remote sensing

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Communication

The Role of Environmental Water and Reedbed Condition on the Response of *Phragmites australis* Reedbeds to Flooding

William Higgisson [1,*], Adrian Cobb [2], Alica Tschierschke [1] and Fiona Dyer [1]

1 Centre for Applied Water Science, University of Canberra, Canberra 2617, Australia; alica.tschierschke@canberra.edu.au (A.T.); fiona.dyer@canberra.edu.au (F.D.)
2 C & M Consulting Pty. Ltd., Canberra 2612, Australia; adrian.cobb@candm-consulting.com
* Correspondence: will.higgisson@canberra.edu.au

Abstract: Globally, wetlands have experienced significant declines in area and condition. Reedbeds are a key attribute of many wetlands and are typically composed of *Phragmites australis* (common reed), a globally distributed emergent aquatic perennial grass. Environmental water is increasingly used to support functioning river and floodplain ecosystems, including reedbeds, where maintaining wetland vegetation condition is a common objective. Drone-based remote sensing allows for the consistent collection of high-quality data in locations such as wetlands where access is limited. We used unoccupied aerial vehicles (UAVs) and convolutional neural networks (CNNs) to estimate the cover of *Phragmites australis* and examine the role of reedbed condition and prior environmental watering in the response of reedbeds to flooding. Data were collected from a large inland reedbed in semi-arid western New South Wales, Australia between October 2019 and March 2021 using UAVs and processed using CNNs. Prior to the flood event, sites that had received environmental water had a significantly greater cover of *Phragmites australis*. The sites that were not managed with environmental water had very low cover (<1%) of reeds prior to the flood event and transitioned from a Critical condition to a Poor or Medium condition following flooding. Using UAVs and CNNs we demonstrated the role environmental water plays in filling the gaps between large flood events and maintaining the condition and resilience of reedbeds.

Keywords: wetland condition; floodplain; hydro-ecology; reedbed; neural networks; environmental water

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1. Introduction

River regulation combined with modern land management practices (such as grazing and land clearing) have led to natural wetlands experiencing significant declines in area, with global losses in wetland area estimated to be as high as 87% since 1700 AD [1]. Inland wetlands have been disproportionately affected, experiencing greater losses at a faster rate compared with those observed in coastal wetlands [1]. A major cause of wetland loss has been the regulation of large inland rivers to provide for the demands of humans for reliable water supplies, which has disrupted the patterns and processes that structure river–floodplain systems [2]. Such regulation changes the volumes of water in river systems and the patterns of flow variability, reducing the magnitude, frequency and interannual variability in high and low flow events [3]. This results in less available water for floodplains, which has reduced the frequency, duration and extent of connections between river, floodplain and wetland habitats [4–6].

Functioning wetlands provide a range of important ecosystem functions and services including supporting biodiversity, maintaining and improving water quality, flood abatement and carbon regulation [7,8]. The ability of wetlands to provide these functions and services is dependent on key wetland attributes including wetland area, water volume and retention and the presence of emergent and floating vegetation [8]. Tracking and comparing these attributes through time or space provides vital information on the ability of wetlands to provide these important functions and services.

Reedbeds are a key attribute of many wetlands that provide important habitat, trap and process sediment and nutrients and contribute to maintaining water quality [9–11]. Reedbeds are typically composed of *Phragmites australis* (common reed) (Cav.) Trin. ex Streud., which is a globally distributed emergent aquatic perennial rhizomatous grass (Poaceae) species [12]. Declines and dieback in *Phragmites australis* have been observed in Europe and Australia, attributed to changes in hydrology, erosion, grazing pressures, mechanical damage and direct destruction [13–15]. Conversely, *Phragmites australis* has experienced range expansions in North America attributed to changes in hydrology and nutrient regimes [16,17]. The growth and condition of *Phragmites australis* is mediated by flooding frequency [18], which is particularly evident in drier environments (such as semi-arid climates) [19,20]. Therefore, changes to flooding regimes (resulting from the reduced frequency and duration of bankfull events) in dryland river systems [5,21] pose a significant threat to *Phragmites australis* growing in these climates.

The over-allocation of water supply and competition between human and environmental needs has led to water management reform in some parts of the world, including Australia [22,23] and the USA [24]. One aspect of the reforms is the allocation of water to selected river systems to maintain and restore riverine and floodplain environments. This strategy is based on the premise that the hydrological regime is one of the fundamental drivers of the structure and function of riverine and floodplain ecosystems [25,26]. Water used to restore some of the natural river flow conditions necessary to support functioning river and floodplain ecosystems is commonly referred to as environmental water or environmental flow [27].

Such restorative actions in freshwater systems require knowledge of the environmental water needs of the river system and its biota [28]. Typically, this involves defining important components of the natural flow regime [29] and considering how these translate to ecological outcomes for, e.g., wetland and floodplain plants [30]. Ongoing monitoring and the provision of data are fundamental inputs to the adaptive management of environmental water. Monitoring also provides a way to track progress against objectives or targets [31] and evaluate the public investment in environmental water.

It is likely that the response of reedbeds to flooding will be influenced by the condition of the reeds, as suggested by Bond et al. [6]. One of the challenges in developing an understanding of reedbed responses to water has been that they are difficult environments in which to work. Data collection using field-based methods is often restricted because site access is limited. This constrains the extent of the monitoring that can be undertaken and the understanding that can be developed. Recent developments in the use of unoccupied aerial vehicles (UAVs) to map the extent, cover and condition of *Phragmites australis* provide an opportunity to improve our understanding of how reedbeds respond to water [20]. UAVs, commonly known as drones, are now widely used in ecological monitoring, as they can collect high-quality imagery and data from locations such as wetlands where access is limited or challenging. Computational machine learning techniques such as convolutional neural networks (CNNs) offer a way in which to recognize and group features from remotely sensed imagery through object detection and pattern recognition [32,33], providing a time-efficient and objective way of processing the large amounts of data collected from UAVs. Imagery captured using UAVs, together with CNNs, are becoming widely used to map the extent and cover of plant communities and species [34,35], including *Phragmites australis* [33,36].

Here, we use the method defined in Higgisson et al. [20], which uses UAVs and CNNs to estimate the cover of *Phragmites australis* and other wetland attributes. This work builds on that of Higgisson et al. [20] by using their method to examine the role of reedbed condition and prior environmental watering in the response of reedbeds to flooding. In doing so, it demonstrates a practical application of UAV-based monitoring that can be used to inform the adaptive management of environmental water.

2. Methods and Study Design

2.1. Study Area

The Great Cumbung Swamp is a large terminal reed swamp surrounded by floodplain forests, woodlands and shrublands [37]. It supports one of the largest areas of *Phragmites australis* in New South Wales (NSW) [37]. The size of the Great Cumbung Swamp makes it one of the most important wetlands for waterbirds in southwestern NSW, supporting species listed as threatened under Australian Commonwealth and State legislation as well as species which are recognized in international migratory bird agreements [37,38]. The central reedbeds are dominated by large stands of *Phragmites australis*, which surround bodies of open water along the channel of the Lachlan River and smaller ephemeral flood channels [39,40].

The Great Cumbung Swamp is located in a semi-arid environment, which experiences a mean annual rainfall of 367.4 mm and a mean maximum and minimum temperature of 33.1 °C in January and 15.1 °C in July [41]. It receives water from the Lachlan River system, which has its headwaters in Australia's Great Dividing Range.

The typical time of floodplain inundation in the lower Lachlan River is spring (September–November), but lower-lying parts of the floodplain can connect to the river throughout the year [5]. Water resource development has intensified in the Lachlan River catchment since the construction of the Wyangala Dam in 1935 [4]. Regulation and flow extraction from the Lachlan River has reduced the flow of the Lachlan River (current flows at Oxley near the Great Cumbung Swamp are approximately half those under undeveloped flow conditions [39]), which has changed the behavior and distribution of floodwaters [5,42]. Under current flow conditions, a substantial portion of the reedbeds of the Great Cumbung Swamp are only flooded during particularly wet periods, when the major reservoirs are full as a result of airspace releases, translucent flows or targeted environmental watering actions. Environmental water is typically delivered to the Great Cumbung Swamp and other floodplain wetlands in the lower Lachlan River system via water released from major reservoirs upstream, such as Wyangala Dam.

2.2. Site Selection

A total of nine (50 × 50 m) sites were established in the reedbeds of the Great Cumbung Swamp as part of the Australian Commonwealth-Government-funded Flow Monitoring, Evaluation and Research (MER) Program (Figure 1) [43]. Sites were established across a hydrological gradient and were grouped according to the number of inundation events each site had experienced in 2019, during the first year of monitoring. All nine sites had been flooded during a natural large-scale flooding event in late 2016 (Figure 2). Sites in the watering treatment Flow 0 group were not inundated in 2019 and had not been inundated prior to this since the flood in 2016. Sites in the watering treatment Flow 1 group were watered in June 2019 as a result of an environmental watering action. Sites in the watering treatment Flow 2 group were watered in June 2019 and November 2019 by environmental watering actions. Prior to 2019, sites in the watering treatment Flow 2 group were watered in December 2018 as a result of an environmental watering action (Figure 2).

2.3. Establishing Reedbed Condition

While there is no standard definition for vegetation condition, it is typically expressed along a scale from good to bad [44]. Metrics vary, but the main indicators of reedbed condition are reed cover and height, stem diameter, density and number of flower heads [45–47], with the most commonly used metrics related to cover and height. Overton et al. [48] defined four condition classes (Good, Medium, Poor and Critical) for emergent macrophytes with an annually renewing canopy, including *Phragmites australis* in Australian conditions. In short, reedbeds in the Good condition class have a vigorous appearance, where stands of reeds are dense and medium-to-tall, with high aboveground biomass; those in the Medium condition class have shorter and less dense stands than those the Good condition class, with no lateral expansion; those in the Poor condition class have an appearance of little standing material and no active growth, and the rhizomes are gradually ageing without replacement; and those in the Critical condition class have little or no evidence above ground that the plant is present below ground as rhizomes [48]. Cover is most easily identified from aerial imagery, and in preliminary investigations in the Great Cumbung Swamp is correlated with height and number of flower heads [49]. Here, we use the percentage reed cover as the main indicator of reedbed condition: the greater the percentage cover, the better the condition.

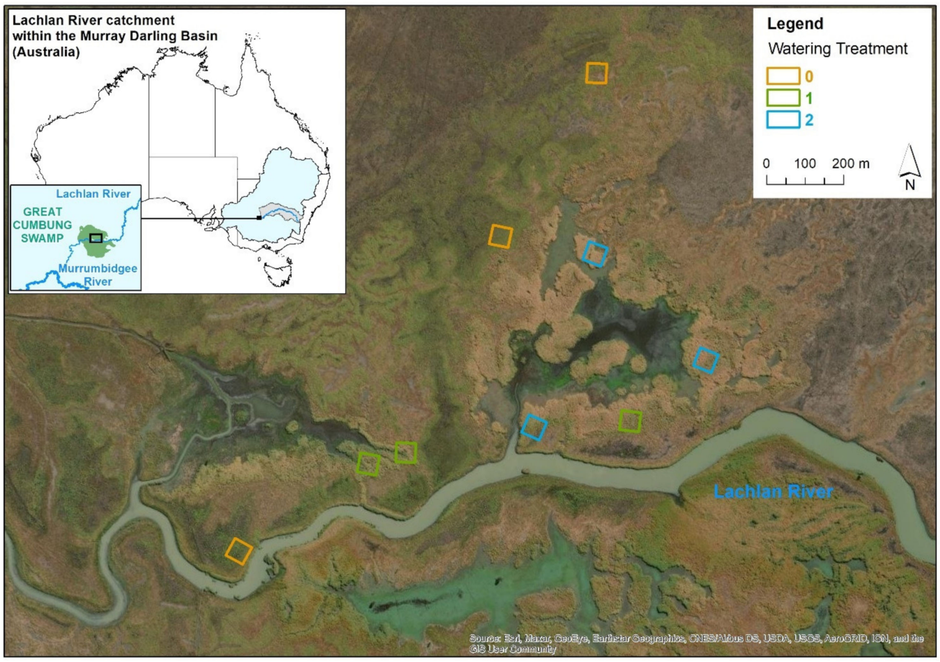

Figure 1. The reedbeds of the Great Cumbung Swamp and the 50 × 50 m plots used as part of this study. Watering treatments are shown as Flow 0 (orange), Flow 1 (green) and Flow 2 (blue), and the location of the Great Cumbung Swamp, the Lachlan River and Murray Darling Basin are shown in the insert map.

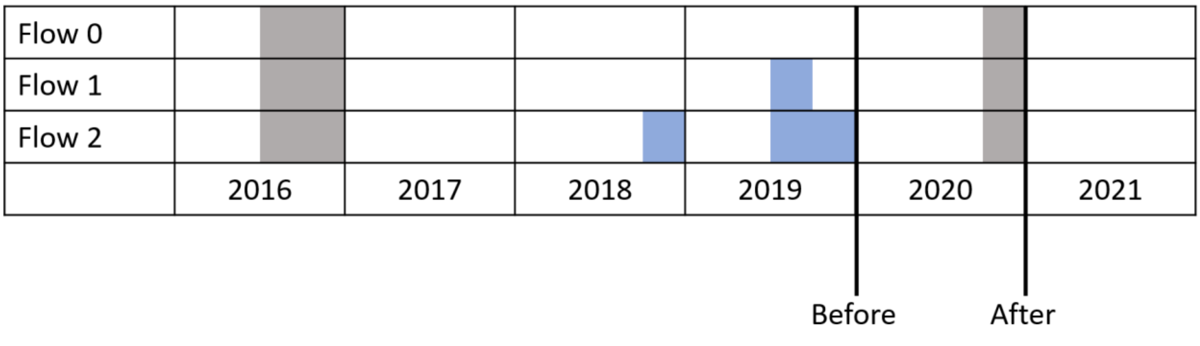

Figure 2. The three watering treatments and the approximate times (on a quarterly basis) of flood events between 2016 and 2021 within each watering treatment. The grey squares represent large natural flood events, and the blue squares represent floods from environmental water. The thick black lines represent the approximate timings (January 2020 and January 2021) of the data used in the analysis comparing situations before and after flooding.

2.4. Monitoring

We conducted UAV surveys seven times at each of the nine sites between October 2019 and March 2021. Survey months were October 2019, January, March, September and November 2020, and January and March 2021. These survey trips captured the reeds over two growing seasons. Aboveground biomass in *Phragmites australis* grows seasonally, and in eastern Australia new shoots start to grow in October and senesce over autumn and winter [30].

All flights complied with Australian Commonwealth regulations and were conducted by a licensed pilot. The UAV was a Phantom 4 Pro quadcopter with a built-in FC6310 camera, using a CMOS sensor. Flights were completed in clear conditions at an altitude of 20 m, with an along- and across-track overlap of 80% and a speed of 2 m per second, which provided a resolution of 0.9 pixels per centimeter. Images were aligned and processed into a single high-resolution true ortho-mosaic for each survey and site using Agisoft Metashape v 1.5.3. The processed ortho-mosaic images were imported into ArcGIS 10.x and clipped using the GPS coordinates for the four corners of each 50 $\times$ 50 m plot. See Higgisson et al. [20] for further details on image collection and processing.

2.5. Data Analysis

Using the single high-resolution true ortho-mosaic images for each survey and site, we estimated the cover of *Phragmites australis* at each site during each survey using the CNN model described by Higgisson et al. [20]. Higgisson et al. [20] described a method to estimate the cover of *Phragmites australis* among other wetland feature classes such as water, bare ground, leaf litter and other vegetation. In short, this CNN model was built using a training data set of 56,900 128 $\times$ 128 mm image portions collected from nine sites over three trips in 2020. These image portions were used to define key wetland feature classes using K-means clustering. Initially, the process involved unsupervised automated clustering, following a supervised manual clustering step where the wetland feature classes *Phragmites australis* reeds, water, bare ground, leaf litter and other vegetation were defined through manual sorting. Using these image portions within the defined wetland feature classes, a process of training, validation, testing and manual verification was undertaken. This method had an overall accuracy of 0.947 and recognized *Phragmites australis* to a very high accuracy (>98%).

Using the data collected, we present the percentage cover data for *Phragmites australis* for the three watering treatments (Flow 0, Flow 1 and Flow 2) across the seven surveys between October 2019 and March 2021. Three sites in October 2019, two in March 2020 and one in March 2021 could not be aligned and processed into a single image due to issues related to stitching and processing the images, and are therefore not included in the data presented in Figure 3.

To determine the effect of prior watering on the reedbed response to a large natural flood, we applied a before-after-control-impact design using the data collected during the surveys in January 2020 and January 2021. All nine sites were flooded by a large-scale natural flooding event in October 2020 (Figure 2). We estimated the percentage cover of *Phragmites australis* as a function of watering treatment (Flow 0, Flow 1 or Flow 2) and whether the survey was before or after the flooding event in October 2020, as well as the interaction between these factors. The January surveys were used because they displayed maximum reed cover for most sites observed during each year, and thus represented the maximum growth for that year. We fitted a linear mixed-effects model, with the watering treatment and the survey date as fixed effects in the model, together with the interaction between these two variables. The site was defined as a random effect in the model, to account for the fact that the sites were measured repeatedly. The Flow 0 treatment in January 2020 (before the flood) was set as the reference class in the model, and the estimated changes in percentage cover were relative to this.

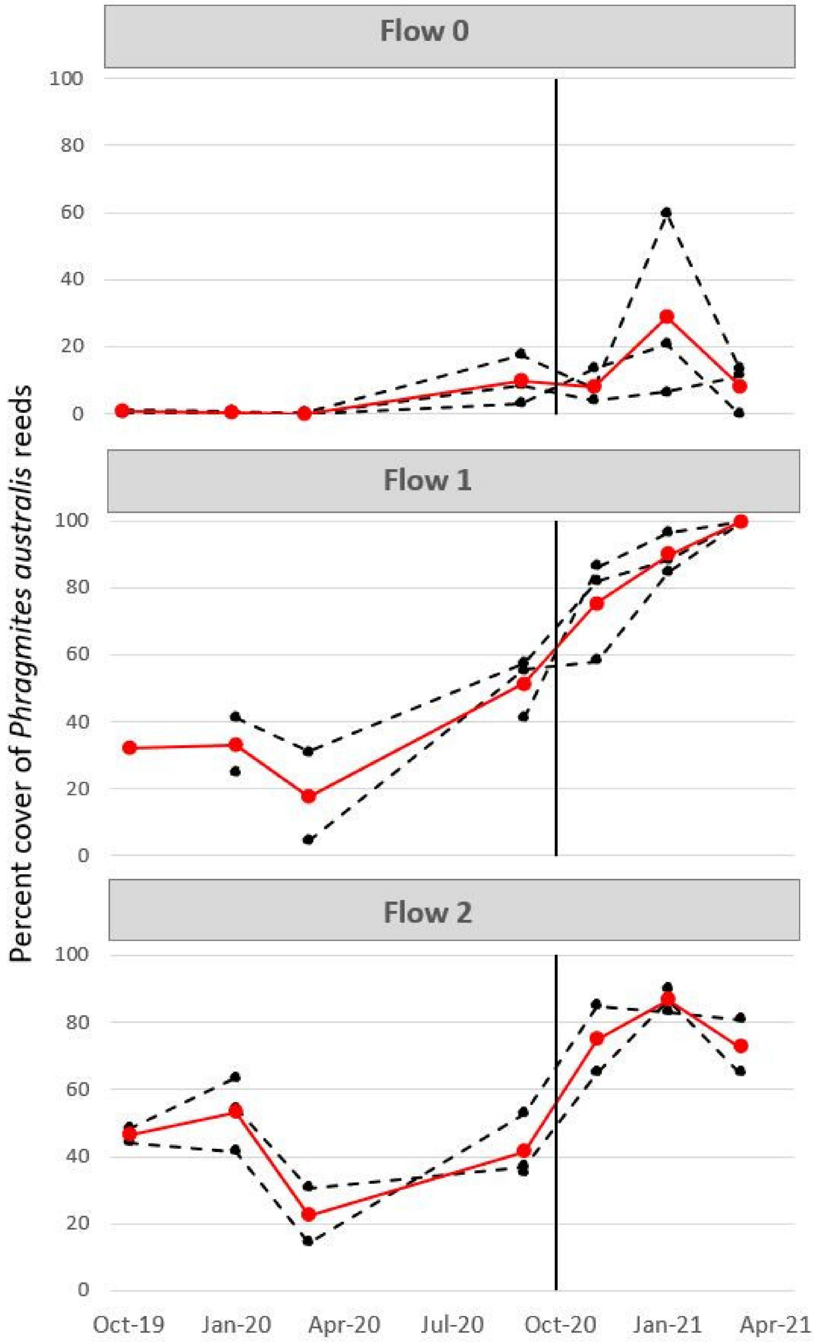

Figure 3. The percentage cover of *Phragmites australis* at each site during the survey, with sites grouped by watering treatment: Flow 0 = sites that had received no water since late 2016; Flow 1 = sites that received environmental water in June 2019; Flow 2 = sites that received environmental water in June 2019 and September 2019. Black circles represent the data for each site and the red circles represent the average across the sites at each survey for each treatment. The black vertical line shows the approximate time of flooding in October 2020, which occurred across all sites.

3. Results

The percentage cover of *Phragmites australis* varied considerably between sites within the three watering treatments during the first growing season (October 2019, and January and March 2020). During this period, sites with watering treatment Flow 0, which had not been flooded for three years prior to this, had very low cover of *Phragmites australis* (<1%, Figure 3) compared to sites with the watering treatments which had received environmental water in 2019 or in both 2018 and 2019. The percent cover of *Phragmites australis* in watering treatment Flow 2, which was the greatest of the three watering treatments during this period, increased between October 2019 and January 2020, which may have been related to the environmental watering action these sites received in November 2019.

The percentage cover of *Phragmites australis* increased for all three watering treatments following the flood event in October 2020. Sites with watering treatments Flow 1 and Flow 2 had a percentage cover of *Phragmites australis* of at least 80% in January 2021. The three sites with watering treatment Flow 0 varied in percentage cover of *Phragmites australis* in January 2021, with one site exhibiting nearly 60% cover and one just 6.6%.

In January 2020, sites with the watering treatments which received environmental water in 2019 (Flow 1 and Flow 2) showed a statistically significantly greater percentage cover of *Phragmites australis* compared to sites which had not been flooded since late 2016 (Flow 0) (Table 1 and Figure 4). In January 2021, sites with watering treatment Flow 0 showed a significant increase in percentage cover of *Phragmites australis* following flooding in October, shown as the effect of flood in Figure 3 (p = 0.0334), compared to the cover observed at these sites in January 2020. While all sites with all watering treatments showed an increase in percentage cover in January 2021, sites in the Flow 1 group showed the greatest increase in cover of *Phragmites australis* following flooding; however, this interaction between the flood and treatment variables was non-significant (p = 0.097; see Table 1). One of the sites within watering treatment Flow 1, for example, increased in cover of *Phragmites australis* from 30.6% in January 2020 to 84.6% in January 2021 (see images of this site taken in January 2020 and 2021 as the graphical abstract).

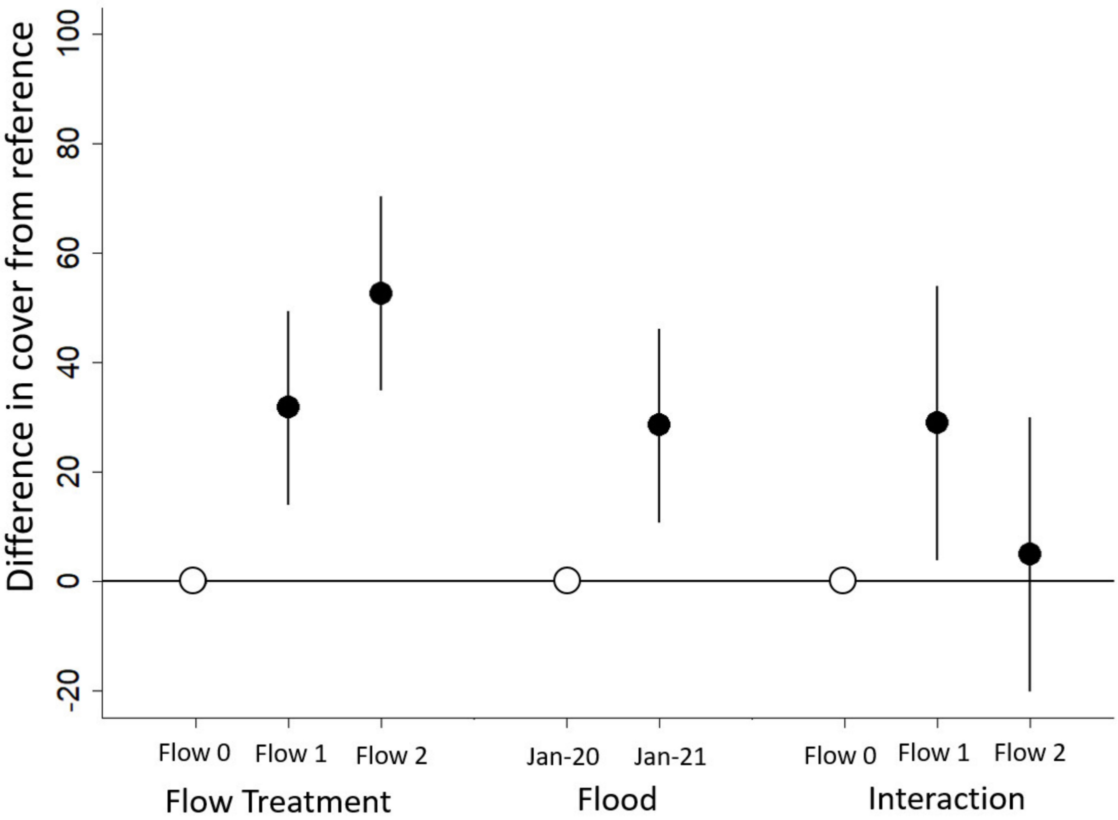

Figure 4. Results from a linear mixed-effects model analyzing the effect of two variables, i.e., flow treatment (Flow 0, Flow 1, Flow 2) and whether a survey occurred before or after a widescale flood event in October 2020, plus the interaction between these two variables, on the percentage cover of *Phragmites australis* reeds. The surveys occurred in January 2020 (Jan-20) and January 2021 (Jan-21). Filled circles show the estimated effects and vertical lines the 95% confidence intervals. All effects are estimated relative to the Flow 0 sites (shown as open circles).

Table 1. Summary of the results of the linear mixed-effects model. The fixed effects were variables classifying each site by flow treatment (Flow 0, Flow 1 or Flow 2) and survey trip as January 2020 (before the flood) and January 2021 (after the flood), plus the interaction between these two factors. The Flow 0 treatment in January 2020 (before the flood) was set as the reference class (intercept). The *p* and *t* values show the statistical significance, where * represents a statistically significant difference at $p < 0.05$.

Fixed Effects	Estimate	Standard Error	*t*-Values	*p*-Values
Intercept	0.3309	7.3857	0.045	
January 2020 Flow 1	31.8371	10.4450	3.048	0.0101 *
January 2020 Flow 2	52.7111	10.4450	5.047	0.0003 *
January 2021	28.5957	10.4401	2.739	0.0339 *
January 2021 Flow 1	29.0043	14.7645	1.964	0.0971
January 2021 Flow 2	5.0118	14.7645	0.339	0.7458

4. Discussion

This study sought to investigate the response of *Phragmites australis* reedbeds following flooding, and the role of reedbed condition and prior environmental watering. In doing so, we demonstrated the application of a drone-based remote sensing method using CNNs to monitor the reedbed condition of *Phragmites australis* reedbeds in a semi-arid floodplain wetland. This method was inspired by the difficulties of site access and field-based data collection in wetlands and the need for accurate and repeatable data. Collecting data via drone allowed for sites to be repeatedly surveyed in an efficient, consistent way, without risk of damaging the sites or species, or risk to field personnel, enabling cover to be tracked over time from a reasonable number of sites. This approach, along with the CNN model established by Higgisson et al. [20], provided a highly accurate method for estimating the percentage cover of *Phragmites australis* reeds, along with other wetland metrics which were not used here.

Since widescale flooding in late 2016, the floodplains of the lower Lachlan River, including the Great Cumbung Swamp, were predominantly in a dry state. The total rainfall for NSW for 2019 was the lowest on record (55% lower than the average) (http://www.bom.gov.au/climate/current/annual/nsw/archive/2019.summary.shtml accessed 5 January 2022), and that for 2018 was the sixth lowest on record (40% below average) (http://www.bom.gov.au/climate/current/annual/nsw/archive/2018.summary.shtml accessed 5 January 2022). During this period, environmental water was used once in 2018 and twice in 2019 to water the central reedbeds of the Great Cumbung Swamp, flooding the sites in watering treatment categories Flow 1 and 2. These watering actions aimed to maintain vegetation condition and provide drought refuge to native waterbirds during a period in which the river system was experiencing one of the worst sequences of dry conditions on record [49]. In the absence of these watering actions the reedbed would have remained dry for the four-year period between the two large flood events in 2016 and 2020. Under natural flow conditions, despite the dry conditions, the Lachlan River would have had greater volumes of water and more flow variability, and the reedbed would have been connected to the river by flooding for approximately 69 days between July 2019 and June 2020, rather than just 10 days (as the result of the environmental watering actions) [49]. Drought conditions, coupled with the changes in flow regimes resulting from water resource development, have the capacity to cause irreversible changes to freshwater ecosystems [50].

The reedbed sites that had received environmental water had significantly greater cover of reeds during the first monitored growing season compared with those that remained dry. These managed sites continued to improve following the natural flood in October 2020 and had a much greater cover of *Phragmites australis* following the flood compared with those which had not been managed. This illustrates how environmental water can be successfully used to prime the system to better respond to natural events.

The finding that sites that were more frequently flooded (watering treatments Flow 1 and 2) had a greater cover of reeds is consistent with a study by Whitaker et al. [18], where sites in the reedbed of the Macquarie Marshes, New South Wales, Australia that were more frequently flooded were found to have taller *Phragmites australis* reeds with greater biomass compared to sites higher on the floodplain which were flooded less frequently. A water deficit has been shown to reduce leaf production, leaf area and biomass, and to increase leaf shedding in *Phragmites australis* [51]. The rhizomes of *Phragmites australis* are important as a storage organ and a bud reservoir, and are recognized as being key to their condition [48]. It was found in an unpublished study that the biomass of rhizomes (per m^2) of *Phragmites australis* reeds was greater for watering treatments Flow 1 and Flow 2 compared with Flow 0 (James et al., unpublished). Shallow local groundwater may play an important role in maintaining *Phragmites australis* between flooding events [39], and the reduced leaf area reduces water demand [51].

Following the flood in 2020, sites with all three watering treatments showed increased cover of *Phragmites australis*; however, sites in the Flow 1 group showed the greatest increase following the flood. This suggests that the reeds at these sites had energy reserves within their rhizomes and were able to rapidly respond to flooding by lateral expansion and an increase in density. The reason why the response at these Flow 1 sites was greater than in watering treatment Flow 2 sites, which had a greater cover in January 2020 prior to flooding, may be that there was more available space compared to sites in the Flow 2 group, as there was a lower cover of reeds prior to the flood. Field observations suggest that the maximum possible cover is likely to be between 85 and 100%, with local micromorphology becoming the limiting factor rather than the condition of the reedbeds per se, which limits the possible response of sites in both Flow 1 and Flow 2 treatment groups. Other characteristics of reedbeds (such as number of stems or flower heads) other than percentage cover may continue to improve/change once cover has peaked.

Maintaining wetland vegetation condition is a typical aim or objective in the use of environmental water for riverine and floodplain vegetation in Australia [52]. Reedbeds can transition between condition classes either on a declining trajectory based on the number of consecutive years with no floods or a recovery trajectory based on the number of consecutive years with floods [6,48]. Here, we defined approximate condition classes for *Phragmites australis* reedbeds based on the work of Overton et al. [48] and Bond et al. [6] and the results from this study (Figure 5). Environmental water maintained reedbeds in a Medium condition, and these reeds then transitioned into a Good condition following the larger flood event. The reedbeds which were not managed with environmental water appear to cycle between the Critical condition and the Medium condition following the larger floods (Figure 5). These reeds in poorer condition are less resilient to current and projected changes to flow regimes, climate and other stressors and drivers [31], and further flooding is needed to continue improvements in condition at these sites.

As predicted by Bond et al. [6] and Overton et al. [48], we showed that *Phragmites australis* reedbeds in a Medium condition can transition to a Good condition following one flood, whilst reedbeds in a Poor or Critical condition require two consecutive years with floods in order to transition to a Medium condition. Providing environmental water to inundate parts of the reedbed during dry conditions provides the conditions that support vigorous growth following a natural flood, contributing to river–floodplain resilience. This is further evidence of the role of the small and medium-sized floods that have been lost to many inland river systems due to regulation (e.g., [5]).

Cover of reeds was variable in January 2021 in watering treatment Flow 0, suggesting a variable response to flooding, possibly related to variation in flooding duration across the floodplain, as well as rhizome condition and starch storage reserves at these sites. While Roberts and Marston [30] recommended flooding every one to two years for maintenance of *Phragmites australis*, these results support the predictions of Bond et al. [6] that reedbed condition prior to restoration influences the ecosystem's response to flooding and that sites in a Poor or Critical condition need consecutive years with floods to improve reedbed condition.

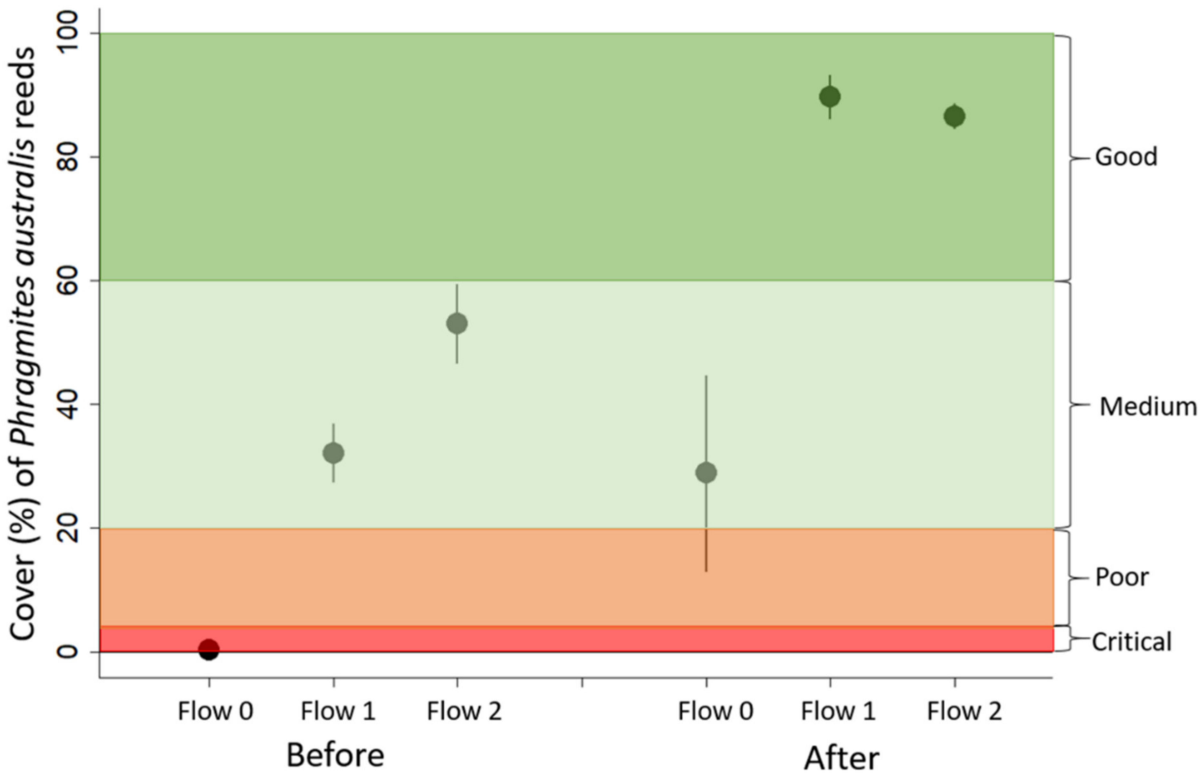

Figure 5. Mean (standard error) percentage cover of *Phragmites australis* reeds in January 2020 (before) and January 2021 (after), for the three watering treatments before and after a large flood event in October 2020, which flooded sites in all treatments. Shading represents estimates of condition based on those defined for emergent macrophytes by Overton et al. [48].

5. Conclusions

Here, we built on the method developed by Higgisson et al. [20] by demonstrating its application in estimating reedbed cover over time at sites under different environmental water management regimes, and showed how these sites responded following a widescale flooding event. We demonstrated the important role that environmental water played in filling the gaps between large flood events and, in doing so, maintaining the condition and resilience of reedbeds. Whilst parts of the reedbed that were managed with environmental water transitioned into a Good condition following a large-scale flood, the reeds that were not managed with environmental water were in a Critical condition prior to the flood. Water resource development has reduced the frequency and extent of flooding in floodplain wetlands in Australia [4], including in the Great Cumbung Swamp [5]. This study highlights the fact that, without environmental water, extended dry phases pose a significant risk of widescale losses to reedbeds. The use of UAVs and CNNs in this study provided a way to obtain data that demonstrate the effects of prior watering and capture the response of the reedbeds to a large natural flood. As such technology becomes increasingly accessible, it is likely to have far wider application and will be a useful tool in helping water managers to better manage floodplain wetland systems.

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Data Availability Statement: The data used as part of this study is stored at the Centre for Applied Water Science, University of Canberra and maybe provided upon reasonable request.

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