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Historical Analysis of Riparian Vegetation Change in Response to Shifting Management Objectives on the Middle Rio Grande

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Abstract: Riparian ecosystems are valuable to the ecological and human communities that depend on them. Over the past century, they have been subject to shifting management practices to maximize human use and ecosystem services, creating a complex relationship between water policy, management, and the natural ecosystem. This has necessitated research on the spatial and temporal dynamics of riparian vegetation change. The San Acacia Reach of the Middle Rio Grande has experienced multiple management and river flow fluctuations, resulting in threats to its riparian and aquatic ecosystems. This research uses remote sensing data, GIS, a review of management decisions, and an assessment of climate to both quantify how riparian vegetation has been altered over time and provide interpretations of the relationships between riparian change and shifting climate and management objectives. This research focused on four management phases from 1935 to 2014, each highlighting different management practices and climate-driven river patterns, providing unique opportunities to observe a direct relationship between river management, climate, and riparian response. Overall, we believe that management practices coupled with reduced surface river-flows with limited overbank flooding influenced the compositional and spatial patterns of vegetation, including possibly increasing non-native vegetation coverage. However, recent restoration efforts have begun to reduce non-native vegetation coverage.

Keywords: riparian ecosystems; remote sensing; climate fluctuation; land cover change; river management

1. Introduction

Riparian ecosystems of the Southwestern United States (USA) are considered particularly valuable for ecosystem services and diverse species habitats, yet are susceptible to various land use changes [1]. Sitting at a transition zone between the terrestrial uplands and aquatic ecosystems, riparian areas are highly productive and provide habitat for a disproportionately large range of plant and animal species in relation to their spatial coverage [2–4]. Many species are dependent on southwestern riparian habitats for survival and shelter [5], including endangered and threatened species like the Rio Grande silvery minnow (*Hybognathus amarus*) [6], southwestern willow flycatcher (*Empidonax traillii*) [7], and western yellow-billed cuckoo (*Coccyzus americanus occidentalis*) [8].

Ecological benefits of riparian ecosystems extend beyond the natural ecosystem, providing humans with services such as water for agriculture, flood and sediment control, and recreation [9,10]. River systems have long been managed by humans for their personal benefit and consumption. The Rio Grande, like many rivers of the Southwest, attracted early human settlement [11], which resulted in anthropogenic-centered river management based around agricultural land uses, including the construction of irrigation infrastructure and flood control measures. Until the late 1800s, however, many of these management practices were small-scale and susceptible to extensive damage from large floods, including the traditional New Mexican irrigation canals—acequias—and local dams [12,13].

Beginning in the late 19th century and extending into the early 20th century, water management practices were adapted as U.S. water laws began to change, highlighting the river as both a risk to society and a natural and economic resource [14,15]. This helped establish river legislative accords such as the 1922 Colorado River Compact [16] and 1938 Rio Grande Compact [17], among others. This shifting management approach additionally led to the establishment of many large-scale river management projects, including the 1905 construction of the first Bureau of Reclamation (BOR) dam, the Elephant Butte Dam [18].

These large-scale structural changes often result in undesirable, surprising, and potentially unknown consequences for the natural riparian ecosystem. In addition to direct flow alteration, river systems of the Southwest are vulnerable to biophysical changes surrounding them. These include groundwater depletion and expansion of invasive vegetation species [1]. Periods of drought have also shifted river flow patterns, particularly in arid environments such as the Southwest [19]. Hydrologic modeling on the San Pedro River in southeastern Arizona indicates that direct negative impacts of drought, such as reduced base flows, could have significant impacts on the riparian vegetation [20]. Inversely, precipitation driven flood events are essential at providing benefits for riparian zones, such as sustaining herbaceous cover and species richness [21], impacting successional cycles for the vegetation [22], as well as providing essential nutrients [23]. Although each river system will be impacted differently depending on the respective management practices applied to them and localized climate patterns, such practices will likely diminish the integrity of the larger riparian ecosystems [15].

The overarching purpose of this research was to address questions about how riparian vegetation has responded to shifting management activities and fluctuating climate by applying remote sensing and spatial analysis techniques to quantify land cover dynamics in the context of management phases [22,24,25]. We hypothesized that changes in both the native and non-native vegetation temporal and spatial dynamics were tied to changing distribution of water resources due to management decisions and fluctuating precipitation. This research addressed these questions from a coupled management and land cover approach and consequently examined the landscape response. First, we quantified the spatial, temporal, and compositional change of riparian vegetation along a portion of the Rio Grande in central New Mexico, USA over four management phases spanning a combined 79 years (1935–2014). Second, we reviewed and identified the main management activities and climate fluctuations during the study period and interpreted the extent at which these particular human and riparian systems are coupled based on temporal, compositional, and spatial change. Because river systems in the Southwest are essential for human agency, understanding the specific reasons behind these historical management decisions and their impacts on vegetation can help land managers and researchers improve efforts to conserve and restore riparian vegetation as well as predict potential future impacts. Additionally, within a Coupled Human and Natural Systems (CHANS) framework, applying this approach allows for a better understanding of not only how river management affects a natural riparian ecosystem, but reciprocally, how the changing riparian ecosystem potentially affects river management and the ecosystem services received by society [26,27]. Understanding the objectives and connected decisions behind varying management practices is an essential part of placing this research within a CHANS framework [28,29]. Petrakis et al. (2015) initially presented this research in thesis form [25].

2. Materials and Methods

2.1. Study Area

The research study area was the San Acacia Reach (SAR) of the Middle Rio Grande (MRG). The MRG is a 160-km portion of the Rio Grande in central New Mexico, stretching from Cochiti Reservoir (35°37′27″ N, 106°19′25″ W) to San Marcial (33°41′18″ N, 106°58′50″ W), and is considered to be one of the most extensive stretches of cottonwood–willow riparian forest remaining in the Southwestern USA [30]. The SAR, the southern portion of the MRG, stretches from the San Acacia Diversion Dam (34°15′20″ N, 106°53′14″ W), north of Socorro, to San Marcial, a length of roughly 65 km (Figure 1). The MRG is dependent on both summer monsoonal precipitation, which can be extreme but isolated and can provide large short-term river flows, and winter and spring storms from the Pacific Ocean, which provide snow for the surrounding high-elevation mountain ranges and result in more consistent and large spring and early summer river flows [30,31]. Climate variability can exist from season to season and year to year within the southwestern USA as a result of mid-latitude and subtropical atmospheric circulation regimes [32].

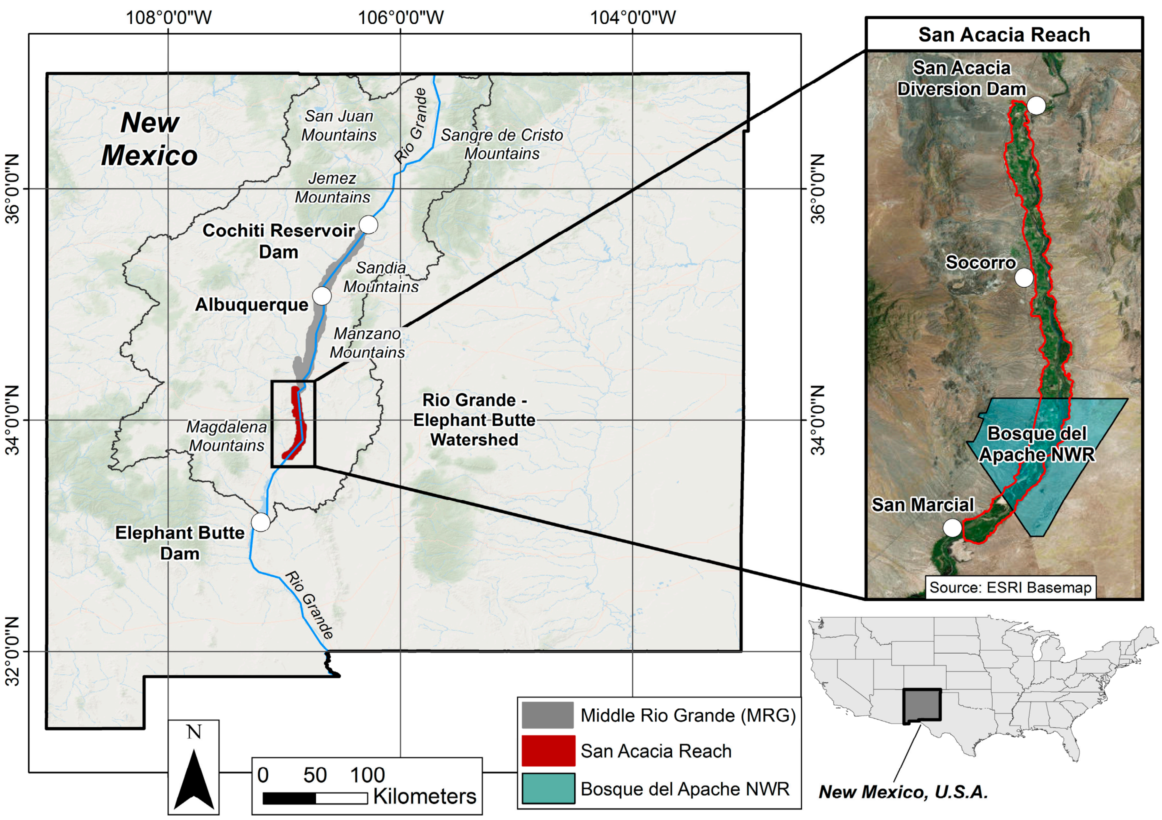

Figure 1. A satellite view of the San Acacia Reach (SAR), including the Bosque del Apache National Wildlife Refuge (NWR) and location of the San Acacia Diversion Dam, which is the focus of this research. The study area and the upstream Cochiti reservoir are located along the Middle Rio Grande (MRG) in central New Mexico, USA.

Typically dominated by the Rio Grande cottonwood (*Populus deltoids* ssp. *wizlizenii*), many other native and non-native species are present within the MRG and the SAR. Common native species considered in this study include the Coyote willow (*Salix exigua*), Goodding's willow (*Salix gooddingii*), alkali sacaton (*Sporobolus airoides*), and other ephemeral and annual grasses. Non-native species include saltcedar (*Tamarix* spp.) and Russian olive (*Elaeagnus angustifolia*), along with other invasive grass and herbaceous species [33,34].

Wildfire is a natural agent of change within riparian vegetation; however, it has become a modern disturbance, with fire recurrence intervals decreasing over the full study period leading to increased severity in areas no longer experiencing natural fire. Within the SAR, humans cause most of the floodplain fires, accounting for four out of five large fires (over 20 hectares in size) in the last 40 years of detailed documentation (personal communication). Nevertheless, riparian zones are documented to recover more quickly from fire due to ecological factors such as capability of basal sprouting, windborne seeds, and easier access to water [35]. However, the location of wildfires within the SAR is not well documented over time and was not assessed as a direct driver of riparian change in this research.

Historically, the MRG was a meandering and braided system that provided extensive irrigation for the early residents of the MRG [36]. However, sedimentation and high groundwater levels contributed to flooding and salinity issues beginning in the late 1800s, which developed into a major economic problem facing the residents [36]. Additional large-scale, weather-driven flooding events caused major damage to agriculture and infrastructure during this period [37]. In the 1920s and 1930s, levees were constructed along the historic floodplain to reduce the risk of flooding and better drain the soils, however, limiting the lateral movement of the river channel [30]. This stabilization of the overall system helped establish the current bosque vegetation [30], where 'bosque' is the Spanish and most commonly used term for the gallery forest that surrounds the river. These physical changes were among the first large-scale management practices documented within the reach, and mark the initial start date of this study.

2.2. River Management Phases

We reviewed the major coupled human and natural system interactions that have occurred within the SAR from 1935 to 2014, as well as the drivers of these interactions, in order to make interpretations and to try to better understand the spatio-temporal manifestations during each of the four specific management phases outlined in Figure 2 and discussed in detail below. This process was completed through a detailed evaluation of the temporal changes in large-scale management practices within the SAR and the distinct hydro-periods of fluctuating precipitation and river flow from 1935 to 2014. Due to the complex interactions of these human and natural agents, this mixed method review consisted of a qualitative literature review and an informal interview process, as well as a quantitative assessment of the natural river and climate dynamics [38] to build a historical geography of the Rio Grande management activities and climate fluctuations [39].

We identified four distinctive management phases as a means to isolate the effects of changing land use on the riparian vegetation. The following qualities were consistent for each phase: (1) begins and ends with a snapshot of the land cover structure and is defined by the aggregate land cover change across the entire multi-year phase; (2) aligns with major physical and/or structural changes within the reach, as well as climate fluctuations; and (3) highlights a shift in the mindset surrounding both the river and the bosque. Specific drivers are introduced for each phase as well as larger themes that are interconnected with the adjacent phases (Figure 2). Despite the unique drivers of each phase, there was a consistent requirement of water delivery for both human and agricultural uses throughout the full study period. This greatly influenced many of the management approaches used throughout the study period.

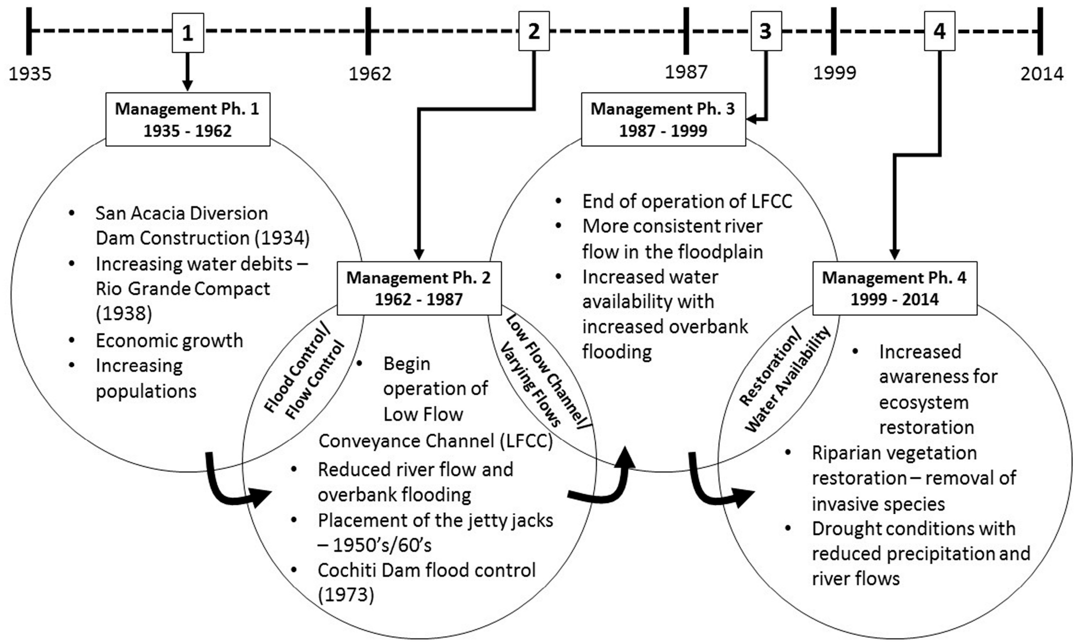

Figure 2. This diagram shows the environmental and management factors that are considered to frame and drive each of the four management phases (Ph.) between 1935 and 2014. Years along the top axis indicate image dates for change detection. Overlapping circles indicate management themes that connect each of the transitions between respective phases.

2.3. Land Cover Classifications

2.3.1. Land Cover Classification System

The classification system used in this study is a version of the National Vegetation Classification Standard (NVCS) scheme modified to represent the vegetation structure and composition within a semiarid southwest riparian ecosystem [40,41], as well as represent the vegetation species that are present within the bosque of the SAR. The hierarchy is structured by six formation classes (Table 1), the highest unit of vegetation rank and defined by dominant growth forms [40]. From there, twenty formation subclasses were included, highlighting more general diagnostic growth forms [40]. It was also imperative for this study to differentiate between cottonwood-willow dominant and non-native dominant vegetation types. Non-native vegetation influences the social, ecological, and aesthetic value of the riparian ecosystem, which is a major concern within the SAR [42]. It is well researched that flow regimes and land use contribute to non-native vegetation expansion in Southwestern riparian areas [43]. To accommodate this distinction, five additional formations, including both cottonwood and invasive dominant types, were included (Table 1). Both vegetation composition and density of cover were considered to differentiate between formation subclasses and formations (see Supplementary Materials—Tables S1/S2).

To help determine the level of tree or shrub density, we used the Crown Density Scale from the Forest Health Monitoring Field Methods Guide document produced by the Environmental Protection Agency [44]. Overall, each of the formation classes were initially based on varying levels of vegetation density to structure the subsequent formation subclasses and formations. The amount of general density of the riparian vegetation formation classes, excluding agriculture and general, is assumed to decrease in the following order: (1) forest/woodland; (2) shrubland; (3) savanna; and (4) sparse. The formation subclasses and formations are hereafter considered on the same level and referred to as 'classes' throughout the remainder of the manuscript.

Table 1. This table shows the twenty-five different land cover classes utilized in this study. There are six formation classes, twenty formation subclasses, and five formations. Depending on the outcome of the various classifications, a different number of available classes were included.

Formation Class	Formation Subclass	Formation
General		
	11—Active River Channel	
	12—Non-Active River Channel	
	13—Cleared Floodplain	
	14—Disturbance/Canal	
	15—Island	
	16—Open Water	
	17—Urban	
Agriculture		
	21—Active Agriculture	
	22—Fallow Agriculture	
Forest/Woodland		
	31—Mixed Riparian Forest	
		32—Cottonwood Riparian Forest
		33—Invasive Riparian Forest
	34—Mixed Riparian Woodland	
		35—Cottonwood Riparian Woodland
		36—Invasive Riparian Woodland
Shrubland		
	41—Mixed Wooded Shrubland	
		42—Invasive Wooded Shrubland
	43—Shrubland	
Savanna		
	51—Herbaceous	
	52—Tree Savanna	
	53—Shrub Savanna	
	54—Grassland Savanna	
	55—Wetland	
Sparse		
	61—Barren	
	62—Shrub Barren	

2.3.2. Land Cover Classification Methods

Two methods were applied to construct the land cover classifications from historical and contemporary imagery. The first method was visual interpretation and digitization of aerial imagery from 1935, 1962, and 1987. The second method used a Classification and Regression Tree (CART) model to classify multispectral Landsat satellite imagery from 1987, 1999, and 2014.

The aerial imagery from 1935, 1962, and 1987 was provided by the BOR Remote Sensing and GIS Group in Denver, CO. Multiple high-resolution single band images were mosaicked together to create full, reach-wide images for each classification year. For 1962 and 1987, the imagery was centered on the river channel and often did not span more than 2 km, which reduced our aerial coverage of vegetation for areas greater than 1 km away from the active river channel. Furthermore, the 1987 aerial imagery extends from San Marcial to ~3 km north of the town of San Antonio, reducing the aerial coverage for the SAR by roughly half. The active river channel and land cover classes on both sides of the channel were digitized, while upland vegetation was not.

The full suite of classes was used for both 1962 and 1987 digitized classifications (Table 1). Complete descriptions of the various digitized classes can be found in the Supplementary Materials

(Table S1). However, due to inconsistencies and spectral variability in the available 1935 aerial imagery, we completed two additional steps for the 1935 digitized classification. First, differences between dominant cottonwood and dominant invasive species were not discernable; therefore, we limited the possible classes to only the formation classes and formation subclasses (Table 1). Second, we used a basin structure dataset from 1935, showing the geomorphic structure of the basin, to better map the active river channel and bar islands within the floodplain [45].

CART models have proven useful for creating general vegetation classifications from spectral data [46] and have been successfully used to model changes in arid land riparian vegetation [47,48]. Additionally, they are rule-based and use independent variables and datasets to determine the correct class membership for each pixel [46]. This research utilized the SEE5 regression tree model software within ERDAS Imagine [49]. Landsat imagery, which was not available for earlier classifications, provides a less-subjective pixel-by-pixel analysis as well as greater availability of dates to capture changing phenology and temporal characteristics of the vegetation. This semi-automated methodology can support continued updating of land cover and can facilitate additional future research and monitoring in the study area.

A combination of Landsat 5 Thematic Mapper (TM) and Landsat 8 Operational Land Imager (OLI) atmospherically corrected surface reflectance imagery and vegetation indices, including the Normalized Difference Vegetation Index (NDVI) [50], Soil Adjusted Vegetation Index (SAVI) [51], and Normalized Difference Moisture Index (NDMI) [52], were downloaded (http://espa.cr.usgs.gov/) and used as the independent variables in the CART model along with the six-band Landsat image. The three vegetation indices were included because they provide unique signatures for each of the vegetation classes assessed in this research. The dependent variables were the training points that were collected for each class.

The phenology of non-native species varies from that of native species; therefore, a multi-temporal image approach was employed in order to exploit seasonal spectral variability between different vegetation types. To identify Landsat dates that matched green-up, peak vegetation, and senescence, we used 16-day composite NDVI time series data for 2000–2014 (http://daac.ornl.gov/MODIS/) acquired from Moderate Resolution Imaging Spectroradiometer (MODIS). Three 250 m pixels were selected for three vegetation types—salt cedar, cottonwood, and grass/shrub savanna. Due to spatial heterogeneity of vegetation within the riparian corridor, locations were limited and had the potential to include minimal amounts of other vegetation types. NDVI values were averaged by each 16-day composite date for each vegetation type. Relative Landsat dates were chosen to match green-up (mid-May), peak vegetation (mid-July), and senescence (late September) of the vegetation.

Training points (n = 35–405 per class) were collected for each vegetation class using a combination of aerial imagery provided by the BOR, National Agriculture Imagery Program (NAIP) color imagery provided by the Geospatial Data Gateway (https://gdg.sc.egov.usda.gov/GDGOrder.aspx), and Landsat imagery, depending on available dates. The points were placed at locations that represented the respective vegetation class in the imagery. SEE5 calculates an error value during the classification process using a percentage of the training points as validation points, instead. Depending on the year of the classification and the number of available training points, either 45% or 50% of the points were used as validation. An error value was then produced that we used as an accuracy assessment value for the classification. We provided the error values within the results and discussion section. Due to spectral similarities for multiple vegetation classes, a reduced number of land cover classes were classified using the CART modeling approach by merging classes provided in our land cover classification structure (Table 2). Complete descriptions of the various classes for the CART modeling approach can be found in the Supplementary Materials (Table S2).

Table 2. Showing the land cover classes used within the CART classification scheme.

Class ID	Class Name
	10—General
11	Active River Channel
14	Disturbance/Canal
	20—Agriculture
21	Active Agriculture
	30—Forest/Woodland
31	Mixed Riparian Forest/Woodland
32	Cottonwood Riparian Forest/Woodland
33	Invasive Riparian Forest/Woodland
	40—Shrubland
41	Mixed Wooded Shrubland
43	Shrubland
	50—Savanna
51	Herbaceous/Grassland Savanna
52	Tree/Shrub Savanna
55	Wetland
	60—Sparse
61	Barren

The second method, using a CART modeling approach, was considered as the primary method to complete the land cover classifications because it allows for a broader application to other regions of the MRG as well as other riparian systems. However, Landsat imagery was not available for the 1935 and 1962 classifications. Additionally, accurately digitizing aerial imagery is time intensive, dependent on the quality of source photos, and is challenging within a monitoring approach. A comparison of both 1987 classification methods was completed to assess comparability between methods.

2.4. Land Cover and Management Impact Analysis across Space and Time

A land cover change analysis was applied to determine the overall spatial and compositional change of the vegetation and channel structure. Each digitized land cover map was converted to a 30-m raster format to match the Landsat classifications. Land cover maps were then aggregated by management phase (i.e., 1935–1962, 1962–1987, 1987–1999, and 1999–2014). Within the aggregation, only areas of spatial overlap between the combined years were assessed. Both the digitized and CART classifications were used for 1987 to match the other classification method for the specific management phase. Change data were derived to represent change across each phase and compared to expected land cover changes from the qualitative review of management impacts in order to derive interpretations about the observed relationships.

A crosswalk was completed for 1987 to evaluate comparability and overall agreement between the digitized and CART classification methods [53]. Additionally, the extent to which the two classification methods mapped different vegetation types could also be quantified. The vegetation class structures for the CART and digitized classifications for 1987 were simplified to match directly. Therefore, a percentage of direct agreement, where both classification methods mapped the same vegetation class, could be quantified. Average agreement was quantified by calculating the average percentage of agreement between the direct agreement of the digitized classifications and the direct agreement of the CART classifications. This process could only be completed for 1987, because that was the only year in which both classification methods were completed.

3. Results and Discussion

3.1. Land Cover Accuracy Assessment

Figure 3 shows each of the land cover classifications of each year in the study period. Error values varied for the 1987 (error = 21%), 1999 (7%), and 2014 (15%) CART classifications. Based on the crosswalked map comparison, the 1987 digitized and 1987 CART classifications had a direct agreement of 50%. The active river channel and active agriculture classes had the highest percentage of agreement while tree/shrub savanna had the lowest (Table 3). The digitized classification contained over twice as much invasive riparian forest as the CART classification (Table 3), which decreased the overall agreement between the two classification techniques and suggests that pixel-based classifications may under-map non-native vegetation types such as *Tamarix* species or *Elaeagnus angustifolia*. For cottonwood riparian forest within the digitized classification, 53% was mapped as mixed riparian forest within the CART classification. When the mixed riparian forest, cottonwood riparian forest, and invasive riparian forest classes were merged to create a class that consisted of all forest types, the comparability of the combined forest class increased from an average of 39% to 75% (Table 3).

The digitized classification had greater coverage of invasive forest, active agriculture, and herbaceous/grassland savanna than the CART classification, with less coverage in the remaining classes (Table 3). This was possibly a result of in-class variability that was observed by the CART classification and not by the digitizing approach. Although fallow agriculture was digitized as an agricultural formation subclass, it was entirely classified as lower density riparian vegetation within the CART classification, including shrubland and herbaceous/grassland savanna. The active river channel and active agriculture classes have more unique spectral signatures that can be used to derive the classes within the CART model, while the riparian vegetation classes are more difficult to map based on more similar spectral responses, adding to the overall lack of agreement. Additionally, there was more pixel-by-pixel variation due to the object-based versus pixel-based techniques that were applied, which also reduced the direct agreement of the classifications. Clarification of the crosswalk technique is provided in Appendix A.

Table 3. Showing the results from the crosswalk between the object-based digitized classification for 1987 and pixel-based CART classification for 1987. The full class comparison is listed above with the simplified forest class (all three forest classes aggregated) below.

Full Class Comparison					
Class ID	**Class Name**	**Digitized Coverage (ha)**	**CART Coverage (ha)**	**Direct Agreement (ha)**	**Average of Digitized/CART Agreement (%)**
11	Active River Channel	575	623	484	81%
14	Non-active/Disturbance/Canal	512	594	336	61%
21	Active/Fallow Agriculture	952	789	731	85%
31	Mixed Riparian Forest	445	814	181	31% **
32	Cottonwood Riparian Forest	315	352	89	27% **
33	Invasive Riparian Forest	1919	876	720	60% **
41	Woodland/Wooded Shrubland	1001	1159	399	37%
43	Shrubland	23	468	14	31%
51	Herbaceous/Grassland Savanna	200	196	80	41%
52	Tree/Shrub Savanna	178	251	15	7%
** Average Forest Class Agreement = 39%					
Aggregated Forest Class					
31/32/33	Riparian Forest	2678	2041	1728	75%

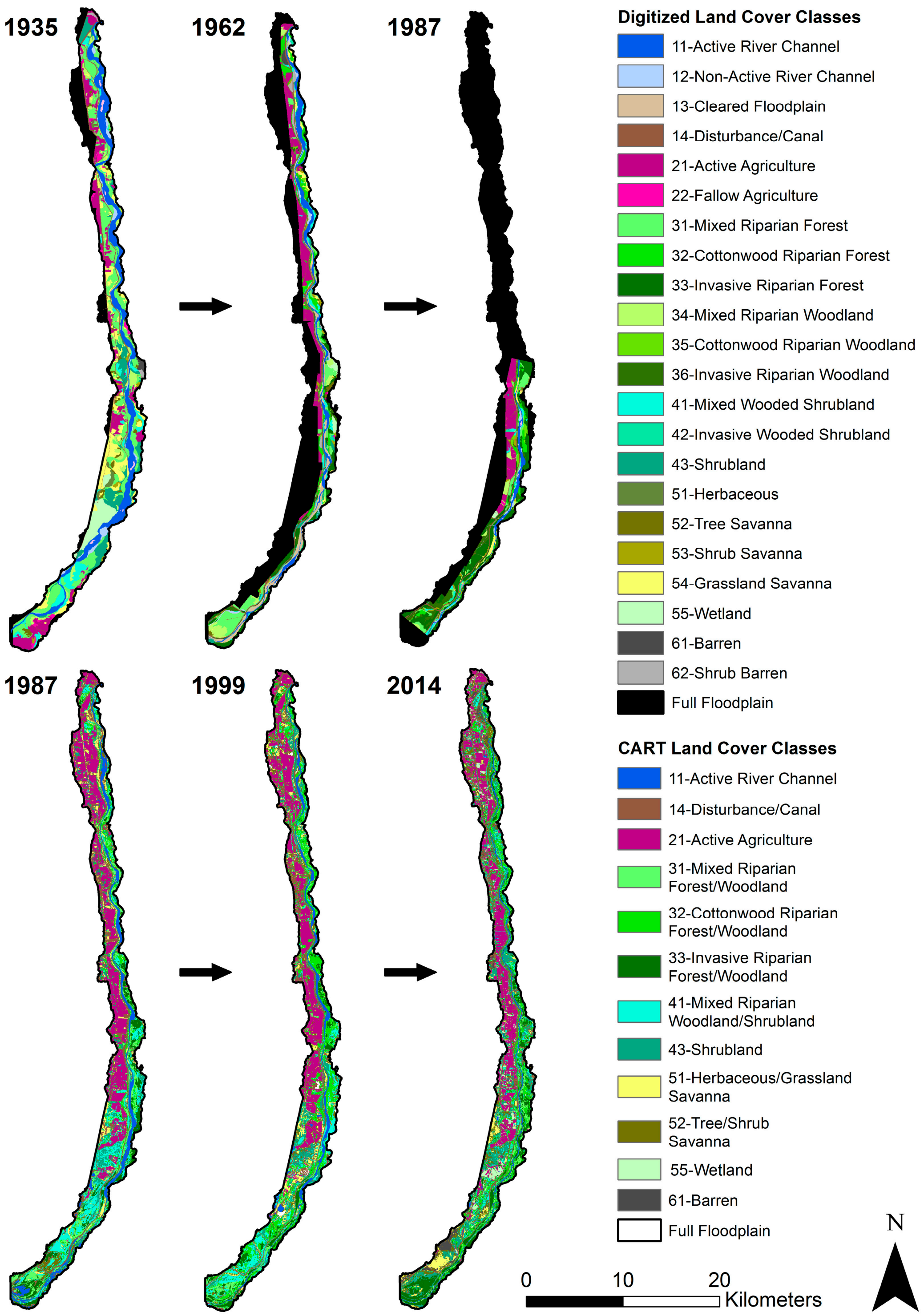

Figure 3. Showing the land cover classifications for 1935, 1962, 1987, 1999, and 2014. For the digitized classifications, the full historic floodplain was displayed in the background using black.

3.2. Shifting River–Vegetation Interactions

3.2.1. The First Management Phase: 1935–1962

The first management phase, 1935–1962, represents the time between the construction of the San Acacia Diversion Dam and the construction and initial operation of the Low-Flow Conveyance Channel (LFCC). Although designed to boost economic growth, provide sedimentation and flood control, and divert river flows down auxiliary channels [54], the San Acacia Diversion Dam also had unintended outcomes including increased riparian vegetation growth and density downstream and a locally enlarged floodplain with reduced lateral flow. Additional occasional large-scale flooding events occurred, the most severe being in 1941 and 1942, each delivering increased sedimentation from areas north of the reach. With these circumstances, coupled with severe drought in the 1950s [11], the state of New Mexico and the Middle Rio Grande Conservancy District (MRGCD) were not meeting the obligations of water compact deliveries to Texas, derived from the Rio Grande Compact of 1938. The requirements of the Rio Grande Compact were a major driving factor within this phase and throughout the full study period. In response, the BOR and the Corps of Engineers began construction of the LFCC to directly channel water from the San Acacia Diversion Dam to Elephant Butte Reservoir. Because construction of the LFCC began at the end of the first management phase, only the structural changes were in place by 1962, while impacts from the initial operation of the LFCC on the vegetation classes likely had not yet become established. We observed four major physical land cover changes during this first phase.

First, the disturbance/canal class, including the LFCC, increased by nearly 200% (1935 coverage: 351 ha–1962 coverage: 1051 ha; or, 351–1051 ha). The LFCC extended the entire length of the SAR and was ~150 m wide when combined with the pre-constructed levee. Additionally, the LFCC was mostly converted from both active river channel (hereafter referred to as the active channel) and riparian forest as it cut through the historic floodplain to straighten and limit the lateral movement of the 1962 active channel. The land on the inside of the LFCC became known as active floodplain while the land outside of the LFCC became detached.

Second, active agriculture increased by an additional 486 ha during this phase (1133–1619 ha), or 43%. Two factors likely helped institute this change. First, the San Acacia Diversion Dam and levee systems provided immediate greater overall flood control. Second, by lowering the water table, the LFCC reduced salinization issues that had plagued the reach for years, although the drop in the water table also negatively impacted the native vegetation [11]. Additionally, lesser local canal systems scattered throughout the historic floodplain, including both acequias and interior drains, also helped drain the soils and reclaim waterlogged agricultural fields [12]. Much of this agriculture was converted from riparian forest (753 ha), some of which had been separated from the active floodplain due to the LFCC construction.

Third, beginning in the late 1950s and early 1960s, the MRGCD and BOR undertook the mechanical clearing of a ~200 m width area of land immediately surrounding portions of the active channel. It is believed this was completed to destabilize the active floodplain, allow for a less incised channel, and remove salt cedar [55–57]. This area of land is referred to as the cleared floodplain.

Present only in the 1962 classification, the cleared floodplain covered 934 ha. The specific location for this clearing did result in the reduction of both the riparian vegetation and the active channel in those areas. First, vegetation was cleared within this management practice, as portions of shrubland (116 ha) and riparian forest (102 ha) were converted to cleared floodplain. Second, 388 ha (14%) of the 1935 active channel was converted to cleared floodplain. An additional 124 ha of active channel was converted to the non-active channel, which was not manually cleared and exhibited spectral or textual indicators of being part of the natural flow of the river. Overall, the non-active channel and cleared floodplain had a combined increase of 304% by 1962 (282–1140 ha).

Fourth, the installation of the jetty jacks along the river banks also occurred during this phase. These 3 m tall metal structures trapped debris and sediment on the outside of their line, limiting the width and movement of the channel by creating higher banks restricting overbank flooding as well as holding seed for vegetation growth by stabilizing large vegetation banks [11]. With these impacts, an increase in the general density of the vegetation coverage was expected [58]. This likely did occur during this phase and was evidenced by changes in certain classes from lower density vegetation formation classes to classes within higher density formation classes. For instance, shrubland and grassland savanna experienced conversion to higher density classes. For shrubland, 238 ha were converted to riparian forest types including mixed forest (74 ha), cottonwood forest (23 ha), and invasive forest (141 ha). For grassland savanna, 160 ha were converted to a mixture of mixed forest

(18 ha), cottonwood forest (18 ha), invasive forest (78 ha), mixed wooded shrubland (15 ha), invasive wooded shrubland (24 ha), and shrubland (7 ha).

A confusion matrix listing the full changes that occurred to and from each class throughout the phase can be found in the Supplementary Materials (Table S3).

3.2.2. The Second Management Phase: 1962–1987

The second phase from 1962 to 1987 represents the time in which the LFCC was in operation. Two major shifts in river flow patterns likely influenced vegetation change during this phase. First, the LFCC often held the entire flow of the river, resulting in intermittent flows within the active river channel with extended periods without surface water through the early 1980s. This was a management approach to increase water deliveries and meet obligations of the Rio Grande Compact and resulted in the elimination of base flows and the reduced reoccurrence of overbank flooding. With the bed of the LFCC sitting 3–6 m below the river surface, it also pulled groundwater from the aquifer [11], although this was exacerbated due to river aggradation over time and was not as extreme during this phase. These river flow characteristics are known to directly impact vegetation composition [59] and are positively correlated with increases in invasive vegetation such as *Tamarix* sp., which have deeper and thicker roots [60]. Second, beginning in the late 1970s, yearly precipitation increased and river flows reached higher levels (Figure 4) resulting in the filling of Elephant Butte. The Palmer Drought Severity Index (PDSI), a regional index of both drought length and intensity [61,62], documents 121 of the 192 months (63%) between 1962 and 1977 as below normal, or experiencing drought conditions (Figure 4), likely leading to reduced river discharge. Starting in the late 1970s, 96 of the 120 months (80%) between 1978 and 1987 had documented above normal conditions (Figure 4), indicating non-drought conditions for a majority of the time, which would likely equate to increased river discharge. Many factors led to this, including a strong El Niño in the winter of 1982 (Figure 4). By 1985, all diverted flows down the LFCC were cancelled due to excess sedimentation in the Elephant Butte delta [63], although the LFCC continued to draw from the groundwater throughout the study period. With more consistent and less severe spring run-off flooding, cottonwood trees can become established [64]. However, periods of intermittent tributary inflows during the monsoon season can also lead to an increase in invasive species, especially *Tamarix* species [65]. This phase likely experienced a mixture of river flow conditions that we believe would allow for limited cottonwood tree establishment as well as increases in invasive species coverage.

Simultaneously, two physical changes were occurring within the larger MRG basin during the 1970s. First, the construction of Cochiti Reservoir Dam (Figure 1) in 1973 provided flood and sediment control within the full MRG and the SAR [66]. Second, the mechanical clearing of vegetation along the floodplain was discontinued during this phase, although an exact date is unknown (personal communication).

These two physical changes and coupled shifts in river flow patterns had four major impacts on the riparian vegetation within the management phase. First, invasive forest more than doubled in coverage during this phase (682–1562 ha). This includes conversion from nearly all other classes, likely in response to reduced flow dynamics during this phase. Combined, 191 ha were converted from active channel (53 ha), non-active channel (9 ha), cleared floodplain (100 ha), and disturbance/canal (29 ha). Conversion from mixed forest (358 ha) and limited amounts of cottonwood forest (25 ha) also occurred. Additionally, vegetation types from lower density formation classes, such as mixed wooded shrubland (72 ha), invasive wooded shrubland (47 ha), and shrubland (106 ha) were converted to invasive forest. Shrub savanna (19 ha) and wetland (66 ha) accounted for conversion from the savanna formation classes. Coverage of invasive woodland also increased by 225% during this phase (100–325 ha). For invasive woodland, 200 ha were converted from mixed riparian forest. We believe that these increases in invasive forest and woodland were a direct result of the lack of surface water availability and reduced overbank flooding within the first portion of this phase, and indirectly driven by river flow controls such as the LFCC and the Cochiti Reservoir Dam.

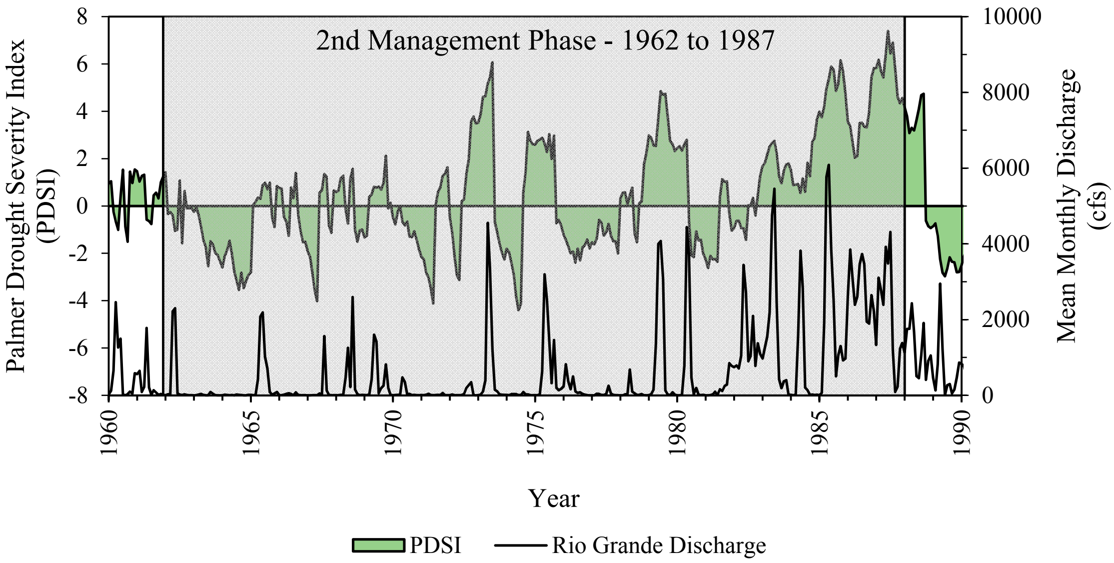

Figure 4. Monthly average Palmer Drought Severity Index (PDSI) data for New Mexico and mean monthly river discharge (cubic feet per second) for the second management phase (1962–1987) at the San Acacia Floodway gage located at the San Acacia Diversion Dam. Occasional flooding events led to periods of large river discharge (>2500 cfs). More consistent higher discharge rates align with above normal PDSI conditions beginning in the late 1970s.

Second, cottonwood forest only increased slightly during this phase (212–309 ha), possibly due to increased river flows near the end of the phase. Cottonwood forest was converted mainly from mixed forest (118 ha), invasive forest (18 ha), as well as cleared floodplain (21 ha) and disturbance canal (12 ha). Some cottonwood forest was also converted to other classes during this phase. Cottonwood riparian woodland coverage also increased (8–103 ha). Conversion to cottonwood woodland occurred from mixed wooded shrubland (33 ha), mixed forest (16 ha), cottonwood forest (24 ha), and invasive forest (16 ha). We believe that over the full length of the second management phase, the long-term physical and management processes likely helped establish increased coverage of invasive vegetation rather than cottonwoods, as evidenced by a larger increase of invasive vegetation.

Third, the mechanical clearing of the historic floodplain, a region known as the cleared floodplain, ceased during this phase, allowing for 15% of the historic floodplain to eventually convert to other land cover types. The largest conversion from the cleared floodplain was to the active channel (210 ha). This was a direct response to its management design to increase lateral movement and reduce incising of the active channel [57, personal communication]. However, much of the cleared floodplain was also converted to vegetation classes of varying densities. For instance, it was converted to low to moderate density vegetation classes including grassland savanna (108 ha), mixed wooded shrubland (46 ha), and shrub savanna (21 ha). Conversion to high density classes including invasive forest (100 ha), mixed forest (21 ha), and cottonwood forest (21 ha) also occurred. Overall, this managed clearing left large portions of open land directly along the active channel vulnerable to the growth and densification of the vegetation and spatial expansion of the active channel.

Finally, general increases in the vegetation density also occurred during this phase. Shrub savanna coverage more than doubled (48–97 ha), including conversion from the cleared floodplain (14 ha), active channel (19 ha), non-active channel (24 ha), and shrub barren (24 ha). Inversely, some shrub savanna was converted to other classes during the phase including conversion to mixed forest (10 ha) and invasive forest (19 ha). Grassland savanna also experienced a large increase in overall coverage (37–151 ha). Similar to shrub savanna, 123 ha were converted to grassland savanna from the former active channel (15 ha) and cleared floodplain (108 ha). We believe that these increases in vegetation density were influenced by both a reduction of surface water on open areas of land, allowing for periods of vegetation growth on portions of the active channel that are not washed out by high river flows, as well as increased precipitation beginning in the late 1970s. Due to the construction of the Cochiti Reservoir Dam, increased precipitation did not commonly result in flows capable of removing vegetation.

A confusion matrix listing the full changes that occurred to and from each class throughout the phase can be found in the Supplementary Materials (Table S4).

3.2.3. The Third Management Phase: 1987–1999

The third phase, 1987–1999, was characterized by increased precipitation and water availability within the SAR. Beginning in the early 1980s and extending through the late 1990s, the volume of Elephant Butte Reservoir ranged between 60% and 100% capacity (1200–2000 thousand acre feet). Additionally, overbank flooding was occurring more often during this phase. At the U.S. Geological Survey San Marcial floodway river gage (33°40′44.7″ N, 106°59′49.2″ W), overbank flooding, above 2500 cubic feet per second (cfs), likely occurred on at least 12 instances within this phase, occasionally lasting more than two months. Native vegetation species within Southwestern riparian ecosystems, such as the cottonwood, are adapted to spring run-off overbanking or flooding [67,68]. Flooding and overbanking is necessary for establishment of cottonwood seedlings [65,67,69].

With increased water availability, the vegetation changes were threefold. First, cottonwood forest coverage increased by a total of 166% (1151–3059 ha). Much of this conversion (88%, or 1686 ha) was from either mixed forest (653 ha), invasive forest (405 ha), or mixed woodland/shrubland (628 ha). With overbank flooding occurring yearly, for multiple months at a time, this provided an opportunity for new cottonwood establishment, particularly in less densely vegetated areas including shrubland and woodland. Young seedlings can germinate within 2–3 days and are supported by consistent water availability in the following years [69]. Along the Rio Grande, cottonwood forest establishment can occur within three years of an overbank flooding event, usually located in areas moistened specifically due to flooding events [69–71]. Conversion to cottonwood forest also occurred from disturbance/canal (188 ha), active agriculture (236 ha), and shrubland (208 ha).

Second, invasive forest decreased (1481–998 ha). A total of 405 ha of invasive forest was converted to cottonwood forest with 272 ha converted to mixed woodland/shrubland. This was possibly due to the river flow patterns during this phase, increasing cottonwood forest establishment. Non-dominant invasive forests can convert to native forests over time during optimal water flow conditions [72], such as during this phase. Additionally, because most contemporary high flows are not sufficient to remove large tracts of mature forest, either cottonwood or non-native, it is possible that potential mapping issues of invasive forest in either the 1987 or 1999 classifications influenced this number, as pixel-based classifications may under-map non-native vegetation types. Because it would take several years for the imagery to clearly show developed cottonwood vegetation, some of the cottonwood establishment may also have occurred during the wetter, later years of the second management phase. Finally, fire could lead to a conversion to mixed woodland/shrubland. Unpublished data from the New Mexico State Forestry Division shows that the return interval for bosque fires is approximately seven to 10 years in areas with dense, primarily non-native *Tamarix* spp. stands (personal communication). However, fire was not directly addressed in this research.

Third, lower density vegetation classes, including herbaceous/grassland savanna and tree/shrub savanna, increased in total vegetative density. This was evidenced by conversion of these classes to higher density vegetation classes such as shrubland and mixed wooded shrubland. Similarly, shrubland was mostly converted to cottonwood forest (208 ha) and mixed woodland/shrubland (246 ha), both suggesting increased tree density in these locations. Coverage of both of the herbaceous dominant classes also increased across this phase, with herbaceous/grassland savanna increasing by 137 ha (1140–1277 ha) and tree/shrub savanna increasing by 577 ha (513–1090 ha). Much of this increase for both herbaceous classes was from a combination of disturbance/canal and active agriculture, which was likely an artifact of the introduction of ground-level vegetation on former agricultural lands and previously barren disturbance and canal sites.

Inversely, highlighting a decrease in density, shrubland was also converted to herbaceous/grassland savanna (148 ha) and tree/shrub savanna (189 ha). Factors driving a decrease in density of shrubland may include the presence of fire on the landscape as well as natural vegetation change, including successional changes due to periods of drought or even large-scale flooding events capable of removing shrub vegetation.

A confusion matrix listing the full changes that occurred to and from each class throughout the phase can be found in the Supplementary Materials (Table S5).

3.2.4. The Fourth Management Phase: 1999–2014

Initiated during the third management phase was a rational shift surrounding riparian ecosystem restoration [73,74]. Despite the timing in this shift, much of the resulting impacts on the vegetation did not become apparent until the fourth phase, 1999–2014. This was also a period of low precipitation, drought, and decreased spring run-off flow duration and intensity. Limited overbank flooding did occur, particularly during the strong spring run-off of 2005 and summer monsoons of 2005 and 2006, but it was less prevalent and tended to favor single monsoon events. With this reduction in spring run-off flows, similar to the second management phase, changes in the amount of coverage of moderate to low density riparian vegetation classes and increases in invasive forest and woodland were expected.

By 2014, invasive forest nearly doubled (998–1975 ha). Much of this increase in invasive forest was from mixed woodland/shrubland (274 ha) and mixed forest (293 ha). Furthermore, cottonwood forest decreased during this phase (3059–1906 ha). Cottonwood forest was mostly converted to the other forest classes including mixed forest (335 ha) and invasive forest (476 ha). Additionally, large areas of cottonwood forest were also converted to woodland/shrubland (248 ha) and shrubland (447 ha). Indicating a decrease in density, this was potentially due to wildfires within the bosque, uncertainties in the mapping process, or limited die-offs due to occasional periods of no river flow, which had been documented for a limited amount of 10–20 year-old trees during this phase.

Increased amounts of vegetation restoration also occurred during this phase, particularly in response to existing or expanding salt cedar stands. Although limited management efforts to control salt cedar have been ongoing since the 1940s along other major southwestern rivers including the Salt River, the Colorado River, and the Rio Grande [57], project scale and funding were limited. This began to change in the 1990s, particularly in the Bosque del Apache National Wildlife Refuge (NWR) where locally large-scale restoration was underway. Researchers, the public, and land managers became invested in not only restoring the most effective stream and flow patterns, as done in early restoration efforts, but also helping re-establish the ecosystem to represent the time before anthropogenic disturbance, which consisted mostly of a mosaic of wetlands, forests, shrubland, and savannas [30,74]. This highlighted a shift within the CHANS framework, in which changes in the riparian ecosystem began to affect river management practices. Although water delivery for both human and agricultural needs was still considered, the focus of riparian restoration shifted to environmental or ecological aspects. Much of this evolution towards more modern restoration efforts was initiated due to reasons including awareness of fire risk, aesthetic habitat experience, water conservation, species protection, and bank stabilization [75]. Nevertheless, much of all river or riparian restoration efforts are small and spatially limited within the overall riparian systems [75].

Within the SAR, private land owners have begun to invest in restoration efforts with help from local groups and federal agencies. In an area south of Socorro known as the Rhodes Property, which covers 215 hectares, there was the systematic removal of invasive shrubland along the privately owned area on the east side of the floodplain (personal communication). This clearing occurred in 2008, two years after a fire burned the vegetation in 2006 (personal communication). Pre-fire, this region consisted of cottonwood and mixed riparian forest with patches of invasive forest. Post-fire, this region showed high invasive vegetation re-growth. Cleared mechanically in 2008, the restoration work developed habitat consisting of a mixture of shrubland, mixed woodland/shrubland, herbaceous/grassland savanna, and cottonwood forest by 2014.

Similarly, in a riparian area inside the active floodplain region of the Bosque del Apache NWR, an area of invasive forest was cleared in 2007 (personal communication). As a result of previous restoration work in an adjacent area in the 1990s, it was expected that the vegetation would be converted to a mixture of grassland and denser native forest, if overbank flooding occurred [76]. Large overbank flooding did occur in 2008 on this plot (personal communication), resulting in a mosaic of native grasses, shrubs, and young cottonwoods. Although these examples are limited in geographic scale, similar projects have been aimed at reducing invasive vegetation coverage across the reach.

A confusion matrix listing the full changes that occurred to and from each class throughout the phase can be found in the Supplementary Materials (Table S6).

4. Conclusions

Our goal here was to make interpretations based on the observed changes in spatial, compositional, and temporal land cover and the shifts in management and climate fluctuations derived during our qualitative review. We believe that each of the four river management phases reviewed in this research revealed unique representations of the coupled larger ecosystem and the general human needs. In response, vegetation and the surrounding land cover classes were also impacted differently depending on the multifaceted goals of the management efforts and the climate fluctuations. The first phase, focusing on anthropogenic control of the river, resulted in a more efficient floodplain with limited lateral movement and increased agricultural coverage within the floodplain. The modern bosque was established during this phase. The second phase was driven by the requirements of the Rio Grande Compact coupled with drought conditions, resulting in extended periods without surface water flow. This is likely when a large amount of non-native vegetation was established within the bosque. The third phase was defined by a period of spring overbank flooding, due to increased precipitation and mountain snowpack, which likely led to increases in the native cottonwood riparian forests and an increase in the overall density of the vegetation. The fourth phase, similar to the second phase, experienced drier periods with reduced snowpack run-off and increased monsoonal inflow resulting in documented increases in non-native invasive forests and mixed woodland/shrubland. Additionally, restoration projects began to be initiated during this phase. The addressed changes for each phase are shown in Figure 5.

Despite the occurrence of the expected responses to the management efforts and climate fluctuations, this research highlighted the dynamic characteristics of this riparian ecosystem along the Rio Grande and its ability to maintain some basic resilience, as the native gallery forest continued to exist throughout the study period despite the shifts in management and fluctuations in climate. With the various changes that occurred over the full study period, the ecosystem retained much of its vegetative diversity and adapted in a variety of ways. However, there are many questions surrounding the ability of riparian ecosystems to continue to exist in the same manner. Climate projections from the 2013 National Climate Assessment for the Southwest indicate increased temperature, decreased snowpack and runoff, as well as increased risk of monsoonal flooding events in the Southwest [77]. Each of these climate projections can directly impact the river flow regimes [78], which can influence riparian vegetation change as well as influence anthropogenic-based management practices within the Rio Grande and other Western river systems. With increased groundwater pumping due to expanding urban centers and the increased risk for intermittency of the river, drastic vegetative changes within the ecosystem may occur [79].

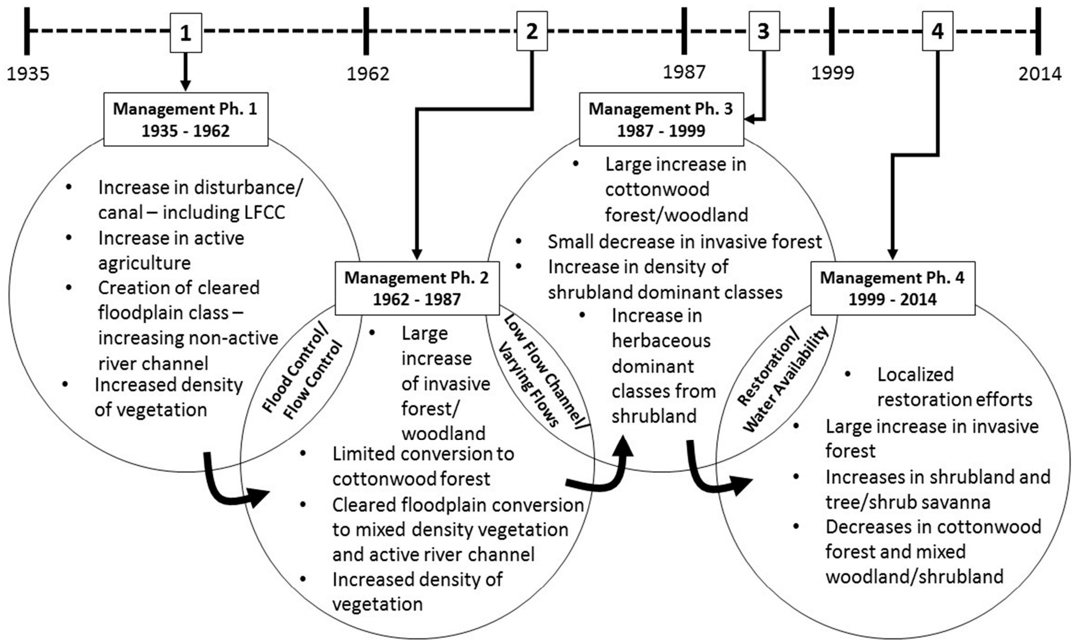

Figure 5. This diagram lists the main vegetative and structural changes that occurred within each management phase, summarized from the land cover results (circles). It also includes the themes that interconnected the adjacent management phases.

Many of the management practices assessed in this research were responsible for changing the physical structure within the system. Figure 6 shows a comparison between the generalized cross-sections of the pre-1935 and 2014 floodplains. The most significant change was in relation to the limitation of both the river channel and the extent of the active floodplain. With the construction of the LFCC, the active floodplain was limited to its boundary with the LFCC on the West and the upland areas on the East. This resulted in a more incised river channel with limited or no lateral movement, which influenced both the vegetation structure and composition. Similar structural controls are occurring or have occurred in many Southwestern river systems including the Santa Cruz and Colorado rivers in Arizona [39,80].

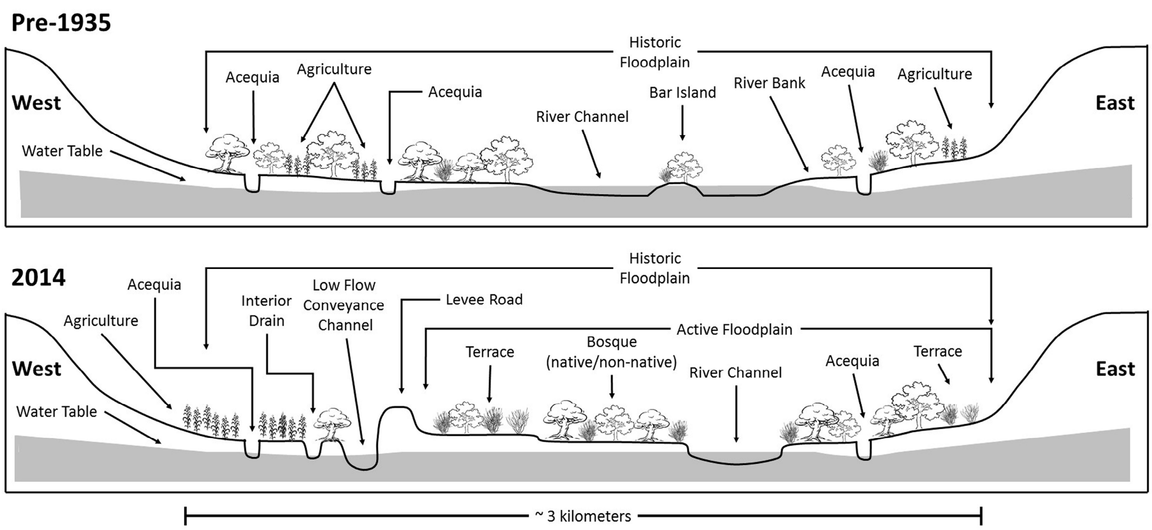

Figure 6. This figure shows a generalized cross section of the floodplain for both pre-1935 and 2014, looking north. There were major changes within the structure of the floodplain across the full study period.

This research applied a multi-method approach to determine the land cover classifications and change. This was necessary due to the imagery available for each specific classification year. A large effort was required to accurately digitize three different years. This object-based method was subjective in nature, and considered the vegetation within the larger surrounding natural contexts. It was less detailed than the pixel-based CART modeled classifications. Although less subjective, depending on the collecting of training points, the CART models classified each pixel separately. The major difference in method between the two classification approaches likely resulted in a moderate percentage of agreement, particularly due to the inconsistences of forest and shrubland mapping.

Although this research reviewed changes in large-scale management practices, each lasting multiple decades, it is important to highlight that management is continuous. The agencies that manage the qualities of human, river, and ecosystem health along the Rio Grande, continuously make changes in response to the current characteristics of the river. As management goals shift to highlight more of a coupled human and natural system in which habitat restoration and species protection are fundamental, influences on the vegetation will also shift. Furthermore, with the initiation of small-scale restoration efforts located along specific and confined locations of the reach, large-scale changes will likely become less pronounced.

We believe that future application of these or similar methods should be applied along this and other portions of the Rio Grande as a way to quantify the impacts of shifting management objectives. Additional spatial layers, including Light Detection and Ranging (LiDAR), may provide more accurate classifications of this riparian ecosystem [81]. With world-wide Landsat coverage, this approach can be also applied globally. Despite the focus on one river system, the larger goal of this research was to highlight the history of management practices and their influence on the spatial and compositional aspects of riparian vegetation of an arid, Southwestern river basin. As crucial ecosystems for human recreational use, aesthetics and service, as well as plant and animal species, understanding how these habitats have evolved in response to management and climate fluctuations is essential. It also allows for land managers, policy makers, and stakeholders to have the opportunity to communicate and determine better methods for management practices [82].

Supplementary Materials: The land cover classifications were published in ScienceBase by the U.S. Geological Survey (DOI tag: 10.5066/F7154F84). The following are available online at www.mdpi.com/2073-445X/6/2/29/s1. Table S1: overview of the land cover class structure for the digitized classifications, including descriptions of the vegetation characteristics that define each class. Table S2: overview of the land cover class structure for the CART based classifications, including descriptions of the vegetation characteristics that define each class. Tables S3–S6 provide confusion matrices of all the changes that occurred between classes for each phase. Table S3 shows changes that occurred during the first management phase. Table S4 shows changes that occurred during the second management phase. Table S5 shows changes that occurred during the third management phase. Table S6 shows changes that occurred during the fourth management phase.

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Appendix A.

We used a crosswalk technique to determine the comparability between the digitized and CART classifications [45]. We merged the two classifications together to quantify the amount of overlap between each land cover class as well as the total area of disagreement between the two classifications. The crosswalk spatial agreement and disagreement is shown in Figure A1. Only regions of direct overlap could be compared; therefore, we were limited by the extent of the 1987 digitized classification, which was limited by the extent of the available 1987 aerial imagery (Figure A1). Much of the disagreement between the two classifications was a result of the variability in mapping techniques, by comparing a homogeneous polygon map with a pixel-based raster image (Figure A1).

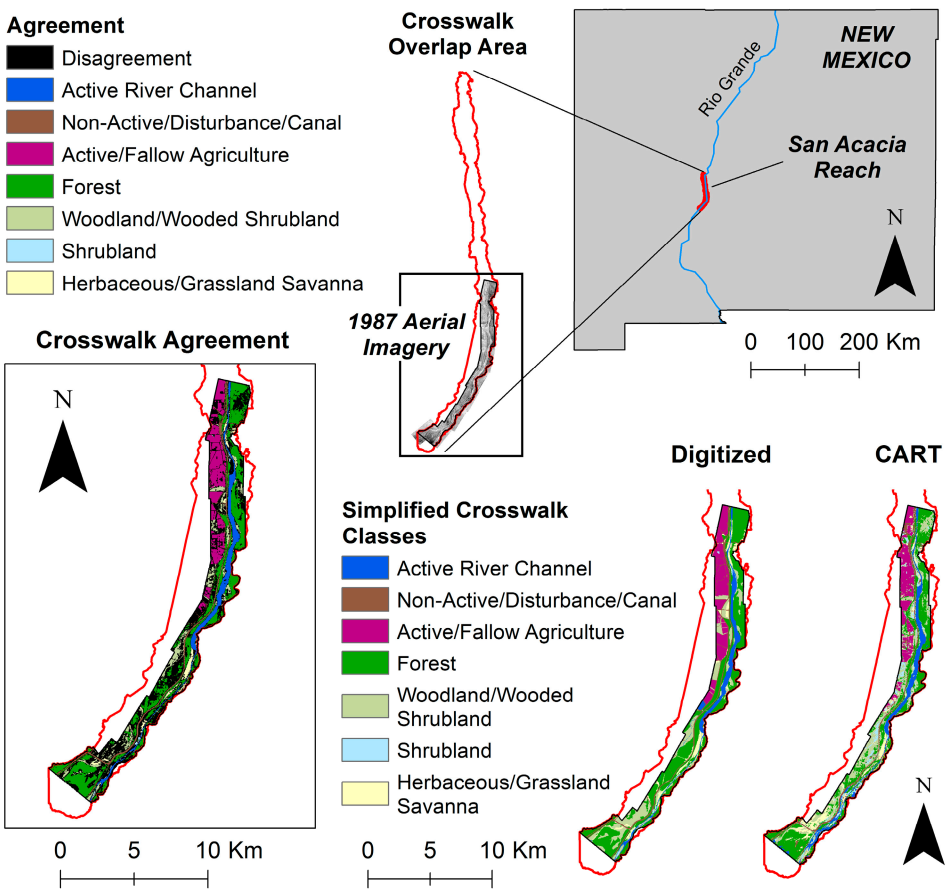

Figure A1. Showing the extent of both the 1987 digitized classification and the 1987 CART classification in relation to the full San Acacia Reach, which were used to complete the crosswalk. The simplified forest class structure is displayed as well as the extent of the 1987 aerial imagery, which are the limits of the spatial extent of the data crosswalk.

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