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Post-Fire Restoration and Deadwood Management: Microsite Dynamics and Their Impact on Natural Regeneration †

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† This manuscript is part of a Ph.D. Thesis by the third author Niccolò Marchi, available online at https://www.research.unipd.it/handle/11577/3426837.

Abstract: After large and severe wildfires, the establishment of tree regeneration, particularly for species without specific fire-adaptive traits, can be challenging. Within harsh environments, the presence of favorable microsites, as those provided by deadwood, enhancing microclimatic conditions, is crucial to the re-establishment of forest cover and thus to foster recovery dynamics. Active restoration strategies can have an impact on these dynamics, altering or hindering them. The main hypothesis of this study is that manipulating deadwood in terms of quantity and spatial arrangement can result in differences in natural regeneration density and composition. Post-disturbance regeneration dynamics and the role played by deadwood over time in the creation of safe sites for seedling establishment were investigated in an area affected by a high-severity wildfire that underwent different post-fire restoration treatments along a gradient of increasing deadwood manipulation, spanning from salvage logging to non-intervention. Two inventories were performed 5 and 11 years after the fire. Ground cover proportion was significantly different among treatments, with lower values of lying deadwood in salvaged sites. A higher probability of regeneration establishment close to deadwood was found in both surveys, confirming the facilitating role of deadwood on post-fire forest regeneration. Microsite dynamics resulting from deadwood facilitation were highlighted, with establishment probability and anisotropic relationships between deadwood elements and seedlings changing over time, as recovery processes slowly improved environmental conditions. In dry mountain areas affected by stand-replacing wildfires, by removing deadwood, salvage logging reduces the number of safe sites for regeneration, further impairing the ecosystem recovery. Passive management should be the ecologically preferred management strategy in these conditions, although intermediate interventions (e.g., felling without delimbing, leaving deadwood on the ground) could be effective alternatives, accelerating snag fall dynamics and immediately increasing favorable microsite availability.

Keywords: salvage logging; deadwood; regeneration; post-fire forest management; forest restoration

Citation: Lingua, E.; Marques, G.; Marchi, N.; Garbarino, M.; Marangon, D.; Taccaliti, F.; Marzano, R. Post-Fire Restoration and Deadwood Management: Microsite Dynamics and Their Impact on Natural Regeneration. *Forests* **2023**, *14*, 1820. https://doi.org/10.3390/f14091820

Academic Editors: Ioannis Spanos, Stefanos Tsiaras and Theano Samara

Received: 1 August 2023
Revised: 2 September 2023
Accepted: 4 September 2023
Published: 6 September 2023

1. Introduction

Different ecological factors, the impact of which is strictly related to species life-history traits [1,2], can affect regeneration with varying relative effects according to the regeneration stages [3]. Seed availability and dispersal, seed and seedling predation, inter and intraspecific competition and abiotic conditions are considered among the most crucial factors playing a role in regeneration dynamics [2,4,5]. Indeed, mechanisms operating at smaller scales can exert a relevant influence in limiting seedling density and performance [6,7]. In this context, conditions resulting from large and high-severity wildfires can be very harsh for vegetation recovery, causing restricted regeneration, particularly for species without specific fire-adaptive traits [8–10]. Fire regime alterations and direct climate change effects are producing noticeable consequences and endangering tree regeneration in dry conifer forests of the Alps, where *Pinus sylvestris* L. is often the dominant species [11–14]. Similarly to other non-serotinous pine species lacking direct post-fire regeneration mechanisms, this obligate seeder has been defined as a fire tolerator, able to survive lower intensity surface fires but highly sensitive to crown fires [15,16]. Despite being a pioneer species, *P. sylvestris* younger seedlings are highly dependent on a sufficient water supply and require a stable soil moisture regime [17]. Successful recruitment is thus mainly related to sufficient seed availability and favorable abiotic conditions. Post-disturbance seed availability for obligate seeder species can be guaranteed by the presence of a buried seed bank able to survive the disturbance or by seed rain from green islands, survived trees and intact surrounding forest edges [18,19]. To establish, the seed should be viable, avoid post-dispersal predation, and land on a suitable microsite. This latter should provide appropriate seedbed and microclimate requirements for seed germination, and good environmental conditions and protection to ensure seedling survival [20]. Several external factors (e.g., solar radiation, wind exposure, vegetation cover) can influence microsite characteristics, possibly creating safe sites for seedlings both directly and indirectly. Indeed, facilitation between plant species is one of the most relevant processes acting in stressful environments [21], thus playing a fundamental role in post-disturbance vegetation recovery. Facilitative mechanisms between different plant species can be essential in degraded or harsh sites, where a net positive effect over competition mechanisms has often been found [22]. However, facilitative mechanisms can change through the ontogenetic development of both nurse species and beneficiaries [23], sometimes shifting to competition when environmental conditions change [24,25]. Non-living woody elements can instead provide microsite amelioration without competing for resources (e.g., water, nutrients) with seedlings [7,26,27], and finally constitute a suitable substrate after decay processes have occurred [28]. This positive effect offered by deadwood elements can be vital in the first stages of the forest recovery pathway, with its importance gradually decreasing as the saplings develop a wider and deeper root system or the crowns escape browsing height [20,27].

After disturbances, deadwood elements are often the most represented [29] among biological legacies (sensu Franklin et al. [30]), and they greatly contribute to shaping microsite availability and type. Any post-disturbance intervention removing, altering, or displacing deadwood elements may therefore greatly modify the availability of microsites for seedling establishment. Salvage logging, consisting of removing dead, dying or damaged trees after disturbances [31], is a widespread practice after large and severe disturbance events. It is adopted for several reasons, mainly related to recovering the economic value, safety issues, or for reducing the risk of subsequent disturbances (i.e., fuel reduction, limiting insect outbreaks, . . .) [32]. However, this practice has been demonstrated to negatively affect ecosystem processes in different ways, with consequences potentially lasting for several years, also resulting from the interactions between the natural and anthropogenic disturbance, which has still not been thoroughly investigated [7,33]. It can also lead to a simplification of the post-disturbance habitat structure, producing negative impacts on species diversity, especially for species related to deadwood presence [34]. Moreover, natural regeneration through seed dispersal can be negatively affected in relation to the extent of the salvage logging operations, as this can increase the distance from seed sources [35].

Although the role of deadwood in enhancing tree regeneration establishment in post-disturbance environments was investigated in a few studies [7,20,36,37], information is still lacking on how deadwood management can alter microsite characteristics over time and how this can affect different regeneration stages. Assessing how the role of deadwood in modifying microsite abiotic factors regulating seedling performance may change during the years can result in stage-specific management strategies, thus improving post-fire forest recovery.

Our main hypothesis is that manipulating deadwood in terms of quantity and spatial arrangement after a large and severe fire disturbance can result in differences in natural regeneration density and composition. The main goal of this research was thus to investigate post-disturbance regeneration dynamics in an area affected by a high-severity wildfire that underwent different post-fire restoration treatments along a gradient of increasing deadwood manipulation. Two inventories of regeneration density and characteristics along an 11 years chronosequence were used to assess changes in the role provided by deadwood and derive management implications.

2. Methods

2.1. Study Site

The study site is located in the municipality of Verrayes (Aosta Valley—NW Italy) in proximity to the Bourra pasture (45°46′21″ N, 7°29′55″ E). In March 2005, a stand-replacing wildfire severely affected 160 ha of a Scots pine (*Pinus sylvestris* L.) forest (total burned area 257 ha). The burnt area ranges between 1650 m and 1800 m a.s.l. with a prevalent southern aspect and an average slope inclination of 25°. The bedrock is ophiolite and schist and the soils are entisols (Soil Taxonomy USDA). The mean annual temperature is 5.6 °C and mean annual precipitation is approximately 750 mm (less than 250 mm for JJA) with the driest month being February, coinciding with the main peak of the fire season. The tree vegetation prior to the wildfire consisted almost entirely of dense even-aged secondary *P. sylvestris* stands, with a sporadic presence of *Larix decidua* Miller, *Picea abies* (L.) Karst, *Quercus pubescens* Will., *Populus tremula* L., *Betula pendula* Roth and *Salix caprea* L.

A post-fire salvage logging project was approved in December 2005 and logging operations started during autumn 2007 [7]. Within the salvaged area, 5 ha of the burned forest were left untouched until 2009, when a sub-section of them (2 ha) was subjected to a cut and release intervention adopting two different procedures. In one area (1 ha), the standing dead trees were cut close to the ground and released following random felling directions (Random Directions—RD); in the other area (1 ha, Fishbone—FB) the snags were cut at 1 m height, delimbed and the resulting logs were oriented on the ground at about 45° to the maximum slope line according to a fishbone scheme. The remaining 3 ha of the experimental area were passively managed with no human intervention (Passive Management—PM). In the surrounding area, the conventional salvage logging operations adopted in the Region took place [38], consisting of felling all trees close to the ground, removing all trunks and large branches, and piling the slash. In order to reduce uncontrolled effects, in this study we considered only a 5 ha salvaged area (Salvage Logging—SL) adjacent to the other treatments with the aim of having a total study area of 10 ha, characterized by similar pre-fire conditions and affected by a similar fire behavior, resulting in the same fire severity.

2.2. Field Sampling

To assess regeneration dynamics over time, we combined data collected in 2010 by Marzano et al. [7] with those collected in a second field campaign conducted in summer 2016. Both field surveys adopted a two-scale approach, site and microsite, with the second one specifically addressing the facilitative role of deadwood.

At site-scale, the 60 circular sample plots (6 m radius; approximately 113 m^2) established in Marzano et al. [7] were remeasured. Twenty plots were located within the salvaged area (SL), 20 in the passive management (PM), and 10 each in both RD and FB (Figure 1). In these plots, we recorded regeneration characteristics (species, seed or sprout origin, height), the number of Coarse Woody Debris (CWD) elements (diameter larger than 10 cm and length longer than 1 m), and ground cover. This latter was visually estimated to the nearest 5% for the following classes: litter, lying deadwood (fine and coarse woody debris, FWD and CWD, respectively), bare soil, grasses, forbs, shrubs, and gravel. In accordance with Marzano et al. [7], we considered all tree seedlings as regeneration without applying a minimum height or age threshold.

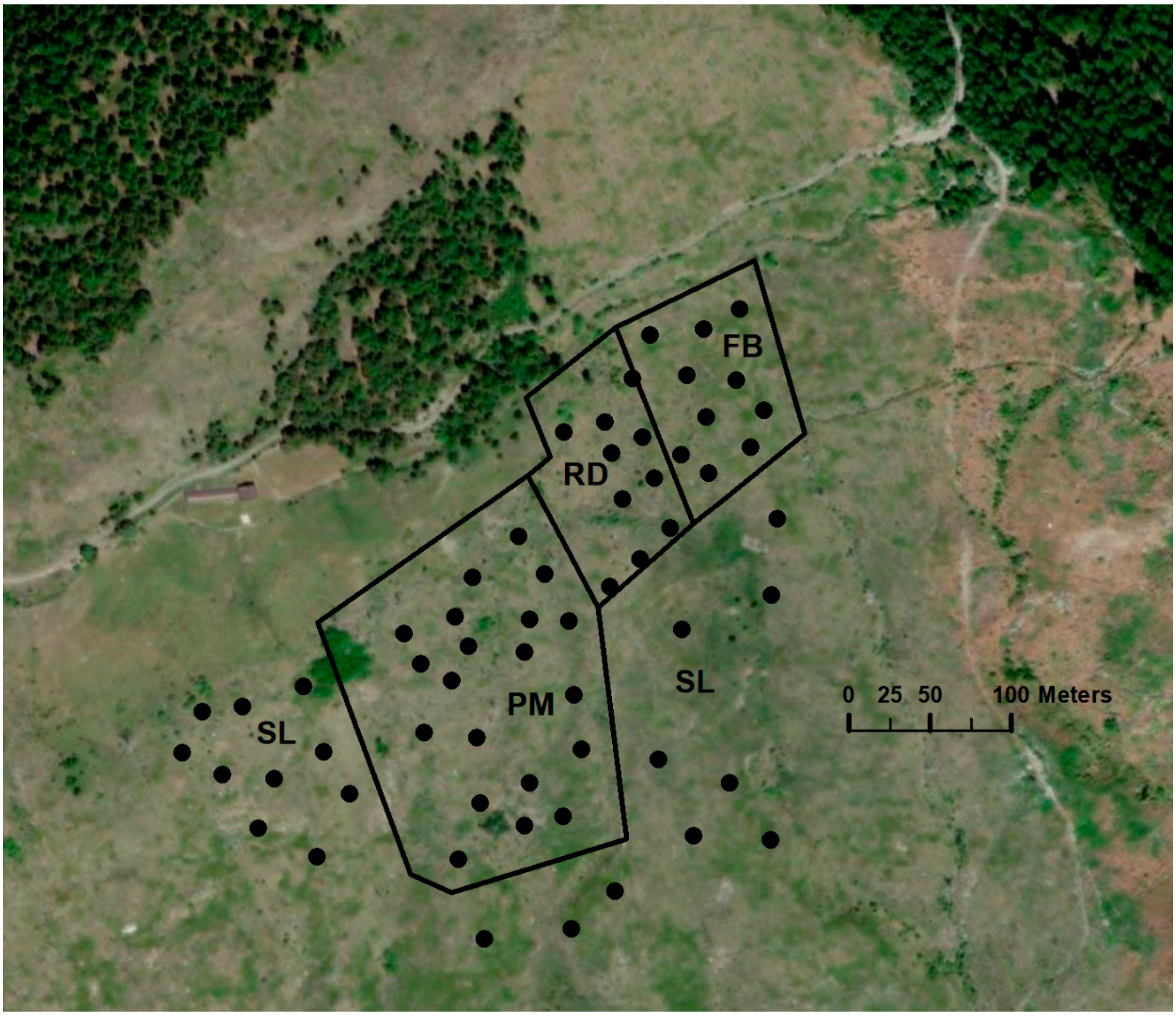

Figure 1. Location of sampling plots (•) inside the different treatments (SL = Salvage Logging; FB = Fishbone; RD = Random Directions; PM = Passive Management).

For each seed-origin seedling found in the plots, a microsite-scale survey was applied (matched case–control design). A 20 cm × 20 cm quadratic plot was centered on each seedling and then matched with a microsite of the same size but without seedlings (control) located 1 m to the east of it on the same contour line for comparison.

Besides collecting the characteristics of the seedling (species, height) in the case plot, the microsites were characterized according to Marzano et al. [7] in both case and control plots, defined as follows: (1) seedbed type (litter, rotten wood, bare soil, grasses, forbs, shrubs and gravel); (2) presence and relative position (distance, azimuth) of standing or lying deadwood elements within 1 m from the microsite center; (3) presence and relative position of rock elements (height > 10 cm).

In order to allow further monitoring, each microsite position was recorded with a submetric GPS device.

2.3. Data Analysis

For each plot, we derived topographic variables from LiDAR data acquired in June 2011 [39]. Since the use of LiDAR technologies was to be an efficient approach to obtain reliable data over large areas, also for deriving parameters on deadwood [40], we computed indices both directly from point clouds (10 points m^{-2}) and from a LiDAR-derived DTM (1 m resolution) to characterize the presence of deadwood lying on the ground. Three different indices describing the roughness of a surface were computed for each treatment and compared with the number of CWD elements recorded in the field inside the plots. From the surface raster, we computed the Terrain Ruggedness Index (TRI), which is the mean of the absolute differences between the value of a cell (elevation, m) and the value of its 8 surrounding cells, and the Roughness index, which is the difference between the maximum and minimum values of a cell and of its 8 surrounding cells [41]. The indices were computed in R using the rgdal package (gdaldem). We further derived directly from the point clouds the Rumple index of roughness (R_3), which is the roughness of a surface computed as the ratio between its area and its projected area on the ground. If the input is a gridded object (lasmetric or raster), the function computes the surfaces using Jenness's algorithm [42]. If the input is a point cloud, the function uses a Delaunay triangulation of the points and computes the area of each triangle. Before running the algorithms, all points higher than 3 m from the ground were removed in order to detect only the deadwood lying on the ground.

In order to describe local environmental characteristics, we computed the Heat Load Index (HLI) from the DTM, which combines both slope and aspect [43], averaging the value inside each plot.

To understand if regeneration establishment and survivorship were affected by different parameters in the time-period analyzed, we computed total regeneration density (n ha^{-1}) and seed-origin regeneration density in 2010 and 2016 (Reg2010, Reg2016, Seed2010, Seed2016) and the differences in regeneration density between the two surveys (ΔReg; ΔSeed).

In order to define the influence of different environmental factors on the regeneration, we ran generalized linear models (GLM) considering treatment (T), Heat Load Index (HLI), deadwood presence (R_3) and distance from the seed source (DIST) as fixed parameters, along with their possible interactions. The GLM analyses were run using STATGRAPHICS centurion XVII (Statpoint Inc., Warrenton, VA, USA, 2014). Model simplification was accomplished by computing the Akaike information criterion (AIC). Starting from the full model, the minimal adequate GLM was obtained by sequentially removing any non-significant model term until no further reduction in AIC was observed.

In order to relate the occurrence (case) and absence (control) of seedlings to microsite variables, we applied a Conditional logistic regression analysis for matched-pairs data. We tested the role of deadwood or rock elements in close proximity to microsites considering both simply presence/absence and in the 4 main cardinal directions. We ran the analyses for both the 2010 and 2016 seedling inventory. The conditional logistic regression was performed using the SPSS 17 statistical package (SPSS Inc., Chicago, IL, USA, 2008).

3. Results

Total tree regeneration density slightly decreased between the two inventories, but the number of seed-origin individuals increased instead. The mean regeneration density in 2010 was 606 individuals ha^{-1} (±198), of which 179 seedlings ha^{-1} (±30.3) considered only seed-origin individuals. In 2016, the mean regeneration density was 590 individuals ha^{-1} (±182.6), of which 244 seedlings ha^{-1} (±50.8) originated from seeds. Agamic regeneration was the dominant strategy in both surveys, but the percentage of sprouting was significantly different (χ^2 test; $p < 0.001$) with higher values (70.4%) in 2010, while in 2016, sprouts accounted for 58.6% of the whole regeneration.

Ground cover proportion was significantly different among treatments in both years (χ^2 test; $p < 0.001$). The ground cover classes that showed significant differences between treatments in 2010 were only lying deadwood and bare soil (Kruskal–Wallis test, $p < 0.01$), with significantly lower values in PM and RD, respectively. Six years after (2016), the ground cover classes, which showed significant differences between treatments, were shrubs, lying deadwood and grasses (Kruskal–Wallis test, $p < 0.01$). Shrubs (the great majority of which were *Arctostaphylos uva-ursi* (L.) Spreng.) became more abundant in FB and SL compared to RD and PM (Figure 2a; Dunn's-post hoc test, $p < 0.05$), while grass cover was higher in RD (Figure 2c) and significantly different from SL, PM and FB

(Dunn's-post hoc test, $p < 0.05$). In 2016, the lying deadwood cover was significantly lower in SL (Figure 2b; $p < 0.05$). Bare soil cover was relatively low and homogeneous in all the treatments in 2010, reducing its share even more in the following years (Figure 2d).

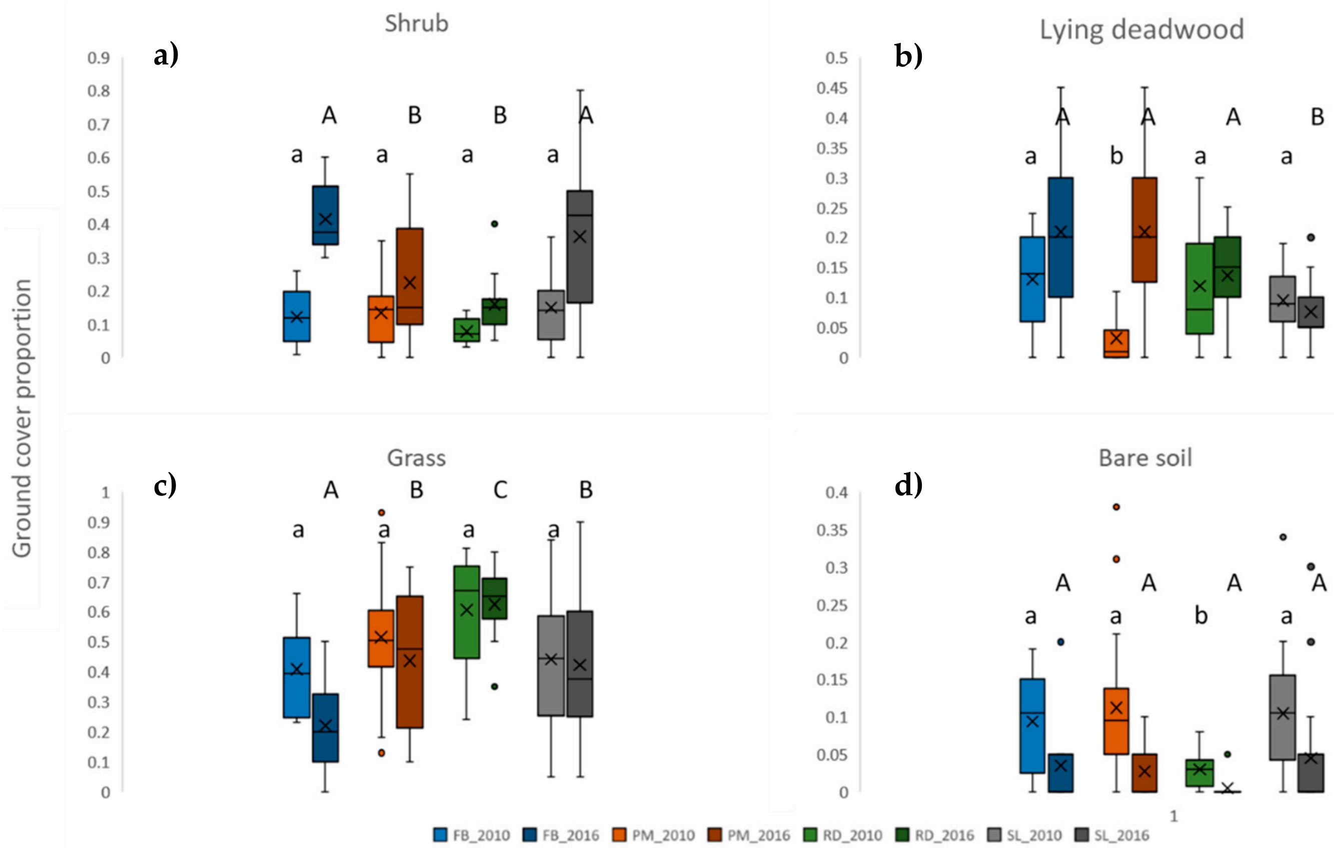

Figure 2. Ground cover proportion for the more represented categories (**a**—Shrub; **b**—Lying deadwood; **c**—grass; **d**—bare soil cover) in the different treatments (SL = Salvage Logging; FB = Fishbone; RD = Random Directions; PM = Passive Management) and years (2010, 2016). The X represents the mean value. Different letters indicate significant differences between treatments among years (lowercase 2010, uppercase 2016; Dunn's post hoc test; $p < 0.01$).

Among the LiDAR derived indices, the Rumple index (R_3) was the only one showing significant differences (Kruskal–Wallis test; $p < 0.01$) among treatments and was significantly and positively correlated to the number of CWD elements (Spearman rho; $p > 0.01$). The PM and RD treatments showed similar values (Figure 3) with significantly higher deadwood heterogeneity compared to the FB and SL ones (Dunn's post hoc test; $p < 0.01$).

GLMs provided significant results only for Seed2010 and Seed2016. When sprouters were included (Reg2010, Reg2016) the models obtained were not significant ($p > 0.05$). No significant models ($p > 0.05$) were generated for the differences in regeneration density (ΔReg; ΔSeed). The models for seed-origin regeneration showed that different factors affected seedlings in a complex pattern in both years (Table 1). The final model for Seed2010 (F = 2.22; $p = 0.0181$) showed that only treatment (T) had a significant impact alone (df 3, 31; $p < 0.01$), along with its interaction with R_3 (df 1, 31; $p < 0.01$) and HLI (df 3, 31; $p < 0.05$). In the final model, Seed2016 (F = 2.50; $p = 0.0072$) both T (df 3, 31; $p < 0.05$) and R_3 (df 1, 31; $p < 0.05$) were significant singularly. In both cases (Seed2010 and Seed2016), the HLI and DIST were only important when combined with deadwood presence (R_3) and with treatments (Table 1).

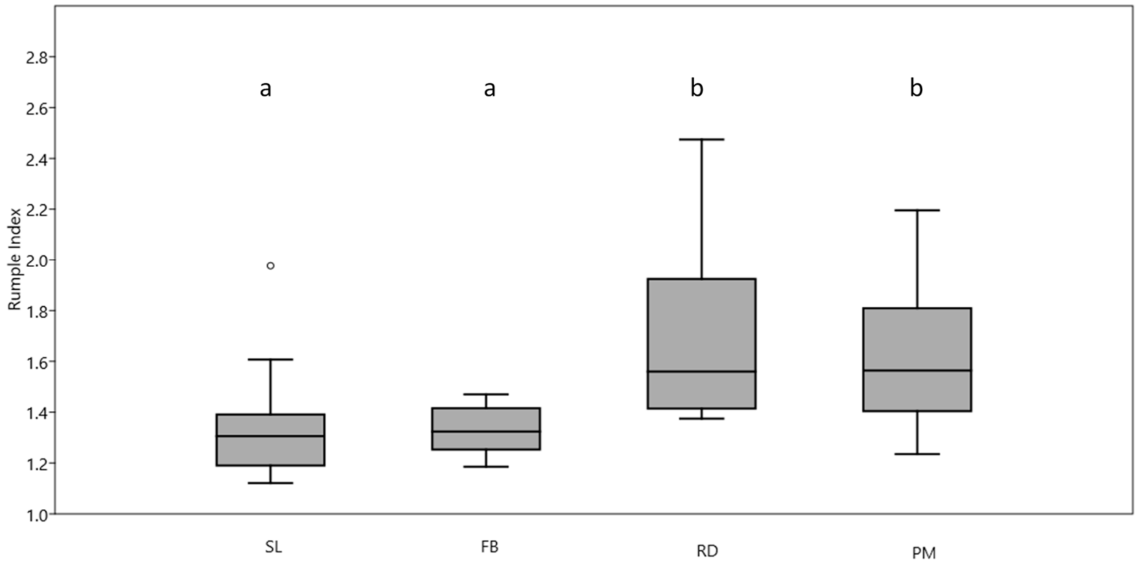

Figure 3. Deadwood availability and heterogeneity as defined by the Rumple Index of roughness for the first 3 m above ground in the different treatments. (SL = Salvage Logging; FB = Fishbone; RD = Random Directions; PM = Passive Management). Different letters indicate significant differences (Dunn's post hoc test; $p < 0.01$).

Table 1. Summary of the GLM for seed-origin regeneration in 2010 (Seed2010) and 2016 (Seed2016). T = treatment; R_3 = Rumple Index; HLI = Heat Load Index; DIST = distance from the closest living mature trees. Significant results are indicated in bold ($p < 0.05$).

Survey	Predictor Variable	df	F	*p*
2010	T	3, 31	4.75	**0.008**
	R_3	1, 31	1.45	0.239
	HLI	1, 31	2.22	0.147
	DIST	1, 31	2.94	0.097
	T × R_3	1, 31	4.81	**0.008**
	T × HLI	3, 31	4.15	**0.015**
	T × DIST	3, 31	3.34	0.335
	R_3 × HLI	1, 31	3.58	0.069
	R_3 × DIST	1, 31	5.55	**0.026**
	DIST × HLI	1, 31	3.81	0.061
	T × R_3 × HLI	3, 31	4.23	**0.014**
	T × HLI × DIST	3, 31	3.77	**0.022**
	T × R_3 × DIST	3, 31	3.74	**0.022**
	R_3 × HLI × DIST	1, 31	5.40	**0.028**
	T × R_3 × HLI × DIST	3, 31	4.12	**0.015**
2016	T	3, 31	3.86	**0.019**
	R_3	1, 31	5.25	**0.029**
	T × R_3	3, 31	4.19	**0.013**
	T × HLI	3, 31	3.91	**0.018**
	T × DIST	3, 31	3.81	**0.019**
	R_3 × HLI	1, 31	7.63	**0.009**
	R_3 × DIST	1, 31	6.39	**0.017**
	T × R_3 × HLI	3, 31	4.10	**0.015**
	T × HLI × DIST	3, 31	3.90	**0.018**
	T × R_3 × DIST	3, 31	4.15	**0.014**
	R_3 × HLI × DIST	1, 31	5.09	**0.031**
	T × R_3 × HLI × DIST	3, 31	4.15	**0.014**

The conditional logistic regression analysis for matched-pairs on the presence–absence of seedlings showed a significant role of deadwood elements (Table 2). The proximity to a deadwood element increased the probability of regeneration presence in both surveys by more than threefold (odds ratio 2010 = 3.7, $p < 0.01$; odds ratio 2016 = 3.6, $p < 0.01$). However, the relative importance of cardinal directions changed among the surveys, reducing the importance of deadwood elements located south in 2016 ($p > 0.05$).

Table 2. Results of the conditional logistic regression analysis. Rocks value for cardinal directions are not presented since no significant values were found. Significant *p*-values are reported in bold ($p < 0.05$).

	Explanatory Variable Proximity to:	Beta	S.E.	*p*-Value	Odds Ratio	95% Confidence Interval for Odds Ratio Lower	Upper
Regeneration 2010 n = 268	Rocks	0.707	0.548	0.197	2.027	0.692	5.934
	Deadwood	1.320	0.274	**<0.001**	3.743	2.188	6.404
	Deadwood_N	0.865	0.318	**0.007**	2.376	1.2727	4.435
	Deadwood_E	1.394	0.354	**<0.001**	4.029	2.0147	8.058
	Deadwood_S	1.311	0.335	**<0.001**	3.709	1.9244	7.148
	Deadwood_W	0.037	0.402	0.926	1.038	0.4722	2.282
Regeneration 2016 n = 147	Rocks	0.511	0.516	0.323	1.667	0.606	4.586
	Deadwood	1.299	0.651	**0.046**	3.667	1.023	13.143
	Deadwood_N	1.485	0.650	**0.022**	4.417	1.236	15.785
	Deadwood_E	1.343	0.657	**0.041**	3.829	1.057	13.875
	Deadwood_S	0.803	0.633	0.204	2.233	0.646	7.716
	Deadwood_W	0.861	0.624	0.168	2.366	0.696	8.045

4. Discussion

More than a decade after the wildfire, post-fire forest recovery is still slow, and concern is growing about the availability of adequate conifer regeneration. Indeed, post-fire deadwood management strategies affected seedling recruitment, altering both site conditions and affecting already established regeneration. Delayed intervention can rewind the successional processes and eventually deviate the recovery pathways [44], even though delaying logging by 2–4 years after the occurrence of natural disturbances was found to partially mitigate negative ecological impacts [45].

Post-fire tree regeneration dynamics showed a different pattern in time as a consequence of the different strategies applied. Concerning species composition, five years after the natural disturbance, sprouter species were dominant with Eurasian aspen (*Populus tremula* L.) being the most represented species. Its root sucker production is induced by disturbances that remove the tree apical dominance [1,46], and it can be further enhanced by subsequent events [47], including silvicultural interventions. However, the dense patches of Eurasian aspen established right after the wildfire tended to self-thin quite early, reducing the dominance of sprout-origin individuals in the regeneration over time. Obligate seeders, particularly when the species lack fire-adaptive traits and direct post-fire regeneration mechanisms, as in the case of non-serotinuos pinewoods, are more sensitive to the post-fire environment. In these cases, the presence of resprouting species can speed up forest recovery, but also deeply influence its trajectories [48,49]. In order to understand and mitigate ecosystem transformations, any factor potentially affecting post-fire forest recovery in conifer-dominated stands should be examined. Future success of conifer regeneration is actually being questioned by several studies, with conifer recruitment being impaired by climate change and altered disturbance regimes (e.g., increasing size, severity and frequency of wildfires), not only for fire-sensitive or independent species, but also for fire-adapted ones [50–52]. In this context, the implications of post-disturbance management should be also carefully evaluated.

Within our study area, despite its relatively low density, seed-origin regeneration increased its relative importance during the 11 years since the wildfire, possibly taking advantage of microsite amelioration over time.

We observed ground cover changes since the wildfire, but differences were found among management strategies. These differences in ground cover proportion through the years are the result of the recovery process as mediated by management. As expected, a general increase in shrubs cover was found and bare soil cover decreased within all treatments. Significant differences in lying deadwood cover emerged over time, with consequent implications for safe site availability.

The increase in shrub cover revealed a general enhancement of environmental conditions, but it can also eventually play a mulching role for the recruitment of new seedlings. Indeed, bearberry (*Arctostaphylos uva-ursi* L.) is known to recover rapidly after disturbances by vegetative resprouting [53], and to have chemical allelopathy potentially limiting seedling establishment [54], similarly to other members of the *Ericaceae* family. However, in the harsh conditions created after a fire, especially at higher elevation, resprouting bearberry shrubs can act as an inoculum of ectomycorrhizal fungi available for the seedlings since they share similar communities [55]. The lack of an ectomycorrhizal community can hinder seedling growth and competitiveness with annual grass species [56].

When salvage logging is carried out in dry sites, as in the Mediterranean mountains, seedling establishment is greatly affected by an increased water stress [37,57]. Logging operations produce a general soil degradation, directly by compaction based on the type of logging system adopted, and indirectly by vegetation removal [58]. In southern exposed slopes suffering from reduced precipitation, the herbaceous layer and especially the shrub layer can ameliorate moisture availability [59]. After a high-severity fire, in the early post-fire environment, bare soil is widespread, organic matter is consumed and superficial erosion is potentially extensive until vegetation recovery [60]. This period, known as the "window of disturbance" [61], can last for years in dry sites, even a decade in the Mediterranean area [62]. Vegetation recovery can afterwards enhance soil conditions and mitigate the harsh environment. In the Bourra site, five years after the fire, fine and coarse woody debris were lower in the passive management area. Fine woody debris on the ground was consumed by combustion and branches were still held on the burned crowns of standing dead trees. Indeed, when falling dynamics started, involving entire snags or branches, new FWD and CWD inputs to the ground were provided. Different dynamics resulted instead from salvage logging, where lying deadwood was scarcely present and mostly consisting of smaller material being generated by logging operations, going through more rapid decay processes on the ground a few years later. Soon after the intervention, deadwood presence on the ground was not yet effective in promoting regeneration, but its effects started to be evident at a later stage. The first survey was actually performed one year after the treatments implementation, and the differences in regeneration occurrence were probably more related to the presence (FB, RD, SL) or absence of direct impacts on already recruited seedlings (PM). Those differences that were highlighted six years after their implementation were instead the result of deadwood generating a complex mosaic of microsites characterized by improved microclimatic conditions (in terms for instance of lower insolation and higher soil moisture content [37].

The collapse of snags over time can create heterogeneous habitats, resulting from lying or standing dead trees or woody debris, with different size, type and decay status [31,63]. Both standing and lying deadwood are biological legacies that showed a positive influence on conifer seedling establishment in different post-fire environments [64,65]. Microsite enhancement is largely produced by deadwood close to the ground, lying or suspended on it, with an increasing amount when standing dead trees are beginning to break and fall down.

In the short time we found deadwood to be particularly important in more xeric sites, enhancing seedling survival. The soil underneath or in proximity to downed logs stores a higher water content [37,66] and has a lower soil surface temperature [20] than in open sites. Eleven years after the fire, in a relatively milder environment, deadwood impact in promoting seedling establishment decreased, although always being a facilitative factor. Indeed, other studies recognized deadwood as less relevant in moist sites [28]. Changes in the anisotropic relationships over time between deadwood and seedlings were probably the result of the general improvement in environmental conditions.

The downed crowns with their branches can increase soil moisture and nutrient availability, but also act as perching site for birds, hamper browsing and reduce competition with ericaceous species [65,67]. Intermediate interventions (i.e., FB and RD) are artificially accelerating this process, potentially creating a greater availability of safe sites for seedling establishment in a shorter period, compared to natural fall dynamics.

Moreover, the higher surface heterogeneity (as measured through LiDAR derived roughness indices) increased the availability of seed traps for wind dispersed seeds and provided perching sites or hidden caches for birds and rodents. Post-dispersal predation can be a limiting factor [68], and deadwood manipulation can increase predation rates by creating safe sites for seed predators [26,69], but it can also provide safe sites for emerging seedlings against browsing [20,70]. Dead crowns or piles of branches on the ground were found to be particularly effective in creating favorable light and moisture conditions and in protecting seedlings from browsing [71].

The interaction between natural disturbances and salvage interventions can produce a cumulative effect that can limit ecosystem recovery, eventually defining new successional pathways [72,73]. By removing biological legacies, salvage logging operations can deeply halter habitat suitability, affecting several biotic groups [74].

Besides its controversial ecological impact, particularly in fire-affected areas, from the economic point of view, salvage logging was also the most expensive strategy in our study site [37], as found in other Mediterranean mountain areas where the income generated from salvaged timber was low [75]. Intermediate treatments instead reduced costs, especially if not involving delimbing or debarking operations. From the ecological point of view, they mimic natural dynamics, simply by accelerating the process of snags falling. Even if the gradual recruitment of lying deadwood can provide positive microclimatic conditions resulting from the co-occurrence of both standing and lying deadwood elements for a longer period [76], felling dead trees retaining all the branches can maintain a higher site heterogeneity, comparable to the one promoted by natural falling processes. Removing branches makes the logs lie closer to the ground with a total roughness similar to that of salvaged areas, as found in the FB strategy.

Facilitation mechanisms between species are dynamic relationships that can vary over the ontogenetic stages of trees [77]. In the same way, shelter elements can have a different influence on tree regeneration as seedlings grow [78]. The positive effects provided by deadwood can be crucial at the establishment stage (emergence) or can become more significant at later stages of seedling development [79]. The lack of seed-origin post-fire regeneration in obligate-seeder pine forests is actually a critical issue in many Mediterranean environments, particularly when experiencing a shift in fire regime [56,80] and with a dominance of non-serotinous species.

Distance from seed sources, i.e., surviving trees, green islands or the intact forest edges, emerged as a crucial driver in determining the speed of the recovery process [18,35]. In the Bourra site, distance alone was not significant in both surveys, becoming significant only in combination with other variables, probably due to the generally low number of seed origin seedlings and the small scale of our experiment. Reduced seed availability and frequent post-fire drought are considered as key constraints to conifer recruitment [52].

To ensure successful regeneration and a stable population structure, a sufficient number of emerged seedlings and a high survival rate are required [81,82]. Thus, after a disturbance, any intervention hindering the ability of seedlings to establish or decrease their fitness should be carefully considered. Indeed, an assisted natural regeneration approach increases opportunities for recovery by improving the number and spatial distribution of favorable microsites through deadwood management, which is highly advisable.

5. Conclusions

Several studies reported the importance of deadwood for implementing nurse-based restoration practices [7,27,83], but still the release of a large amount of deadwood is a practice barely applied. These shelter elements could contribute to buffer the harsher conditions provided by climate change, especially in a dry environment, guaranteeing safe sites for seedling recruitment and development by enhancing microclimatic conditions [20,76,84].

The justification of salvage logging interventions in order to avoid the occurrence of a secondary disturbance (e.g., insect outbreaks, fire, . . .) has recently been highly debated and several pieces of evidence show contrasting results [85,86]. However, post-fire management also has to deal with the human dimension [87], with affected forests being part of socio-ecological systems; societal and political requirements sometimes overcome scientific evidence. In mountain forests affected by severe wildfires, in the absence of prescriptions related to public safety, passive management should be pursued, totally or partially based on the amount of time deemed acceptable to recover forest functions (e.g., according to the window of protection gap [88]). If interventions are required, intermediate treatments should be preferred, restricting salvage logging operations only to specific areas. In steep terrains, besides enhancing microsite availability for tree regeneration establishment and survival, lying logs can increase soil roughness and constitute barriers, thus limiting surface erosion, avalanche formation and rockfall propagation [89,90].

After a major disturbance, a rapid recovery is advocated, often by promoting salvage logging interventions followed by afforestation (depending on budget availability). This is often the case with mountain forests in the Alps, particularly if they also exert a protective function. Intermediate interventions, such as partially releasing the amount of deadwood, should be a more-preferred option, especially in dry sites. In large disturbed areas and at landscape level, a mosaic approach should be adopted, alternating salvaged, unsalvaged and partially salvaged patches. However, each site should be analyzed separately and decisions taken considering all the dimensions (i.e., ecological, economic, and societal) that can greatly vary locally. Further studies are still needed on the combined effects of natural disturbances, different post-disturbance interventions and longer chronosequences to assess their impact.

Author Contributions: Conceptualization, E.L., R.M. and M.G.; methodology, E.L., N.M., R.M. and M.G.; software, E.L., G.M. and N.M.; formal analysis, E.L., G.M., N.M. and R.M.; investigation, E.L., G.M., N.M., D.M., F.T. and R.M.; data curation, G.M., N.M. and E.L.; writing—original draft preparation, E.L., R.M. and M.G.; writing—review and editing, E.L., R.M., M.G., G.M., N.M., D.M. and F.T.; supervision, E.L.; funding acquisition, E.L. All authors have read and agreed to the published version of the manuscript.

Funding: This research was funded by the Interreg Alpine Space project "MOSAIC" (ASP0100014), by PNRR PE3-RETURN Multi- Risk Science For Resilient Communities Under A Changing Climate—(RETURN) SPOKE 4—VS4 Environmental Degradation (CUP: C93C22005160002).

Data Availability Statement: The datasets used and/or analyzed during the study are available from the corresponding author upon request.

Acknowledgments: We are grateful to E. Marcolin, F. Meloni, A. Marigo, for field work, and G. Cesti, W. Chapellu, and the Administration of Regione Valle D'Aosta for providing logistic support.

Conflicts of Interest: The authors declare no conflict of interest. The funders had no role in the design of the study; in the collection, analyses, or interpretation of data; in the writing of the manuscript; or in the decision to publish the results.

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