Developing an Updated Multi-temporal Landcover Classification to Assess Riparian Conservation and Inform Decision-Making in the Upper San Pedro Watershed: A Classification and Regression Tree (CART) Model Approach

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Abstract:

Effective transboundary watershed management and ecological conservation in arid regions with growing water demands remain significant challenges facing land and water managers worldwide. In transboundary watersheds, previous research confirms that it is imperative to understand and integrate decision-making about water and land resources on both sides of international borders, as the processes impacting water supply and demand are deeply interlinked in systems with shared surface and groundwater supplies. In arid and semi-arid region watersheds in particular there exists the further challenge of meeting diverse and often competing demands for natural resources, particularly land and water.

The Upper San Pedro watershed, spanning southeastern Arizona and northeastern Sonora, is experiencing increasing demand for limited water supplies, including urban consumption in the cities of Sierra Vista, Arizona and Cananea, Sonora, the military installment at Fort Huachuca, industrial copper production in Sonora, irrigated agriculture, livestock production, and the often overlooked water demand from natural vegetation via evapotranspiration. Exacerbating the existing challenge of achieving scientifically informed, integrated decision-making in a transboundary context is the fact that policy-makers often lack scientific information that is up-to-date and well suited to their needs. Landcover and landuse maps are critical tools for assessing, analyzing, and managing ecological change and for catalyzing conversations and enhanced communication among stakeholders and decision-makers. However, accurate, up-to-date land-cover maps are often lacking. For example, although the Upper San Pedro watershed has undergone considerable changes in landcover over the last decade, the most recent widely used landcover map is based on satellite imagery from the year 2000.

This paper presents the methodology conducted to create an updated, high-accuracy landcover map at 30-meter spatial resolution for the bi-national San Pedro River watershed for the year 2010 and land cover change analysis that assesses the effectiveness of riparian conservation efforts in the Arizona portion of the watershed. Utilizing multi-temporal imagery from NASA's Landsat 5 Thematic Mapper (TM), we followed a sophisticated Classification and Regression Tree (CART) approach to derive an updated, high accuracy (kappa=0.7511), 10-class landcover classification that matches the landcover categories used in the previous series of landcover maps. The 2010 classification provides an opportunity to analyze recent land use and land cover changes in the watershed at an enhanced level of detail and brings the temporal coverage of the entire dataset for the Upper San Pedro River watershed to 37 years (1973-2010). The land cover change analysis provides a spatially explicit assessment of changes at both the binational watershed scale and within the San Pedro Riparian Conservation Area (SPRNCA), established in 1988 to protect the unique, highly biodiverse dryland riparian corridor in southern Arizona.

Key Words: Remote sensing; land cover change detection; riparian conservation; San Pedro River; binational watershed management

1. INTRODUCTION

In the arid and semiarid southwestern region of the United States, unique dryland riparian areas and associated uplands play host to a number of diverse and critical ecosystem processes and services (Newman et al. 2006). In addition to being vital habitat for diverse flora and fauna species, riparian corridors, defined as "the area from the edge of the stream bank to the external visible line of the canopy where an abrupt change in vegetation height, types, and amount occurs" (Johansen and Phinn 2006), assist in controlling non-point source pollution, help to maintain cool water temperatures through shading, and afford numerous cultural, recreational, and aesthetic values (Bagstad et al. 2005; Ashraf et al. 2010; Wang et al. 2010). Riparian vegetation structure also provides protection against flooding, by attenuating peak discharge (Forzieri et al. 2010). In ecosystems, such as the Sonoran Desert, that experience intense seasonal precipitation due to the North American Monsoon (NAM), riparian areas act as a first line of defense to moderate damage from flooding.

However, in these water-limited systems, various natural and anthropogenic stressors are causing major shifts in the environment, leading to degradation of critical aquatic habitats and changes in land cover (Steiner et al. 2000). The social-ecological transformations, including wholesale habitat alterations and restructuring, that accompany these land cover changes can result in impacts at multiple scales, such as habitat fragmentation, altered carbon and nutrient cycles, and changes in global climate patterns (DeFries et al. 2004a). Land cover classification and change detection through remote sensing present effective methods for identification and analysis of social and environmental change over time at multiple spatial scales.

In the binational Upper San Pedro River watershed, located in the US-Mexico border region of southeastern Arizona and northeastern Sonora (Figure 1), there are a number of competing demands for limited water supplies, including urban consumption in the cities of Sierra Vista, Arizona and Cananea, Sonora, the military installment at Fort Huachuca, increasing copper production at the massive Buenavista del Cobre copper mine in Cananea, irrigated agriculture, livestock production, and the often overlooked water demand from natural vegetation via evapotranspiration. In the riparian corridor, biodiversity is critically dependent on hydrologic processes, specifically surface flow, shallow groundwater, and water quality, which are influenced in complex ways by both direct human intervention and broader climatic and landscape-scale processes. For example, aquifer depletion, due to excessive groundwater pumping to sustain agriculture and urban areas, reduces the amount and timing of water available to ecological communities in the riparian corridor. Fernandes et al. (2011) detail the impacts of adjacent land use change on riparian systems. The authors argue that the surface water extraction, groundwater pumping, grazing, nutrient inputs, and replacement of riparian forest with crops often concomitant with agricultural production, can result in a loss of riparian habitat complexity, increased stand mortality and decreased growth rates, and impacts on the ability of species, such as cottonwoods that depend on seasonal flooding, to successfully reproduce. Urban development is responsible for increased runoff and sediment, replacement of riparian habitat with roads and infrastructure, habitat fragmentation, increased levels of point and non-point pollution, and the introduction of exotics. Importantly, the distributions of streamside plants, which comprise the structure vital for providing many ecosystem services, are dependent on numerous factors, including depth to the water table, rooting characteristics, and the riparian substrate geology, all of which are sensitive to the processes of land use change, including proximal agricultural and urban development (Amlin and Rood 2002).

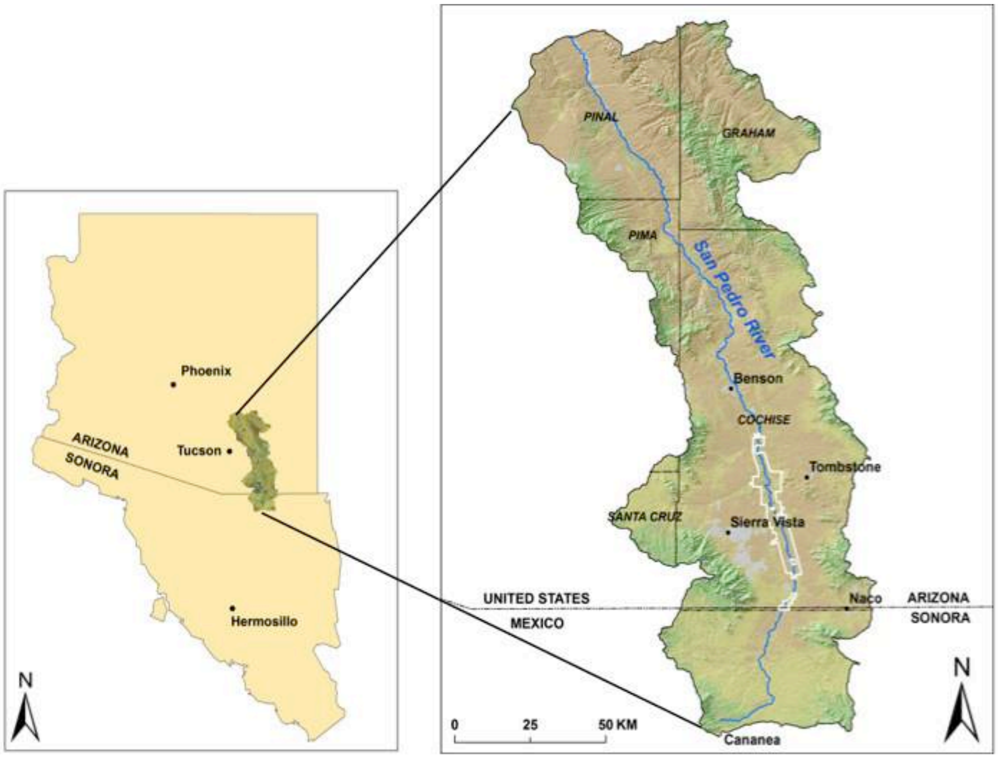

Figure 1. Map of the binational Upper San Pedro watershed. The study area ranges from the headwaters of the river in Cananea, Sonora, Mexico to the USGS gaging station near Redington, Arizona. The white outline on the right-hand map denotes the San Pedro Riparian National Conservation Area (SPRNCA).

In 1988, in an attempt to mitigate the environmental degradation occurring to the San Pedro River riparian corridor due to groundwater exploitation and urban development, the US Congress federally designated the San Pedro Riparian National Conservation Area (SPRNCA), a 40-mile riparian zone conservation area (Steiner et al. 2000, Brookshire et al. 2010) (Figure 2). The purpose of the designation was to protect and enhance 56,000 acres of key desert riparian ecosystem, which is home to 84 mammal species, 14 fish species, 41 reptile and amphibian species and 100 bird species. The San Pedro River is also a critical flyway for North-South migration of birds; the SPRNCA provides habitat to over 250 migratory bird species, making it one of the top ten birding destinations in the world. Furthermore, the Upper San Pedro watershed contains critical habitat for seven threatened and endangered species, five of which rely on wetlands (Steinitz et al. 2003).

However, a number of factors, both negative and positive have impacted the success of the SPRNCA, including steady population growth in the city of Sierra Vista, intensive groundwater withdrawal, decreasing precipitation, and the removal of grazing and most of the irrigated agriculture from the riparian area. However, the success of the SPRNCA remains contentious. For example, the USGS reported that between 1913 and 2002 (14 years after the SPRNCA designation), surface water levels at the Charleston, Arizona gauge dropped a precipitous 90% in July and 80% in the months of August and September.

Figure 2: Left hand photo depicts the cottonwood-willow riparian canopy forest along a dry stretch of the once perennial San Pedro River within the SPRNCA. Right hand photo depicts a wet stretch of the San Pedro River within the SPRNCA.

In light of the importance of riparian ecosystems and the abundance of critical functions that they provide it seems obvious that the protection, restoration, and monitoring of these ecosystems would be a paramount concern. However, there is limited understanding of the combined effects of anthropogenic land use change and natural climate variability on the resilience of complex riparian systems in semi-arid regions. Although, efforts to conserve these areas have been become a priority in the United States, Jones et al. (2010) note that little is actually known about whether or not cumulative efforts to restore and protect riparian zones are succeeding in affecting rates of riparian habitat preservation nationwide. Similarly, Goetz (2006) argues that approaches are needed to monitor changes that are taking place in riparian vegetation, to target restoration activities, and to assess the success of previous management activities.

Over the last several decades, agriculture, grazing, and mining have been decreasing in the United States portion of the Upper San Pedro watershed, but have remained stable or have increased in the Mexican portion of the basin (Steinitz et al. 2003). The impact of these activities, as well as the effect of natural stressors to the environment, has been tracked through analysis of land cover datasets. As part of the North American Landscape Characterization (NALC) project, Kepner et al. (2000) developed a series of four 10-class land cover datasets for the Upper San Pedro watershed that spans from the headwaters of the San Pedro River in Cananea, Sonora to the USGS gaging station near Redington, Arizona. The datasets were developed for 1973, 1986, 1992, and 1997 using a combination of NASA Landsat MSS and TM imagery resampled to a 60m spatial resolution prior to classification. These classifications relied on the ISODATA procedure to perform an unsupervised classification. Overall accuracies for the four datasets were assessed by Skirvin et al. (2004) and ranged from 68% to 75%. Due to the consistencies of the classification methodologies and type of imagery used, these four maps allow for a detailed quantitative examination of land cover change in both the United States and Mexican portions of the Upper San Pedro watershed for a 24 year span (Kepner et al. 2002). Notable trends seen in the watershed from 1973 to 1997 include transitions in native vegetation assemblages and changes in agriculture and urbanized areas (Kepner et al. 2002, Miller et al. 2002). Between [2002 and 2014], this land cover dataset and the associated 10-class classification system has been widely cited and utilized by numerous government agencies and stakeholder groups, including the US Geological Survey (USGS), the Bureau of Land Management (BLM), the U.S. Forest Service, the Friends of the San Pedro, and The Nature Conservancy. However, all of the studies that have utilized these data for analyses have been limited by its 60m spatial resolution, while comparable land cover datasets, such as the National Land Cover Dataset (NLCD), are classified at a 30m spatial resolution.

The goal of this research is to characterize and assess the more recent changes to the landscape of the binational Upper San Pedro watershed at a regional scale and within the SPRNCA riparian conservation area. To meet this objective, we performed a land cover classification reflecting conditions in 2010 to complement the existing four map series of land cover dataset, last updated in 1997. In the 13 years between 1997 and 2010, Arizona has experienced rapid population growth and urbanization (Brown et al. 2005, Lang and Nelson 2007, MacDonald 2010). Using the methods from a recent high-accuracy land cover classification in a neighboring watershed as a starting point (Villarreal et al. 2011), imagery from NASA's Landsat 5 TM satellite and additional spatial data from various sources were used as inputs to develop a robust Classification and Regression Tree (CART) classification method to generate a 30 meter resolution land cover dataset in the Upper San Pedro watershed for the year 2010. The same 10-class landcover scheme (Kepner et al. 2002) was used in our new classification, making it possible to compare landcover across the series of classifications. The landcover classes include: forest, oak woodland, mesquite woodland, grassland, desertscrub, riparian, agriculture, urban, water, and barren (Figure 3) (see Kepner et al. (2000) for detailed land cover class descriptions).

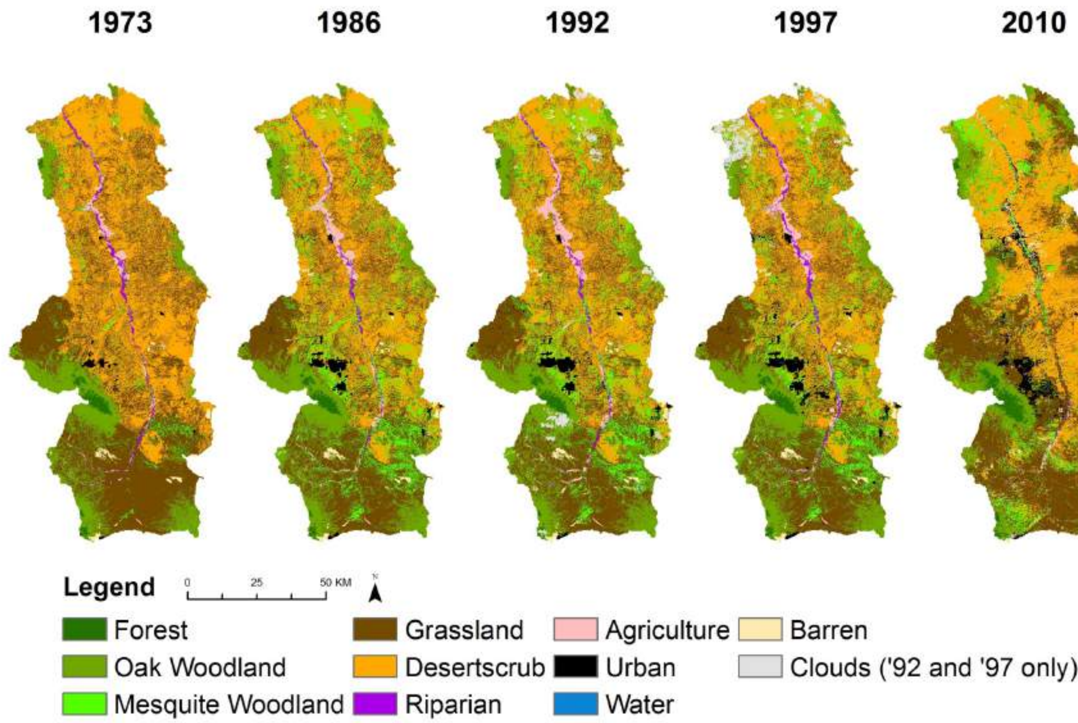

Figure 3: 1973 to 2010 series of land cover maps for the San Pedro River watershed.

2. INFORMING CONSERVATION POLICY THROUGH ASSESSMENT OF LAND COVER CHANGE

2.1 Conservation of Binational Natural Resources

Worldwide, the effective and equitable management of transboundary watersheds and ecosystems, hydraulically or ecologically connected systems that traverse national boundaries, present significant challenges to natural resource governance (Megdal et al. 2012; Lopez-Hoffman et al. 2009). Implementing conservation of sensitive transboundary resources, such as biodiversity hotspots, surface and sub-surface water flows, and migratory land and air routes for endangered species, represent highly contested processes that often end in inequitable or unsustainable outcomes (UN 2009). In transboundary riparian corridors, such as the San Pedro River, ecological functions and processes flow across national political borders and boundaries, yet management regimes are bounded and asymmetric, producing a tapestry of interwoven, overlapping, and at times contradictory, rules, rights, and resource extraction and conservation practices. The existence of multiple agencies and organizations with distinct management priorities, ideologies, and political-economic settings challenges the establishment of integrative natural resource management. In the US-Mexico border region, transboundary collaboration for natural resource management is complicated by unilateral decision-making, institutional asymmetries, uneven topographies of power relations, distinct historical contexts and systems of property relations, diverse natural resource management ideologies, multiple, competing demands for resource use, distrust between actors, and uneven access to information about the social and biophysical characteristics of the natural resource system (Browning-Aiken et al. 2004; Wilder et al. 2010; Megdal and Scott 2011; Varady et al. 2013).

Because the decisions of actors to use and extract resources, such as water, both depend on and influence institutional possibilities and constraints, policy makers must consider the political, social, and ecological contexts in which resources are extracted, utilized, and traded, including current and historical configurations of ownership and access rights, market mechanisms, and norms of resource use. To achieve sustainable watershed management, serious attention to human conditions, characteristics, and interactions is necessary for successful creation, implementation, and long-term functioning of policy (Davenport and Seekamp 2013). Identifying and characterizing catalyst events that initiate collaboration and galvanize collective action behavior leads to increased understanding of the mechanisms that encourage and promote collective action (Prokopy et al. 2014). In border communities, it is also necessary to account for are the rights, returns, relationships, and responsibilities among diverse stakeholder positions (Petursson et al. 2011). Understanding the intricate and complex relations between stakeholders is critical for what Ebbin (2014) refers to as the problem with problem definition. The author argues that in the context of natural resource-based conflicts, discursive problem framing is a key variable because how and by whom problems are framed impacts perceptions of causation, assigning of blame, and narrows the range of possible solutions. At the center of the problem of problem framing are conflicting ideologies that structure understanding of human-environment relations, including appropriate forms and levels of resource allocation, resource use behaviors, and conservation priorities.

The establishment of conservation areas and the implementation of conservation policy also transform how nature is understood. Differing and competing conservation ideologies create and may exacerbate conflicts between actors as local management and governance of the resource is affected. In relation to the transboundary San Pedro River watershed, previous research argues that any renewed binational sustainability effort focus on social equity and the development of institutions that strengthen the collective action of binational actors and the integrated management of water resources, rather than imposing conservation strategies inappropriate to the local context (Ruiz et al. 2011). Ruiz et al. (2011) argue that central to achieving effective collective action is the active participation of diverse social actors. Furthermore, the decision-making context is altered by the existence of diverging visions and interpretations held by the actors and users on both sides of the border with respect to natural resource (land and water) rights.

2.2 Scenario Planning and Analysis for Policy and Decision Making

In natural resource management situations prone to conflict and with high uncertainty about future conditions, scenario planning and analysis is a powerful method for catalyzing conversations and enhancing communication among diverse stakeholders in a transboundary context, including land and water managers, urban planners, and conservationists (Scott et al. 2012). Scenarios represent a range a possible futures based on plausible forecasts of climate change, land-use change and population growth, which can be utilized to investigate how future water demand will change as a result of changes in climate and land-use patterns. The benefits of participatory scenario planning are well documented, including improved understanding of future uncertainties, integration of local knowledge, co-creation of a diverse set of robust adaptation options, and development of trust among managers and communities (NCA 2013; Scott et al. 2012). The result of these benefits is increased acceptance of the model outputs among stakeholders and improved likelihood that the model will be adopted by land and water managers and integrated in decision-making.

Yet, scenario planning and analysis requires tools that are user-friendly, up-to-date, and appropriately scaled (Steinitz et al. 2005; Kepner et al. 2004), such as high resolution, high accuracy, current land cover and land use data. Although previous research has explored a range of alternative future scenarios for impacts to groundwater in the Upper San Pedro River basin (Steinitz et al. 2003), these scenarios did not sufficiently consider the individual influences and impacts of water and land management activities occurring in the upstream portion of the San Pedro River watershed in Sonora, Mexico. Furthermore, the results of the scenario modeling and analysis were not spatially-explicit.

In the transnational San Pedro River, the existence of competing demands for water, multiple, diverse agencies, organizations, and actors, and overlapping legal and territorial systems of property rights creates a complicated and conflict-ridden water management situation with detrimental consequences. The 1,875 square mile, bi-national San Pedro River basin encompasses diverse and ecologically sensitive ecosystems, including the SPRNCA, varied topography, and an eclectic mix of populations, ranging from the urban center of Sierra Vista to the military base, Fort

Huachuca, to the rural cotton farmers and cattle ranchers of the rapidly shrinking agricultural lands. The combination of highly variable precipitation patterns, heavily irrigated agriculture, rapid population growth, and the junior status of San Pedro River permit holders to Colorado River water from the Central Arizona Project, has resulted in steadily increasing groundwater withdrawals that currently exceed the natural rate of recharge (Browning-Aiken et al. 2004; Kepner et al. 2004). Within the United States portion of the basin groundwater pumping has caused the San Pedro River to lose over half of its historical perennial surface water flow, a condition aggravated by high well densities in close proximity to the river (Browning-Aiken et al. 2007).

3. METHODOLOGY

Satellite remote sensing is an important tool for classifying land cover, assessing change over time, and monitoring conservation effectiveness, such as spatially-explicit characterizations of increasing or decreasing riparian vegetation. Moderate- and high-resolution satellite and aerial imagery has been previously used in a number of applications with regard to assessing vegetation presence, structure, biomass, and land cover change in riparian corridors (Johansen and Phinn 2006). To analyze how land cover has changed in the San Pedro River watershed over time, we performed a land cover classification of the binational San Pedro watershed for the year 2010. This land cover map provides an update to the existing series of four land cover maps (Kepner et al. 2000), which stretch across 24 years from 1973 to 1997. The landcover classification followed a sophisticated Classification and Regression Tree (CART) protocol that previously has shown success in producing high accuracy classifications in the nearby Santa Cruz watershed (Villareal et al. 2011). The final 2010 land cover map product (Figure 4) has an overall accuracy of 77.6%, which a review of the remote sensing literature found to be acceptably high, as 85% represents the highest accuracy possible with current methods.

3.1 CART Model Training and Reference Data

Training data were collected by digitally delineating polygons of homogeneous land cover for each class based on high spatial resolution basemap imagery made available by Environmental Systems Research Institute (ESRI) and 1-meter resolution imagery from the National Agriculture Imagery Program (NAIP) for 2010 (Table 1). Training data were collected across the entire extent of the watershed in order to capture the variance in elevation and slope gradients that occur across the watershed. During the development of the classification, several classes exhibited high variation, resulting in overestimation of occurrence. To improve classification accuracy, we split these classes into subclasses for training purposes. Urban was divided into high density and low density urban. Agriculture was split into three classes, corresponding to strong Normalized Difference Vegetation Index (NDVI) signals for (I) both scenes, (II) May only, and (III) September only. NDVI is an indicator of green leaf biomass, with higher values corresponding to higher amounts of green vegetation (Tucker 1979). These classes were recombined after the CART process was completed prior to any other post processing or execution of the accuracy assessment. A summary of the training data used to create the final land cover classification is presented in Table 2.

Table 1. Reference imagery used to collect training and accuracy assessment data for the 2010 land cover classification of the Upper San Pedro watershed.

Date	Resolution (m)	Accuracy (m)	Country	Data Source
5/20/2010	0.3	4.08	USA	ESRI High Resolution Basemap Imagery
11/8/2010	0.3	2.72	USA	ESRI High Resolution Basemap Imagery

11/8/2010	0.3	4.08	USA	ESRI High Resolution Basemap Imagery
5/20/2010	0.3	5.4	USA	ESRI High Resolution Basemap Imagery
3/28/2010	0.5	10.2	Mexico	ESRI High Resolution Basemap Imagery
4/11/2010	0.5	10.2	Mexico	ESRI High Resolution Basemap Imagery
6/5/2010	1	5	USA	Cochise County NAIP
6/14/2010	1	5	USA	Santa Cruz County NAIP
8/14/2010	1	5	USA	Graham County NAIP
6/14/2010	1	5	USA	Pima County NAIP

Table 2. Summary of the final training dataset used for the 2010 land cover classification of the Upper San Pedro watershed.

Class	Polygons	Pixels	Area (m^2)
Forest	50	8802	7921800
Oak Woodland	100	10042	9037800
Mesquite Woodland	254	5655	5089500
Grassland	211	29126	26213400
Desertscrub	199	10108	9097200
Riparian	116	2074	1866600
Agriculture I	43	1659	1493100
Urban- High Density	102	12421	11178900
Water	31	849	764100
Barren	113	3661	3294900
Urban- Low Density	67	4970	4473000
Agriculture II	25	1061	954900
Agriculture III	43	4480	4032000
Total	**1354**	**94908**	**85417200**

3.2 CART Model Input Data

The land cover classification methodology of the nearby Santa Cruz watershed performed by Villarreal et al. (2011) was used as a starting point for this classification of the Upper San Pedro watershed. See5, a data mining software (Quinlan 2012) was used in combination with ERDAS IMAGINE (Intergraph 2010) to construct the decision tree used in the CART process and to perform the classification. A key feature of the input data used to develop the classification was the exploitation of the phenological changes resulting from the North American Monsoon (NAM). The NAM is marked by an increased amount of precipitation in the region during the summer months

(Forzieri et al. 2011). The Upper San Pedro watershed, for example, receives approximately 65% of annual precipitation during the summer monsoon season, between July and September (Steinitz et al. 2003). Due to differences in the response to precipitation pulses (Hultine et al. 2004), different vegetation classes may experience varying levels of greening.

For the Upper San Pedro watershed, the phenological changes resulting from the NAM were striking (Figure 4). These changes were exploited using the multitemporal Kauth-Thomas (MKT) transformation, which shows differences in brightness, greenness, and wetness between dates (Collins and Woodcock 1996). Other layers used in the classification include Normalized Difference Vegetation Index (NDVI), which approximates the greenness of vegetation; Slope, derived from a 30m digital elevation model from the National Elevation Dataset (Gesch et al. 2002, Gesch 2007); and Band 3 (red) Texture, generated from a moving 3*3 standard deviation window on the Landsat imagery. The full list of input layers used in the final CART classification is presented in Table 3.

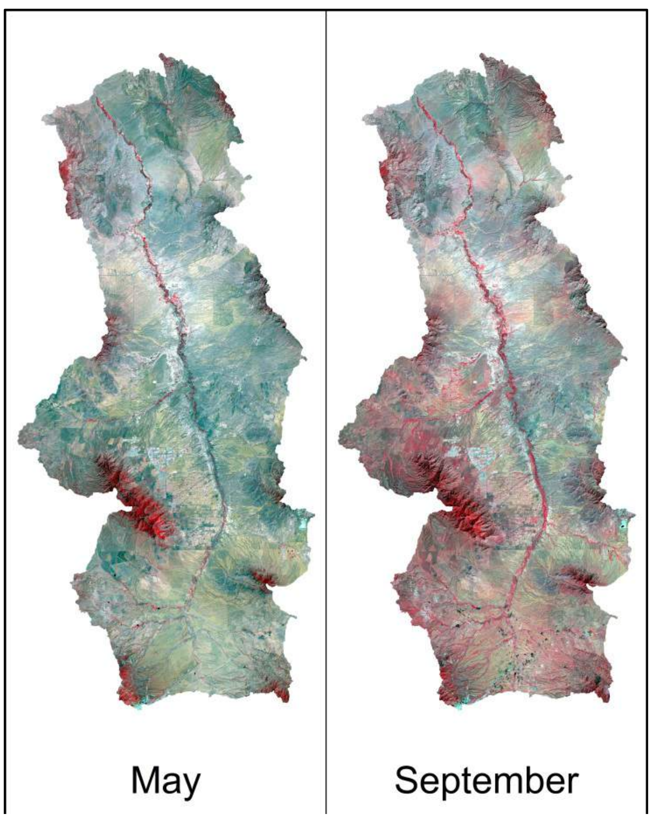

Figure 4. False color composite (FCC) images of May (left-hand image) and September (right-hand image) depicting the phenological change as a result of the summer monsoon in the Upper San Pedro watershed. Band 4 (near-infrared) is displayed as red, band 3 (red) as green, and band 2 (green) as blue. Green vegetation appears as shades of red.

Table 3. Input layers for the 2010 land cover classification of the Upper San Pedro watershed. All input layers were raster files (.img) in signed 16-bit format at a 30m spatial resolution. Landsat imagery and its derivatives were atmospherically corrected using ATCOR. Band 24 was eliminated during the winnowing process of developing the regression tree.

Bands	**Description**
1-12	12 band stack of May & September 2010 Landsat scenes (Path 35, Row 38/39)
13-24	12 band Multi-temporal Kauth-Thomas
25	NDVI May 2010
26	NDVI September 2010
27-32	6 band Principal Components Analysis of the 12 band Landsat scenes
33	Slope raster derived from DEM
34	Red band standard deviation texture May 2010
35	Red band standard deviation texture September 2010

3.3 Post Processing

After the CART classification was completed, we manually examined the output dataset to identify areas of obvious misclassification. Many areas initially classified as Urban stood out as being incorrectly classified, especially dry streambeds that are spectrally similar to roads. The Automated Geospatial Watershed Assessment (AGWA) Tool's Land Cover Modification Tool (Burns et al. 2004) was used to manually correct such areas of obvious misclassification. A majority filter with a 3*3 moving window was then used to reduce noise and create a smoother classified surface. We also used the Smart Eliminate Tool from the NLCD Mapping Tools (Homer et al. 2007) to enforce a one-acre minimum mapping unit (five pixels).

3.4 Accuracy Assessment

A stratified random sampling approach was taken to perform the accuracy assessment of the 2010 land cover map. Strata were defined by the output land cover map, with 50 random accuracy assessment points per class, for a total of 500 points in accordance with recommendations of Congalton and Green (2008). During the accuracy assessment process, the identifying land cover information was not accessed. We manually identified each accuracy assessment site using the same ESRI basemap imagery and NAIP aerial photography used for training data collection. Due to potential georegistration errors between the Landsat and basemap imagery, only center pixels of 3*3 homogeneous areas of land cover were used as accuracy assessment sites. Areas covered by the training dataset were removed from consideration for the accuracy assessment. To ensure data objectivity and to remove any potential bias, a Python script was written to automatically generate and randomize the accuracy assessment sites. Final accuracies were assessed by analyzing overall accuracy, producer's accuracy, user's accuracy, and a kappa analysis (Story and Congalton 1986; Congalton and Green 2008).

3.5 Change Analysis

Land cover change over time is assessed through two methods. First, we present a comparison of the relative proportions of each land cover class at the watershed scale and within the San Pedro Riparian National Conservation Area (SPRNCA) for each of the datasets. This method is used to assess broad trends in land use and land cover change over several decades, but has the disadvantage of not being spatially-explicit. Thus, it is not possible through this type of comparison to evaluate where the changes in land cover are occurring nor to assess the type of tradeoffs between land cover types. The second type of land cover change analysis presents the results of a post-classification change detection. This method of change detection demonstrates a spatially-explicit assessment of land cover change at both the regional, watershed scale and within the boundaries of the SPRNCA riparian conservation area. The change detection analysis is particularly useful to assess the effectiveness of riparian conservation activities because it highlights specific areas of change on the landscape between the pair of years, 1986 (2 years before the establishment of the SPRNCA) and 2010 (22 years after the establishment of the SPRNCA). Assessing change within the established riparian conservation area over this 24-year time period provides valuable insight for locating areas of successful intervention in the riparian corridor, such as areas of increase in land cover types that are positive for riparian ecosystems, and for evaluating the specific conversions of one land cover type to another, such as the conversion of grassland to mesquite, which has been occurring in some areas of the SPRNCA.

4. RESULTS & DISCUSSION

4.1 Accuracy Assessment

We assessed the accuracy of the 2010 land cover classification using the well-established indicators of kappa analysis and overall, user's, and producer's accuracies. The accuracy assessment results were organized into an error matrix (Table 4). Analysis of the error matrix revealed an overall accuracy of 77.60% and a kappa of 0.7511. Individual class accuracies varied from 53% to 100%, (Table 5). By comparison, Skirvin et al. (2004) reported an overall accuracy of 72% and a kappa of 0.65 for the 1997 land cover classification of the same area.

Table 4. Error matrix for the 2010 land cover classification of the Upper San Pedro watershed. Overall accuracy is 77.60%, kappa 0.7511.

		Reference Data										
	Class ID	1	2	3	4	5	6	7	8	9	10	Total
Map Data	1. Forest	**36**	14									50
	2. Oak Woodland	1	**40**	5	2	2						50
	3. Mesquite Woodland		1	**40**	6	3						50
	4. Grassland		1	9	**38**	2						50

5. Desertscrub		1	4	8	**35**			2			50
6. Riparian			8	1		**40**	1				50
7. Agriculture			2	4	1		**43**				50
8. Urban			3	1	2		1	**41**		2	50
9. Water									**48**	2	50
10. Barren		3	5	5	8			2		**27**	50
Total	37	60	76	65	53	40	45	45	48	31	500

Table 5. Individual class Producer's and User's accuracies for the 2010 land cover classification of the Upper San Pedro watershed.

Class	Producer's Accuracy	User's Accuracy
1. Forest	97%	72%
2. Oak Woodland	67%	80%
3. Mesquite Woodland	53%	80%
4. Grassland	58%	76%
5. Desertscrub	66%	70%
6. Riparian	100%	80%
7. Agriculture	96%	86%
8. Urban	91%	82%
9. Water	100%	96%
10. Barren	87%	54%

With an overall accuracy of 77%, the 2010 land cover map has a slightly higher accuracy than the maps created for the NALC project (Skirvin et al. 2004). The greatest single source of confusion came from areas of Oak Woodland misclassified as Forest (Table 4). Both communities are dominated by evergreen trees, with the primary difference coming from overall canopy cover and structure (Kepner et al. 2000). This level of detail may be difficult to discern at a 30m pixel resolution. Mesquite woodland had the lowest producer's accuracy at 53%. The largest contribution to this was confusion with Grassland, possibly due to the influence of a largely grass understory in such areas. Another large source of error with Mesquite Woodland came with its misclassification as

Riparian. The broad definition of Mesquite Woodland (crown coverage ≥15%; Kepner et al. 2000) is a likely culprit for why Mesquite Woodland is misclassified as both Grassland and Riparian areas, despite these classes being vastly different from one other. For the accuracy assessment of the prior NALC classification, Skirvin et al. (2004) had similar difficulties with Mesquite Woodland accuracies, attributing potential sources of error to insufficient spatial and spectral resolutions, as well as differences in interpretation among the groups performing the classification and accuracy assessment. Splitting the Mesquite Woodland into Mesquite Shrubland and Mesquite Bosque in future classifications could potentially improve overall classification accuracy. Barren had the lowest user's accuracy (54%), with the largest confusion from Desertscrub. This confusion is understandable, as Desertscrub has sparse foliage surrounded by large amounts of barren ground. Likewise, the previous NALC classifications also had difficulty with classifying Barren, notably having 0% accuracy (both user's and producer's) for the 1986 map (Skirvin et al. 2004).

4.2 Land Cover Change I: A Comparison of Relative Proportions of Land Cover Types

The updated land cover map we developed for the binational San Pedro River watershed for the year 2010 (Figure 5) demonstrates significant changes in land cover (Table 6). At the entire watershed scale, there has been a marked increase in urban cover and desert-scrub, accompanied by a decrease in cottonwood-willow riparian gallery forest, agriculture, and mesquite bosque. Grassland (35.50% of total land cover) and desertscrub (35.32% of total land cover) were the dominant land covers in the Upper San Pedro River watershed, which is in line with the prior classifications of the watershed. Due to the improved spatial resolution of our 2010 classification (30-meters vs. 60-meters), the updated 2010 classification is able to detect major roads and dry riverbeds, two features that could not be discerned in the previous classifications. Throughout the watershed, urban cover has increased, particularly in the area around Sierra Vista. In contrast, agriculture has noticeably decreased, especially in the US portion of the watershed. Riparian areas have also decreased throughout the watershed, down to 26% of the area reported in the 1997 classification.

Table 6. Proportional land cover extent as percent for the Upper San Pedro River Watershed over time (1973, 1986, 1992, 1997, and 2010). Classifications from 1973, 1986, 1992, and 1997 percentages are based on the results of Kepner et al. (2000).

Class	1973 (%)	1986 (%)	1992 (%)*	1997 (%)*	2010 (%)
Forest	1.00	1.00	0.95	0.95	2.10
Oak Woodland	12.55	12.57	12.05	12.09	12.57
Mesquite	2.74	14.14	14.01	13.41	7.71
Grassland	41.35	35.28	34.57	34.81	35.50
Desertscrub	38.99	32.11	31.25	30.26	35.32
Riparian	1.14	0.82	0.85	1.21	0.32
Agriculture	1.15	1.80	2.40	1.91	0.66
Urban	0.45	1.36	1.65	2.22	4.05
Water	0.04	0.01	0.05	0.06	0.03
Barren	0.60	0.91	0.94	0.92	1.74

[*1.28% and 2.15% cloud cover in predominantly Forest and Oak Woodland areas (1992 and 1997 classifications, respectively)]

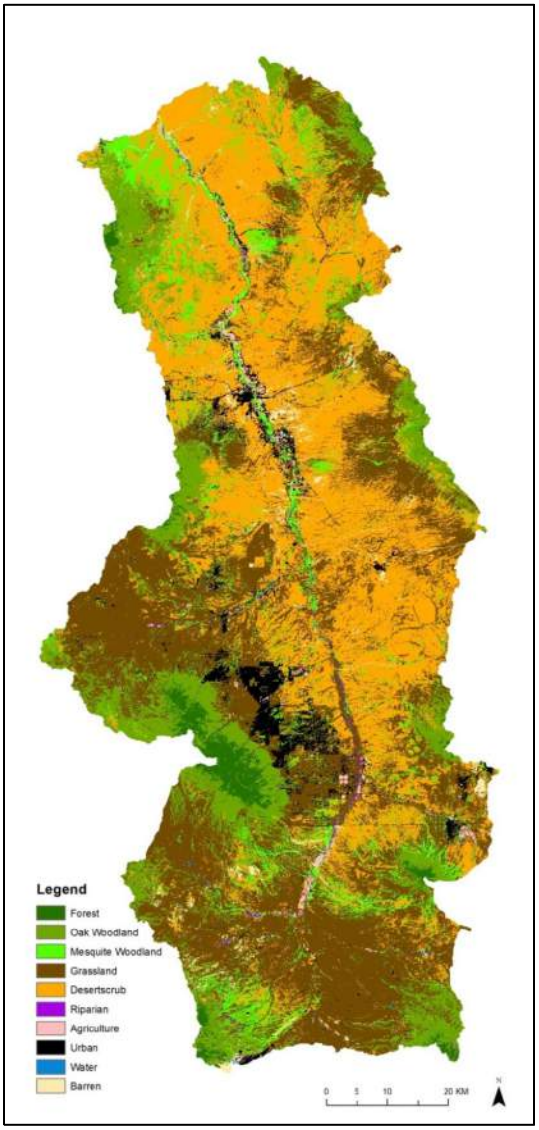

Figure 5. Land cover map for the Upper San Pedro watershed reflecting land cover conditions in 2010.

The 2010 classification reveals several trends in land cover changes that occurred throughout the watershed between the 1997 land cover map and our 2010 classification. In line with the general pattern of increasing urbanization in the Southwest (Steiner et al. 2000), particularly exurban, or "wildcat" development, there has also been a large increase in already existing urban areas. This trend is reflected in the growing populations of the urban centers in the watershed, such as the 16.2% population increase (6,113 people) in Sierra Vista seen between the 2000 and 2010 U.S. Census. Agriculture has seen a significant decrease, especially near Benson, AZ and Sierra Vista, AZ, in line with observations by Steinitz et al. (2003). Another significant trend is a significant decline in Riparian areas. Though riparian vegetation decline from 1.91% in 1997 to 0.32% can potentially be linked to a number of factors, such as natural vegetation transitions or an improvement in the classification accuracy, it may also be indicative of a declining water table due to excessive groundwater pumping, which is a trend seen in the regional aquifer (Pool and Dickinson 2006). According to the class definition we used for classification, riparian cover is principally composed of cottonwood (*Populus fremontii*) and Goodding willow (*Salix gooddingii*), which are both phreatophytic trees that require access to groundwater for their survival. If the groundwater table declines too rapidly or becomes too deep, widespread mortality of the riparian gallery forest may occur (Stromberg et al. 1996). Because the riparian gallery forest plays host to a wide array of critical ecosystem processes and functions (Stromberg and Tellman 2009), this apparent decline in the extent of the riparian corridor is cause for concern.

An interesting trend seen in comparisons between the old classifications and the new is the apparent decline in Mesquite Woodland from 13.41% in 1997 and 7.71% in 2010. Research on the persistence of woody plants (such as mesquite) and their dominance over herbaceous vegetation (Miller 2005) indicates that such a decline is unlikely to be related to an actual decrease in Mesquite Woodland on the ground. Analysis of the classification errors for the Mesquite Woodland class between the 1997 and 2010 classifications show similar producer’s accuracies (54% and 53%, respectively), but different user’s accuracies—48% in 1997, 80% in 2010. Because user’s accuracy refers to the probability that a classified pixel’s identity will correspond to the same class on the ground in reality (Story and Congalton 1986), the improvement in user’s accuracy and decline in coverage suggests the difference in Mesquite Woodland coverage between 1997 and 2010 could be attributed to the large difference in reliability of the mapped Mesquite Woodland class in the 1997 classification versus the 2010 classification.

The addition of the 2010 classification provides an opportunity to analyze recent land use and land cover changes in the watershed at an enhanced level of detail. One application is looking at the impact of the establishment of the SPRNCA in 1988 (Table 7; Figure 6). The 1986 classification shows the status of the landscape prior to the establishment of the SPRNCA. While the 1997 classification shows the status of the landscape nine years after the establishment of the SPRNCA, the 2010 classification presents a vastly different picture of the SPRNCA landscape. Though some of the change may represent an improvement in classification accuracy and detail, one impact of establishing the protected SPRNCA can be seen in the elimination of Agriculture from the area and the increase in Grassland along the river.

Table 7. Proportional land cover change within the San Pedro Riparian National Conservation Area (SPRNCA) over time.

Land Cover Class	1986	1997	2010
Mesquite Woodland	21%	20%	10%
Grassland	17%	16%	34%
Desertscrub	44%	41%	46%
Riparian	12%	14%	3%
Agriculture	7%	6%	0%
Urban	1%	2%	4%
Barren	0%	0%	3%

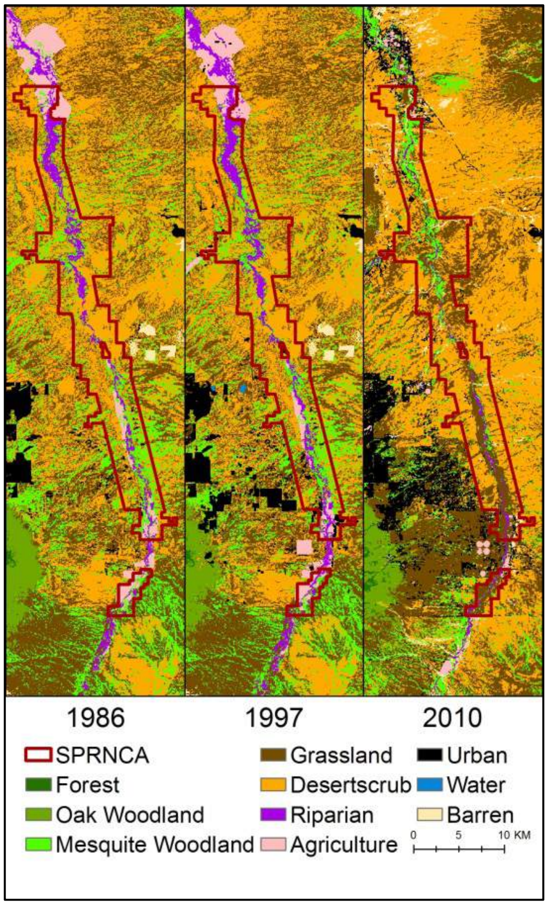

Figure 6. Land cover change in the San Pedro Riparian National Conservation Area (SPRNCA). The 1986 panel displays conditions prior to the SPRNCA's establishment (1988), while the 1997 and 2010 panels show how the landscape in and around the SPRNCA has changed since its establishment. Land cover classifications for 1986 and 1997 were created by Kepner et al. (2000).

4.3 Land Cover Change II: Spatially-Explicit Post-Classification Change Detection

The changes assessed across the entire watershed in the above section (4.2) have specific spatial distributions, which influence ecosystem function, water availability, and riparian health at the watershed scale and at a local scale, within the SPRNCA. To assess the spatial location and distribution of significant land cover change, we conducted post-classification change detection between the 2010 land cover map and the 1986 cover map, which allows for the comparison of classified land cover maps from different dates. Although the 2010 map and the previous land cover maps share an identical 10-class classification scheme, the spatial resolution of the images are not identical (60m v. 30m resolution). Thus, conducting change detection required that we re-scale the spatial resolution of the 2010 landcover classification to 60m, losing some of the image detail in the process.

The results of the change detection between the pair of years 1986 (pre-SPRNCA establishment) and 2010 (22 years after SPRNCA establishment) are presented for riparian change (Figure 7, Table 8), agricultural change (Figure 8, Table 9), and urban change (Figure 9, Table 10). The northern portion of the basin, near Fairbank, has experienced significant decreases in agriculture in the river zone, while agriculture has increased along the international border and in the riparian zone in Sonora, on the land managed by the collective ranching community, Ejido María Morelos (Figure 8). Based on the results (Table 9), the transition away from agricultural land cover is dominated by three landcover classes that are replacing agriculture: desertscrub (26%), urban (25%), and grassland (23%). The largest increases in urban cover are visible in Arizona, encompassing both urban expansion of Sierra Vista and exurban, rural development, while Sonora remains highly rural and undeveloped with the exception of two urban centers, Naco and Cananea (Figure 9). A total of 3 percent of the pixels in the image changed to urban cover between 1986 and 2010, led by the conversion of grassland to urban cover (34%) and desertscrub to urban (33%) (Table 10).

Change in riparian cover is negative in almost all areas of the river (Figure 7, red color), with the exception of small areas of increased riparian forest cover on river tributaries (Figure 7, green color), and areas of no change (Figure 7, yellow) within the SPRNCA boundaries. Only 0.22% of pixels in the image changed to riparian between 1986-2010 (Table 8), with 32% mesquite to riparian transition, 18% grassland to riparian transition, and 18% agricultural to riparian transition. The decrease in riparian cover presents a challenge to the success of establishing the SPRNCA. The largest land cover transition away from riparian was to mesquite cover (56%) (Table 8). However, the drivers of riparian change are multiple and complex and some of the change may also represent an improvement in the detail and classification accuracy of the 2010 map update, making it difficult to establish the influence of the conservation area on riparian change.

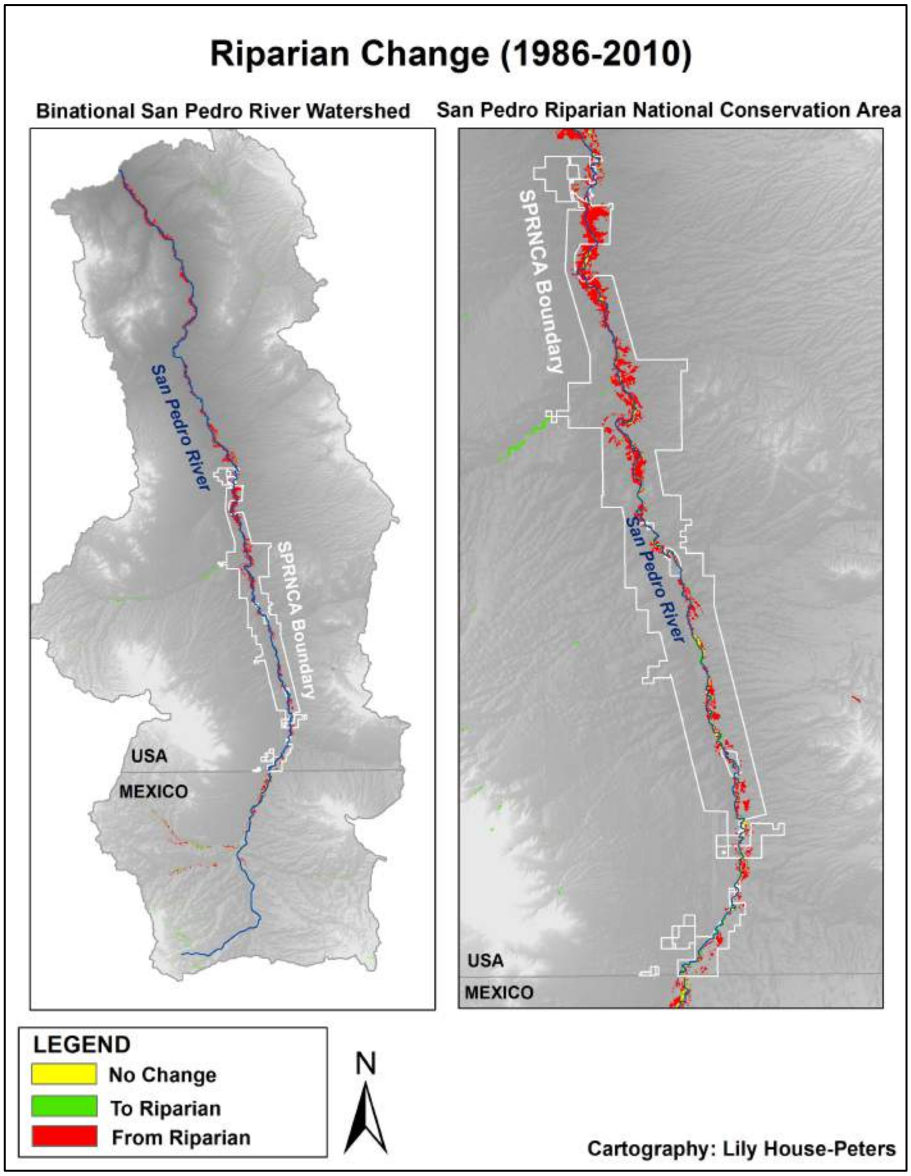

Figure 7. Left image: Riparian land cover change at the watershed scale; Right image: Riparian land cover change within the SPRNCA

Table 8. Results of the post-classification change detection for “change to” riparian and “change from” riparian between 1986 and 2010.

	Number of Pixels	**Percent of Total**	**Percent of Total Riparian Change**
Total change to Riparian:	**4739**	**0.2242**	
From Forest to Riparian	33	0.0016	0.6963
From Oak to Riparian	1007	0.0476	21.2492
From Mesquite to Riparian	1551	0.0734	32.7284
From Grassland to Riparian	882	0.0417	18.6115
From Desertscrub to Riparian	270	0.0128	5.6974
From Agriculture to Riparian	858	0.0406	18.1051
From Urban to Riparian	94	0.0044	1.9835
From Water to Riparian	9	0.0004	0.1899
From Barren to Riparian	35	0.0017	0.7386
Total change from Riparian:	**15384**	**0.7278**	
From Riparian to Forest	5	0.0002	0.0325
From Riparian to Oak	85	0.0040	0.5525
From Riparian to Mesquite	8718	0.4124	56.6693
From Riparian to Grassland	3180	0.1504	20.6708
From Riparian to Desertscrub	800	0.0378	5.2002
FromRiparian to Agriculture	579	0.0274	3.7637
From Riparian to Urban	920	0.0435	5.9802
From Riparian to Water	19	0.0009	0.1235
From Riparian to Barren	1078	0.0510	7.0073
No change:	1976	**0.0935**	

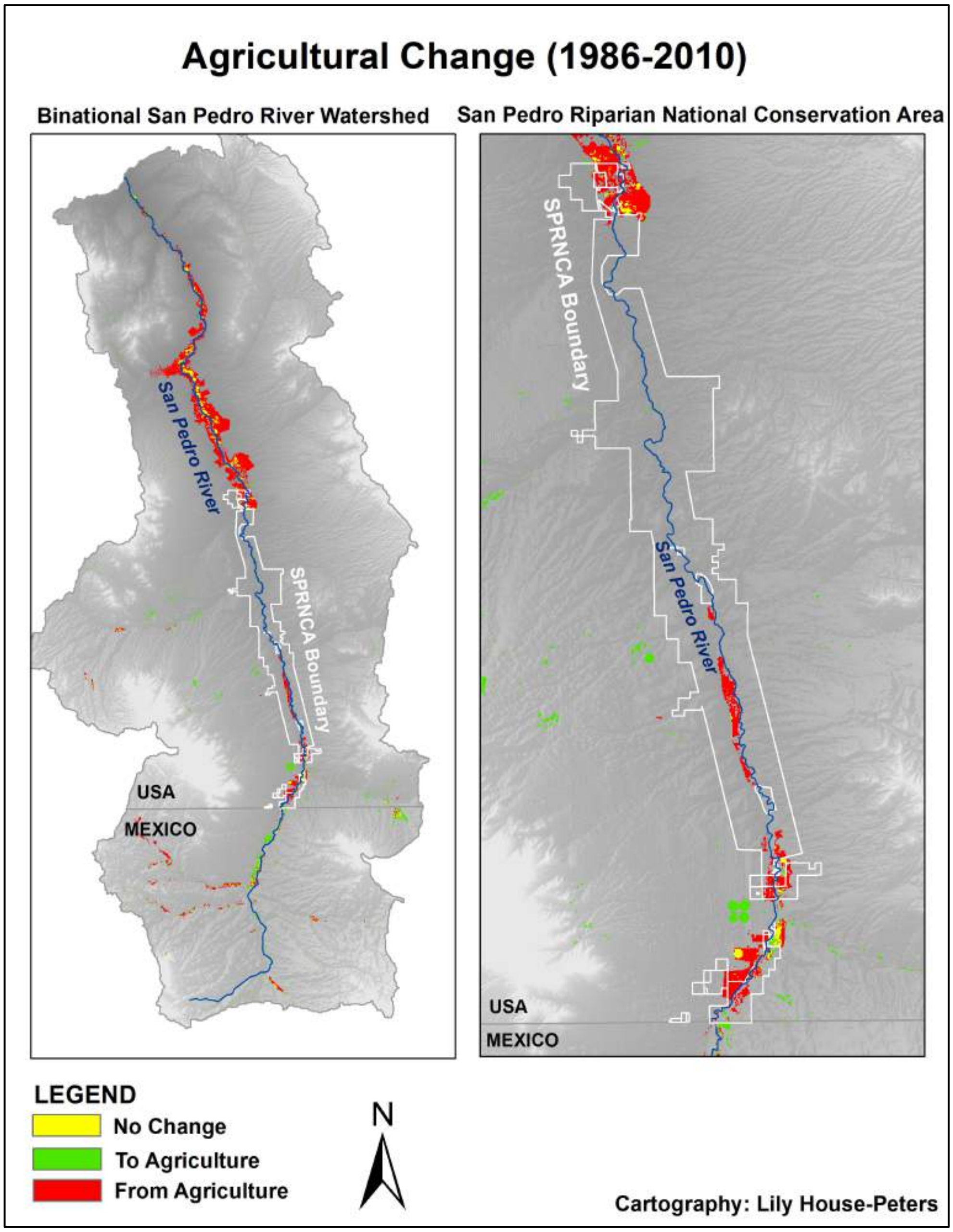

Figure 8. Left image: Agricultural land cover change at the watershed scale; Right image: Agricultural land cover change within the SPRNCA

Table 9. Results of the post-classification change detection for "change to" agriculture and "change from" agriculture between 1986 and 2010.

	Number of Pixels	Percent of Total	Percent of Total Ag. Change
Total change to Agriculture:	**8700**	**0.4116**	
From Forest to Agriculture	21	0.0010	0.2414
From Oak to Agriculture	433	0.0205	4.9770
From Mesquite to Agriculture	2164	0.1024	24.8736
From Grassland to Agriculture	3579	0.1693	41.1379
From Desertscrub to Agriculture	1387	0.0656	15.9425
From Riparian to Agriculture	579	0.0274	6.6552
From Urban to Agriculture	453	0.0214	5.2069
From Water to Agriculture	20	0.0009	0.2299
From Barren to Agriculture	64	0.0030	0.7356
Total change from Agriculture:	**32879**	**1.5554**	
From Agriculture to Forest	18	0.0009	0.0547
From Agriculture to Oak	185	0.0088	0.5627
From Agriculture to Mesquite	4367	0.2066	13.2820
From Agriculture to Grassland	7724	0.3654	23.4922
From Agriculture to Desertscrub	8653	0.4093	26.3177
From Agriculture to Riparian	858	0.0406	2.6096
From Agriculture to Urban	8427	0.3987	25.6303
From Agriculture to Water	120	0.0057	0.3650
From Agriculture to Barren	2527	0.1195	7.6858
No change:	**5184**	**0.2452**	

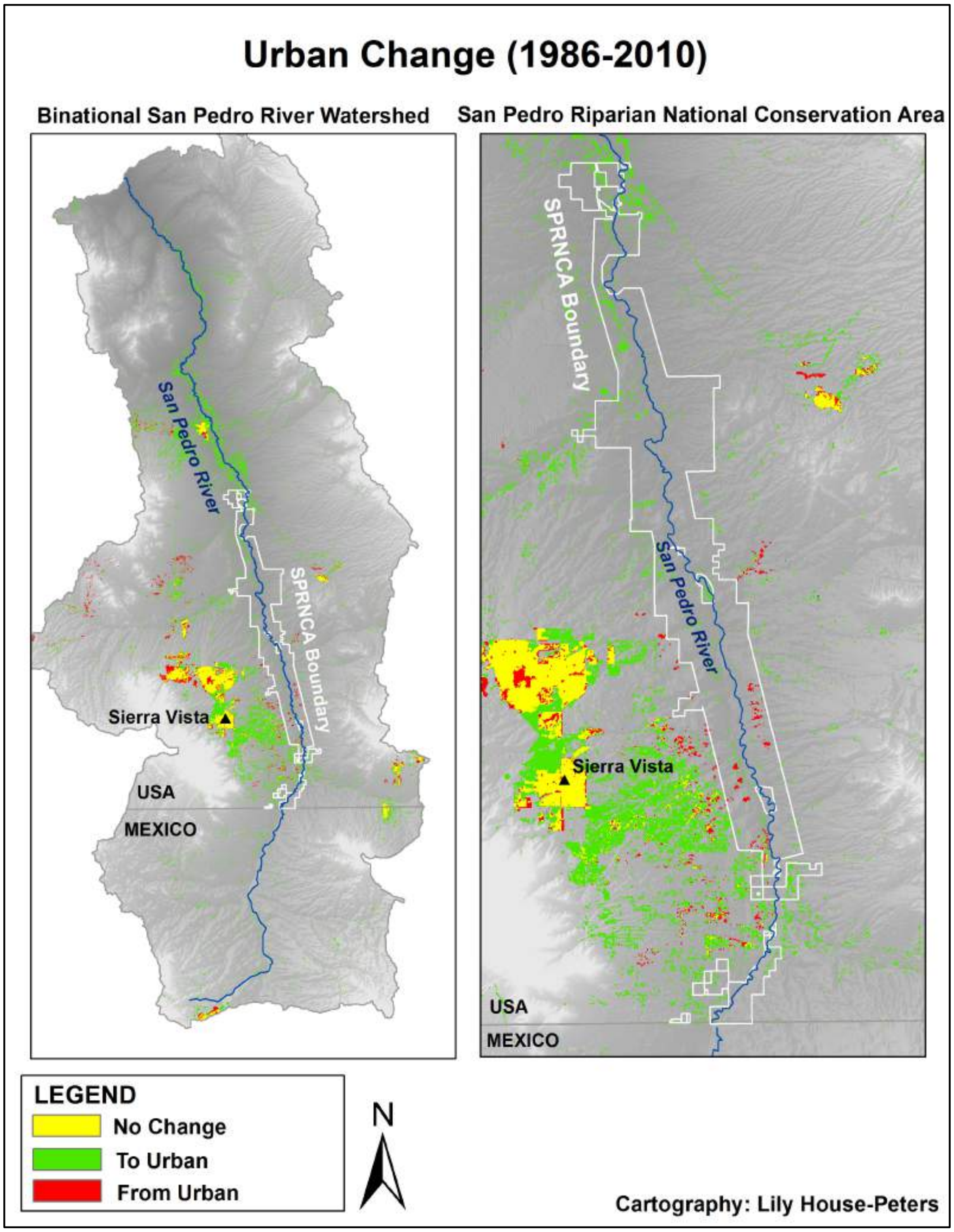

Figure 9. Left image: Urban land cover change at the watershed scale; Right image: Urban land cover change within the SPRNCA

Table 10. Results of the post-classification change detection for "change to" urban and "change from" urban between 1986 and 2010.

	Number of Pixels	**Percent of Total**	**Percent of total Urban Change**
Total Count of pixels:	2113841		
Total change to Urban:	**66991**	**3.1692**	
From Forest to Urban	7	0.0003	0.0104
From Oak to Urban	1756	0.0831	2.6212
From Mesquite to Urban	10208	0.4829	15.2379
From Grassland to Urban	22751	1.0763	33.9613
From Desertscrub to Urban	22183	1.0494	33.1134
From Riparian to Urban	920	0.0435	1.3733
From Agriculture to Urban	8427	0.3987	12.5793
From Water to Urban	12	0.0006	0.0179
From Barren to Urban	727	0.0344	1.0852
Total change from Urban:	**10056**	**0.4757**	
From Urban to Forest	2	0.0001	0.0199
From Urban to Oak	498	0.0236	4.9523
From Urban to Mesquite	957	0.0453	9.5167
From Urban to Grassland	4898	0.2317	48.7072
From Urban to Desertscrub	2719	0.1286	27.0386
From Urban to Riparian	94	0.0044	0.9348
From Urban to Agriculture	453	0.0214	4.5048
From Urban to Water	9	0.0004	0.0895
From Urban to Barren	426	0.0202	4.2363
No change:	**18641**	0.8819	

5. CONCLUSION

This research presents the results of a methodology to update a multitemporal land cover dataset that brings the temporal coverage of the entire dataset to 37 years. The results of the research provide insight into the changes that have occurred in the binational San Pedro watershed since the establishment of a local riparian conservation area in 1988. This analysis is useful for assessing spatially explicit change at multiple scales and can be integrated into scenario planning and analysis activities to inform policy and decision-making. In US-Mexico borderland region of southern

Arizona, achieving conservation goals is especially challenging. In transboundary watersheds, previous research confirms that it is imperative to understand and integrate decision-making about water and land resources on both sides of international borders, as the processes impacting water supply and demand are deeply interlinked in systems with shared surface and groundwater supplies. In arid and semi-arid region watersheds in particular there exists the further challenge of meeting diverse and often competing demands for natural resources, particularly land and water.

Despite the difference in inputs and methods between the new and old land cover datasets, comparing them can yield valuable insights and understandings as long as the limitations of comparison are kept in mind. Although consistent input data and methods across all maps in a multitemporal land cover dataset allows for a spatially explicit examination of land cover change, as seen in the NALC series of land cover maps (Kepner et al. 2000), data availability changes and methodologies change and improve over time. This work demonstrates the possibility of adding to a multitemporal land cover dataset using different techniques and input data, while still retaining some capacity to analyze trends in land cover change.

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