
		Consensus Model Network (126 playa features)					
Scenario		A1*		A2*		B1*	
Year		2020	2050	2020	2050	2020	2050
Average % of Surrounding Impervious Surface Among Playa Features In A Network (Ranges)		13.51 (0.53–36.61)	16.71 (0.63–44.37)	12.93 (0.36–36.61)	16.40 (0.63–44.37)	13.37 (0.53–36.61)	16.72 (0.63–41.87)
% of Playas in Developed Land Use (Number of playas)		85.7 (108)	100.0 (126)	90.4 (114)	100.0 (126)	82.5 (104)	100.0 (126)

### 2.3. Connectivity analyses

Using graph theory-based terminology, each of these three modeled networks consisted of nodes (i.e., playa centroids) and the links (i.e., Euclidian distances) between them; connectivity among the nodes was examined for each of the three networks. Connectivity was quantified for each of the three model networks using methods derived from Ruiz et al. (2014) with the *igraph* package (Csardi & Nepusz, 2006) in R 3.3.2 (R Development Core, 2014). We compared the networks using seven graph theory metrics pertaining to size and connectance. Four of these metrics quantified the network as a whole whereas the other three metrics defined the roles of individual playas within each network. The four whole-network metrics were (1) coalescence distance (threshold distance at which the network becomes connected into a single cohesive grouping of nodes; this distance may be thought of as the farthest distance between neighboring nodes an organism must travel to traverse the network); (2) graph density (bidirectional linkage density, or the ratio of links present to the number of all possible links among nodes); (3) average nodal connectance (number of connections that a node has with other nodes at coalescence; known as average path length in *igraph*, higher values of this metric indicate more path redundancy through the network); and (4) graph diameter (the number of links forming the longest path through the network). The metrics investigating individual playas included the degree to which each node in each network played a role as a (5) stepping-stone that facilitates connectivity among habitat patches, (6) hub that is connected to more patches relative to other patches, or (7) cutpoint that with removal would increase network fragmentation (increase coalescence distance). Stepping-stones are those nodes with high values of betweenness centrality, which is highest for those nodes along the most direct paths through a network. Hubs were identified as having high Kleinberg's centrality scores, which quantifies the relative number of links per

node. Finally, cutpoints (articulation points in *igraph*) are those nodes that, if removed, fragment the network into clusters that are farther apart than the previously identified coalescence distance. For these three individual-node metrics, we identified and mapped the distribution of the top 10 stepping-stones, top 10 hubs, and all cutpoints (Drake, Griffis-Kyle, and McIntyre, 2017a, 2017b; McIntyre, Drake, & Griffis-Kyle, 2016; Ruiz et al., 2014). Because a network is defined solely by its nodes and links, the global connectivity metrics are indicators of properties of one or both of these and as such are somewhat correlated (e.g. average node connectance is positively associated with the number of links). The individual-scale metrics, however, are based upon their placement within the network and as such are only obliquely related to the global metrics. Therefore, using both global and individual metrics provided a more comprehensive examination of the playa network.

## 3. Results

### 3.1. Overall results

A side by side comparison of differences in expected urban playa locations by 2050 in relation to large urban centers reveals the unique connectivity structure of each of the model networks (Fig. 4).

Both ICLUS and FORE-SCE projected increased urbanization in the south-central Great Plains, with amounts differing by model and climate scenario. Between the years 2020 and 2050, average impervious surface coverage surrounding all playas was projected to increase in the ICLUS scenarios A1 (by 0.03% over all 89,798 playas), A2 (0.03%), and B1 (0.02%). Total developed land use for the entire playa region was also projected to increase in each of the FORE-SCE scenarios A1B (by 0.0054% or 19604.00 km<sup>2</sup> over the entire playa region as depicted in Fig. 4), A2 (0.005% or 17731.50 km<sup>2</sup>), and B1 (0.0037% or

**Table 3**

Summary of differences in average basin size and connectivity metrics for urban playas within the ICLUS Model Network, the FORE-SCE Model Network, and the Consensus Model Network by 2050.

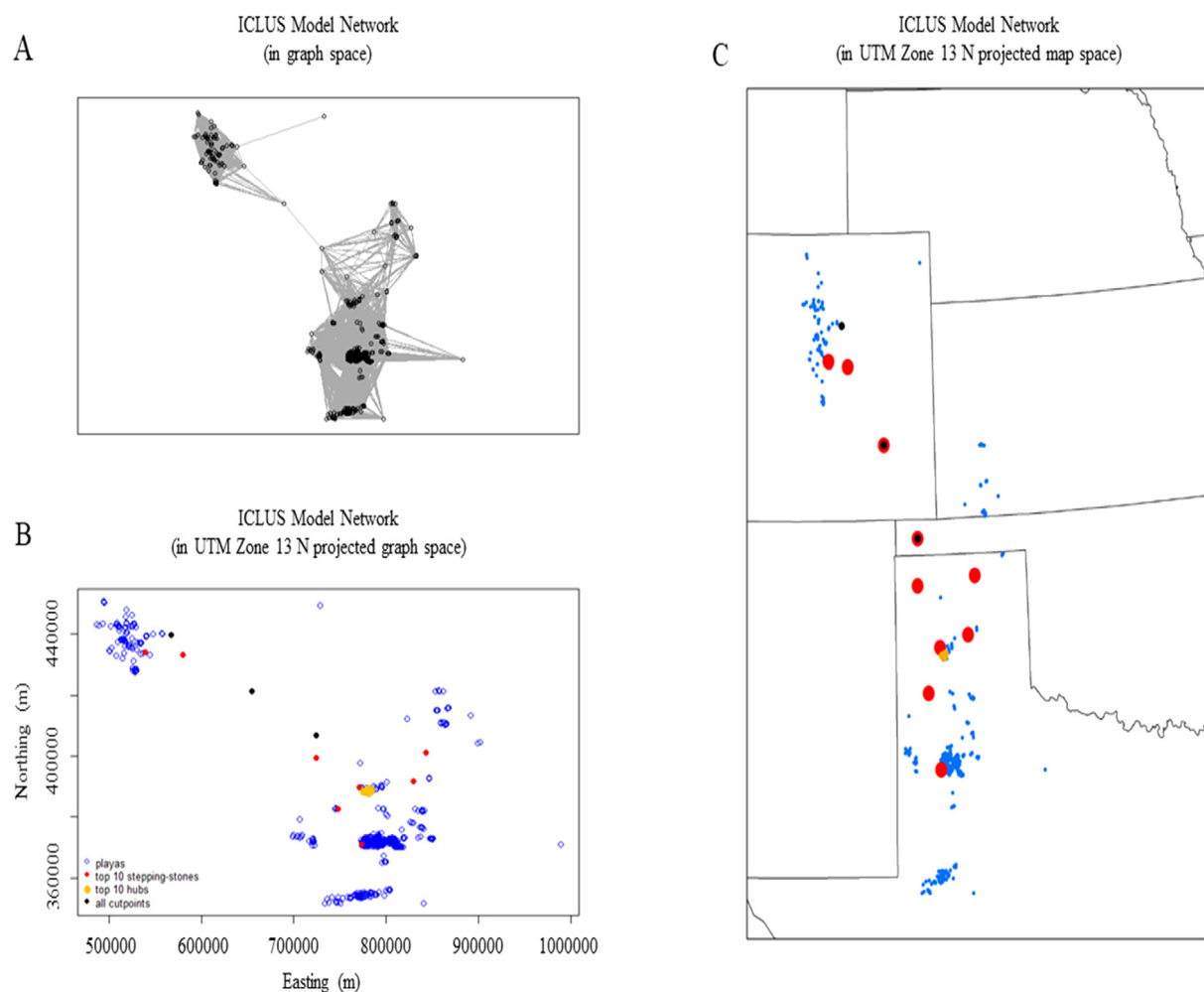
	ICLUS Model Network	FORE-SCE Model Network	Consensus Model Network
# of Playa Features	795	420	126
Average Playa Feature Basin Sizes & Ranges (ha)	3.35 (0.05–47.30)	2.22 (0.01–45.27)	2.25 (0.07–13.41)
# of Links	196,710	29,677	6085
Coalescence Distance (km)	189.08	163.46	363.50
Graph Density	0.62	0.33	0.77
Average Node Connectance	2.08	3.31	1.63
Graph Diameter	9	12	6

13504.50 km<sup>2</sup>). However, these increases in projected impervious surface coverage and developed land use were not completely concordant among scenarios or models: some playas are expected to experience a projected increase in surrounding impervious surface yet not be classified within developed land use. Similarly, playas may be projected to occur within developed land use yet not experience an increase in surrounding impervious surface (Table 2).

Furthermore, assays of playa size differences and structural connectivity among the scenarios depicted vastly different playa network topologies during the same time span. Despite the overlap in ranges of

basin sizes among model networks (Table 3), the ICLUS Model Network contained playas that were significantly larger than those in either the FORE-SCE Model Network or the Consensus Model Network (Fisher's LSD,  $p = < 0.0001$ ). However, there was no significant difference in basin sizes between the FORE-SCE Model Network and the Consensus Model Network. With respect to structural connectivity, differences among network models were especially pronounced for metrics evaluating coalescence distance, graph density (Table 3), and the distribution of stepping-stones, hubs, and cutpoints (Supplementary Material).

Supplementary data associated with this article can be found, in the



**Fig. 5.** ICLUS Model Network connectivity: (A) Top left image depicts linkages in unprojected graph space (gray lines) among playas (black rings). (B) Bottom left image depicts locations of locations of playas (blue rings), top 10 stepping-stones (red circles), top 10 hubs (orange circles), and all three cutpoints (black circles) in UTM Zone 13 N projected graph space. (C) Right image depicts the same information as in B, but is projected in UTM Zone 13 N projected map space. Note: due to icon overlap and close proximity (clustered in northern Texas), most hubs are not apparent. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)



online version, at <https://doi.org/10.1016/j.landurbplan.2019.02.007>.

### 3.2. ICLUS Model network

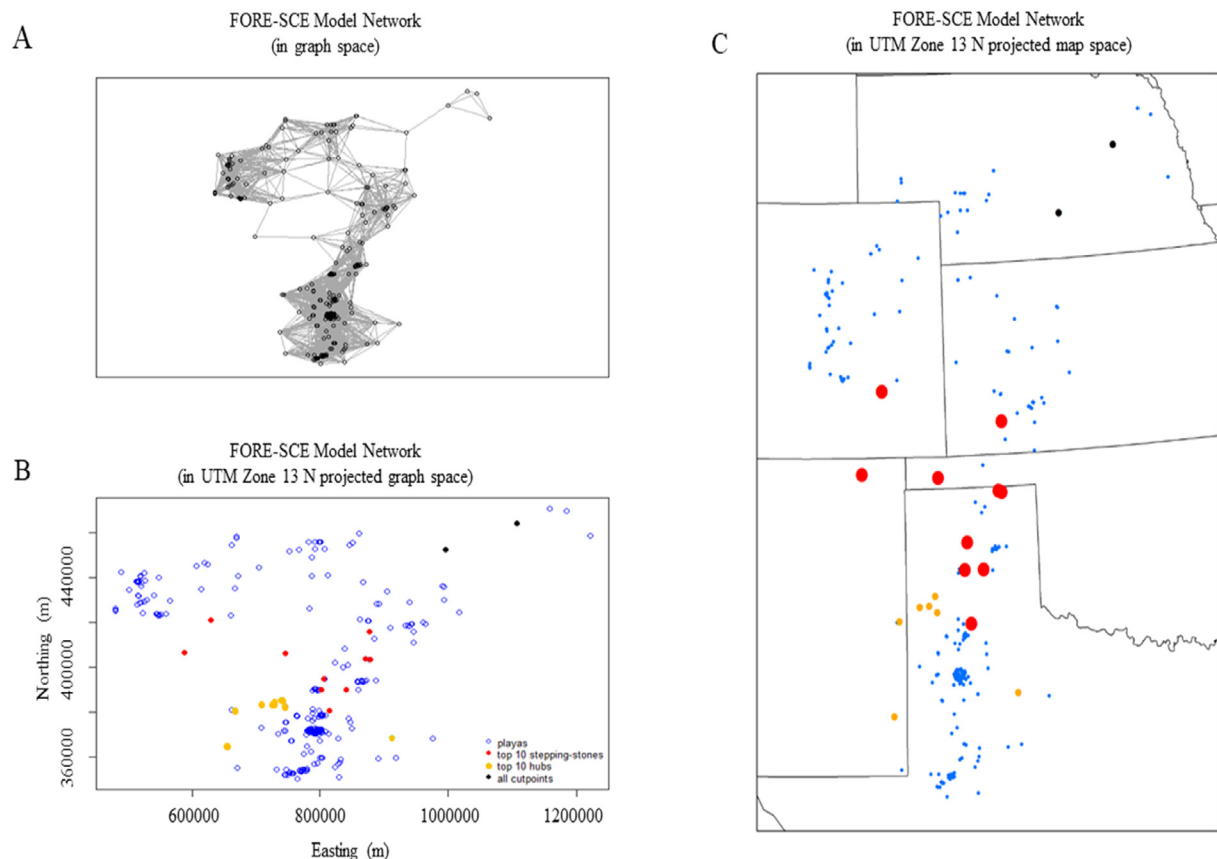
A total of 795 playa features within 626 basins were projected to experience an increase in impervious surface coverage between 2020 and 2050 in all three climate scenarios. These 795 playa features had an average basin size of 3.35 ha (range: 0.05–47.30 ha) and were located in Texas (660 playa features), Colorado (108), Kansas (26), and Oklahoma (1). The ICLUS Model Network was characterized by a couple of dense groupings of playas (along the Front Range of Colorado to the north, and a larger cluster south of the Arkansas River). Of the three modeled networks (ICLUS, FORE-SCE, and Consensus), this one had most nodes and links. Although we predicted that this network would have the most nodes, the coalescence distance of this network was not the shortest. Graph density was larger but average node connectance and graph diameter were smaller than for the FORE-SCE Model Network (Table 3). The top 10 stepping-stones were concentrated in the larger southern cluster, with two stepping-stones also functioning as cutpoints (located in the towns of La Junta, Colorado, and Boise City, Oklahoma) connecting the northern and southern clusters. A third cutpoint was present within the northern cluster, linking a single playa in far eastern Colorado near the town of Holyoke. The top 10 hubs were highly concentrated in the region with the greatest density of playas in the Texas Panhandle (Fish, Atkinson, Mollhagen, Shanks, & Brenton, 1998) in the southern cluster (Fig. 5).

### 3.3. FORE-SCE Model network

A total of 420 playa features within 353 basins were projected to exist within developed land-use by 2050 in all three of the climate scenarios. These 420 playa features had an average basin size of 2.22 ha (range: 0.01–45.27 ha) and were located in Texas (296), Colorado (55), Kansas (36), Nebraska (25), New Mexico (6), and Oklahoma (2). The urban playas forecast under FORE-SCE form a diffuse network, unlike the denser network from ICLUS. Contrary to our predictions, the FORE-SCE Model Network featured the lowest coalescence distance, despite having the second-fewest numbers of playas and linkages. These fewer playas were more diffusely spread, resulting in a lower graph density and larger graph diameter than the ICLUS network (Supplementary Materials). This network had the highest average nodal connectance of any of the three networks, indicating the highest amount of path redundancy. The top 10 stepping-stones, top 10 hubs, and all of the cutpoints were different playas than in the ICLUS Model Network. The stepping-stones occurred primarily in Nebraska, Oklahoma, and the northern Texas Panhandle. A smaller group of stepping-stones also connected western Oklahoma with New Mexico and Colorado. Two cutpoints linked playas in far eastern Nebraska with the main network. The top 10 hubs were concentrated near the southwestern periphery of the network in New Mexico and Texas (Fig. 6).

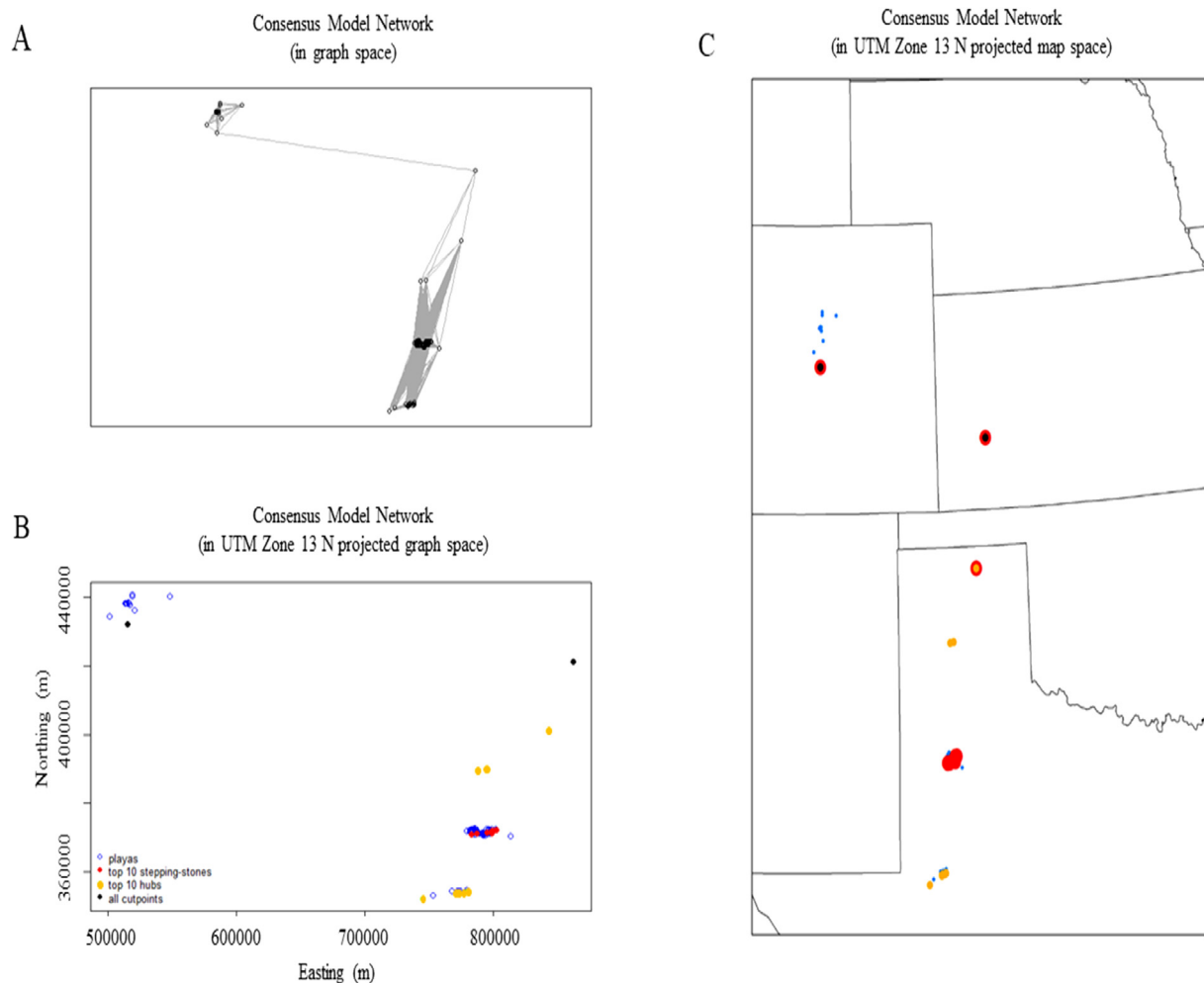
### 3.4. Consensus Model Network

The consensus network of playas that will be influenced by



**Fig. 6.** FORE-SCE Model Network connectivity: (A) Top left image depicts linkages in unprojected graph space (gray lines) among playas (black rings). (B) Bottom left image depicts locations of locations of playas (blue rings), top 10 stepping-stones (red circles), top 10 hubs (orange circles), and all three cutpoints (black circles) in UTM Zone 13 N projected graph space. (C) Right image depicts the same information as in B, but is projected in UTM Zone 13 N projected map space. Note: due to icon overlap and close proximity (clustered in northern Texas), some hubs are not apparent. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)





**Fig. 7.** Consensus Model Network connectivity: (A) Top left image depicts linkages in unprojected graph space (gray lines) among playas (black rings). (B) Bottom left image depicts locations of locations of playas (blue rings), top 10 stepping-stones (red circles), top 10 hubs (orange circles), and all three cutpoints (black circles) in UTM Zone 13 N projected graph space. (C) Right image depicts the same information as in B, but is projected in UTM Zone 13 N projected map space. Note: due to icon overlap and close proximity (clustered in northern Texas), some hubs are not apparent. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

urbanization and climate change was comprised of a north-south line of playas positioned around existing cities and towns in Texas and Kansas, and ones associated with the Denver (Colorado) metropolitan area. These 126 playa features within 93 basins were projected to experience an increase in surrounding impervious surface coverage and developed land-use by 2050, regardless of climate scenario. Averaging 2.25 ha (range: 0.07–13.41 ha), they were located in Texas (110), Colorado (15), and Kansas (1). Because this network had the fewest nodes and links, it is not surprising that it had the greatest coalescence distance and lowest average nodal connectance. The top 10 stepping-stones were concentrated in Texas in the vicinity of Lubbock. The top 10 hubs were all located in Texas in the southern cluster, mostly near the relatively large population centers of Amarillo, Midland, and Odessa. One hub occurred in extreme northern Texas near the town of Spearman. Cutpoints were observed near Garden City, Kansas, and Gleneagle, Colorado (Fig. 7). These consensus stepping-stones, hubs, and cutpoints are the most appropriate candidates for management, as they are the most likely to be influenced by urban growth regardless of focus (impervious surface or land use) or climate scenario.

#### 4. Discussion

Urban playas represent important ecological (Collins et al., 2014; Ruiz et al., 2014; Starr et al., 2016) and cultural resources (Smith, 2003;

Young, 2015). Although urban playas have garnered some scientific interest with respect to their contamination chemistry (Arefeen, 1995; Faust et al., 2012; Heintzman et al., 2015; Huang, 1992) and microbial communities of human health concern (Huddleston, Zak, & Jeter, 2006; Moorhead et al., 1998; Warren et al., 2004), our study is the first to our knowledge to document quantitatively the projected rates and distribution of future playa urbanization and their associated influence on network connectivity under various climate and land-use change scenarios. Such scenarios are also in play for ephemeral wetland networks in other parts of the world.

With respect to playa management, climate-mitigation actions are intractable for much of the Great Plains playa region, but what can be managed more effectively is land-use change, including urban growth. Focusing on impervious surface is one way to project urban growth; using developed land use is another. These two ways do not converge upon similar outcomes in terms of which playas will be considered urbanized, however. Both small-scale changes in impervious surface development (road construction), and larger-scaled, municipal projects (urban parks development) will increase the number of playas that are urbanized, which will influence individual playa ecology (i.e., hydrology, habitat quality, and biotic communities) and overall playa network structure (i.e., routes through the landscape based on the topological distribution of hubs, stepping-stones, and cutpoints). Because urban playas are more consistently inundated compared to non-urban

playas, especially during drought (Collins et al., 2014), the distribution of urban playas may foster channelization of migratory movement pathways across the network (due to the presence of fewer options rather than the more diffuse full network of playas), as drought conditions are projected to occur more frequently within the region (McIntyre et al., 2014).

Urban playas, with their characteristic hydrology, biotic communities, and municipal importance, are expected to play important roles as stepping-stones, hubs, and cutpoints in supporting movement through this wetland network in the future. However, the ability of urban playas to function in these roles is influenced by the biotic constraints of species that use the system and the topology of the network as a whole. With coalescence values ranging from 189 to 364 km, the distances among urban playas may be insurmountable for some species (e.g. amphibians) yet be accessible for others (e.g. migratory birds). Also of importance is the spatial arrangement of playas within the network; as predicted, the ICLUS Network Model had the greatest number of nodes, but the FORE-SCE Network Model had the greatest average nodal connectance (i.e., greatest path redundancy), a potentially important component of local habitat selection for migratory species. Lastly, and with respect to both graph density and graph diameter, linkages were depicted as Euclidean, but the actual movements of species will likely not be (Albanese & Haukos, 2016; Drake, Griffis-Kyle, & McIntyre, 2017a, 2017b; Haig, Mehlmann, & Oring, 1998; Pittman, Osbourn, & Semlitsch, 2014; Smith & Green, 2005; Sinsch, 2014), further complicating management actions.

Because our study examined connectivity within the network of playas expected to be directly affected by urbanization, which are < 1% of the total number of all playas in the Great Plains, our results are best viewed as highly conservative with respect to overall potential connectivity changes that are anticipated to occur within this region. However, while other studies have described connectivity of the playa network as a whole in other contexts (Albanese & Haukos, 2016; McIntyre, Collins, Heintzman, Starr, & van Gestel, 2018; Ruiz et al., 2014), these assessments did not examine land use, nor future projections of land use and climate change. Although the overall number of playas expected to experience future urbanization is small, those playas that are so affected may exert ecological influences at larger spatial extents than would be immediately inferred by their abundance on the landscape.

Relatedly, our results suggest that effects of urbanization may differ by playa basin size. In the ICLUS Network Model, larger basins were expected to be incorporated into urban environments, whereas the FORE-SCE Network Model and Consensus Model Network predicted urbanization of smaller playas. Because urban development and basin size differences can prolong playa hydroperiod, urban playas may represent an attractive opportunity and effective target to mitigate against anticipated climate changes (Hayhoe & Wuebbles, 2007; U. S. Global Change Research Program, 2014) by maintaining accessible aquatic resources on the landscape, albeit at the potential reduction in regional productivity of playas (Haukos & Smith, 1994). Potential effects of compromised urban water quality on these events, and decreased ecological functionality of smaller playas via development, however, remain to be examined.

The ecological threats posed by urbanization and climate change to playa network functionality are similar to those affecting isolated wetland systems worldwide (Calhoun et al., 2017; Cohen et al., 2016; Rains et al., 2016), with both urban development and climate change expected to accelerate in the foreseeable future. Indeed, an examination of recent aerial imagery of urban development in the Great Plains playa region has revealed that several playas that we included in our models have already been effectively removed from the landscape, being replaced entirely by built-up areas. Unfortunately, direct mitigation practices for threats associated with urbanization is scant, based on limited legal protections of playas (Haukos & Smith, 2003). The future trajectories of urbanizing playas will be influenced by both economic

factors (municipal land acquisition and zoning) and social factors (preferences for conservation and park spaces), which are themselves subject to change across the urban environments of the Great Plains and thus are outside the realm of this study. However, our research can inform ecologists and managers about playa conservation under a suite of climate and development scenarios, by identifying exactly which playas are likely to be affected by additions of impervious surface, land-use development, or both (Supplementary Material). Additionally, our results can be used to identify and direct the establishment of sites to better review the effects of rural urban development, a potentially fruitful application of urban ecological principles and landscape planning. Finally, the 126 Consensus Model Network playa features are the best candidates for longitudinal monitoring of the effects of future urban growth on aquatic ecosystems in the Great Plains because they are likely to be affected by urbanization regardless of how that is defined. Geographic coordinates of these playas are provided in the Supplementary Material. These playas will likely show impacts from growth in both impervious surface and developed land-use, which could be compared to patterns in playas that are predicted to be affected by only one of the variables. For the majority of the urban playas, the two projection models did not converge; therefore, a more targeted approach to urban playa management will be needed for these other playas by first identifying the more likely changes to occur for a given locality (impervious surface expansion or land-use development), which would then allow one to focus on the projected outcomes from the appropriate model (ICLUS or FORE-SCE). By comparing patterns that emerge from the Consensus Model Network playas with those from the ICLUS and FORE-SCE Model Network playas, we may be able to tease apart which defining feature of urbanization—impervious surface or development—is the more influential. Our findings are, essentially, predictions that need to be tested.

## 5. Conclusions

Playas in urban contexts, as a consequence of their altered hydrology, may in the future become increasingly important and consistent components in maintaining regional connectivity under projected climate and land-use changes in the Great Plains (Burris & Skagen, 2013; Hayhoe & Wuebbles, 2007; McIntyre et al., 2014). The Great Plains playa wetland network has been and will continue to be altered by the coincident challenges of land-use and climate changes; however, the projected distribution and degree of the effects of these changes are subject to model inputs and may not reflect on the ground decision-making by stakeholders. Since urbanization makes playas more likely to contain water, enhancing connectivity in this region may involve embracing urban development, which is a rather novel concept and one that is contrary to traditional conservation thinking. Our study can be used to inform the actions of stakeholders and guide larger-scale landscape planning. The current existence and expected increases in both the number and distribution of urban playas has resulted in altered ecological function of the playa network and may yet provide opportunities for mitigation of some climate-based management practices both at local and regional scales. Thus, although our study is limited to discussion of urban playas, it provides a basis by which to better appreciate the connectivity difficulties put forth by urbanization and climate change. All future landscape modelling is subject to uncertainty; however, by identifying which playas are most likely to be affected by projected landscape changes, our findings can be used to help guide regional development and conservation strategies across the Great Plains. Finally, our study emphasizes that a broader appreciation of urban ecology is needed, focusing not just on traditionally large urban areas, because even small or modest changes in impervious surface or developed land use may have large effects in rural urban areas.

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