Impacts of wildfires on the survival of *Phragmites australis*: Results of a field experiment.

Report within the framework of LIFE Project “Prespa Waterbirds” LIFE15 NAT/GR/000936

PRESPA WATERBIRDS

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Acknowledgements

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Contents

Summary

Aim: Wildfires are major drivers of vegetation change in reedbed communities of lakes with similar impacts on helophyte species. Depending on the time of the year that they occur, with respect to the water level, they can change the vegetation structure in different ways. At Prespa National Park, NW Greece, wildfire events occur regularly in reedbed communities in late winter/ early spring, while the water level of Lesser Prespa Lake fluctuates 85 cm on average within the year. The effects of fires are expected to vary depending on whether they are followed by a rise of the water level, resulting in the submersion of the remaining parts of reed culms.

Methods: In order to test the effects of culm submergence after fires, a replicated field experiment was conducted in eight selected sites of pure reedbed in Prespa National Park (Greece) with reed harvest treatments simulating fire. In every site, three plots of 25 m^2 were marked with metal piles. One plot was left intact to serve as a control and two different treatments were applied; (i) cutting *P. australis* stems <5cm above-ground and (ii) cutting them ≈ 1 m above-ground. All plots were cut in November 2017 while the reedbed structure was measured in August 2018. In every plot, 8 quadrats of 0.4x0.4 m were sampled. Structural (density, culm height and diameter) and water level parameters were recorded.

Analysis: Linear mixed models were used to statistically identify differences between the two treatments and the control.

Results: Culm density of *P. australis* was 70% lower in the “Ground” harvest treatment than in the “Meter” treatment (*p-value* < 0.001) and 71% lower than in the “Control” (*p-value* < 0.001) while no significant differences were detected between the last two. Additionally, *P. australis* maximum culm height, maximum diameter, random height and random diameter were significantly smaller in the “Ground” treatment than in the “Meter” treatment and in the “Control”. The topographic position of the plot had a relatively small but significant additional effect on the density of *reed* culms with a marginally significant interaction between land elevation and treatment.

Main conclusions: Cutting followed by no flooding or flooding of short duration has little if any impact on the structure of the reedbed. In contrast, long flooding after cutting results in strong decrease of the density of *P. australis* culms and can significantly modify all of its important structural parameters. The combination of these two factors could explain the decrease of *P. australis* in large patches in recent years: wildfires reduced *P. australis* culms to ground level, as in the ground cutting treatment.

Keywords: *Phragmites australis*, common reed, Prespa National Park, reedbed, hydrology, culm submergence, wildfires, vegetation management, vegetation cutting

I. Introduction

During the last 25 years, large scale changes have been observed in the reedbed communities of Prespa National Park (PNP), especially around the area of Vromolimni (Grillas *et al.,* 2018a; Sakellarakis *et al.*, 2018). More specifically, narrowleaf cattail (*Typha angustifolia* L.) has largely encroached upon common reed [*P. australis* (Cav.) Trin. ex Steud.] stands. Further expansion of these cattail communities could deprive Dalmatian pelican *Pelecanus crispus* and the great white pelican *Pelecanus onocrotalus* of their nesting substrates, since these bird species are known to create their nests only on islands consisting of *P. australis* rhizomes in PNP (Catsadorakis & Crivelli, 2001). Both species are included in Annex I of the 2009/147/EC Directive and are listed as Vulnerable in the Greek Red Data book (Catsadorakis *et al.,* 2009a; Catsadorakis *et al.,* 2009b). Accordingly, the evaluation of any factor that undermines the pelicans' survival in PNP is imperative and must be carefully addressed.

P. australis and *Typha* spp. are among the most common plant species in tall helophytic vegetation and they tend to create almost monospecific communities, poor in plant richness. They are found in a wide range of environmental conditions (soil, climate & nutrients). In shallow water the two species form a common plant association – both in their native range and in areas where they are characterized as aliens (Mason & Bryant 1975). Tall helophyte species are generally highly competitive and tend to form monospecific stands (Grillas *et al.,* 2018a). In addition, they are resilient to changes of ecological conditions and can encroach upon neighboring habitats – including other helophyte species – due to their clonal growth form (Hara *et al.* 1993; Amsberry *et al.,* 2000; Luo *et al*., 2014). The replacement of one helophyte species by another is usually explained by changes in physical conditions (notably changes in hydrology, anoxia or eutrophication), disturbance or encroachment of invasive alien species (e.g., Boar *et al.,* 1989; Coops *et al*., 1999; Boers & Zedler 2008; Uddin & Robinson 2018).

Studies of reedbeds in PNP have shown that the ecological niches of *P. australis* and *T. angustifolia* are almost fully overlapping (Grillas *et al.,* 2018a). Based on this result, the hypotheses that the changes in their spatial distribution in Prespa were mostly driven by a recent nutrient inflow (e.g., eutrophication), or different environmental conditions in their stands, were rejected. Another possible explanation may well be that stochastic processes drove the encroachment of *T. angustifolia* (Grillas *et al.*, 2018b). An example of such a process is wildfires, which typically occur on an almost yearly basis mainly in late winter/ early spring (February – March) in the study area and are deliberately set by people, in combination with an environmental extreme, such as flooding or drought events.

Typha species are fast colonizers producing thousands of viable seeds (70.000-200.000 per individual), each with numerous hairs that facilitate wind dispersal when dry (Vaccaro 2005). In contrast, colonization of *P. australis* by seedlings is rare (Engloner 2009), with regeneration usually occurring mostly through clonal growth (Koppitz 1999) after drawdown and when shallow water prevails (Mauchamp *et al.,* 2001). Thus, where there has been large-scale destruction of common reed stands, there will be favorable ground for the invasion of *Typha* spp. if the species is present in the species pool or the seedbank of the area.

Wildfires are known to have variable effects on *P. australis* growth and stand development (Van der Toorn & Mook 1982; Thompson & Shay 1985; Ostendorp 1999). Post-fire vegetation growth is controlled by several factors: (i) the intensity of the fire, (ii) the time of the year that wildfires occur with respect to the phenological state of *P. australis*, e.g., if the fire occurs while the above-ground organs of *P. australis* are dead or alive; and (iii) the water-level, if the reedbed occurs in occasionally or constantly flooded areas. Little or no impact has been noticed on *P. australis* when fire occurred on wet or flooded soils and before the emergence of new culms (Van der Toorn & Mook 1982) and then the effects of fire are similar to those of winter harvesting (Ostendorp 1999; Sinnassamy & Mauchamp 2000). Instead, burning during the emergent period in both wet and dry treatments resulted in the mortality for the majority of culms.

Changes in water levels are also important to consider after the fire (Motivans & Apfelbaum 1987; Sinnassamy & Mauchamp 2001). If the wildfire event occurs during the annual water minimum, then the burnt *P. australis* culms may stay submerged in spring during the gradual increase of the water-level. This can affect the rhizome survival of *P. australis* since its culms must remain above water for the needs of respiration, elimination of toxic gases and oxidation (Armstrong *et al.*, 1999). The negative impact of suppressing this oxygen fluxes from aerial parts increases when soil oxygen is depleting (Weisner & Granéli 1989). It is therefore crucial to elucidate the post-cut as well as post-fire development of *P. australis* in the absence and presence of subsequent flooding. This can shed light into the possible reasons underlying the *T. angustifolia* encroachment and help in preventing any further expansion of this helophyte species. Additionally, understanding the ecology and responses of *P. australis* can also serve as guidelines for the active management of its stands in the study area.

Fire and harvesting have similar impacts on reedbeds as long as the fire has not impacted the soil surface and below-ground parts of the plants (Sinnassamy & Mauchamp 2000). Because fire is difficult to control in reedbeds we have tested experimentally *in situ* the impact of reed harvest on the structure of *Phragmites*-dominated reedbeds.

The objectives of this experiment were to assess the effects of winter[1] fires, when the above-ground organs of *P. australis* are dry, on the density, height and diameter of culms of *P. australis*

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[1] As winter we define the period in which the above-ground organs of *P. australis* are dry. At Prespa, culms start to turn “yellow” from the end of October.

stands. Cutting of *P. australis* was employed, as a mean of simulation of fires, both in the presence (ground cutting) or absence (1 m above-ground cutting) of subsequent winter flooding.

II. Methods

II.a. Experimental design

Experiments were established in November 2017. Ten sites were selected in the SW area of Prespa National Park where large monospecific *P. australis* dominated reedbeds occur on the littoral zone of Lake Lesser Prespa (Figure 1). Two criteria were used to select each site: (i) its ecological and floristic homogeneity; and (ii) elevation characteristics, as the site had to be flooded with the lake's water for at least one month each year.

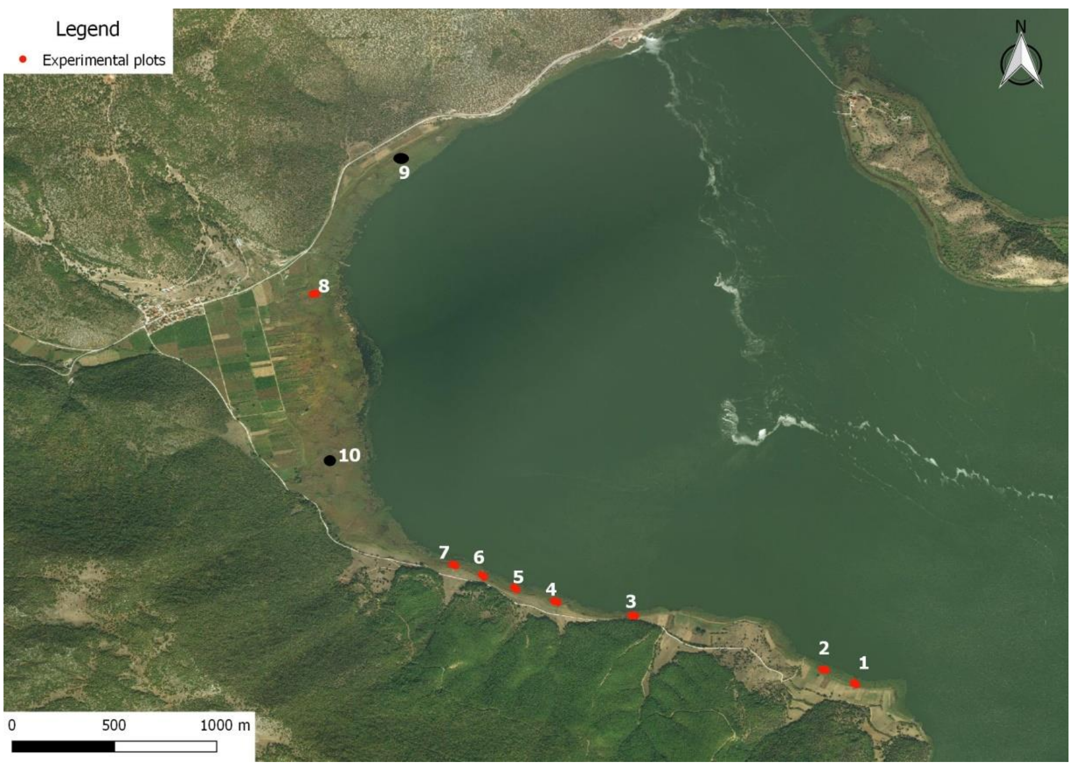

Figure 1. Position (red dots) of the experimental sites. Site nos. 9 and 10 (black dots) were burnt during fire events that took place in February-March 2018 and were removed from the final dataset of the analysis.

During the experiment, the water level of Lesser Prespa was low at the time of cutting (November 2017) and remained low in the beginning of 2018 (January-April) in comparison with the mean of the last 23 years (1995-2018; Figure 2). However, during the rest of the year it reached higher levels.

Three 25m² (5 X 5m) plots were established in each of the 10 sites (although as described above two were burnt by fire, so only eight contributed to analyses). Plots were at least 3 m apart. Each plot corner was marked with a metal pole and the position of the pole was recorded with a handheld GPS device (GARMIN 64 BASEMAP) to ensure it could be relocated for monitoring.

Three different treatments were implemented: (a) a clear cut of *P. australis* culms 2-4 cm from the ground level – hereafter termed "Ground" treatment, (b) cut of *P. australis* culms one meter above-ground – hereafter termed "Meter" treatment– and c) uncut ("Control") reedbed. The treatments were chosen in order to simulate two different submergence conditions following fires (in this case harvesting): (a) the "Ground" treatment was expected to simulate long submergence of culms after fire events and (b) the "Meter" treatment was expected to simulate a short submergence period after fires, i.e. cutting in November at 1-m above ground would not allow submergence of the culms in early spring of next year, thus simulating conditions following the patterns of reed burning in Prespa in late winter/ early spring. The implementation of the "Meter" treatment in November was chosen over harvesting in late winter/ early spring for practical reasons, as conditions in late winter/ early spring may not allow for harvesting to be undertaken (e.g. frozen lake/ high water level).

In each site, each treatment was randomly allocated to one plot. Cutting was implemented during November 2017 with the use of mechanical machinery. All the cut *P. australis* biomass was removed from harvested plots.

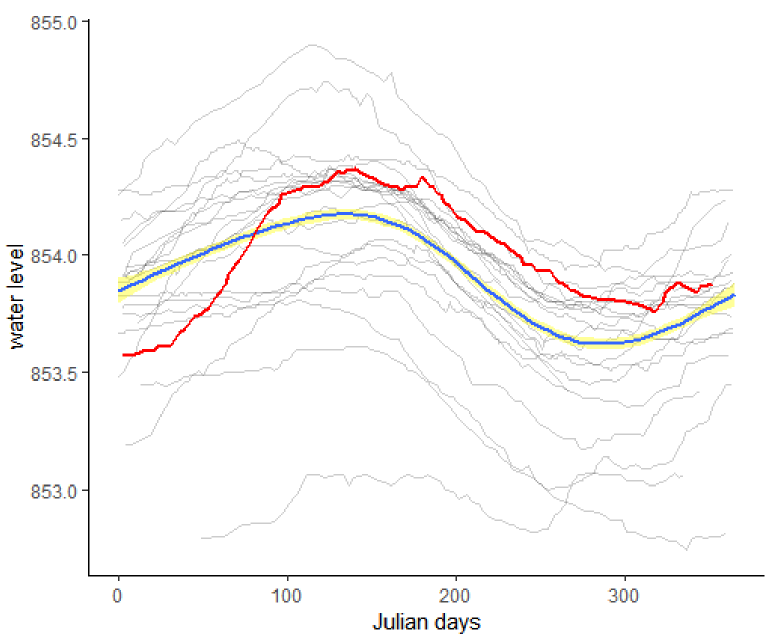

Figure 2. Water level (m) of Lesser Prespa during 2018 (red line is the year of the experiment implementation; Julian day 0 is the 1st of January). Water level fluctuations from 1995 are depicted. Blue line corresponds to the mean and with the yellow intervals the standard error.

II.b. Monitoring

Reedbed structure was measured across 4 successive days in early August 2018. A 4 m transect was established in the middle of every plot to avoid any edge effects. Eight quadrats (0.4x0.4 m) were sampled in every plot with a distance of 0.5 m from each other. In one site (no. 1), 6 instead of 8 quadrats were sampled for every treatment and the control. In every quadrat, *P. australis* culm density and the height and basal diameter of the tallest and of a random culm (the closest individual to the lower right corner of the quadrat) were recorded. Seventy-eight quadrats were assessed for every treatment.

Four measurements of the water level, one in every corner of each quadrat, were recorded. The length of 5 year-old cut culms was measured in every quadrat in the Meter treatment for the calculation of the submergence duration. The duration of flooding of soil surface for all the plots and the duration of submergence of culms cut at the Meter treatment plots, was calculated by using available water level data for Lesser Prespa Lake (data courtesy: Society for the Protection of Prespa). Correlation between the water level fluctuation in the plots and the sluice gate of Lesser Prespa, where the water level measurement took place, was cross-validated during spring (April-May) 2018 and summer (July-August) 2018. All plots in every site were found to have the same water fluctuation as the lake's water level.

The elevation of every quadrat was calculated by the water level data (4 measurements per quadrat) and the water level of the lake on the day that the fieldwork was performed.

II.c. Data analysis

Two sites (no. 9 & 10) were removed from the dataset before analysis because they were burnt during wildfires events in February-March 2018.

All dependent variables were tested for normal distribution with the Shapiro-Wilk test (Shapiro & Wilk 1965). Linear mixed models were then used to investigate the effects of treatments (i.e., Ground, Meter and Control); within-site and within-treatment correlations in all dependent variables were treated with random intercepts, with treatments nested within each site and quadrats nested within plots. The 0.05 criterion was used for statistical significance in all models. For model validation, we plotted the histogram of residuals and the residuals vs the fitted values. Linear mixed models were performed in RStudio (R Core Team 2013) using the 'nlme' package

(Pinheiro 2014). Post-hoc comparisons for each pair of treatments were calculated with Tukey's HSD, using the 'multcomp' package (Hothorn *et al.*, 2013). Confidence intervals were calculated with the 'effects' package (Fox *et al.,* 2018). Conditional and marginal R^2 were calculated using the package 'MuMIn' (Bartón 2015). Colinearity between variables was investigated using Spearman's correlation with the 'corrplot' package (Wei & Wei 2016). Graphs were produced using R functions and the package 'ggplot2' (Wickham 2009).

III. Results

III.a Physical conditions

All plots were constantly flooded from early January 2018 until the date of the measurements (August 2018). Culms in Ground treatment plots were constantly submerged, while the submergence duration for culms in the Meter treatment varied. Differences in the submergence duration do not reflect micro-relief heterogeneity of the plots but are a result of the uneven cutting of the culms that was implemented with the mechanical machinery during the establishment of the experiments. No statistical difference (Control-Meter *p-value*= 0.938, Control-Ground *p-value*= 0.116, Meter-Ground *p-value*= 0.227) was found for water-level between the three different treatments (Figure 3). Additionally, no statistical difference was found for the land elevation between the treatments (mean land elevation: 853.6 ± 0.12 m).

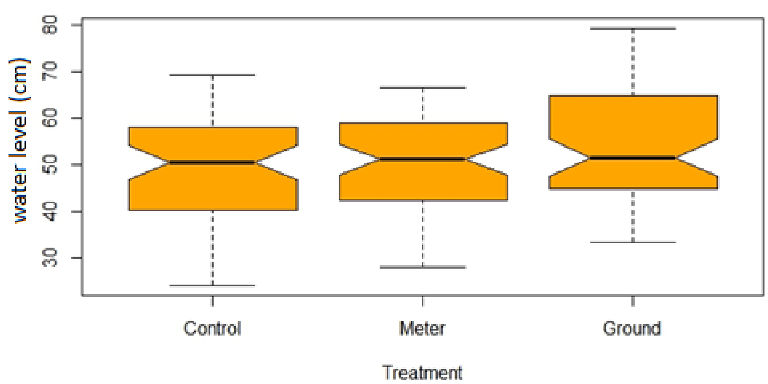

Figure 3. Notched boxplots of the water level of the plots in all eight sites that were included in the analysis. The box shows the interquartile range (IQR) of the data while the whiskers add 1.5 times the IQR to the 75 percentile and subtract 1.5 times the IQR from the 25 percentile. Bold

black line corresponds to the median. Notches display the 95% confidence interval around each median.

III.b. Effect of the treatments on the structural parameters of *P. australis* stands.

The density of culms was significantly impacted by the cutting treatment explaining most of the variance, but also by the elevation of the plot; the interaction of these variables was marginally significant (p=0.05, Table 1, Figure 4). Even though there was a correlation in the density of the culms of *P. australis* with increasing elevation in the Meter treatment and the Control, no effect was found in the "Ground" treatment. Reedbeds in Prespa tend to be denser in higher elevation where the flooding period tends to be shorter (Sakellarakis & Grillas, unpublished). The absence of a trend in the Ground treatment most probably is a result of the constant flooding and submergence of *P. australis* culms.

The height and the diameter of the culms were significantly impacted by the treatment with no significant influence of the elevation of the plots (Table 1).

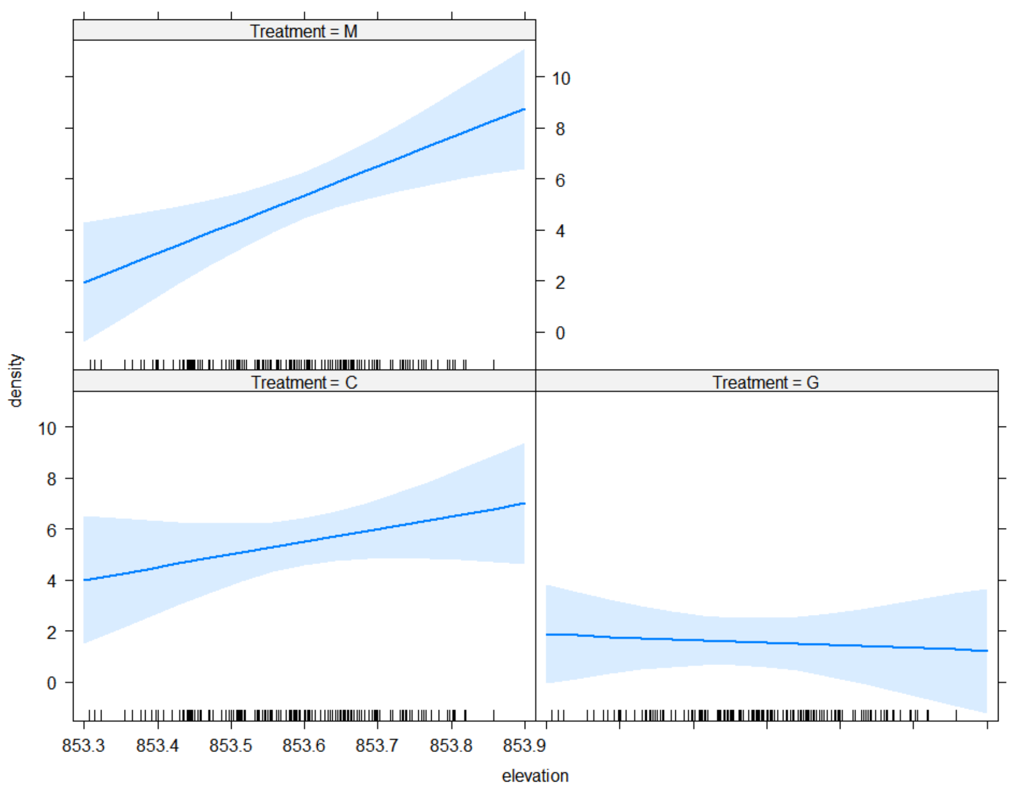

Figure 4. Density of culms (individuals/quadrat) versus elevation (m asl) in the two different treatments and the control.

Table 1. Results of the Analysis of Variance testing the effects of treatment (G, M and C), nested in site and plot, elevation and their interaction. Significant effects are in bold character.

	Sum of squares	Mean square	Degree of Freedom	F value	P value
Density of culms					
Cutting treatment	**249.0**	**124.5**	**2**	**25.4**	**0.000**
Elevation	**24.8**	**24.8**	**1**	**5.1**	**0.034**
Treatment X Elevation interaction	**34.2**	**17.1**	**2**	**3.5**	**0.050**
Maximum height of culms					
Cutting treatment	**124.9**	**62.5**	**2**	**7.5**	**0.005**
Elevation	6.3	6.3	1	0.8	0.390
Treatment X Elevation interaction	3.6	1.8	2	0.2	0.809
Maximum diameter of culms					
Cutting treatment	**25.5**	**11.8**	**2**	**10.0**	**0.001**
Elevation	0.6	0.6	1	0.5	0.491
Treatment X Elevation interaction	1.1	0.5	2	0.5	0.632

The culm density in the Ground treatment was 70% lower (10 ± 9.9 culms/m²; Picture 1) than in the Meter treatment (*p-value* = 0.003; 33 ± 17 culms/m²) and the Control (*p-value* = 0.002; 35 ± 19 culms/m²). Cutting at 1 meter height had no significant effect (not different from the Control treatment) on the culm density (*p-value* > 0.05; Figure 5 & 6, N=62 for every treatment). Conditional R^2 (the variance explained by the fixed and random factors) was 0.49 and marginal R^2 (the variance explained by the fixed factors) was 0.33.

The tallest culm per quadrat (Figure 7) was significantly shorter in the Ground treatment plots (2.4 ± 1.9 m) than in the Meter treatment (*p-value* < 0.000; 4.3 ± 0.8 m) and in the Control plots (*p-value* < 0.000; 4.2 ± 0.9 m). The culm height was not significantly different between the Meter treatment and the Control (*p-value* = 0.954). Conditional R^2 was 0.53 and marginal R^2 was 0.31.

Furthermore, the height of a random culm (Figure 7) was significantly smaller in the Ground treatment (5.9 ± 4.7 mm) than in the "Control" (*p-value* = 0.003; 8.3 ± 2.4 mm) and in the Meter treatment (*p-value* < 0.001; 9.4 ± 2.8 mm); the height of random culm did not differ between the Meter treatment and Control (*p-value* = 0.885). Conditional R^2 was 0.44 and marginal R^2 was 0.21.

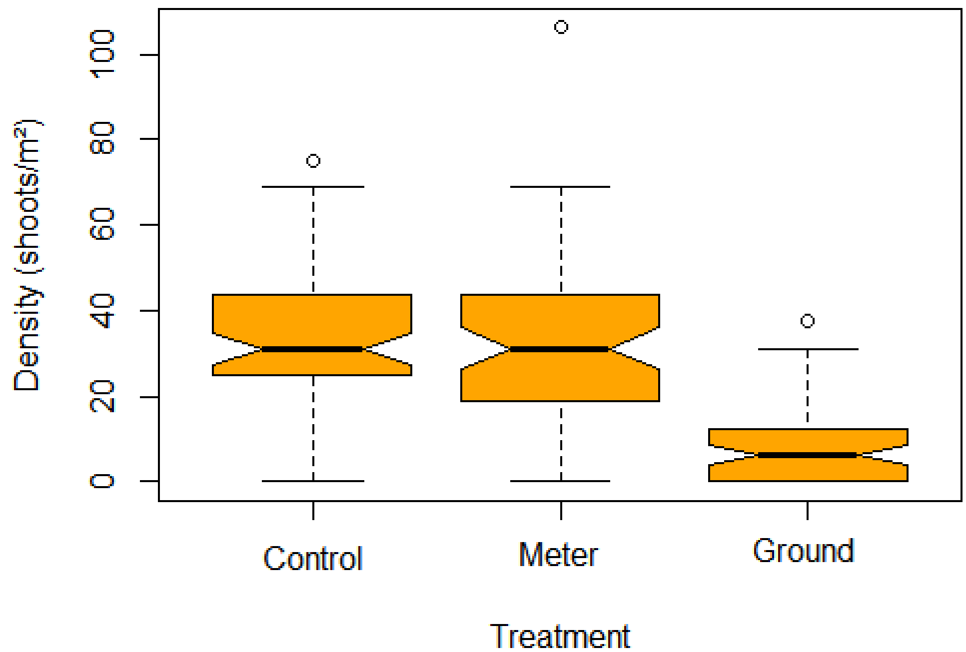

Figure 5. Notched boxplot of densities (culms/m²) per treatment.

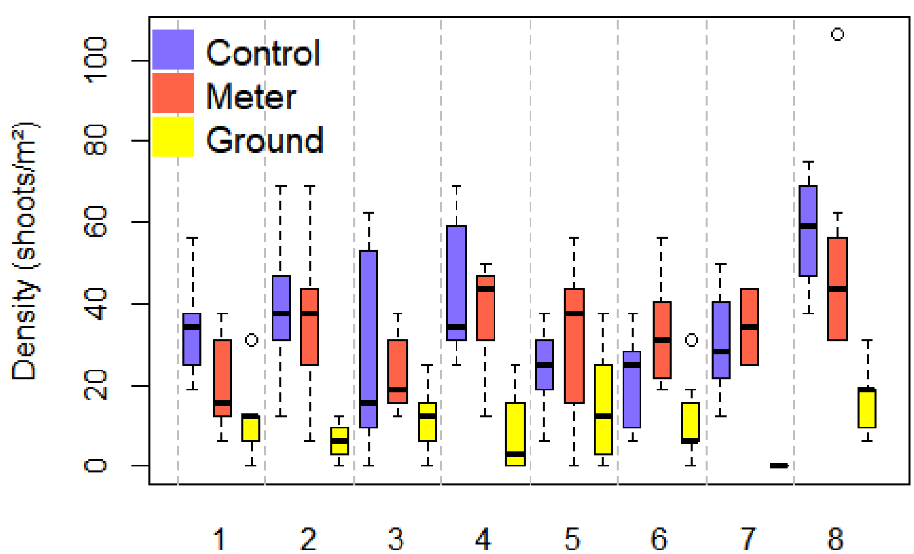

Figure 6. Boxplots of densities (culms/m²) per treatment and per site.

Picture 1. Treatment effect at site no. 7 (photographs captured on the 8th May 2018): (A) Ground treatment & (B) Meter treatment. Culms in the Meter treatment are approximately 4 m tall.

The diameter of the tallest culm per quadrat (Figure 7) was significantly smaller (about 40% decrease) in the Ground treatment (6.5 ± 5 mm) than in the Meter treatment (*p-value* < 0.000; 11.21 ± 2.3 mm) and the Control (*p-value* = 0.003; 10 ± 2.3mm) but no significant difference was detected between the last two (*p-value* = 0.54). Conditional R^2 was 0.47 and marginal R^2 was 0.25

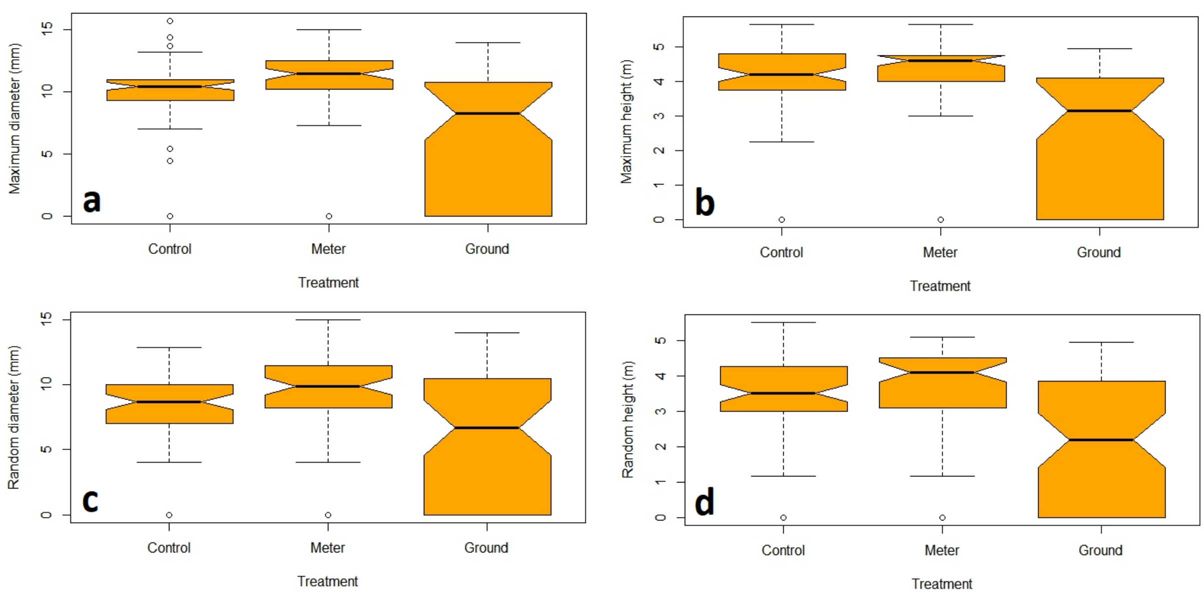

Figure 7. Notched boxplots of (a) maximum diameter; (b) maximum height; (c) random diameter and (d) random height.

IV. Discussion and Conclusions

All parameters of the structure of the reedbed were impacted by cutting the culms at Ground level while cutting culms at 1 m height had no effect on the structure of the reedbed. Many studies have shown that winter harvesting (or fire) per se, results in various impacts on reedbed structure:

- An increase in culm density (Ingram *et al.,* 1980; Björndahl 1985; Thompson & Shay 1985; Granéli 1989; Cowie *et al.,* 1992; Ostendorp 1999; Poulin & Lefebvre 2002; Schmidt *et al.,* 2005).
- A decrease in culm height (Ingram *et al.,* 1980; Björndahl 1985; Granéli 1989; Cowie *et al.,* 1992; Ostendorp 1999); however other studies did not find significant differences (Granéli 1989; Schmidt *et al.,* 2005).
- No significant effect was found by Poulin & Lefebvre (2002) on culm diameter after winter harvesting, however, a significant decrease of culm diameter was found after summer harvesting by Karunaratne *et al. (*2004).

The impact of harvesting is cumulative over years in reedbeds but the mechanisms are still not completely understood. Hypotheses include damage to the superficial rhizomes, change in the microclimate and reduction of apical dominance (Ostendorp 1999).

However, in our results, the impact of winter harvest occurred only when culms were harvested at ground level and not when they were cut one meter above-ground. Thus, it was not harvesting itself which impacted the regrowth of the reedbed but most probably the complete flooding of the culms of the remaining culms which results in oxygen deprivation of rhizomes.

Belowground parts (tubers, rhizomes, taproots, etc.) of helophytes, when dormant, can tolerate deprivation of oxygen even though the duration can significantly differ between helophyte species; for *P. australis* and *Typha* spp. it can be up to one month (Crawford 1987). Mechanisms for the internal transfer of oxygen to belowground parts exist through lacunar parenchyma (e.g. *Schoenoplectus lacustris*) or leaves (*Typha* spp.). In *P.australis* the entrance of air (and oxygen) occurs mainly in living culms through the stomata on leaf sheets and in dead culms from culm section to the underground rhizomes (Brix 1990, Armstrong & Armstrong 1991; Brix *et al.,* 1996, Armstrong *et al.,* 1996). Total submergence of *P. australis* affects their survival of all stages in its life cycle (Armstrong *et al.,* 1999; Mauchamp *et al.,* 2001; Mauchamp & Methy 2004).

Therefore, the strong impact of harvesting at ground level is explained by the long duration of submergence (>7 months) of the remaining parts of the culms when the water level of the lake rose in spring (rainfall and snow melt). The importance of elevation on culm density is highlighted by the positive relationships between density and elevation, higher elevation corresponding to lower duration and depth of flooding. The lack of significant relationship between the duration of submergence of culms in the meter treatment is thus not consistent with the global effect of elevation measured (and corresponding differences in submersion). This could be explained by the fact that the number of emerging culms was sufficient to provide oxygen to the underground network of rhizomes.

Although the height and diameter of culms were impacted by cutting at the ground level, they were not significantly impacted by the topographic level of the plots. The impact on the height and more particularly on the diameter of the ground treatment culms resulted in weaker culms than in not cut (or not burnt) reedbeds making them more sensitive to mechanical damage or predation (Ostendorp *et al.,* 1999).

Recovery after cutting can be relatively rapid (1-2 years) but repeated cutting at ground-level/followed by flooding can have a cumulative impact resulting in a strong decline of reedbeds (Ostendorp *et al.,* 1999). Fires in Prespa have a similar effect on reed culms to our Meter treatment: removing above-ground culms at a time (February to April) when limited submergence will follow within the remaining winter and early spring. Therefore, repeated fires during late winter/ early spring, even when they are followed by water level increase, in Prespa reedbeds will probably not weaken *P. australis* stands. On the contrary, the impact in weakening the reedbed would be most severe if fires occurred early in the winter in dry years and followed by a long duration of flooding of the remaining basal parts of the culms. The establishment and maintenance of fire breaks could prevent the propagation of wildfires.

The main conclusions of the analyses are:

a) Fires in late winter/ early spring (or harvesting above-ground) probably have a limited direct effect on the structure of the reedbed in the following growing season.

b) Submergence of cut *P. australis* culms decreases the density of the culms as well as their diameter and height. The mechanism of impact is oxygen deprivation of the rhizomes since their gas exchange in winter is made through the standing dead culms. The duration of submergence is an important factor determining the impact on plants, probably with non-linear effects, but could not be studied in details (different durations).
c) Fire events often take place during the late winter/ early spring period, when the water level of Lesser Prespa is on the rise. The rate of the rising water level in spring does not allow the long-term submergence of the burnt culms, and does not lead to a reduction of their density, diameter and height, as shown for cut culms (in the Meter treatment of our experiment).

d) On the contrary, after long submergence, *P. australis* culms are generally shorter and with smaller diameter, thus stands have less above-ground biomass. Fires followed by flooding reduces the strength of the culms.

e) Conclusively, the impact of fire followed by submergence will be varying with the date of fire events related to the fluctuation of the water level and the phenology of the plants. The impact will increase along with the duration of culm submergence before new shoots can emerge. As soon as new shoots emerge they provide oxygen to the underground parts. Most fire events in Prespa reedbeds had probably a limited impact on *Phragmites* populations.

f) It should be noted however, that the dramatic change from *P. australis* to *T. angustifolia* (1988-1998) has been attributed to the successive occurrence of intense drought and probably fires followed by "historically" high and long lasting water levels thus resulting in long duration of submergence. Wildfires, in combination with *P. australis* culm submergence, may partially explain the encroachment of *T. angustifolia* against common reed. *T. angustifolia* spreads mostly using seeds, so may be better able to colonize burnt reedbeds than *P. australis*, which spreads mostly through rhizomes from neighbouring clones. The absence of a close and competitive reedbed dominated by the common reed might favor the germination and the establishment of the cattail.

g) As shown in the "Ground" treatment, reedbed management aiming at the control of *P. australis* encroachment in littoral areas (e.g. the restoration of wet meadows), could be applied at ground level even during early winter (November/ December) with a strong direct effect (i.e. a reduction of helophytes by 70%), as far as it is followed by long-term submergence.

V. Literature

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Appendix I. Tables of the models

Table 1. Results of the linear mixed model (water level plots).

	Value	Std.Error	DF	t-value	p-value
(Intercept)	49.3129	3.9555	162	12.4668	0
treatment G	4.9709	2.5067	14	1.9830	0.067
treatment M	0.8514	2.5067	14	0.3397	0.739

Table 2. Summary of fixed and random effects for the linear mixed model of water level and treatment.

fixed effectes	**Density**		
	Ground	**Meter**	**Control**
Est (95 %CI)	54.28 (41.10 - 67.49)	50.16 (36.98- 63.34)	49.31 (41.50 - 57.12)
$\sigma\varepsilon^2$	22.14		
Random effects			
σt^2	22.25		
σs^2	100.03		
p-value	<.000		

Table 3. Multiple comparisons of means with Tukey Contrasts (water level plots).

	Estimate Std.	Error	z value	Pr(>|z|)
G - C	4.9709	2.5067	1.983	0.116
M - C	0.8514	2.5067	0.34	0.938
M - G	-4.1195	2.5067	-1.643	0.227

Table 4. Results of the Nested ANOVA (density of culms).

	Value	Std.Error	DF	t-value	p-value
(Intercept)	5.5416	0.5226	162	10.6039	0
treatment G	-3.9449	0.5896	14	-6.6911	0
treatment M	-0.2095	0.5896	14	-0.3553	0.728

Table 5. Summary of fixed and random effects for the linear mixed model of density and treatment.

fixed effectes	Density		
	Ground	**Meter**	**Control**
Est (95 %CI)	1.6 (-0.7 - 3.89)	5.34 (3.03 - 7.63)	5.54 (4.51 - 6.57)
$\sigma\varepsilon^2$	5.03		
Random effects			
σt^2	0.74		
σs^2	0.79		
p-value	<.000		

Table 6. Multiple comparisons of means with Tukey Contrasts (density of culms).

	Estimate Std.	Error	z value	Pr(>|z|)
G - C	-3.9449	0.5896	-6.691	**<1e-05 *****
M - C	-0.2095	0.5896	-0.355	0.933
M - G	3.7354	0.5896	6.336	**<1e-05 *****

Table 7. Results of the linear mixed model (Maximum diameter).

	Value	Std.Error	DF	t-value	p-value
(Intercept)	10.0959	0.7654	162	13.1896	0
treatment G	-3.5413	1.0825	14	-3.2714	0.006
treatment M	1.1444	1.0825	14	1.0572	0.308

Table 8. Summary of fixed and random effects for the linear mixed model of maximum diameter and treatment.

fixed effectes	Density		
	Ground	**Meter**	**Control**
Est (95 %CI)	6.56 (2.72 - 10.38)	11.24 (7.04 - 14.8)	10.1 (8.58 - 11.61)
$\sigma\varepsilon^2$	8.46		
Random effects			
σt^2	3.59		
σs^2	0.0000		
p-value	< 0.000		

Table 9. Multiple comparisons of means with Tukey Contrasts (Maximum diameter).

	Estimate Std.	Error	z value	Pr(>|z|)
G - C	-3.541	1.083	-3.271	**0.003 ****
M - C	1.144	1.083	1.057	0.541
M - G	4.686	1.083	4.329	**< 1e-04 *****

Table 10. Results of the linear mixed model (Maximum height).

	Value	Std.Error	DF	t-value	p-value
(Intercept)	4.2145	0.2987	162	14.1107	0
treatment G	-1.8066	0.4224	14	-4.2772	0.001
treatment M	0.1229	0.4224	14	0.2909	0.775

Table 11. Summary of fixed and random effects for the linear mixed model of maximum height and treatment.

<table>
<tr><th rowspan="2">fixed effectes</th><th colspan="3">Density</th></tr>
<tr><th>Ground</th><th>Meter</th><th>Control</th></tr>
<tr><td>Est (95 %CI)</td><td>2.4 (2.13 - 2.67)</td><td>4.33 (4.06 - 4.6)</td><td>4.2 (3.93 - 4.47)</td></tr>
<tr><td>$σε^2$</td><td colspan="3">1.19</td></tr>
<tr><td colspan="4">Random effects</td></tr>
<tr><td>$σt^2$</td><td colspan="3">0.56</td></tr>
<tr><td>$σs^2$</td><td colspan="3">0.0000</td></tr>
<tr><td>p-value</td><td colspan="3">< 0.000</td></tr>
</table>

Table 12. Multiple comparisons of means with Tukey Contrasts (Maximum height).

	Estimate Std.	Error	z value	Pr(>|z|)
G - C	-1.8066	0.4224	-4.277	**<1e-04 *****
M - C	0.1229	0.4224	0.291	0.954
M - G	1.9295	0.4224	4.568	**<1e-04 *****

Table 13. Results of the linear mixed model (Random diameter).

	Value	Std.Error	DF	t-value	p-value
(Intercept)	8.2880	0.7765	162	10.6742	0
treatment G	-2.2984	1.0948	14	-2.0993	0.054
treatment M	1.2079	1.0948	14	1.1032	0.289

Table 14. Summary of fixed and random effects for the linear mixed model of random diameter and treatment.

fixed effectes	Density		
	Ground	Meter	Control
Est (95 %CI)	5.99 (2.21 - 9.86)	9.48 (5.61 - 13.38)	8.28 (6.75 - 9.82)
$\sigma\varepsilon^2$	8.83		
Random effects			
σt^2	3.647		
σs^2	0.0282		
p-value	<.000		

Table 15. Multiple comparisons of means with Tukey Contrasts (Random diameter).

	Estimate Std.	Error	z value	Pr(>|z|)
G - C	-2.298	1.095	-2.099	0.09
M - C	1.208	1.095	1.103	0.512
M - G	3.506	1.095	3.203	**0.0040 ** **

Table 16. Results of the linear mixed model (Random height).

	Value	Std.Error	DF	t-value	p-value
(Intercept)	3.5101	0.3012	162	11.6519	0
treatment G	-1.3936	0.4260	14	-3.2711	0.006
treatment M	0.2007	0.4260	14	0.4711	0.645

Table 17. Summary of fixed and random effects for the linear mixed model of random height and treatment.

Fixed effects	Density		
	Ground	Meter	Control
Est (95 %CI)	2.12 (0.62 - 3.63)	3.71 (2.21 - 2.99)	3.51 (2.91 - 4.10)
$\sigma\varepsilon^2$	1.334		
Random effects			
σt^2	0.5527		
σs^2	0		
p-value	<.000		

Table 18. Multiple comparisons of means with Tukey Contrasts (Random height).

	Estimate Std.	Error	z value	Pr(>|z|)
G - C	-1.3936	0.426	-3.271	**0.0030 ****
M - C	0.2007	0.426	0.471	0.885
M - G	1.5943	0.426	3.742	**< 0.001 *****