remote sensing

Article

Remote Sensing Measures Restoration Successes, but Canopy Heights Lag in Restoring Floodplain Vegetation

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Abstract: Wetlands worldwide are becoming increasingly degraded, and this has motivated many attempts to manage and restore wetland ecosystems. Restoration actions require a large resource investment, so it is critical to measure the outcomes of these management actions. We evaluated the restoration of floodplain wetland vegetation across a chronosequence of land uses, using remote sensing analyses. We compared the Landsat-based fractional cover of restoration areas with river red gum and lignum reference communities, which functioned as a fixed target for restoration, over three time periods: (i) before agricultural land use (1987–1997); (ii) during the peak of agricultural development (2004–2007); and (iii) post-restoration of flooding (2010–2015). We also developed LiDAR-derived canopy height models (CHMs) for comparison over the second and third time periods. Inundation was crucial for restoration, with many fields showing little sign of similarity to target vegetation until after inundation, even if agricultural land uses had ceased. Fields cleared or cultivated for only one year had greater restoration success compared to areas cultivated for three or more years. Canopy height increased most in the fields that were cleared and cultivated for a short duration, in contrast to those cultivated for >12 years, which showed few signs of recovery. Restoration was most successful in fields with a short development duration after the intervention, but resulting dense monotypic stands of river cooba require future monitoring and possibly intervention to prevent sustained dominance. Fields with intensive land use histories may need to be managed as alternative, drier flood-dependent vegetation communities, such as black box (*Eucalyptus largiflorens*) grasslands. Remotely-sensed data provided a powerful measurement technique for tracking restoration success over a large floodplain.

Keywords: land use; floods; environmental flows; cultivation; chronosequence; drought; fractional cover

1. Introduction

Wetlands are important and valuable ecosystems, yet, since 1900, more than 50% of wetlands have been lost worldwide [1–3]. This loss has prompted increased investment and support for landscape-scale restoration, such as reconnecting rivers with their floodplains and restoring wetland inundation [4,5]. These projects are costly and achieve variable success [5–7]. There is a strong imperative to track success or failure and to better understand restoration processes to improve management [5,8,9].

Remote sensing provides a considerable opportunity to efficiently assess vegetation restoration in wetlands over large spatial and temporal scales [10–14]. These analyses offer cost-effective methods for managers to track trajectories of vegetation change over time for improving management [9,13,15,16] and extending ground-based monitoring, with its more detailed, but spatially-restricted view of ecological responses. Further, the application of different remote sensing techniques provides complementary data on vegetation change [14]. For example, Landsat (TM/ETM+/OLI) provides data on vegetation reflectance (an outcome of vegetation composition and productivity), while LiDAR (light detection and ranging) data inform changes in structure [17]. Landsat analysis can be used to assess changes in the extent and type of wetland vegetation, with free data availability over large spatial and long temporal scales [3,9,12,15,18].

Vegetation indices may be derived from Landsat data, while canopy height models (CHMs) can be built from LiDAR data. Normalised Difference Vegetation Index (NDVI) data are often used, but pixels that contain mixtures of vegetation and water or bright soil can compromise measurement accuracy [19,20]. Spectral unmixing and fractional cover analyses provide a viable alternative, where each pixel is assumed to be a mixture of covers, but linear unmixing processes, similar to multiple regression, quantitatively divide pixels into percentages of endmembers: photosynthetic vegetation (PV; green vegetation), non-photosynthetic vegetation (NPV; dead/woody vegetation) and bare soil [18,20,21]. With fractional covers, similarity measures can discriminate individual pixels to average vegetation classes and quantify similarity and change over time [22]. These similarity measures enable quantification of differences in fractional cover (indicative of community composition and productivity) between restoration areas and undisturbed target vegetation communities (representing restoration goals). When degraded and target vegetation have contrasting fractional cover signatures, inferences can be made on restoration progression. Further, independent LiDAR data can be used to build CHMs to effectively measure changes in wetland vegetation structure in response to disturbances and degradation [17,23–26], which affect canopy heights and vertical structure [24].

Assessing and demonstrating the relative success of restoration is essential for setting realistic management goals and identifying effective management approaches. With dense time series of Landsat data, trajectories of change can be tracked over large wetlands [3,13]. Monitoring wetlands is challenging due to their high spatial and temporal heterogeneity, with complex vegetation communities [27]. Most time series analyses of vegetation changes using remote sensing have focused on short time series or comparisons of only two dates, until the free release of Landsat data [3,15]. Dense time series (using many dates as opposed to a two-date comparison) can detect both the large changes that two-date time series can (e.g., wetland loss) and more subtle incremental changes (e.g., conversion from forested to herbaceous wetland [15]). Recent efforts to restore floodplains of regulated rivers using environmental flows and river reconnection provide opportunities for such monitoring (environmental flows [4,5,28]). For example, one of the largest and most expensive restoration projects in the United States involved reconnecting the deltaic floodplain and diverting water from the Mississippi River for nearly 20 years. Assessment, using spectral unmixing of Landsat data and time series analysis, showed that vegetation cover failed to increase and that ecosystems were less resilient to natural disturbances (e.g., hurricanes [7]).

We used remote sensing to evaluate the success of a wetland restoration project in a previously cultivated floodplain wetland on a regulated river. Before restoration, the area was divided into fields with different intensities of land use (clearing, cultivation with frequencies varying from 1–25 years). Restoration interventions included levee removal and culvert enhancement for river-floodplain reconnection and inundation through environmental flow to supplement natural flood events. We analysed time series data of remote sensing across the different fields to determine the effects of different land use practices (clearing and cultivation) on restoration trajectories in vegetation cover and height. We established three time periods: before agricultural land use (pre-development; 1988–1997), after agricultural land use, but before restoration (peak-development; 2004–2007) and post-restoration through inundation (post-restoration; 2010–2015). Our objectives were: (1) to use time series of Landsat sensor data to evaluate the rates of convergence of fractional cover for each field with respect to those of uncultivated wetlands; and (2) to measure changes in the vegetation canopy height of each field based on LiDAR.

2. Methods

2.1. Study Site

The Macquarie Marshes are a wetland of national and international significance in the Murray-Darling Basin in north-central New South Wales (NSW), with semi-regular inundation from spring flows. These flows are dependent on upstream rainfall events as the Marshes have an average rainfall of less than 500 mm [29]. Average temperatures in the Marshes range from 3–6 °C in winter to 30–36 °C in the summer [29]. Vegetation consists of a mosaic of wetland types, dependent on inundation frequency and extent [10,30]. In the most frequently-inundated areas (once every year), monotypic stands of reed beds and cumbungi (*Phragmites australis* and *Typha* spp.) occur with patches of water couch (*Paspalum distichum*). In more elevated, but still frequently-inundated places (2–7 years), there are river red gum forests (*Eucalyptus camaldulensis*), lignum swamps (*Duma florulenta*), stands of river cooba (*Acacia stenophylla*) and areas of these three species interspersed. Sparse stands of black box (*Eucalyptus largiflorens*) and terrestrial grasslands occur high on the floodplain where they are rarely inundated (up to once every 10 years [10,31]). Inundation generally occurs from late winter into spring (August–November), but can also be highly variable, occurring in winter or lasting through summer. Only around 10% of the Marshes are protected within a nature reserve, where all of the major vegetation communities are relatively undisturbed by farming, but nonetheless degraded through reduced inundation due to river regulation [10,30–32].

In 2009, the state and federal governments purchased 2436 hectares of the Pillicawarrina property (a wheat and cotton enterprise), adjacent to the Macquarie River, along with the property's 8658ML (megalitre) water license (AUD $10.5 million [33]), with the goal of wetland restoration. This land was developed for agriculture at intensities (fields) ranging from virtually no intervention to chain cleared once 33 years ago to continuous cultivation for 23 years (Figure 1, Table 1). This gradient of land use intensity across fields provided a structure for exploring the effects of land use intensity on restoration of vegetation communities. For the analysis, we grouped three similar fields together as one (cleared 1982 and 1998, cleared 1986 and 1998 and cleared 1997 were labelled cleared 1997/1998; Table 1). Additionally, a field that was cleared in 1982 was divided into two as ground surveys observed that one section was recovering (healthy river red gum, lignum and river cooba mixes), while the other section consisted mainly of dead river cooba (no living trees regenerating underneath) and a grassland understory, we termed these fields cleared (1982-healthy) and cleared (1982-dead) respectively (Table 1, Figure 2).

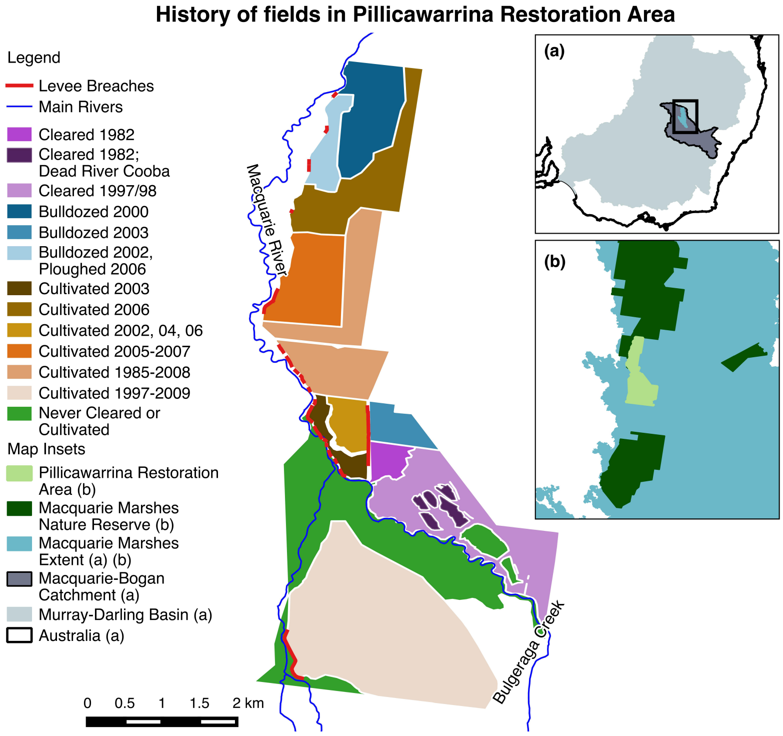

Figure 1. Map of the fields on Pillicawarrina used in the analysis. The Macquarie Marshes are in southeast Australia, within the Murray-Darling Basin (**a**); with about 10% of these extensive floodplain wetlands gazetted as a protected area, including Pillicawarrina (**b**), the focus for restoration.

Table 1. Names and history of fields in the restoration area.

Field Name	Land Use History
Never cleared	Never cleared or cultivated
Cleared (1982-healthy)	Chain cleared 1982
Cleared (1982-dead)	Chain cleared 1982, but now comprised of dead river cooba overstorey and grass understory
Cleared (1997/1998)	Comprised of three similar fields cleared in: (1) 1982 and 98, (2) 1986 and 98, (3) 1997
Bulldozed (2000)	Bulldozed 2000
Bulldozed (2003)	Bulldozed and ploughed 2003
Bulldozed (2002 and 2006)	Bulldozed 2002, ploughed 2006
Cultivated (2003)	Levelled 2002, cultivated 2003
Cultivated (2006)	Levelled 2006, cultivated 2006
Cultivated (2002, 2004, 2006)	Levelled 2002, cultivated 2002, 2004 and 2006
Cultivated (2005–2007)	Bulldozed 2002, levelled 2003, cultivated 2005–2007
Cultivated (1997–2009)	Levelled 1997, cultivated 1997–2009
Cultivated (1985–2008)	Cleared 1985, cultivated 1985–2009

Figure 2. Pictures taken at random locations (based on field sites) within the two sections of the cleared 1982 field. These sections were analysed separately due to the different extant vegetation condition and to explore how this division occurred.

2.2. Restoration and Floodplain Vegetation

The Pillicawarrina restoration plan aimed to restore historic wetland vegetation communities and used vegetation maps from 1949 and 1963 [34] as target historical communities [35]. Yet, these historical vegetation maps are inconsistent in describing the location and extent of vegetation communities [34,36,37], and the effects of river regulation (1967) also altered the vegetation distribution across the Marshes [10]. The maps used in the restoration plan identified more extensive river red gum and water couch communities, especially in southern Pillicawarrina, than in later maps. Vegetation communities entirely unaffected by river regulation may not provide a realistic restoration goal for communities post-regulation. Additionally, these maps were digitised from old aerial photography (unavailable in this analysis), producing some misalignment. For these reasons, we used the later vegetation community maps from 1981 [36] and 1991 (Figure 3; [37]). While former, probably inadequately recorded extensive river red gum communities, based on local knowledge [38] and shown in the later map of Wilson [37], this does cover some areas missing in the 1991 map and indicates the distribution of an extensive lignum understory. On this basis for most of the property, we adopted a restoration goal of a mosaic of three vegetation types: river red gum forests; lignum patches; and a mixed marsh understory, except for southern Pillicawarrina, which was historically sparsely forested grasslands with some river red gum. Restoration began in 2010 with the breaching of levee banks at strategic points (Figure 1) and improving culverts to allow and enhance the passage of flood waters across the floodplain [33].

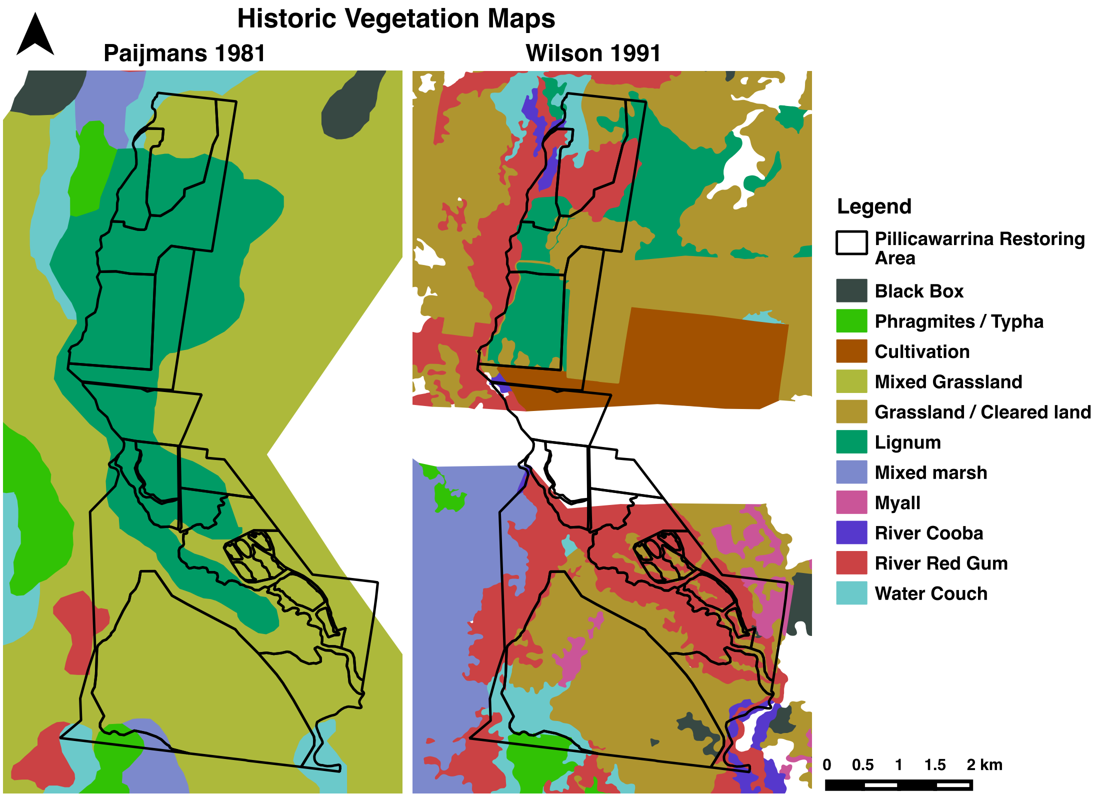

Figure 3. Two historic vegetation maps indicating historic vegetation communities on Pillicawarrina produced in 1981 Paijmans [36] and 1991 Wilson [37], with the white gap reflecting missing imagery data used to classify vegetation.

The timing of restoration efforts on Pillicawarrina coincided with the end of the Millennium Drought (2001–2009), a ten-year drought in southeast Australia. The Millennium Drought had a large impact in the Macquarie Marshes, with loss of wetland vegetation types, expansion of terrestrial communities (e.g., *Sclerolaena* spp.) and a decline in the health of river red gum communities

(Figure 4; [10,29–32]). These changes were strongly correlated with decreased inundation from both the drought and river regulation [29,30]. The extent of degradation has now been formally recognised as a change to ecological character, meaning that Australia is failing to meet its obligations under the Ramsar Convention [30].

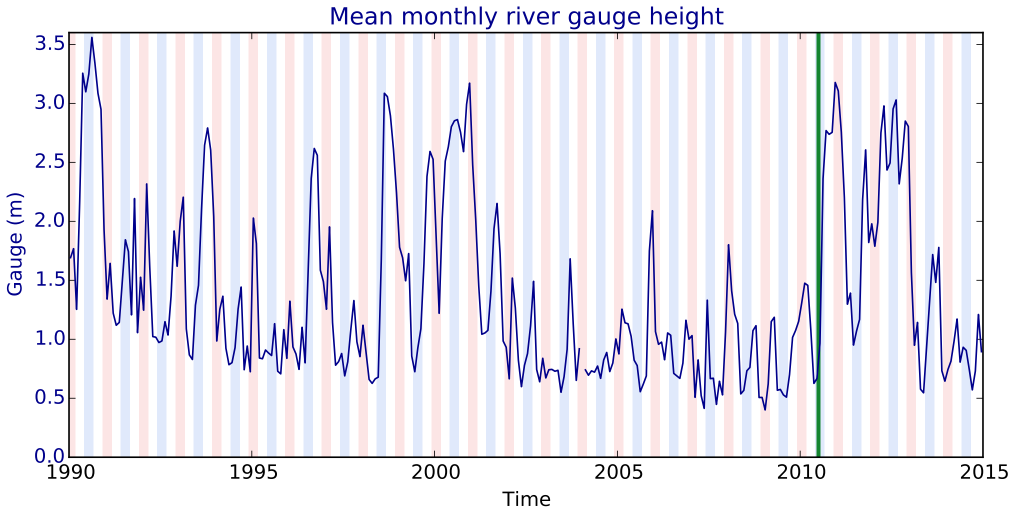

Figure 4. Mean monthly river gauge height at the Pillicawarrina gauge 1990–2015. The green line indicates when restoration actions (levee breaching and culvert improvements) were undertaken. Records were sourced from the Australian Bureau of Meteorology.

2.3. Fractional Cover Similarity Analysis

We compared vegetation communities using similarity analyses applied to time series of fractional cover derived from Landsat [3,7,21]. Orthorectified Landsat data were downloaded from the United States Geological Survey's Earth Explorer website (http://earthexplorer.usgs.gov) as 30-m resolution Level 1T processed scenes. All cloud-free Landsat 5 Thematic Mapper (TM), Landsat 7 Enhanced Thematic Mapper (ETM+) and Landsat 8 Operational Land Imager (OLI) data were used, except for Landsat 7 ETM+ data where the scan line corrector was missing. All data were atmospherically corrected and adjusted for bi-directional reflectance and topography, representing surface reflectance with a nadir-view and a 45° incidence angle [39]. Elevation data were obtained from the 1-second Shuttle Radar Topography Mission [40,41]. Each image that contained flooding in target vegetation or areas undergoing restoring within flooding maps produced by Thomas et al. [10,29,31] were identified and excluded, as the fractional cover algorithm does not apply to pixels containing surface water. Additionally, to avoid phenological (e.g., seasonal vegetation) cycles interfering with the analysis, we also only used scenes from autumn and early winter (March–June), the season with most non-flooded scenes. Fractional cover was derived from each surface reflectance image, using the pixel unmixing method described in Guerschman et al. [16]. The model determined spectral end-members by regressing Landsat bands and interactive terms against coincident field measurements, before unmixing the fractional cover components using a bounded variable least squares (BVLS) method [16]. It produced per-pixel estimates of cover percentages for photosynthetic vegetation, non-photosynthetic vegetation (e.g., woody branches, dead leaves and litter) and bare ground. The model was trained using 1171 field measurements of fractional cover from across Australia, with root mean square error (RMSE) values of 0.11 (photosynthetic vegetation), 0.16 (non-photosynthetic vegetation) and 0.13 (bare ground) [16].

As the flood-dependent vegetation growth responds closely to inundation and drought with seasonal and longer wavelength components (e.g., El-Niño cycles), it was difficult to determine trends in vegetation growth from the fractional cover time series. Instead, the time series of each pixel in the restoration area was compared to the time series of intact vegetation, representative of the desired vegetation through successful restoration. This approach does not require any time series decomposition, and allows the restoration to be assessed using time series similarity metrics [42].

Intact target vegetation communities (from Paijmans [36] and Wilson [37] maps) of river red gum and lignum were identified in other parts of the Marshes that were not degraded, using the most recent map of vegetation, the 2008 map (and field observations [43]). The time series of mean fractional cover values was used to define the characteristics of the target vegetation communities desired by restoration. Three time periods of interest were defined as: pre-development (1988–1997, containing 25 images), peak-development (2004–2007, containing 14 images) and post-restoration (2011–2015, containing 10 images; Table S1). These periods captured changes in land use, although some fields were developed earlier than the pre-development period (i.e., cleared 1982-healthy, cultivated 1982-dead and cultivated 1985–2008). The peak-development period captures the time when all fields had either been cleared or were being cultivated, providing a baseline for the condition during agricultural development. The only exception to this is the cultivated 2006 field, which was not cleared until half way through the peak-development period, and there may have been some recolonisation of those fields that were cleared pre-2000.

For each time period, we calculated a Euclidean distance measure [42] between the fractional cover time series of the target vegetation $g_p(t)$ and that of every restoring pixel $f_p(t)$ in each restoring field:

$$D = \frac{1}{n} \sum_{(p=1)}^{3} \left(\sum_{(t=1)}^{n} (f_{pt} - g_{pt})^2 \right)^{\frac{1}{2}}$$

where g_{pt} and f_{pt} are the fractional covers of the target vegetation and restoring pixel, with fractional component p (photosynthetic, non-photosynthetic and bare), at time t; n is the number of observations in the time series; and D is the sum of the Euclidean distances between the three fractional cover time series, normalised by the length of the time-series. The metric D summarises the overall similarity of each pixel to the target vegetation across the time period of interest. We produced three maps showing the dissimilarity of each pixel to the target vegetation type for that pixel, for each time period. By calculating dissimilarity to reference areas at each time step, we were assessing change against a moving target that was similarly affected by drought and inundation as the area undergoing restoration. Similar approaches successfully evaluated spectral characteristics of highly variable vegetation in semiarid regions of Australia and other wetlands [15,16,22].

2.4. Canopy Height Model Analysis

Two airborne LiDAR surveys were acquired in 2008 and 2015, allowing changes in the canopy heights of the fields undergoing restoration to be examined. Many studies show strong relationships (r^2 = 0.85–0.95) between LiDAR-derived tree height and field measurements [44,45]. Although more complex methods of deriving tree height from LiDAR data are possible, subtracting ground elevation from the local maximum LiDAR elevation is the simplest and most robust approach [44].

The LiDAR data were delivered in tiled point clouds, in the LAS file format, with ground points classified. They were both acquired with a Leica ALS50-II at around 1500 m above ground, and both have mean point densities of >1.5 points/m^2. The data were converted to the SPD format with a one-meter spatial grid [46], and a one-meter resolution digital elevation model (DEM) was derived using natural neighbour interpolation of the ground classified points [47]. Height above ground was defined for all laser return points through subtracting the DEM elevation, and a canopy height model (CHM) was derived using natural neighbour interpolation of the maximum first return height in each pixel. The CHM had many 'canopy pits' (pixels where the first return was lower than the top of the surrounding canopy), which were removed by smoothing with a 5 m × 5 m median filter. Although CHMs can generally underestimate tree heights by approximately 1 m [44], their accuracy is difficult to quantify due to difficulties in measuring tree height in the field. We assume that each CHM has an uncertainty of less than 1 m, and changes in canopy height between surveys that are less than 1 m could be the result of the this uncertainty.

No significant differences between the DEMs from the two surveys (2008 and 2015) were observed, ensuring that differences between the CHMs were caused by changed vegetation structure. Although differences are also possible due to the point sampling of the two surveys, these were reduced through the application of the median filter. A visual assessment of the differences in the two CHMs revealed a clear increase in vegetation height across most areas. However, due to changes in the horizontal location of the edges of trees, there were also many pixels showing a decrease or increase in height not related to vegetation growth. These changes limited the application of common methods for quantifying changes in vegetation height between dates through differencing the CHMs [44,48,49]. To minimise their influence, we examined changes to the frequency of canopy heights within each field in the restoration area. For each field, we extracted the number of pixels (density) within every 1-m height interval using the raster package in R [50,51]. We used a maximum threshold of 32 m, as the height of river red gums is generally 30 m with only occasional taller trees [52]. Higher returns were defined as errors, possibly birds. For every field, we then calculated the percentage of pixels in every meter interval above 2 m, providing a focus on shrubs and trees, and compared these values between surveys on each field and to the never cleared field (Table 1, Figure 1), as representative of a functioning river red gum forest.

2.5. Accuracy Testing

Field measurements and observations were conducted across the study area, to describe the vegetation and to assess the fractional cover model. Extensive compositional surveys were conducted during the post-restoration period (2012–2014), but it was only possible to measure fractional cover at 15 field sites in 2015 (May and June). This small sample of field measurements precludes thorough model evaluation or thorough validation, but does prove a preliminary assessment that modelled fractional cover values were within the reported error range [16] and whether there are potential biases in the image-based estimates. Further sampling is needed to extend this preliminary assessment into a validation of the method.

Each fractional cover site was measured with the same methods that calibrated the Landsat fractional cover model [53]. Each field site was located on flat terrain away from disturbances, in homogeneous vegetation (Figure 5). Sites were positioned to sample a variety of vegetation communities and canopy cover densities. At each site, the star transect method was used to record 300 vertical sighting tube observations of the overstorey (woody vegetation >2 m in height), mid-storey (woody vegetation <2 m in height) and understorey (herbaceous plants <2 m in height), from a circular area with a radius of 50 m. The location of each centre point was recorded using differential GPS (<5 m). Fractional cover was calculated from these observations, allowing for overstorey and mid-storey occlusion of the ground using the equations given in Guerschman et al. [16]. Landsat fractional cover was extracted and averaged from the closest 3×3 pixels from the OLI image acquired on 10 June 2015, to allow comparison with the field measurements.

Additionally, extensive surveys were conducted during the post-restoration period (2012–2014). The compositional data were collected to qualitatively compare the vegetation condition in each field with the results from the fractional cover and CHM analyses. Surveys were conducted at nine randomly-selected locations in each field, within a 20 m $\times$ 20 m quadrat for shrubs and trees and five randomly-placed 1 m^2 for herbaceous vegetation. For a qualitative comparison, we reported the principal canopy species and the general vegetation community trends in each field.

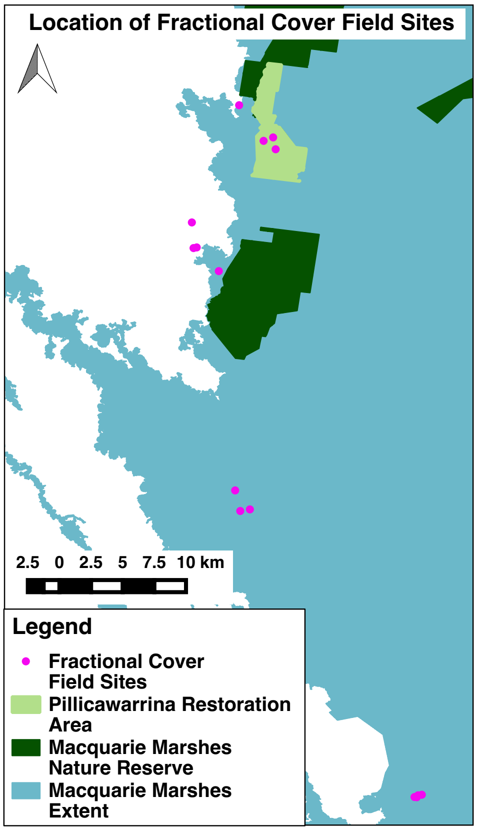

Figure 5. Map of the location of fractional cover field sites within the Macquarie Marshes.

3. Results

3.1. Fractional Cover

There were clear spatial and temporal differences among vegetation types in the dissimilarity measure in our restoration area and the pre-development vegetation communities (Euclidean distance; Table 2, Figure 6). Generally, dissimilarity compared to the target vegetation types increased with development intensity, but then decreased with restoration intervention (i.e., improved relative to desired vegetation type). The reduction in dissimilarity after 2009, and hence the level of restoration success, was clearly linked to the intensity and timing of historical land use in a field

Restoration success was reflected in the individual trajectories of the different fields. Few fields returned to pre-development similarity after the beginning of the restoration intervention period; the never cleared field was more dissimilar to target vegetation after restoration than most of the cleared fields (Table 2). The cleared (1982-healthy) and cleared (1982-dead) fields did not have a pre-disturbance measurement (due to no pre-1982 images), but maintained relatively similar fractional covers between the peak-development and post-restoration periods (Figure 6, Table 2). Fields that were only cleared or bulldozed once (Figure 1, Table 1) moved to their pre-development level of similarity after the restoration intervention and had less extreme departures after development. Fields that were only cultivated for one year or cleared twice had a similar response. The cultivated (2003) field (Figure 1) was the most similar to the desired vegetation peak-development, but also had the smallest improvement in dissimilarity after intervention for restoration (Figure 6, Table 2). Cultivated (2006) field changed the most among the three time periods, becoming the most dissimilar of these three fields to the target vegetation in peak-development and then returning to a state that was more similar to pre-development. The two fields cultivated for three years became more similar to the target vegetation (i.e., improved) a little in the post-restoration period compared to peak-development, but they did not return to their pre-development level (Figure 6, Table 2). The cultivated (1985–2008) field showed no improvement in dissimilarity to target vegetation after the restoration intervention (Figure 6, Table 2). Of the fields with pre-development data, the cultivated (1997–2009) field was the least similar to target vegetation types, but also had the largest difference between pre-development and post-restoration.

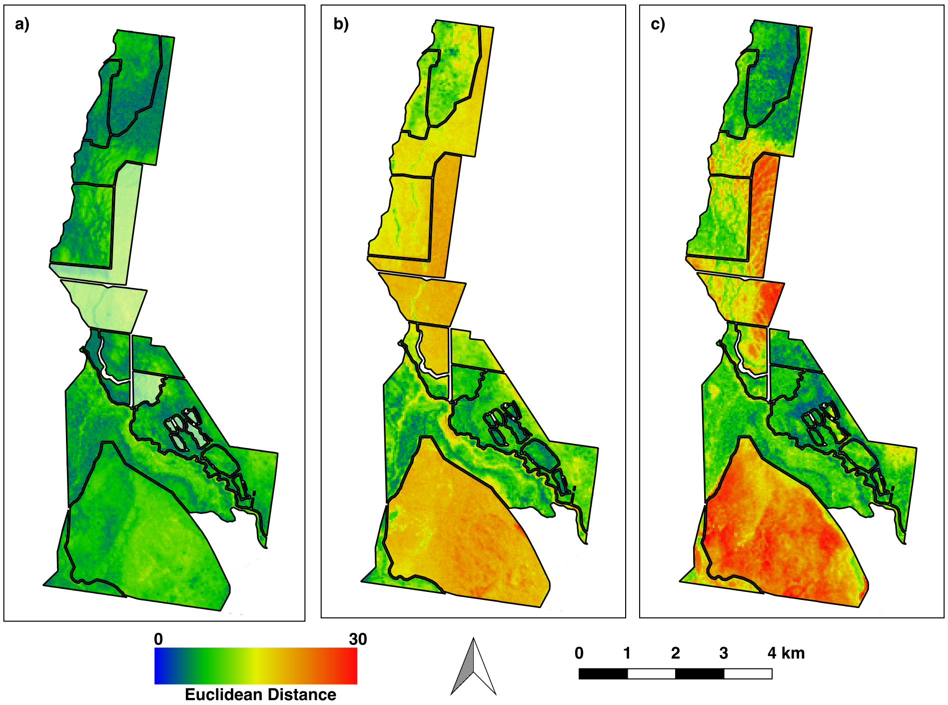

Figure 6. Euclidean distance between desired vegetation types and vegetation types in each of the fields (see Figure 1) for the combined fractional cover of each pixel to the desired vegetation types (river red gum and lignum) for two periods: (**a**) pre-development 1988–1996; (**b**) peak-development 2004–2007 and (**c**) post-restoration 2010–2015. The redder the colour, the greater the dissimilarity to natural vegetation types. Restored fields are identified by black borders, and filling in the pre-disturbance period indicates fields cleared or cultivated during this time; hence, these fractional covers are not representative of the historical vegetation.

Table 2. Dissimilarity between desired vegetation types and vegetation types in each of the fields (Figure 1) described by their mean (±standard deviation) Euclidean distance of combined fractional covers of each field for three time periods (pre-development, peak-development, post-restoration). Δ ED is the difference in mean Euclidean distance between pre-development and post-restoration. Individual pixel values shown in Figure 6.

Field Name	Pre-Devel.		Peak-Devel.			After-Rest.		
	Mean	SD	Mean	SD	Δ	Mean	SD	Δ
Never cleared	7.31	±1.89	9.45	±3.38	2.14	10.15	±3.73	2.84
Cleared 1982 healthy	NA [a]		8.35	±1.44	NA [a]	7.9	±1.23	NA [a]
Cleared 1982 dead	NA [a]		6.22	±1.7	NA [a]	8.83	±3.1	NA [a]
Cleared 1997/1998	7.05	±1.69	9.73	±2.44	2.68	7.84	±2.93	0.79
Bulldozed 2000	5.78	±1.15	12.16	±2.47	6.38	6.06	±1.72	0.28
Bulldozed 2003	6.91	±1.64	12.63	±1.75	5.45	6.65	±2.16	−0.26
Bulldozed 2002 and 2006	5.37	±0.92	12.69	±1.98	7.32	8.17	±2.24	3.34
Cultivated 2003	5.32	±1.11	11.82	±2.77	6.5	9.62	±2.33	4.3
Cultivated 2006	6.14	±1.16	16.29	±1.23	10.15	11.54	±4.91	5.4
Cultivated 2002, 2004, 2006	6.5	±1.52	17.98	±0.88	11.48	15.87	±4.09	9.37
Cultivated 2005–2007	7.12	±1.8	16.17	±1.4	9.05	12.31	±3.23	5.19
Cultivated 1997–2009	8.66	±1.62	19.34	±1.69	10.86	22.58	±3.43	13.92
Cultivated 1985–2008	NA [a]		19.14	±1.33	NA [a]	19.67	±4.94	NA [a]

[a] Indicates disturbed fields in the pre-development period.

3.2. Canopy Height Models

There were obvious differences in canopy height between fields in the heights of canopy vegetation (>2 m; Figure 7). While none of the cleared or cultivated fields had trees as tall as the never cleared field, the cleared fields had more similar vegetation structure to the never cleared field than the cultivated fields (Figure 7). The cleared (1982-healthy) field had a much higher percentage of trees and shrubs less than 15 m high than the never cleared field. There was little change between canopy heights peak-development (2008) compared to post-restoration (2015) of the cleared (1982-healthy) field (Figure 7). The cleared (1997/1998) fields increased in canopy in the below 5-m height classes, reaching similar levels to the never cleared field. Mid-canopy (<10 m) increased in all of the bulldozed fields between 2008 and 2015; for the bulldozed 2003 and bulldozed 2002 and 2006 fields, vegetation canopy (>2 m) increased from almost zero to about 1.5% (Figure 7). In the fields cultivated for one year, there were similar, albeit smaller increases. In fields cultivated for three years or more, vegetation structure only slightly increased in the 2-m and 3-m height categories in cultivated (2005–2007) and in cultivated (1985–2008) (Figure 7). The fields cultivated (2002, 2004 and 2006) and cultivated (1997–2009) had no increase in canopy height (Figure 7). All 9–15-m tall trees present in 2008 persisted until 2015, including in cultivated fields where a few large trees remained, generating low height percentages up to 18 m. In fields that had large changes in the canopy height distribution (bulldozed fields and fields cultivated for one year) over the seven years, the increases were across several metres of height gained and the assumed uncertainty for the CHMs <1 m, supporting these findings.

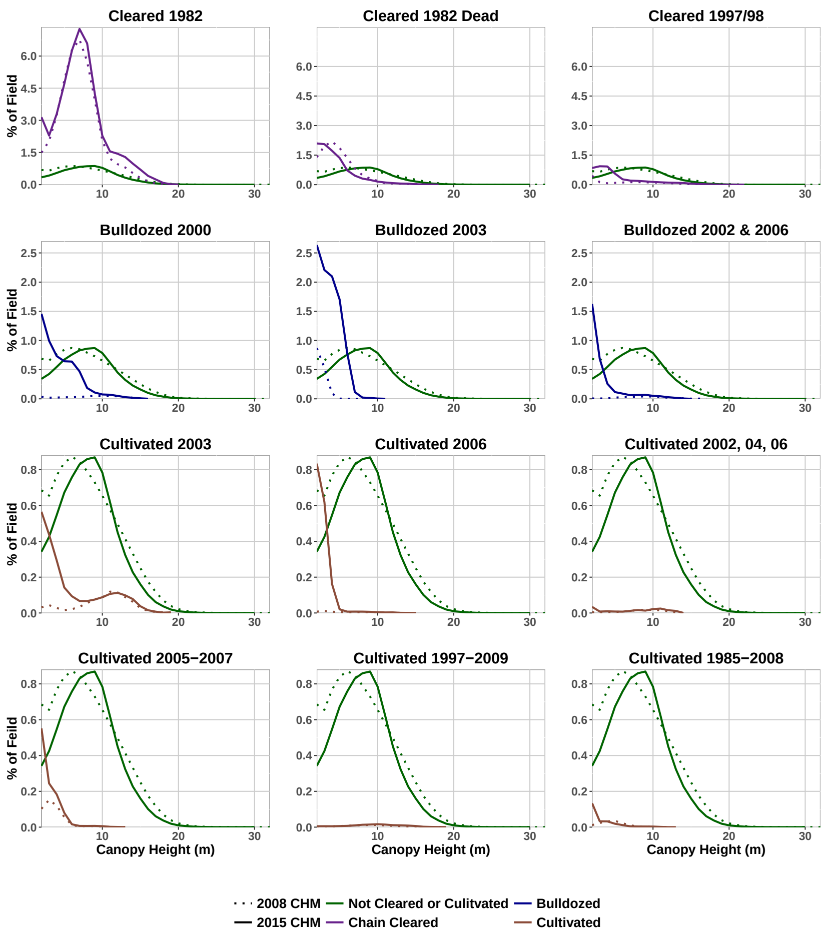

Figure 7. Percentage of canopy height above 2 m in each field, based on the number of pixels in each field in the canopy height models generated from LiDAR data for 2008 (dotted lines) and 2015 (solid lines).

3.3. Accuracy Testing

Due to access constraints imposed by environmental conditions when the measurements were made and the focus on sites containing trees, the 15 fractional cover field sites only covered a limited range of values. Sites were dominated by vegetation cover, with relatively low bare ground cover (0%–22%) and greater photosynthetic (24%–59%) and non-photosynthetic cover (40%–59%). Our limited field sample produced root mean square errors (RMSEs) that were smaller than the errors reported for the Australia-wide model (Figure 8, [16]; Guerschman's values were between 13%–18%). However, the results are suggestive of bias, with all 15 estimates of photosynthetic cover greater than field values and conversely underestimating non-photosynthetic cover. Further work is needed to confirm and quantify this apparent systematic error and, thus, to correct for bias using regression models.

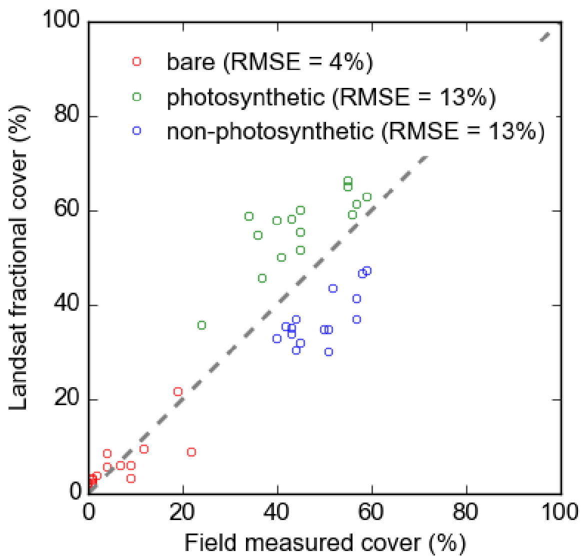

Figure 8. Field measurements compared to fractional cover modelled from Landsat data for fifteen sites, including the root mean square error values (RMSE).

Data from ground surveys conducted from 2012–2014 were qualitatively consistent with both fractional cover and CHM analyses (Table 3). Fields cleared and cultivated for only one year had mixes of species, similar to target vegetation, though they tended to be smaller trees more densely distributed, reflected in the CHM results. Fields cultivated for longer than one year had larger areas of grasslands, fewer of the target vegetation species than less disturbed fields and slower or no recolonisation of canopy species, which aligns with both fractional cover and CHM results. Occasional small patches of lippia (*Phyla canescens*), an exotic dense ground cover that can crowd out native plants, occurred in several fields undergoing restoration.

Table 3. Qualitative description of vegetation in each field.

Field Name	Land Use History
Never cleared	Largely comprised or river red gum forest, with smaller patches of lignum and occasional river cooba stands and grasslands
Cleared (1982-healthy)	Densely forested with river red gum, river cooba and lignum all co-occurring
Cleared (1982-dead)	Densely forested with river cooba, which is largely dead, grassland understorey
Cleared (1997/1998)	River cooba makes up the overstorey majority, with occasional lignum plants and grassland understorey
Bulldozed (2000)	River cooba stands, patches of lignum, some reeds and rushes, some grasslands with (mostly) wetland species, occasional mature river red gum trees
Bulldozed (2003)	Lignum patches and smaller stands of river cooba with grasslands in areas observed to be drier
Bulldozed (2002 and 2006)	Some dense river cooba stands, patches of lignum and river red gum, occasional older trees
Cultivated (2003)	River cooba and river red gum saplings, occasional patches of reeds (observed wetter areas) and grasslands (observed drier areas)
Cultivated (2006)	Some dense river cooba stands, patches of reeds, rushes and grasslands (mainly wetland species), some lignum
Cultivated (2002, 2004, 2006)	Largely grasslands (mostly drier/terrestrial species with some patches of wetland species), occasional river cooba or river red gum trees
Cultivated (2005–2007)	Bulldozed 2002, levelled 2003, cultivated 2005–2007
Cultivated (1997–2009)	Largely grasslands (mostly drier/terrestrial species with some patches of wetland species), a very few river red gum trees from pre-cultivation still present
Cultivated (1985–2008)	Largely grasslands (terrestrial and wetland patches) occasional river cooba or lignum plants, small dense stands of river red gum saplings in channel lines

4. Discussion

We used fractional cover and CHM analyses to evaluate the efficacy of re-introducing inundation (and ceasing cultivation) as a restoration measure in agricultural land. Increased inundation produced a strong recovery in fractional cover, while canopy height showed a less-pronounced response. Following restoration of the river-floodplain connection and inundation that ended the Millennium Drought, there was a gradient of response, closely related to development intensity. Post-restoration, fields that had only been cleared were most similar to the target vegetation community and showed the greatest recovery. Vegetation in fields cultivated for more than 12 years recovered the least. None of the fields, even those cleared 33 years ago, had trees of similar heights to the un-cleared field. The lower tree heights of the cleared fields are indicative of an earlier successional phase, where dense stands of small trees establish before they self-thin to form a mature forest [24].

Our approach in using fractional cover and Euclidean distances to quantify similarity was particularly effective in showing differences among restoration areas, compared to a target restoration goal, in our case, established from intact reference areas and a ten-year baseline prior to development. Others have applied a similar technique (RS time series, some using Euclidean distance) to: show rangeland recovery from grazing (lightly grazed areas recovered faster [21]); track the progress of wetland recovery in Louisiana coastal deltas [7] and monitor more subtle changes in wetland condition [15]. There is considerable opportunity to apply this technique to measure not only restoration success, but also degradation of flood-dependent vegetation.

4.1. Response to Inundation and Restoration Success

Our modelled canopy heights and estimates of fractional cover clearly showed evidence of wetland restoration at the landscape scale, with variations in restoration success related to the development intensity of the fields. The restoration intervention (cultivation cessation, levee breaching and culvert enhancement) allowed inundation of the fields in the restoration area when the river flooded after a long period of drought. The degree of restoration success varied, particularly in relation to the amount and intensity of development. There were clear differences emerging in canopy height between the peak-development period and post-restoration, after several years of inundation, demonstrating the importance of inundation for regeneration and regrowth of wetland vegetation. Fields not affected by levees (i.e., cleared 1997/1998) had fractional cover that somewhat diverged from target vegetation during the drought, but they then partially recovered after inundation. Further, fields only cleared on one occasion returned to fractional covers similar to pre-disturbance levels; any level of disturbance greater than this (i.e., cleared twice or cultivated) and restoration was less successful. This disparity increases with the level of land use, with those areas cultivated for more than three years showing little indication of changing, especially in the cleared 1985–2008 field, if we assume that pre-development it had similar fractional covers to the neighbouring fields (e.g., cultivated 2005–2007, cultivated 2002, 2004, 2006). This may indicate that these fields need direct interventions (e.g., plantings) to restore.

Inundation is clearly effecting the level of restoration observed within the fields. Where flood water remained for relatively long periods based on field observations, such as the northeastern corner (cultivated 2006 field) or the bulldozed (2003) field, strong signs of restoration were exhibited in the fractional cover, becoming more similar to target and pre-development vegetation. In contrast, the dry areas, such as the eastern edge of the cultivated (1985–2008) field, higher elevation parts of the cultivated (2002, 2004, 2006) field and the entire cultivated (1997–2009) field, showed smaller indications of recovery with restoration. The never cleared field became more dissimilar to target vegetation across the three time periods than any of the cleared fields; this may be due to a trend of encroaching of drier vegetation that occurred across the Marshes in the Millennium Drought [10,30] and/or higher elevation parts of this field that support drier vegetation. This may not have occurred in our target vegetation areas as these smaller patches are in the core of target vegetation communities, whereas some of the fields (higher elevation parts described above) that were observed to be drier. These drier edge areas are not accounted for fully in the restoration plan, which may highlight the inappropriateness of the restoration target in some areas.

There are clearly limitations on whether full restoration will be achieved, particularly in fields cultivated for more than one year. The lack of convergence in these restored fields to pre-development states may suggest restoration barriers or lagged responses to renewed flooding. Restoration of wetlands can take many years (up to centuries), but it may be less for this large wetland (>100 ha) in a warm climate where a 40-year recovery time may be possible [5]. Our CHM analysis supported this assessment. Even fields cleared 33 years ago (e.g., cleared 1982 with healthy floodplain vegetation) were still showing signs of successional phases of restoration. There are other scenarios: dense stands of river cooba (observed in some low land use development fields) may die off during prolonged dry periods [54], which is possibly what occurred in the cleared 1982 dead field. There may be other vulnerabilities for restoring wetlands from either abiotic or biotic disturbances. For example, restoring coastal wetlands in Louisiana suffered more from hurricanes than undisturbed wetlands [7].

Other remote sensing studies of ecosystems undergoing restoration have found similar responses to our study. For example, restoring terrestrial pinyon-juniper woodlands in Nevada, United States, had fewer trees than undisturbed woodlands [20]. Further, restored wetlands in Oregon were more herbaceous than the forested natural systems [3]. Restoring deltaic wetlands in Louisiana failed to achieve the vegetation coverage or resilience to natural disturbances (e.g., hurricanes) of natural wetlands after reinstatement of the historical inundation regime [7]. Finally, grazed rangelands in

Queensland with lower grazing pressure recovered more quickly than those where grazing pressure was heavy with high stocking levels [21].

4.2. Remote Sensing for Assessing Wetland Management Actions

We demonstrated how remote sensing in wetland management could be used to assess a specific management action. Remote sensing time series analyses generally classify spatial changes in vegetation (e.g., [3,11,15,27]) or detect when changes in wetlands occur (e.g., [3,55]), whereas our analysis had defined spatial extents (fields) and time periods. We measured the convergence of restored vegetation with reference vegetation based on remote sensing indicators of vegetation cover and height to assess the outcomes of restoration efforts across a range of historic land uses. Although unable to provide information about compositional responses that field-based methods can, our method provides a simple and cost-effective means of measuring the response of vegetation structure. Kearney et al. [7] used a similar methodology to assess the restoration of coastal wetlands in Louisiana, but this is the first time such an analysis has been used on an ex-agricultural floodplain wetland in Australia. This is of importance because governments are currently investing in buying and restoring retired agricultural land on inland floodplain ecosystems.

The use of two remote sensing datasets (Landsat and LiDAR) enabled us to examine different structural features of the vegetation. Fractional cover was the most powerful where there was a strong difference between reference vegetation and degraded initial states [21], separating undesirable grassland from target vegetation types, such as lignum or river red gum. LiDAR was more effective when the spectral signatures of regenerating wetlands were more similar to the target vegetation (e.g., in fields cleared only once), but the canopy height distribution still differed from un-cleared areas.

Our approach was cognisant of the high temporal variability inherent in periodically inundated floodplain wetlands. Analysis of fractional cover to derive Euclidean distances within the same image date ensured that comparison between fields and target vegetation was standardised to the same climatic and soil moisture conditions. Water can have a confounding effect on remotely-sensed signals of vegetation [16], but we overcame this by removing images taken during times of inundation from the analysis. Remote sensing indicators may overestimate restoration success if target vegetation produces similar fractional covers to degraded vegetation.

We focused on trees and shrubs, river red gum and lignum, which we expected to have contrasting signatures to degraded grasslands, although our focus on woody vegetation may not have adequately represented the entire range of 'historical' vegetation communities. Natural stochastic effects of dry and wet periods may have complicated the interpretation of restoration success. The Millennium Drought preceded restoration action, and a period of natural inundation after restoration may have caused an overestimation of restoration success as estimated by fractional cover, if there was a strong photosynthetic response [20,27]. This includes the dissimilarity of the never cleared field to the target vegetation, which is likely due to both the ongoing effects of the Millennium Drought and grassland patches at higher elevations.

The accuracy of the analyses performed was checked against both field fractional cover measurements and field vegetation composition surveys. The Landsat-derived fractional covers performed well when compared to the field measurements, with RMSE values comparable to Australia-wide studies [16]. These field measurements were conducted across a range of representative vegetation types within the Macquarie Marshes. Although we had no field values of vegetation heights, LiDAR-derived heights are generally very reliable [44,45], particularly in flat terrain with open canopies. Further collection of ground fractional cover and height measurements to test accuracy would assist in a more complete accuracy assessment. Although these field measurements were limited (15 sites only), we had compositional surveys allowing us to qualitatively confirm the results of fractional cover and CHM analyses. The ability to detect change will depend on the imagery available for analysis before, during and after restoration treatment. In our analysis, fewer images were available for the later periods of the analysis (pre-development: 25; peak-development: 14; post-restoration: 10).

This can be addressed in the future with more post-restoration images over time; the peak-development time period, however, is limited.

4.3. Restoration of Pillicawarrina

There are lessons for the restoration management of Pillicawarrina. Most importantly, the fields require different management, reflecting the intensity of development disturbance. Where development impacts were low, frequent inundation may be sufficient for recovery, except where dense river cooba establish. Thinning with management of re-establishment from rootstock [54] may be needed to avoid large-scale mortality during drought periods. Additionally, ground cover species will need to be monitored so that action can be taken if lippia begins to dominate understorey communities.

Where there was prolonged development and smaller restoration responses, black box seedlings could be planted to establish grassy black box open woodlands, a community in severe decline across the Murray-Darling Basin [54,56]. Black box communities are suited to low frequency inundation areas with low natural recruitment of this species. This restoration goal would also suit the cultivated 1997–2009 field, given that it historically supported floodplain vegetation less frequently inundated than river red gum and lignum.

The ongoing lag effects of the Millennium Drought underline the importance of frequent inundation for recovery and restoration. Vegetation communities within the Macquarie Marshes Nature Reserve have continuing changes from the Millennium Drought [30] and the increasing dissimilarity of the never cleared field from target vegetation may also be due to this. Any future inundation would have the dual benefit of drought recovery and restoration from agriculture. Overall, river reconnection to floodplains triggered some restoration in most fields, and continued inundation in the future will be vital for long-term restoration success.

5. Conclusions

Remote sensing analyses, generating time series of fractional cover and CHMs allowed us to assess restoration as a function of vegetation structure in our floodplain wetland, at spatial and temporal scales not possible using limited field data. Time series of fractional cover is a powerful method, effectively tracking changes in grazed rangelands [21], as well as wetlands, and building on the value of distance-based metrics for examining change in wetlands [15,22]. Remote sensing for wetland management is an increasingly valuable tool for the assessment of restoration success [3,9,13] and impacts on vegetation communities [16]. Unlike the majority of remote sensing analyses in wetlands, which focus on change detection or spatial classification [3,15], we demonstrate a method to quantify restoration relative to intact reference areas where both the spatial extent of vegetation types and disturbance history are previously known. The technique is applicable to wetlands around the world, given the global availability of Landsat imagery, coupled with the identification of comparative natural areas. Finally, due to the ongoing availability of remote sensing data, studies of this kind have great potential for monitoring wetland restoration in the long term.

Supplementary Materials: The following are available online at http://www.mdpi.com/2072-4292/8/7/542/s1, Table S1: A list of all the image dates used in each period of the analysis.

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Author Contributions: S.K.D., A.F. and R.L. conceived of the study and designed the analysis. A.F. developed the code for the fractional cover analysis and built the CHMs. S.K.D. developed the code to analyse the CHMs. S.K.D, D.K.H and A.F. ran the analyses and developed the figures. R.T.K., J.A.C., D.K. and P.B. gave intellectual input to framing the paper. S.K.D., R.T.K, A.F., D.K.H., R.L., J.A.C., D.K. and P.B. wrote the paper.

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Abbreviations

CHM Canopy height model
DEM Digital elevation model

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