COASTAL IFOA MONITORING PROGRAM

Review of forest recovery in the Coastal IFOA region of New South Wales following the 2019/2020 wildfires and preceding drought

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A report submitted to the New South Wales Natural Resources Commission

Acknowledgement of Country

With respect for Aboriginal cultural protocol and out of recognition that its campuses occupy their traditional lands, Western Sydney University acknowledges the Darug, Eora, Dharawal (also referred to as Tharawal) and Wiradjuri peoples and thanks them for their support of its work in their lands (Greater Western Sydney and beyond). Western Sydney University conducted field surveys on the lands of the Kurnai, Bidwell, Yuin, Ngunawal, Gundungurra, Wiradjuri, Darug, Darkinung, Dainggatti, Gumbainggir and Ngarabal peoples. Western Sydney University also acknowledges and pays respects to Elders past and present.

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CONTENTS

1 Background

1.1 Review context

The Coastal Integrated Forestry Operations Approval (Coastal IFOA) sets out the rules for native timber harvesting in New South Wales (NSW) coastal state forests and establishes the outcomes that must be achieved under the approval. The Coastal IFOA requires that the approval conditions are monitored to ensure they are effective in achieving the required objectives and outcome statements.[1]

The Environment Protection Authority (EPA) and Department of Primary Industries (DPI) jointly approved the Coastal IFOA Monitoring Program proposed by the NSW Forest Monitoring Steering Committee. The program sets out the broad framework to evaluate the effectiveness of priority conditions in meeting the Coastal IFOA objectives and outcomes. It centres on strategies to monitor and research forest health, biodiversity, water quality and aquatic habitat, and wood supply.

Major fires throughout the Coastal IFOA region during 2019/2020 affected around 40 percent of native forests on all tenures, many of which had been suffering from widespread drought in NSW prior to the fires. Above average rainfall in the years following the drought and fires have resulted in broadscale regreening of the forest. This review analyses field-based research and forest monitoring data to quantify the impacts of the drought, fire and subsequent high rainfall on forest survival and structural recovery.

1.2 Review summary

Extensive field surveys conducted across the Coastal IFOA region and adjacent areas following the Black Summer fires of 2019/2020 and prior drought revealed that forests growing on relatively fertile granite-derived soils generally experienced higher levels of tree mortality (completely dead) and topkill (completely dead or basally-resprouting only) than forests growing on lower-fertility sandstone soils. Levels of post-fire seedling recruitment were high across all surveyed substrates/forests. Compared to studies following previous fire seasons, overall levels of tree mortality, topkill and recruitment post-2019/2020 were similar, although the geographic extent of forest affected was far beyond that of any previous fire season, leading to higher losses of trees at a regional scale. High levels of seedling recruitment suggest that

[1] Coastal IFOA Conditions (Chapter 8) and Coastal IFOA Protocols (Protocol 38).

above-average rainfall in the years following the Black Summer fires may have enhanced recruitment, particularly on lower fertility soils where post-fire recruitment has historically been reported to be relatively low. High levels of mortality and topkill were generally restricted to areas burnt at high or extreme fire severity. Trees with pre-existing basal injuries were substantially more prone to experience mortality and topkill. Two independent datasets revealed congruent and atypical 'u'-shaped mortality and topkill response curves, where the smallest and largest trees were more likely to be killed or topkilled. A remote sensing assessment indicated strong alignment with field data analysis results, whereby delayed recovery signals were significantly more prevalent at sites with elevated rates of mortality and topkill, and in wet forests on granite substrate. Higher recovery index values were associated with dense *Acacia* regrowth, high or extreme fire severity, at drier sites, and in dry sclerophyll forests on sandstone substrate.

These findings suggest that the severity of the combined drought and fires may have pushed some forests beyond a hypothetical disturbance threshold that caused large, fire-resistant trees to die, particularly in wet sclerophyll forests, resulting in a demographic shift in some forests toward mid- and smaller-sized trees. For example, around 22-28% of large trees (e.g. >80 cm diameter at breast height, DBH) in wet sclerophyll forests were likely to be dead overall, and this increased to >50% mortality for trees that had pre-existing basal fire scar injuries (e.g. around half of all mature trees). Though analyses were limited to wet and dry sclerophyll forests growing on sandstone and granite substrates, the analysed forest types represented 44% of State Forest tenure within the Coastal IFOA region and some areas were extensively impacted by high severity fire, particularly in the south (e.g. 35-45%). Thus, it can be inferred from the results that forests across all tenures within the Coastal IFOA region, particularly in the south, have experienced substantial mortality and topkill and will be in a state of recovery for many decades due to structural changes primarily caused by losses of large trees. However, the magnitude of structural change is likely to vary with substrate/forest type and depend on the proportion of trees that have basal injuries (e.g. due to fire and potentially harvesting[2]). The key differences between the Black Summer fires and previous fire seasons are the widespread manifestation of 'u'-shaped tree mortality and topkill response curves and the much larger geographic scale of the affected area. This report contains extensive data summaries, analyses, and interpretation of the observed trends and is accompanied by a spatial

[2] Further research including monitoring is required to quantify the long-term effects of basal injuries caused by harvesting operations, for discussion see Bendall et al. (2023).

data product that may assist land managers in locating severely affected areas. A series of recommendations (see **3** Recommendations) are also provided to guide future survey methods.

1.3 Introduction

The capacity of forests to maintain multi-cohort stands has important implications for the sustainability of forest resources, the quality of ecosystem services and the survival of forest fauna (FCNSW, 2015; Gao et al., 2014; Turner et al., 2009; Wood et al., 2017). For example, if stochastic events such as coupled extreme drought and wildfire result in the mortality of many large trees, the population structure may become skewed toward smaller size-classes and affected areas may no longer store as much carbon or provide as many hollows for hollow-dependant fauna over the long term[3], which impacts the goals of land managers and the distribution of natural assets across landscapes. Maintenance of multi-cohort stands is dependent on both effective tree recruitment that compensates for attrition of older trees through disturbances, and on effective management, i.e. maintaining disturbance/harvesting regimes that promote desirable population structures. The rate at which forests recover from drought, fire, harvesting or their combinations is dependent on the severity of the disturbance, preceding disturbance history, post-disturbance environmental conditions, and species composition and life history traits (Heath et al., 2016; McCaw & Middleton, 2015; Oliver & Larson, 1996; Wilson et al., 2021).

Disturbances during the 21st century have had major impacts on forests globally (Allen et al., 2015; Millar & Stephenson, 2015) and within New South Wales forests specifically (Boer et al., 2020; Collins et al., 2021; Nolan et al., 2020a; Nolan et al., 2022). For example, Bradstock et al. (2021) found that almost 60% of State Forest tenure was burned within the Coastal IFOA region during the Black Summer fires, with around 24% burning at high or extreme severity. Broad patterns of high and extreme fire severity were likely related to the relationship between fire weather and topographic position, i.e. exposed ridges generally experienced higher rates of canopy damage than deep gullies (Bowman et al., 2021). The Black Summer fires resulted in major shifts in fire regime across State Forest tenure, with a

[3] Dead and damaged trees are likely to contain hollows (Gibbons et al., 2000) and so provision of hollows may not necessarily decrease in the short term following tree mortality or topkill, although this will depend on the extent of damage/rate of combustion of hollow bearing structures. Monitoring of dead tree hollow stocks under varying disturbance regimes would be required to ascertain the longevity of hollows in dead trees.

substantial reduction in the area considered to be within the tolerable fire threshold[4], from 22.2% to 7.6%, and a substantial increase in the area considered to be vulnerable to becoming too frequently burnt, from 22.2% to 50.4% (Bradstock et al., 2021). When viewed at the level of the vegetation formation (Keith, 2004), wet and dry sclerophyll forests (WSF, DSF) within State Forest tenure, where the majority of harvesting occurs, experienced two to three-fold increases respectively, in the area considered to be vulnerable to being too frequently burnt (Bradstock et al., 2021). A consequence of these major changes to the landscape mosaic of fire histories and widespread reset of time-since-fire is that forests at large are now within a recovery phase and in general will require substantial time to reach an ecological state that could be considered comparable to the pre-2019 state (Le Breton et al., 2022).

The extent of fire during 2019/2020 in different forest types within the Coastal IFOA region and the tolerable inter-fire intervals required for those forests to reach an ecological state capable of sustaining another fire without deleterious impacts to biodiversity (i.e. tolerable fire thresholds[4]), has already been addressed by Bradstock et al. (2021). The purpose of this current report is to document rates of tree mortality (completely dead), topkill[5] (i.e. completely dead or loss of the majority of live above-ground biomass and basally-resprouting only), and recruitment across forests in relation to the variable impacts (i.e. severity) of the Black Summer fires, while considering the preceding megadrought and subsequent extended period of above-average rainfall. This report also provides comparisons of field data against remote sensing signals, which will aid interpretation of statewide indicators of post-fire biomass recovery monitoring by remote sensing. With analysis of these patterns in hand, this report provides estimates of potential structural changes in forests as a function of forest type (e.g. wet or dry forest, different growing substrates) and fire severity, drawing on data obtained from recent field surveys and both published and unpublished literature sources relevant to the Coastal IFOA. Comparisons of the magnitude of impacts to forests between the Black Summer fires and previous fire seasons are presented and discussed in summarised terms. This report focuses

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[4] Tolerable fire thresholds in NSW are based on known plant responses to fire/life history traits, e.g. obligate seeder, resprouter, time to maturation; see Kenny et al. (2004).

[5] In the context of this report, 'topkill' refers specifically to a tree that has either i) died and not resprouted, or; 2) the above-ground biomass has completely died and the tree has resprouted from the base/ground level. A basally resprouting tree must complete a 'secondary juvenile phase' before reproduction can resume (see Fairman et al., 2016, 2019), the length of which is likely to vary widely across species and productivity gradients. The secondary juvenile phase is likely to be shorter than the primary juvenile phase, i.e. the time to reproductive maturity will be shorter. Remaining dead sections of trees may continue to provide resources for hollow-using fauna (Goldingay, 2011) if hollows are present and at rates comparable to dead standing trees.

solely on data relating to changes to tree populations and does not attempt to interpret changes to understorey vegetation or fauna populations, which are beyond the scope of this work.

1.4 Project scope

The agreed scope of this report is to summarise evidence for the response of forests to the 2019/20 wildfires, including impacts of the preceding drought and subsequent above average rainfall, based on field-based research. The report should cover:

- recovery mechanisms of forests within the Coastal IFOA region to different fire severity and drought intensity
- results from available empirical evidence (either grey or published data/literature) of post-fire forest recovery and any observed differences across the Coastal IFOA region (by location, forest type, disturbance history), including rate of recovery by fire severity class
- results from testing how well the post-fire stability index performs against field data
- recommendations for further work to monitor forest recovery in the Coastal IFOA region following the 2019/20 wildfires.

1.5 Response mechanisms of eucalypt forests to fire and drought

Many eucalypt species, from the genera *Eucalyptus*, *Corymbia* and *Angophora*, recover from severe disturbance by resprouting from epicormic buds beneath the bark and/or from an underground storage organ called a lignotuber (Burrows, 2013; Clarke et al., 2015; Nicolle, 2006). Successful resprouting in eucalypts depends on the severity of the disturbance and tree characteristics such as stem size, bark characteristics and previous fire damage (Bendall et al., 2022b; Collins, 2020; Nolan et al., 2020b). For example, very small trees (e.g. <10 cm DBH) typically resprout from a lignotuber as they do not have thick enough bark on the stem to protect epicormic buds. While mature trees typically have thick enough bark to protect epicormic buds from fire, resistance varies among species due to differences in bark type (Bendall et al., 2022b; Collins, 2020; Nolan et al., 2020b). However, there are several obligate seeder species with limited or no capacity to resprout following disturbance; their regeneration is primarily from seed. Although many species can both resprout and recover from seed, some are weak resprouters (Bradford, 2018). Thus, disturbance-related mortality and recovery patterns across forests are likely to vary due to local differences in species composition (Trouvé et al., 2021a).

Several obligate seeders and weak resprouters occur throughout the Coastal IFOA region (Table **1**).

Table 1. List of species within Coastal IFOA region with limited ability to resprout following disturbance.

Name	Resprouting limitations
Eucalyptus oreades (Blue Mountains ash)	Obligate seeder with limited resprouting capacity lignotuber (Little & Gardner, 2003).
E. fraxinoides (white ash)	Obligate seeder with limited resprouting capacity (Little & Gardner, 2003).
E. pauciflora (snow gum)	Limited capacity to resprout epicormically, vulnerable to hydraulic failure, typically able to resprout from a lignotuber (Losso et al., 2022) .
E. grandis (flooded gum)	Facultative seeder with 2-3 year recruitment timeline with limited resprouting capacity (Bradford, 2018) .
E. delegatensis (alpine ash)	Obligate seeder (mainland population only) with limited resprouting capacity (Rodriguez-Cubillo et al., 2020).

Stands dominated by any of the species in Table **1** are in effect subject to stand replacement if fire intensity is sufficient to kill mature trees and subsequent structural recovery is greatly extended in comparison to forests dominated by trees capable of epicormic resprouting. Obligate-seeder dominated forests[6] may require minimum fire intervals of 80 years or more to reach structural maturity (Lindenmayer, 2009; Lindenmayer et al., 2000). Multiple fires at shorter intervals than the primary juvenile phase (e.g. <20 years) may eliminate obligate-seeder eucalypt species from areas of the landscape (Bowman et al., 2014). In contrast, structural recovery of stands dominated by epicormic resprouters is relatively fast as mature trees have a higher probability of surviving fire and rapidly regenerate foliage (Collins, 2020; Pausas & Keeley, 2017); the mixed-age classes that typically occur in resprouting forests also provide continuing opportunities for recruitment, ensuring population stability via adequate replacement rates (Bendall, 2021; Bendall et al., 2022a). However, if fires are too frequent, recruitment bottlenecks may form, i.e. juvenile trees may be prevented from reaching fire-tolerant size classes, potentially leading to population instability in the long-term (Collins et al., 2014; Fairman et al., 2019). For example, Collins (2020) detected a demographic shift toward mature trees following repeated high severity fires at short intervals in mixed-species eucalypt forests, owing to elevated mortality of small stems, when compared to areas that experienced low severity fires. Repeat high severity fires are also known to decrease the

[6] This report primarily focuses on the response of mixed-species eucalypt forests dominated by trees capable of resprouting. The responses of obligate seeder forests to fire are well-documented elsewhere. For the purposes of this report, all obligate-seeder dominated forests that occur within the Coastal IFOA region are assumed to be subject to partial stand-replacement following high severity fire (e.g. see Benyon and Lane 2013).

resprouting success of mature trees (i.e. increased topkill) and decrease levels of seedling recruitment in mixed-species eucalypt forests (Fairman et al., 2019; Fairman et al., 2016).

Eucalypt species vary broadly in their resistance to hydraulic failure and capacity to survive drought (Bourne et al., 2017; Li et al., 2018). Some species appear capable of recovery following drought via epicormic resprouting and growth of new xylem (Gauthey et al., 2022; Losso et al., 2022). Hydraulic failure potentially leads to greater vulnerability to future stressors including fire (Mantova et al., 2022), as stored carbohydrates must be used to recover from disturbance and their depletion has been linked to increased rates of post-fire mortality, e.g. in north American conifer forests (Reed & Hood, 2023). However, disentangling the coupled effects of drought and fire is challenging without experimental controls or repeat measures before and after fire. For example, Nolan et al. (2022) found that unburnt, drought-affected sites experienced substantially less topkill than sites affected by high severity fire and drought (12-55% versus 47-78%), which suggests that the compounding effect of drought and fire on tree resprouting success is more influential than drought impacts alone. Bendall et al. (2022a, 2022b) quantified the impacts of variations in drought severity preceding fire in the Sydney region, although the scale and severity of drought in those studies was generally far less extreme than the 2017-2019 drought. Nonetheless, Bendall et al. (2022b) found that mortality and resprouting patterns in mature trees in dry and wet forests were similar when exposed to both moderate and severe drought prior to fire. However, juvenile mortality was elevated and total juvenile abundance reduced by severe pre-fire drought compared to moderate drought (Bendall et al., 2022a).

Some highly drought-tolerant woodlands in NSW experienced significant canopy dieback due to drought and heatwaves during 2017-2020 (Losso et al., 2022; Nolan et al., 2021b) and recovery in these areas has been variable, despite remaining unburnt during the Black Summer fires (Losso *et al.* 2022, Nichols *et al.* unpub). Similarly, some eucalypt forests in southwestern Western Australia have been observed to suffer mass crown dieback (e.g. >70% of trees) and high mortality rates (e.g. 25%) following prolonged severe drought (Matusick et al., 2013). Projected increased temperatures under climate change are likely to produce more extreme droughts, which could compound impacts on forests (Allen et al., 2021; Naumann et al., 2018).

There have been few field studies of drought responses in eucalypts relevant to the vegetation of the Coastal IFOA region. Table **2** outlines studies that are potentially relevant, although most relate to moderately-xeric dry sclerophyll forests (DSF) or woodlands in tableland areas.

Table 2. Overall results and key findings from a range of studies that examined forest responses to drought in temperate eucalypt forests and woodlands. DSF = dry sclerophyll forest; WSF = wet scleophyll forest; GWD = grassy woodland.

Source	Forest type	Variable measured	Overall results
Losso et al. (2022)	- DSF - GWD	- Canopy health - Hydraulic functions	- 35-37% of basal area with completely dead canopy
Nolan et al. (2021)	- DSF - GWD	- Canopy health - Hydraulic functions	- Hydraulic failure primary mechanism of canopy dieback - Trees with larger DBH and height less likely to experience canopy dieback
Li et al. (2018)	- DSF	- Canopy health - Hydraulic functions	- Hydraulic failure primary mechanism of canopy dieback
Matusick et al. (2013, 2016, 2018)	- DSF	- Mortality - Canopy health - Stand structural changes	- 74% of stems in affected areas had crown dieback - 26% of stems died within 6 months of drought - Demographic shift toward smaller trees - Increase in mortality from 10-55% between low and high severity long-term drought - Increase in crown dieback from 74-96% between low and high severity long-term drought in one species
Pook et al. (1966, 1984, 1986, 1987)	- DSF - WSF	- Litter fall - Change in leaf area - Hydraulic functions - Post-drought survival of epicormic growth	- **DSF**: on shallow stony exposed aspects most vulnerable to drought - **DSF**: 3-30% of trees died during or following drought, varying with species. Likely interactions with insect attack. - **WSF**: Significant leaf shedding during drought
Podger et al. (1980)	- WSF	- Canopy health	- Significant crown or total dieback, possibly related to drought.
Cremer (1966)	- DSF	- Canopy health	- Mortality or severe canopy dieback on stony exposed sites, mostly occurred post-drought - Likely interactions with insect attack.

1.6 Recent studies relevant to the Coastal IFOA region

Widespread increases in vegetation cover and greenness were observed via remote sensing assessments across forests following the Black Summer fires and the antecedent drought, typical of the post-fire flush of epicormic resprouting and new seedling establishment common throughout fire-adapted eucalypt forests of southeastern Australia (DPE, 2023; Gibson & Hislop, 2022; Qin et al., 2022). However, based on remote sensing estimates of post-fire biomass recovery dynamics associated with field measures at high fire severity, short fire interval sites, approximately 19% of the total area that was burnt at high or extreme severity in the 2019/2020 fires in NSW displayed signals of limited or delayed post-fire recovery dynamics (DPE, 2023).

Satellite derived spectral indices that contrast short-wave infra-red (SWIR) and near infra-red (NIR) wavelengths, such as derivatives of the Normalised Burn Ratio (NBR), have strong correlations with field-based measurements of post-fire forest dynamics (e.g. fire severity and post-fire recovery; Gibson et al., 2020; Hislop et al., 2018; Hislop et al., 2019; Gibson et al., 2022; Kennedy et al., 2010; Shvetsov et al., 2019; White et al., 2017; Wulder et al., 2009). However, reflectance-based observations of vegetation cover cannot differentiate between understorey and overstorey vertical structural elements and, consequently, were unable to directly quantify estimates of tree mortality levels among size-classes. For example, Rifai et al. (2022) estimated time to recovery for forests (recovery of remotely-sensed leaf area index) following fires during the last two decades. Rifai et al. (2022) found that forest leaf area index was likely to take up to 15% longer to recover than in previous fire seasons, e.g. within the range ~2-5 years. However, estimates of remotely sensed leaf area index provide limited insights into changes to forest structure, e.g. the proportion of recovery due to new seedling regeneration versus resprouting of surviving trees. Further, remotely sensed leaf area index is known to significantly over-estimate values in some mature eucalypt forests, and over-estimate values where there is significant grassy understorey (Hill et al., 2006). Future studies that combine spectral reflectance data and LiDAR could potentially overcome such issues (see Karna et al., 2020), as can studies that are validated with ground based observations (Gibson et al., 2022). Remote-sensing approaches to quantifying tree mortality have recently been developed for structurally simple northern hemisphere forests (Dixon et al., 2023) but these approaches have not yet been applied to resprouting eucalypt forests. Hence, there is still heavy reliance on expensive and time-consuming ground-based field surveys to provide estimates of tree mortality, recruitment, and demographic change in forests.

A small number of field studies quantifying levels of tree mortality/topkill and recruitment have been conducted following the Black Summer fires. This section summarises the main findings of those studies and indicates their relevance to forests within the Coastal IFOA region, making generalisations where appropriate. This section also discusses findings from studies relating to previous fire seasons relevant to the region. The majority of the studies described here examine aspects of the fire regime (e.g. fire interval, fire frequency) and response variables outside the scope of this report (also see Fairman et al., 2022), in addition to fire severity. Hence, the results summarised here are overall results (reported and derived) with respect to fire severity only.

Three key studies conducted following the Black Summer fires (Bendall et al., 2024; Nolan et al., 2022; Volkova et al., 2022) found overall tree mortality in areas affected by high or extreme severity fire was in the range of 34-41% of stems and topkill in the range of 47-78% of stems. Volkova et al. (2022) and Nolan et al. (2022) found that mortality and topkill was highest for trees <20-30 cm DBH, while Bendall et al. (2024) found a ‘u’-shaped response curve, i.e. elevated mortality and topkill for small trees and the largest trees. The data used in the Bendall et al. (2024) study is reanalysed and discussed in section **0** with specific focus on the Coastal IFOA region, although the published version of the data is listed in Table **3**.

The Volkova et al. (2022) study focuses on DSF dominated by *Eucalyptus sieberi* (silvertop ash) and is relevant to the Eden district within the southern IFOA region. Volkova et al. (2022) found that overall topkill increased by ~10% (from 48% to 57%) when fire severity was high compared to low/moderate. However, Volkova et al. (2022) sampled few trees larger than 60-70 cm DBH, so it is not clear what the consequences for large trees were in that study. The Nolan et al. (2022) study included both DSF and WSF, with the DSF sites being broadly relevant to dry forests on sandstone substrates in sub-coastal areas of the northern IFOA region and the WSF sites being relevant to wet forest on granite substrates in tableland areas of the southern IFOA region. Nolan et al. (2022) found that overall topkill increased by 29-42% (from 34-36% to 65-78%) when fire severity was high compared to low/moderate severity in DSF, but found little difference due to fire severity in WSF (low/mod: 53%; high: 47%). Nolan et al. (2022) had only 25 sites within the high/extreme fire category, spread across four forest types, hence the need for further investigation of sites exposed to extreme severity that was conducted in the subsequent Bendall et al. (2024) study.

Five studies relating to major fire events prior to 2019/2020 examined responses in Victorian forests with comparable species assemblages to those found in the southern IFOA region (Bennett et al., 2016; Benyon & Lane, 2013; Collins, 2020; Fairman et al., 2019; Trouvé et al., 2021a). After the 2009 Black Saturday fires in Victoria, Trouvé et al. (2021a) assessed responses of DSF and WSF (strictly *E. delegatensis*, alpine ash) typical of higher elevations within the southern IFOA region and found that topkill increased from 25% to 80% in DSF and from 50% to 100% in WSF between low and high fire intensity. Collins (2020) assessed responses of DSF comparable to moderate to higher elevations in the Eden district and sub-coastal area west of Batemans Bay. Collins (2020) found that topkill increased from 9% to 21% between low and high severity and that around one fifth of topkill could be attributed to stem collapse caused by fire scars. Collins (2020) also found that seedling recruitment was promoted by high fire severity, increasing from 300-500 seedlings ha^{-1} to 1000 seedlings ha^{-1}. Fairman et al. (2019) assessed DSF exposed to high fire severity in forests similar to that in Collins (2020). Fairman et al. (2019) found mortality in the range 10-30% and that topkill was >50% for trees smaller than about 20-25 cm DBH, decreasing to <10% for trees greater than about 40 cm DBH. Fairman et al. (2019) also found high seedling recruitment rates between 10,000-35,000 seedlings ha^{-1}.

Benyon and Lane (2013) assessed the responses of WSF typical of mature and old growth forests found at higher elevations in the far south of the southern IFOA region. For the resprouter-type WSF, Benyon and Lane (2013) found that low, moderate and high fire severity resulted in <10% mortality, while extreme fire severity resulted in ~25% mortality for mixed-species stands and up to 45% mortality for stands dominated by *E. nitens* (shining gum). For the obligate seeder-type WSF (*E. regnans/E. delagatensis*), Benyon and Lane (2013) found that mortality increased from 61% at moderate severity to 100% at extreme severity. Bennett et al. (2016) assessed the responses of DSF comparable to that found at moderate to higher elevations in the southern IFOA region. At low fire severity, Bennett et al. (2016) found that mortality was <20% for trees >20 cm and up to 45% for trees 10-20 cm DBH. At high fire severity, Bennett et al. (2016) found that mortality was generally >25%, being highest for small trees (e.g. up to 93%) and the largest trees (e.g. up to 46% for trees >70 cm DBH). Thus, the Bennett et al. study detected a 'u'-shaped mortality response curve similar to that in Bendall et al. (2024), resulting from coupled extreme drought and fire.

Three further studies relating to past fire events in the sandstone forests of the Sydney region estimated overall mortality or topkill in DSF burnt at high severity at between 10-33%, and mortality in WSF burnt at low/moderate severity at 25% (Bendall et al., 2022a, 2022b; Nolan et al., 2020b). Bendall et al. (2022a) estimated levels of seedling recruitment at below <1000 seedlings ha^{-1} in both DSF and WSF. These three studies are broadly relevant to coastal/subcoastal areas at the southern end of the northern IFOA region.

Table 3. Overall results and key findings from a range of studies that examined forest responses to fires of different severity with vegetation relevant to the Coastal IFOA region. DSF = dry sclerophyll forest; WSF = wet scleophyll forest; CBS = clearfell, burn and sow silvicultural system (Trouvé et al., 2021b).

Source / other variables		**Fire severity**	**Mortality (%)**		**Topkill (%)**		**Seedling recruitment (seedlings ha^{-1})**		**Notes**
			DSF	**WSF**	**DSF**	**WSF**	**DSF**	**WSF**	
Bendall et al. (2024)		High / extreme	20-23	22-33	34-37	36-41	8300	3600-6800	
Volkova et al. (2022)		Low / moderate			56				Dead trees mostly <20 cm DBH
		High			58				
Nolan et al. (2022) *Excluding western slopes DSF*		Unburnt			12-21	55			Dead trees mostly <20-30 cm DBH
		Low / moderate			36	53			
		High / extreme			65-78	47			
Bendall et al. (2022a,b)	***Mild drought***	Low / moderate		25				0-320	No mortality effect due to drought. Estimates are for trees with fire scars.
		High	33				160-520		
	Severe drought	Low / moderate		25				0	
		High	33				0-1080		
Trouvé et al. (2021a,b)	***Resprouting species***	Low intensity			25				
		Moderate intensity			50				
		High intensity			75				
	Obligate seeder species	Low intensity				50			Recruitment higher/patchier in burnt plots; lower/more spatially uniform in CBS plots
		Moderate / high intensity				100		4000-56,000 (fire)	

Source / other variables		Fire severity	Mortality (%)		Topkill (%)		Seedling recruitment (seedlings ha^{-1})		Notes
			DSF	WSF	DSF	WSF	DSF	WSF	
								3000-25,000 (CBS)	
Collins (2020)		Low			9		300-500		~19% of topkilled trees had collapsed stems due to fire scars; topkilled trees mostly <30 cm DBH
		High			21		1,000-1,500		
Nolan et al. (2020) *Coastal sites only*		Unburnt	10		5				Topkilled trees mostly <30 cm DBH
		High	10		29				
Fairman et al. (2019)		Unburnt	~15		<5		Very low		
		High	10-30		>50% (<20 cm DBH) <10% (>40 cm DBH)		>10,000		
Bennett et al. (2016)		Unburnt	10-24				Very low		Mortality highest for trees 10-20 cm DBH in unburnt/low; 'u'-shaped response in high
		Low	5-45				Very low		
		High	24-93				>10,000		
Benyon and Lane (2013)	***Resprouting species***	Low		<10					
		Moderate		<10					
		High		<10					
		Extreme		25					
	Eucalyptus nitens	Low		<10					
		Moderate		<10					
		High		<10					
		Extreme		45					
	Obligate seeder species	Low		<10					
		Moderate		61					
		High		96					
		Extreme		100					

2 Analysis of impacts to forests following 2019/2020 wildfires and preceding drought

2.1 Aims

(i) To assess levels of tree mortality, topkill and seedling recruitment across a range of forest types and substrates common across the Coastal IFOA region. This aim is addressed through analyses of two pre-existing independently collected field datasets.

(ii) Improve interpretation of remote sensing estimates of post-fire biomass recovery. This aim is addressed by comparing remote sensing signals against field data.

2.2 Methods

2.2.1 Field survey design

Between May 2022 and February 2023, a large-scale field study was conducted by Western Sydney University (WSU) across NSW aimed at determining rates of tree mortality, topkill (i.e. dead tree or tree with basal resprouting only) and post-fire seedling recruitment following the Black Summer fires and preceding drought. Eighty-nine sites were stratified across two forest types (dry sclerophyll forest, DSF; wet sclerophyll forest, WSF) growing on both higher fertility granite substrates and lower fertility sandstone substrates to examine the interactive effects of forest type and substrate (i.e. soil fertility, productivity) on tree responses. Sites were placed along a climatic gradient defined by a climate moisture (aridity) index (CSIRO, 2014) to obtain broad representation of xeric/mesic sites within each forest type (Fig **1**). Site selection was constrained such that all sites: (i) experienced high (class four) or extreme (class five) fire severity in 2019/2020, as defined by data from the Fire Extent and Severity Mapping (FESM) database (DCCEEW, 2024); (ii) experienced severe pre-fire drought (index value of <-1.5) as defined by the Standardized Precipitation-Evapotranspiration Index at 12-monthy time scale (SPEI, Vicente-Serrano et al., 2010); (iii) experienced no fire within the minimum ecological fire threshold for each vegetation type (preceding 10 years in DSF; 25 years in WSF) as defined by Kenny et al. (2004); (iv) had no prior harvesting history[7] and was within the National Park estate.

[7] The majority of sites had no evidence of past harvesting but we could not rule out that some type of historical selective tree removal may have occurred at some sites.

Standard 0.1-hectare (50m x 20m) plot sizes were used, with smaller sub-plots (0.05 ha, 0.025 ha) used to target smaller tree size classes. Diameter at breast height over bark (DBH, 1.3 m) was measured for all trees >2.5 cm DBH. All DSF plots were on ridges or plateaus; all WSF plots were in gullies or on plateaus. Trees were categorised as dead when there was no living foliage present, and topkilled when either dead or basally resprouting from a dead stem. Logs and stumps were included when they were determined to have been trees alive prior to the Black Summer fires and preceding drought. Presence/absence of basal fire scars was recorded, as fire scars are a known contributor to stem mortality (Gill, 1974; Gutsell & Johnson, 1996; Mattheck et al., 1994; Whitford & Williams, 2001). The WSU data was collected post-fire only and could not quantify the number of trees that were completely combusted (i.e. leaving no trace). This means that mortality was likely to be underestimated to some extent. A more comprehensive description of the methods can be found in Bendall et al. (2024).

Additional data was provided to Western Sydney University (WSU) under a license agreement by the NSW Forestry Corporation (FCNSW) and consisted of tree-level measurements of DBH, mortality and topkill (derived[8]) for 261 sites spread across the Coastal IFOA region (Fig **1**). FCNSW sites burnt during 2019/2020 across a range of fire severities, time-since-harvest and previous fire intervals. The majority of sites experienced severe pre-fire drought (e.g. SPEI <-1.5). FCNSW sites were comparable in size to the WSU sites (approx. 0.1 ha) but no downed trees (e.g. logs, stumps) or trees <10 cm DBH were measured (i.e. mortality and topkill were likely to be underestimated in the FCNSW dataset). The FCNSW data was collected post-fire only and could not quantify the number of trees that were completely combusted (i.e. leaving no trace). This means that mortality was likely to be underestimated to some extent. However, additional data may be available from FCNSW could shed light on this aspect of mortality, although it is beyond the scope of this report. FCNSW sites represented DSF and WSF spread across a range of substrates, including sandstone, granite, metamorphic, clastic and clay-rich (e.g. mudstone, siltstone) (Fig **2**). Sites were also spread across a climatic gradient (CSIRO climate moisture index, see above; Fig **1**). Both datasets covered a similar geographic range, although the FCNSW data included a higher proportion of coastal/sub-coastal sites (Figs **1**,**2**).

[8] For the FCNSW dataset, categorical measures of mortality and topkill were derived from multiple classification systems used by FCNSW and made comparable to the classification system used in the WSU dataset.

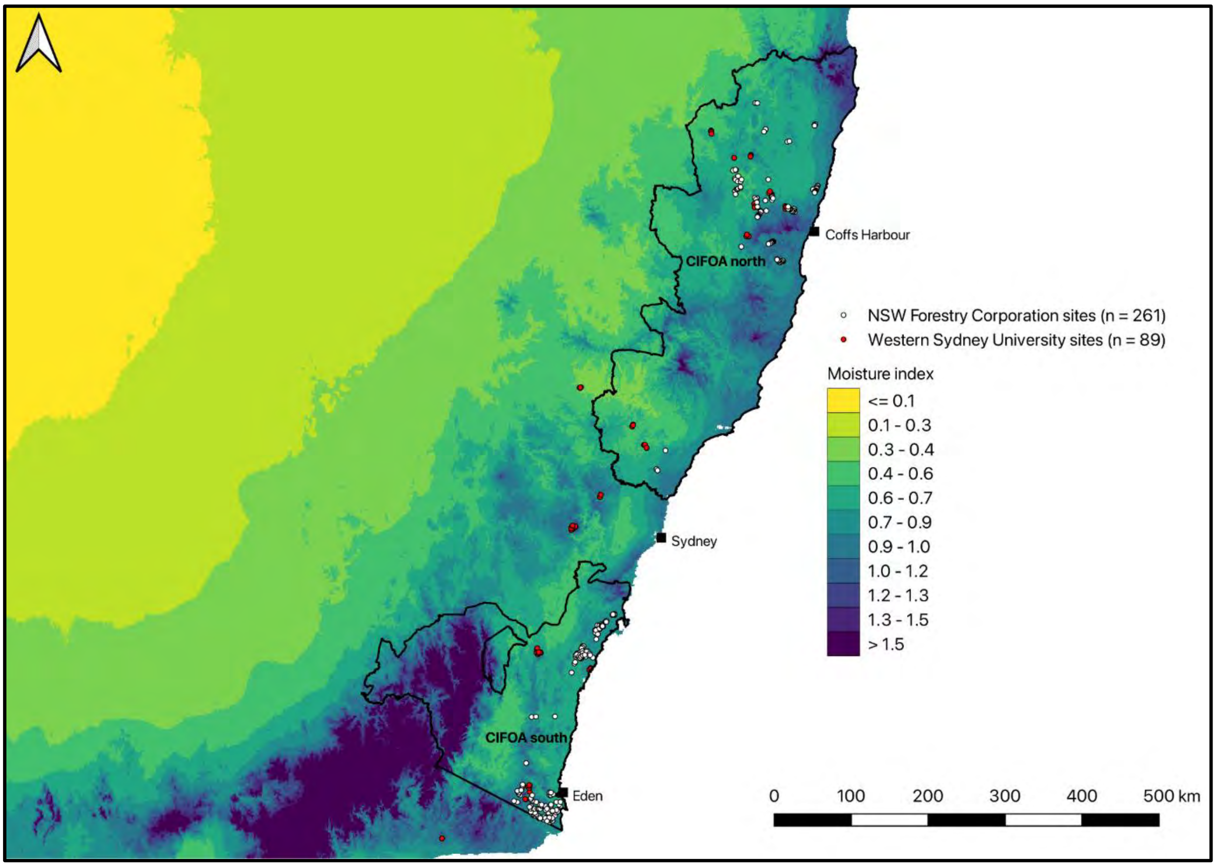

Figure 1. Map of New South Wales and adjacent areas showing locations of sites used in the analyses, in relation to Coastal IFOA region (CIFOA) and a climate moisture (aridity) index representing the mean annual value of monthly ratio of precipitation to potential evapotranspiration (CSIRO, 2014).

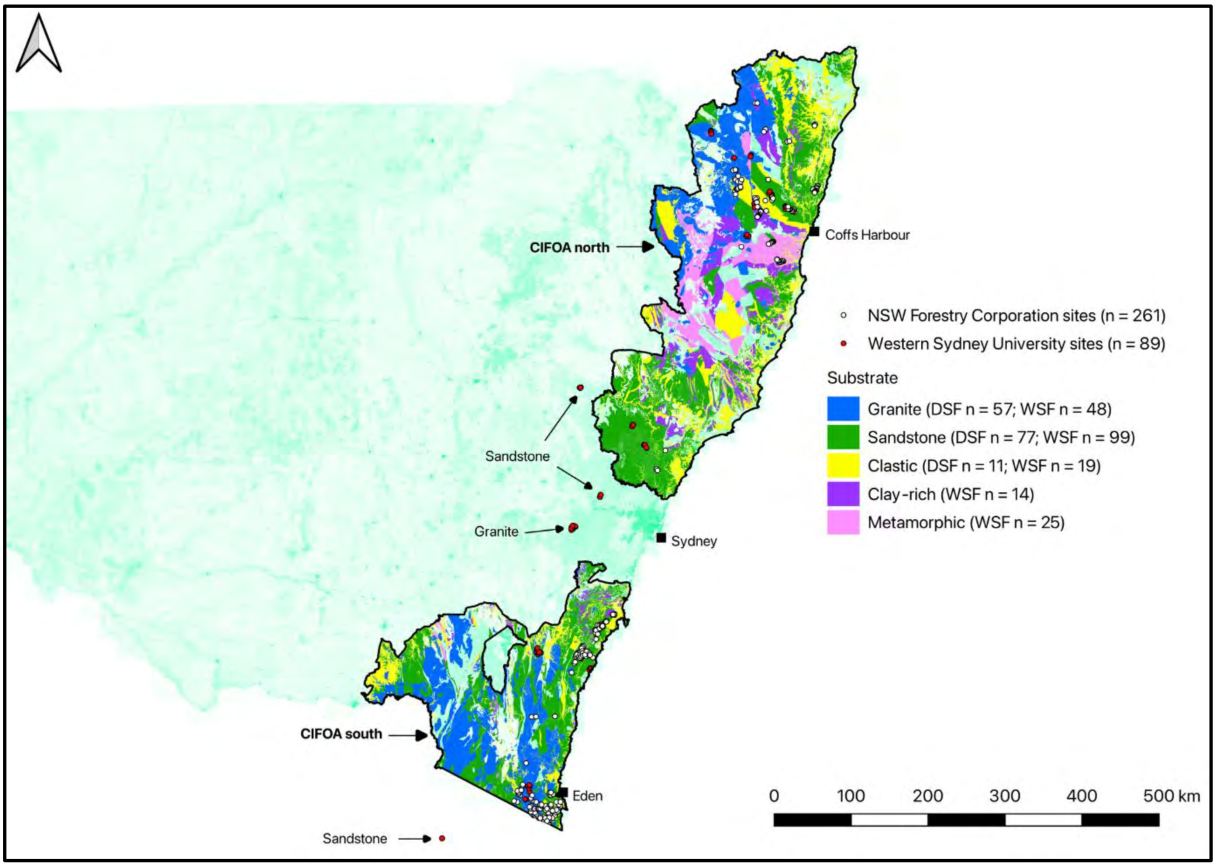

Figure 2. Map of New South Wales showing locations of sites used in the analyses, in relation to the dominant lithology (substrate) and Coastal IFOA region (CIFOA). All WSU sites were located on sandstone or granite substrate and analyses only included FCNSW sites that fell within mapped sandstone or granite substrate. Substrate of sites outside the CIFOA region is indicated by small arrows. Geological data sourced from the NSW Seamless Geology (NSW Government & DRNSW, 2018).

2.2.2 Field data analysis

Raw data was summarised by calculating the mean and standard error of responses and scaled to one hectare (for raw data summaries/stand-level estimates see Appendix **1**). We modelled bounded estimates of response probabilities (Bernoulli-distributed) at the tree level using generalised additive modelling (Wood, 2017). We modelled the WSU data and FCNSW data separately. For the WSU data we excluded trees <2.5 cm DBH from the mortality/topkill analyses, as the identity of dead stems smaller than this was not able to be determined. In all models, DBH was included as a smooth term to account for relationships between tree size and responses and was fit individually for each substrate/forest type combination. In all models Coastal IFOA region was included as a two-level factor (‘north’ = Upper North East and Lower North East Coastal IFOA sub-regions; ‘south’ = Southern and Eden Coastal IFOA sub-regions; as shown in Fig **2**). For models using the WSU data we included fire scar presence as a binary factor (i.e. presence/absence) and fire severity and moisture index as covariates. For models using the FCNSW data we included fire severity as a two-level factor (‘low’ = FESM classes

2, 3; 'high' = FESM classes 4, 5) and the moisture index as a covariate. Previous fire interval and time-since-harvest for the FCNSW data were highly variable and unevenly spread across the other variables. So, we did not include either in the model (i.e. the model assumed that the previous fire interval and time-since-harvest had no effect on responses). Likewise, low observations across four other substrates, including clastic (DSF/WSF), metamorphic (WSF) and clay-rich substrates (WSF) precluded modelling and only data for granite and sandstone substrates was included (n granite/sandstone sites = 192; n other substrate sites = 69). All models included a site-level random factor as tree-level observations were nested within each site. Post-fire seedling recruitment (WSU data only; number of seedlings ha^{-1}) was modelled using a negative binomial distribution. No topographic effect was investigated as i) All DSF and WSF sites in the WSU dataset experienced high or extreme fire severity; ii) all DSF sites in the WSU dataset occurred on ridgetops and the majority of WSF sites occurred in gullies, precluding factorial stratification.

Credible intervals (CI's) were calculated as the central 50% and 95% of model predictions and we visually assessed the level of overlap of CI_{50} where possible as a means of obtaining support for statements relating to whether responses differed between groups. In addition, we selected contrasts from the posterior distribution of the model and calculated the mean difference ('calculated mean difference') between them (e.g. to compare the magnitude of differences between different groups overall or at specific intervals across the range of DBH values). One advantage of using this approach was direct comparison of the largest trees in one group versus the largest in another, even though the maximum DBH could differ between those groups. Models were fitted using the 'brms' package for R version 4.2.1 (Bürkner, 2017; R Core Team, 2023).

2.2.3 Relative risk of delayed recovery

To assist land managers in locating severely affected areas that may be at risk of delayed recovery due to higher rates of mortality and topkill, a set of spatial data products were generated based on results from field data analysis. Substrate/forest type mapping was derived from three input layers; the FESM fire severity mapping from the 2019-20 fire year (DCCEEW, 2024), the NSW State Vegetation Type Mapping (DCCEEW, 2022) and the NSW Seamless Geology (NSW Government & DRNSW, 2018). Relative risk mapping was based on key results from the field data analyses whereby higher rates of mortality and topkill were observed at sites that burnt at (i) high (or extreme) severity; (ii) sites with granite substrate; and (iii) sites in wet sclerophyll forest (see **Results**). Sites burnt at high severity with one of the other factors (wet sclerophyll forests or granite substrate) were ranked with the highest relative risk category ('very high'). Sites burnt at high severity on sandstone substrate in dry sclerophyll forests were ranked with the next highest relative risk category ('high'). All sites burnt at low/moderate severity were ranked in the 'moderate' relative risk, and all unburnt sites were ranked in the 'low' relative risk category. These spatial layers have been limited to State Forest tenure within the north and south CIFOA regions.

2.2.4 Remote sensing assessment against field data

The field data collected in this study did not provide repeated site measures over multiple consecutive years, precluding a comprehensive field validation of the remote sensing method. Nonetheless, the comparison of remote sensing signals against field data will help to quantify the relationship between remote sensing estimates of post-fire biomass recovery in terms of field measures of post-fire mortality, topkill, and vegetative regrowth.

Two indices based on remote sensing signals were utilised:

(1) The post-fire biomass recovery index ('recovery index', Gibson et al 2022, DPE 2023b) estimates rates of biomass change in the post-fire environment, based on the amount and greenness of vegetation cover (see Fig **3a**, 'post-fire recovery classes', which represent binned values of the recovery index). It is based on the concept that a disturbed system will show high rates of change, while undisturbed or recovered system will show near-zero rates of change. Higher values in the index indicate a greater magnitude of increase in biomass compared to the preceding year (i.e. a forest that is still recovering from fire impact). Recovery index values close to zero (-50 – 50) indicate little to no change in biomass. A stable state may be indicated by little to no change in burnt areas, synchronised with local unburnt areas, across multiple consecutive years (see Table **1**, DPE 2023b);

(2) Estimates of areas with limited or delayed recovery ('delayed recovery index') were developed based on field measurements[9] of reduced vigour in biomass response in the first 2 years post-fire at sites burnt at high or extreme severity with 2 sequential short-interval preceding fires, likely due to exhausted soil seedbank or bud bank reserves (DPE 2023b). The binary classification of delayed recovery is determined by high or extreme severity, plus recovery index value of <500 in year 1 and <200 in year 2 (DPE, 2023).

[9] No field data contained in the WSU or FCNSW datasets were used in the development of the delayed recovery index. See DPE (2023) for details on the methodology used to generate the index.

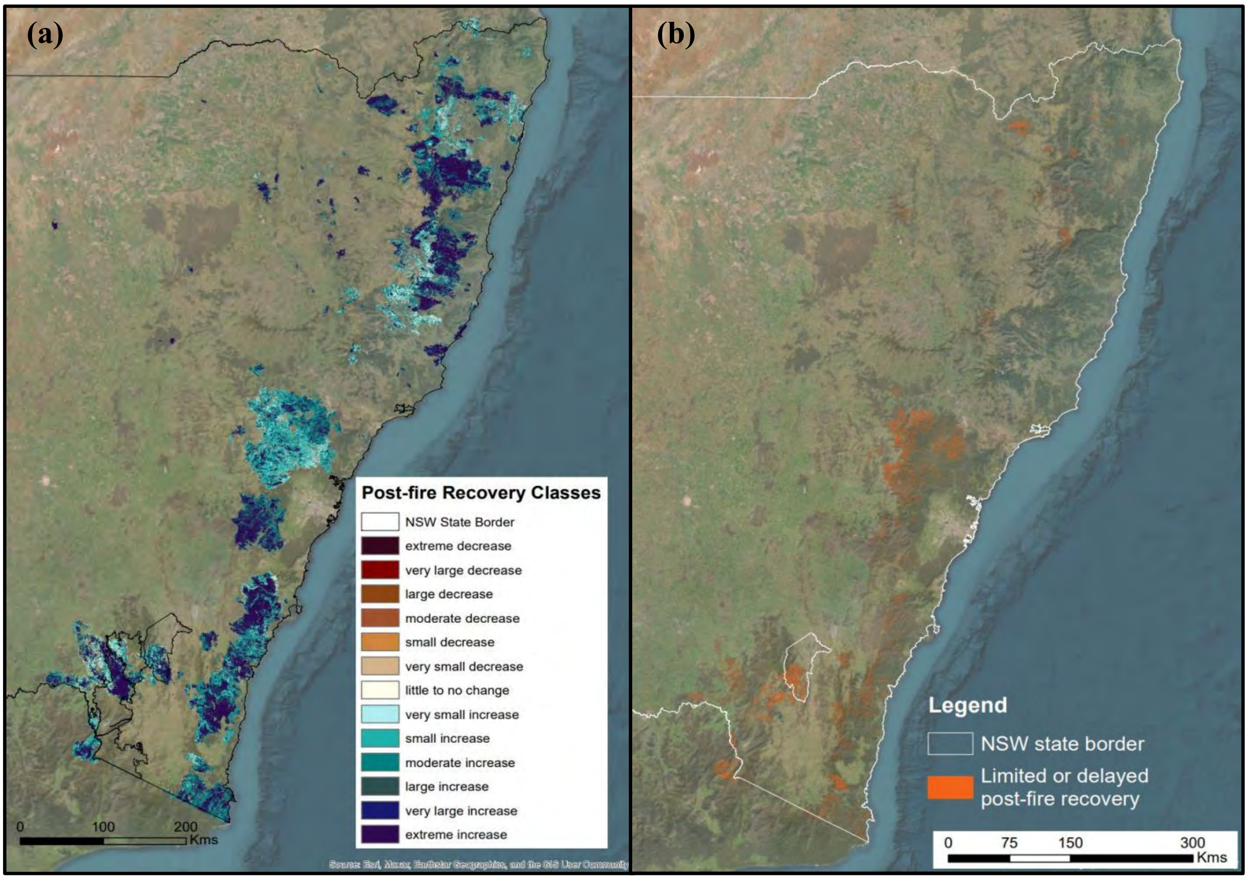

Figure 3. Map of the eastern part of New South Wales showing the post-fire recovery classes (binned values of the post-fire recovery index) at 1 year (panel **a**), and projected areas of limited or delayed post-fire recovery following the 2019/20 fire season in New South Wales, (panel **b**; DPE 2023b). The delayed recovery index was not based on field data in the WSU or FCNSW datasets.

At each field survey site, raw recovery index values for each year (1, 2 and 3) following the 2019/2020 Black Summer fires were extracted as the mean of all pixel values in a 50m radius plot centred on the field survey point coordinates. The severity class and recovery index values for the first 2 years post-fire were used to determine whether the site had signals of delayed recovery. The sum for the three years of the recovery index, and the delayed recovery index values were response variables in separate analyses. The analyses included the same explanatory variables and covariates as described in section **2.2.2**. An additional composite variable was generated for the remote sensing assessment to represent live cover, calculated as the sum of basal, stem, canopy resprouting and seedling basal area for the WSU dataset, and the sum of basal and epicormic resprouting for the FCNSW dataset. Furthermore, two additional indices were generated from the presence (1) or absence (0) of dense *Acacia* regrowth and rock at sites based on qualitative field notes for the WSU dataset.

In the analyses of the delayed recovery index, we examined rates of mortality and topkill, as well as substrate/forest type between sites with and without signals of delayed recovery. In a broader analysis of the recovery index, we assessed the relative influence of the explanatory variables by comparing a suite of candidate generalised linear models (GLMs) and selecting the top models (see Appendix **1**, Table **10**, Table **11**). Akaike's Information Criterion corrected for small sample size (AICc) was used to identify the best set of models (Burnham & Anderson, 2002). We considered any model within 5 AICc points of the top model (i.e. the model with the smallest AICc) to have strong support (Burnham & Anderson, 2002). Only results significant beyond the $p = 0.05$ level were presented. Following the field data analysis method, the WSU and FCNSW datasets were analysed separately. Response and explanatory variables were first transformed to reach a normal distribution and acceptable skewness values. All remote sensing analyses were run in R version 4.3.1 with packages 'bestNormalize', 'ggeffects' and 'AICcmodavg'.

2.3 Results

2.3.1 Site-level data summary

The WSU dataset contained 4,144 standing trees, 509 fallen trees/stumps that had been alive prior to the most recent fire, and 19,097 post-fire seedlings. Tree density was highest for post-fire seedlings (e.g. ~6,800-27,000 ha^{-1}) and lowest for trees in larger size-classes (e.g. ~25-270 ha^{-1}; Table **4**). The FCNSW dataset contained 9,282 standing trees that had been alive prior to the most recent fire and generally contained fewer large trees (e.g. >20 cm DBH) and more medium-sized trees (e.g. 10-20 cm DBH) than the WSU dataset (Table **5**). Insufficient replication across all available substrate/forest type combinations in the FCNSW data reduced the number of trees available for modelling to 7,037 trees (e.g. granite and sandstone substrates only). In both datasets, tree density was generally higher in the southern IFOA region (Table **4**). Additional stand-level data summaries can be found in Appendix **1**.

Table 4. Summary of overall stand demographics and tree density in the WSU datset. DSF = dry sclerophyll forest; WSF = wet sclerophyll forest; SE = standard error.

Variable	DBH category	IFOA region	Sandstone/DSF mean ± SE	Sandstone/WSF mean ± SE	Granite/DSF mean ± SE	Granite/WSF mean ± SE
Live tree density (stems ha^{-1})	≥20 cm	North	186 ± 19	182 ± 20	197 ± 14	196 ± 20
	≥20 cm	South	152 ± 37	253 ± 16	193 ± 21	267 ± 19
	10-20 cm	North	68 ± 10	26 ± 10	65 ± 23	31 ± 6
	10-20 cm	South	110 ± 43	83 ± 18	29 ± 8	67 ± 16
	2.5-10 cm	North	63 ± 10	39 ± 20	51 ± 14	13 ± 10
	2.5-10 cm	South	105 ± 52	56 ± 21	19 ± 5	44 ± 19
Dead standing tree density (stems ha^{-1})	≥20 cm	North	45 ± 8	41 ± 9	57 ± 11	121 ± 21
	≥20 cm	South	32 ± 12	51 ± 11	67 ± 20	59 ± 9
	10-20 cm	North	18 ± 8	11 ± 6	18 ± 7	96 ± 41
	10-20 cm	South	5 ± 5	41 ± 11	17 ± 12	31 ± 33
	2.5-10 cm	North	21 ± 15	2 ± 2	20 ± 8	64 ± 36
	2.5-10 cm	South	42 ± 22	23 ± 11	7 ± 9	22 ± 6.2
Fallen tree density (stems/stumps ha^{-1})	≥20 cm	North	29 ± 4	23 ± 6	28 ± 7	54 ± 10
	≥20 cm	South	22 ± 10	31 ± 6	34 ± 6	36 ± 6
	10-20 cm	North	9 ± 3	8 ± 5	6 ± 2	13 ± 5
	10-20 cm	South	2 ± 2	22 ± 6	4 ± 3	15 ± 5
	2.5-10 cm	North	7 ± 3	2 ± 2	13 ± 6	14 ± 5
	2.5-10 cm	South	35 ± 20	17 ± 9	4 ± 3	14 ± 5
Number of seedlings (stems ha^{-1})		North	4,093 ± 1,521	2,680 ± 1,047	3,507 ± 989	9,754 ± 2,450
		South	6,840 ± 3,687	10,141 ± 2456	27,326 ± 12,912	13,212 ± 4124

Table 5. Summary of overall stand demographics and tree density in the FCNSW datset. DSF = dry sclerophyll forest; WSF = wet sclerophyll forest; SE = standard error.

Variable	DBH category	IFOA region	Sandstone / DSF mean ± SE	Sandstone / WSF mean ± SE	Granite / DSF mean ± SE	Granite / WSF mean ± SE
Live tree density (stems ha^{-1})	≥20 cm	North	117 ± 19	128 ± 9	143 ± 25	93 ± 20
	≥20 cm	South	178 ± 155	163 ± 13	187 ± 20	181 ± 44
	10-20 cm	North	109 ± 22	76 ± 13	123 ± 34	75 ± 29
	10-20 cm	South	185 ± 29	127 ± 24	200 ± 46	143 ± 47
Dead standing tree density (stems ha^{-1})	≥20 cm	North	14 ± 5	13 ± 3	20 ± 7	30 ± 20
	≥20 cm	South	16 ± 3	21 ± 6	15 ± 3	13 ± 3
	10-20 cm	North	27 ± 12	17 ± 5	32 ± 10	20 ± 8
	10-20 cm	South	30 ± 5	86 ± 18	73 ± 16	76 ± 29
Variable	**DBH category**	**IFOA region**	**Clastic / DSF mean ± SE**	**Clastic / WSF mean ± SE**	**Clay-rich / WSF mean ± SE**	**Metamorphic / WSF mean ± SE**
Live tree density (stems ha^{-1})	≥20 cm	North	110 ± 10	165 ± 68	118 ± 8	153 ± 14
	≥20 cm	South	243 ± 43	219 ± 126	90 ± -	-
	10-20 cm	North	65 ± 55	82 ± 19	78 ± 22	43 ± 8
	10-20 cm	South	267 ± 73	139 ± 39	50 ± -	-
Dead standing tree density (stems ha^{-1})	≥20 cm	North	30 ± 20	9 ± 3	15 ± 3	23 ± 5
	≥20 cm	South	33 ± 20	17 ± 7	-	-
	10-20 cm	North	15 ± 5	5 ± 2	22 ± 11	27 ± 6
	10-20 cm	South	93 ± 39	171 ± 53	-	-

2.3.2 Modelling: WSU dataset

2.3.2.1 Mortality: trends

For the WSU dataset, there were elevated levels of mortality and topkill for smaller (e.g. <30 cm DBH) and larger trees (e.g. >60-80 cm DBH) and important differences between trees with and without fire scars and between substrate/forest type combinations. Mortality was more likely overall when a tree had a fire scar (mean probability: 37.6% vs 8.8%) and for the granite forests compared to sandstone forests (mean probability: 21.3-32.4% vs 19.4-23%; Fig **4**). In particular, granite/WSF experienced higher mortality than the other substrate/forest type combinations across the range of DBH (e.g. ~10% higher overall; Fig **4**). Mortality in sandstone/DSF decreased as DBH increased, irrespective of whether a tree had a fire scar (Fig **4**).

2.3.2.2 Mortality: key differences

For trees without fire scars, there was little difference in the probability of mortality among IFOA regions (e.g. <5%; Fig **4**). Small trees (e.g. <10 cm DBH) in granite/WSF were substantially more likely to be dead (e.g. by 20-30%, non-overlapping CI_{50}) than in any other substrate/forest type combination (Fig **4**).

For trees with fire scars, there was only a minor difference in the probability of mortality between IFOA regions, being slightly higher in the northern IFOA region across the range of DBH (42.1% vs 32.7%; Fig **4**). Small trees (e.g. 10 cm DBH) in granite/WSF were substantially more likely to be dead compared to the other substrate/forest type combinations (non-overlapping CI^{50}; calculated mean difference: 21.5-29.2%; Fig **4**). Average-size trees (i.e. 30-60 cm DBH) in granite/WSF were slightly more likely to be dead (e.g. by 6-10%; partially overlapping CI^{50}) compared to the other substrate/forest type combination (Fig **4**). The largest trees in granite/WSF (e.g. 110 cm DBH) and sandstone/WSF (e.g. 140 cm DBH) were substantially more likely to be dead than the largest trees in either DSF forest type (calculated mean difference: 27-28.6%; Fig **4**).

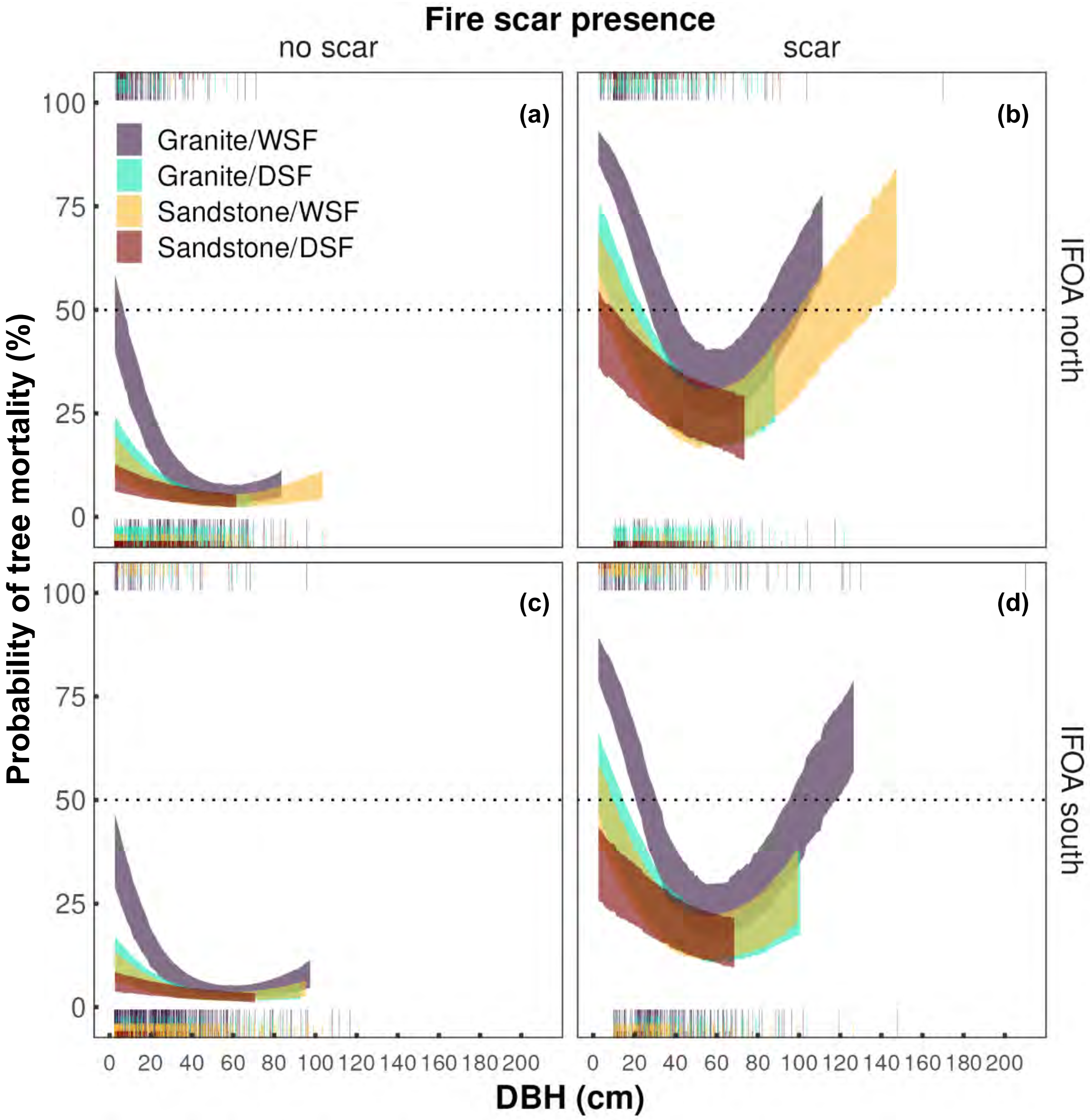

Figure 4. The effect of DBH (x-axis), substrate/forest type combination (coloured ribbons), fire scar presence (left/right panels) and Coastal IFOA region (top/bottom panels) on the probability of tree mortality for trees in forests within and adjacent to the Coastal IFOA region of New South Wales, following combined extreme drought and fire at WSU sites. Coloured ribbons represent 50% credible intervals. The observed range of DBH values varied with substrate/forest type, fire scar category and IFOA region. Predictions for each substrate/forest type are constrained to within the 99th percentile of the observed DBH range for each plot window. Rug plots (narrow coloured vertical bars) at top (dead trees) and bottom (live trees) of plot window represent the observation density across the complete range of DBH values used to inform the model. WSF = wet sclerophyll forest; DSF = dry sclerophyll forest.

2.3.2.3 Topkill: trends

Topkill trends generally followed mortality trends (i.e. there were elevated levels of topkill for smaller (e.g. <30 cm DBH) and larger trees (e.g. >60-80 cm DBH)). Overall, topkill was twice as likely for trees with a fire scar compared to trees without (47.4% vs 23.4%; Fig **5**). When trees were smaller than ~20 cm DBH, the probability of topkill was high irrespective of fire scar presence (e.g. >50%; Fig **5**). The probability of topkill was slightly higher overall in the granite forests compared to corresponding sandstone forests (e.g. by 2-5%; partially overlapping CI_{50}; Fig **5**).

2.3.2.4 Topkill: key differences

For trees with fire scars, there was little difference in the probability of topkill between IFOA regions (calculated mean difference <7%). In the northern IFOA region, large trees in WSF (e.g. 100 cm DBH) were moderately more likely to be topkilled compared to large trees in DSF (e.g. 60-80 cm; calculated mean difference of 12.6-28.8%; Fig **5**). In the southern IFOA region, the largest trees in the granite forest types were substantially more likely to be topkilled than the largest trees in corresponding sandstone forests (calculated mean difference: 21.4-34.4%; Fig **5**).

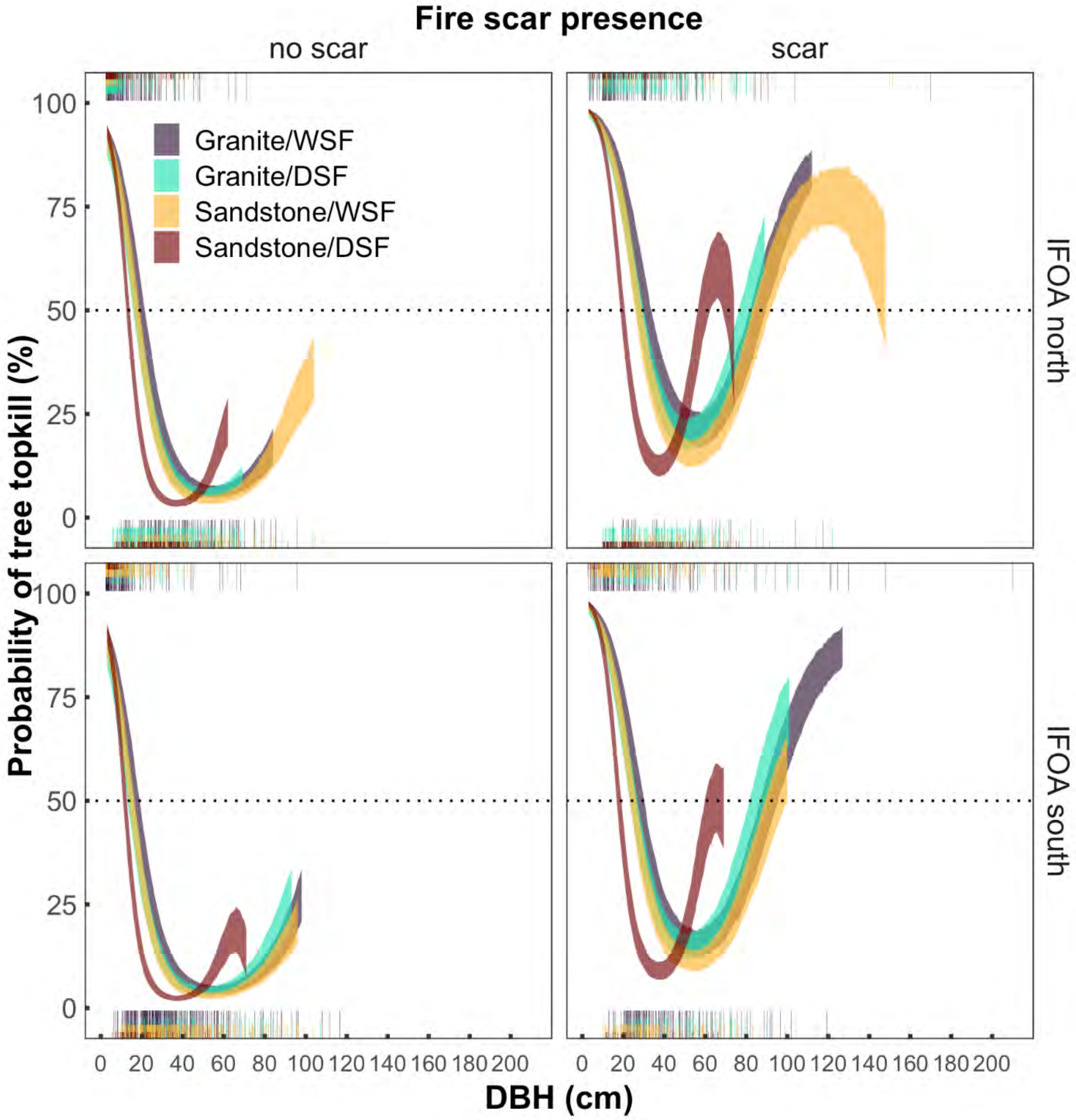

Figure 5. The effect of DBH (x-axis), substrate/forest type combination (coloured ribbons), fire scar presence (left/right panels) and Coastal IFOA region (top/bottom panels) on the probability of tree topkill for trees in forests within and adjacent to the Coastal IFOA region of New South Wales, following combined extreme drought and fire at WSU sites. Coloured ribbons represent 50% credible intervals. The observed range of DBH values varied with substrate/forest type, fire scar category and IFOA region. Predictions for each substrate/forest type are constrained to within the 99th percentile of the observed DBH range for each plot window. Rug plots (narrow coloured vertical bars) at top (topkilled trees) and bottom (not topkilled trees) of plot window represent the observation density across the complete range of DBH values used to inform the model. WSF = wet sclerophyll forest; DSF = dry sclerophyll forest.

2.3.2.5 Post-fire seedling recruitment

Post-fire seedling recruitment was similar across all substrate/forest type combinations and between IFOA regions, indicated by partially to completely overlapping CI_{50} (Fig **6**). The climate moisture index had a moderate effect on seedling recruitment. This explained some of the variability of seedling recruitment among sites, e.g. recruitment was generally higher for all substrates/forest types at higher values of the moisture index (see Appendix **1**).

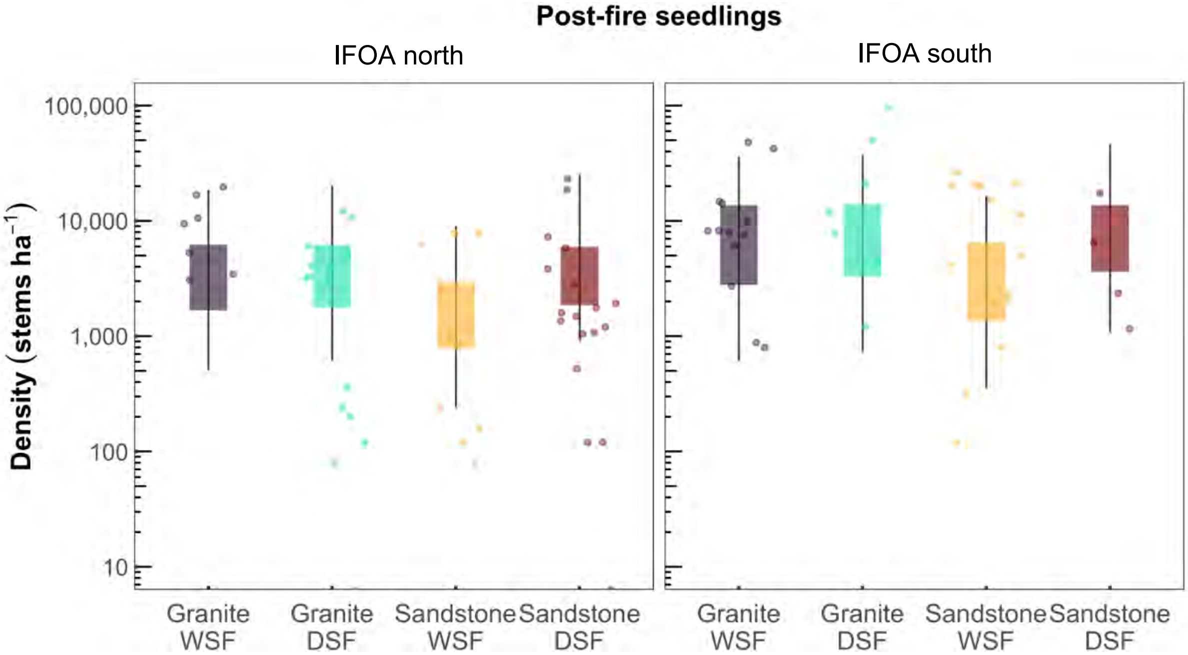

Figure 6. The effect of substrate and forest type (coloured boxplots and whiskers) on the predicted density of post-fire seedlings in each Coastal IFOA region (panels left/right) in forests within and adjacent to the Coastal IFOA region of New South Wales, following combined extreme drought and fire at WSU sites. X-axis represents substrate/forest type combination and points represent raw data. Y-axis is log scale so that extremely high observations of seedling counts can be displayed. Coloured boxplots represent 50% credible intervals and black whiskers represent 95% credible intervals. WSF = wet sclerophyll forest; DSF = dry sclerophyll forest.

2.3.3 Modelling: FCNSW dataset

2.3.3.1 Mortality: trends

Mortality trends in the FCNSW dataset generally followed those in the WSU dataset, with elevated mortality for both small and large trees in granite forests and higher mortality in granite forests (e.g. 8.5-16.6% vs 5-5.5%, Fig **7**). Mortality was higher overall under high fire severity compared to low severity (15.3% vs 3.6%) and in the southern IFOA region compared to the northern IFOA region (11.4% vs 7.3%, Fig **7**). Average-sized trees (e.g. 30-60 cm DBH) generally experienced low mortality rates irrespective of fire severity, IFOA region or forest type (e.g. <12%, Fig **7**).

2.3.3.2 Mortality: key differences

Under low fire severity, small trees (e.g. 10 cm DBH) in both granite forests and sandstone/WSF were slightly more likely to be dead than small trees in sandstone/DSF (partially overlapping CI_{50}; calculated mean difference: 10-12%, Fig **7**). Under high fire severity, substantial differences in the probability of mortality between IFOA regions were only evident for trees smaller than 20 cm DBH or trees larger than ~70 cm DBH (Fig **7**). Small trees (e.g. 10 cm DBH) in sandstone/DSF were substantially less likely to be dead compared to small trees in other forest types (e.g. by 20-30%; non-overlapping CI_{50}; Fig **7**). The largest trees on granite substrates were substantially more likely to be dead than the largest trees on sandstone substrates, particularly in the southern IFOA region (e.g. calculated mean difference: 21-40.9%; Fig **7**). The largest trees in granite/WSF were both substantially more likely to be dead than the largest trees on sandstone substrates (e.g. calculated mean difference: 28.6-40.9%) and substantially more likely to be dead in the southern IFOA region compared to the northern IFOA region (e.g. mean probability: 51% vs 32.7%; Fig **7**).

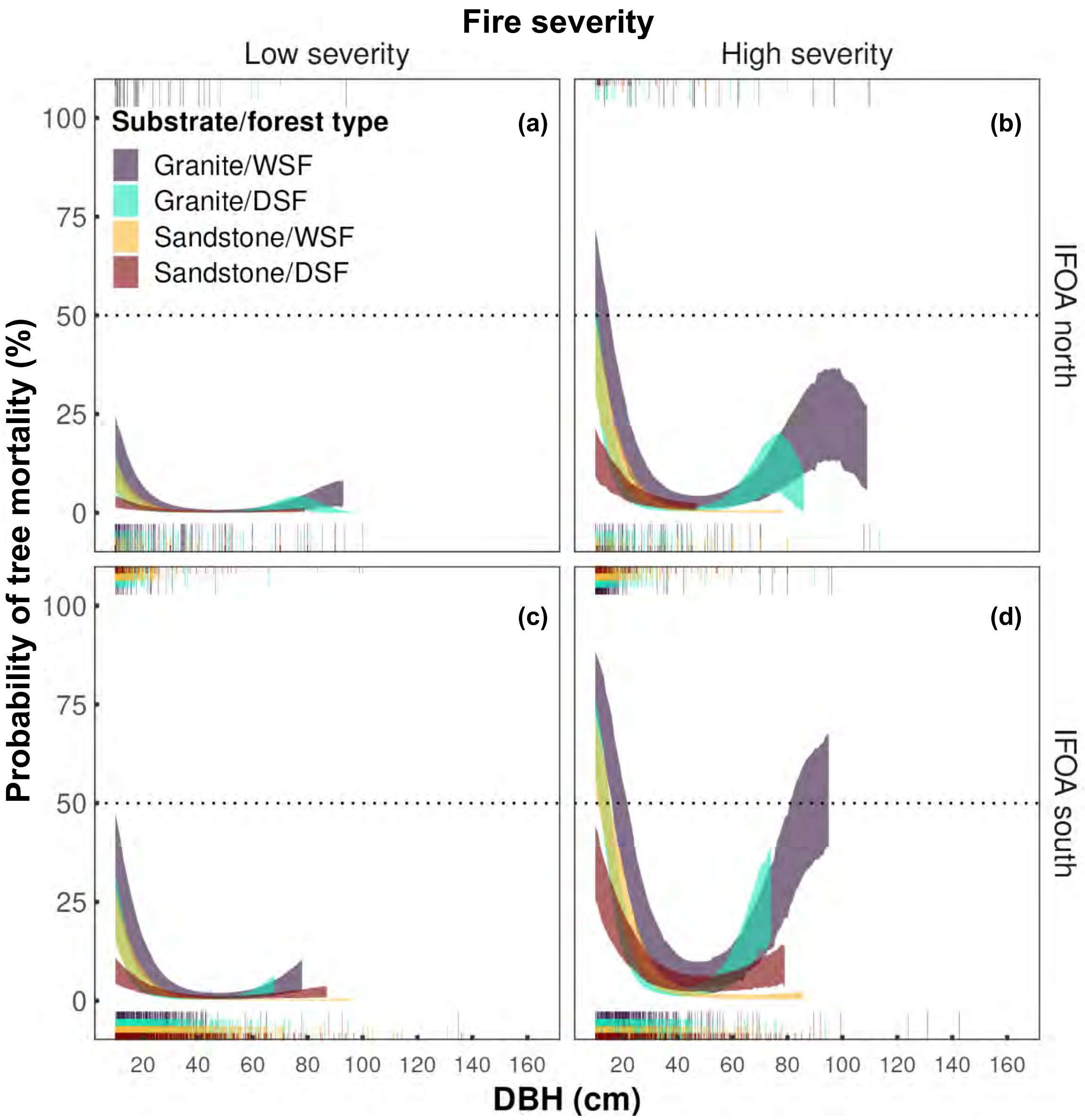

Figure 7. The effect of DBH (x-axis), substrate/forest type combination (coloured ribbons), fire severity presence (left/right panels) and Coastal IFOA region (top/bottom panels) on the probability of tree mortality for trees in forests within the Coastal IFOA region of New South Wales, following combined drought and fire at FCNSW sites. Coloured ribbons represent 50% credible intervals. The observed range of DBH values varied with substrate/forest type, fire severity category and IFOA region.

Predictions for each substrate/forest type are constrained to within the 99th percentile of the observed DBH range for each plot window. Rug plots (narrow coloured vertical bars) at top (dead trees) and bottom (live trees) of plot window represent the observation density across the complete range of DBH values used to inform the model. WSF = wet sclerophyll forest; DSF = dry sclerophyll forest.

2.3.3.3 Topkill: trends

Topkill trends generally followed mortality trends (i.e. there were elevated levels of topkill for smaller (e.g. <30 cm DBH) and larger trees (e.g. >60-80 cm DBH, Fig **8**). Overall, topkill was more likely under high fire severity compared to low fire severity (mean probability of 20.7% vs 4.8%), more likely for granite forest compared to sandstone forests (mean probability of 12.5-19.5% vs 7.4-9.8%) and more likely in the southern IFOA region compared to the northern IFOA region (9.2 vs 16.1%, Fig **8**). The elevated topkill of large trees under high fire severity (e.g. 60-80 cm DBH) was restricted to the granite forests and sandstone/DSF (Fig **8**).

2.3.3.4 Topkill: key differences

In the northern IFOA region under high fire severity, large trees (e.g. 70 cm DBH) in the granite forests were moderately more likely to be topkilled compared to the largest trees in corresponding sandstone forest type (calculated mean difference 13.5-14.7%, Fig **8**). However, this difference became smaller as DBH increased to the maximum size in the granite forests (e.g. 90-110 cm DBH, Fig **8**). In the southern IFOA region under high fire severity, the largest trees in granite/DSF (~75 cm DBH) were moderately more likely to be topkilled than the largest trees in corresponding dry sandstone forests (calculated mean difference: 18.9%), while the largest trees in granite/WSF (~95 cm DBH) were substantially more likely to be topkilled than the largest trees in corresponding wet forests on sandstone (calculated mean difference: 43.7%, Fig **8**).

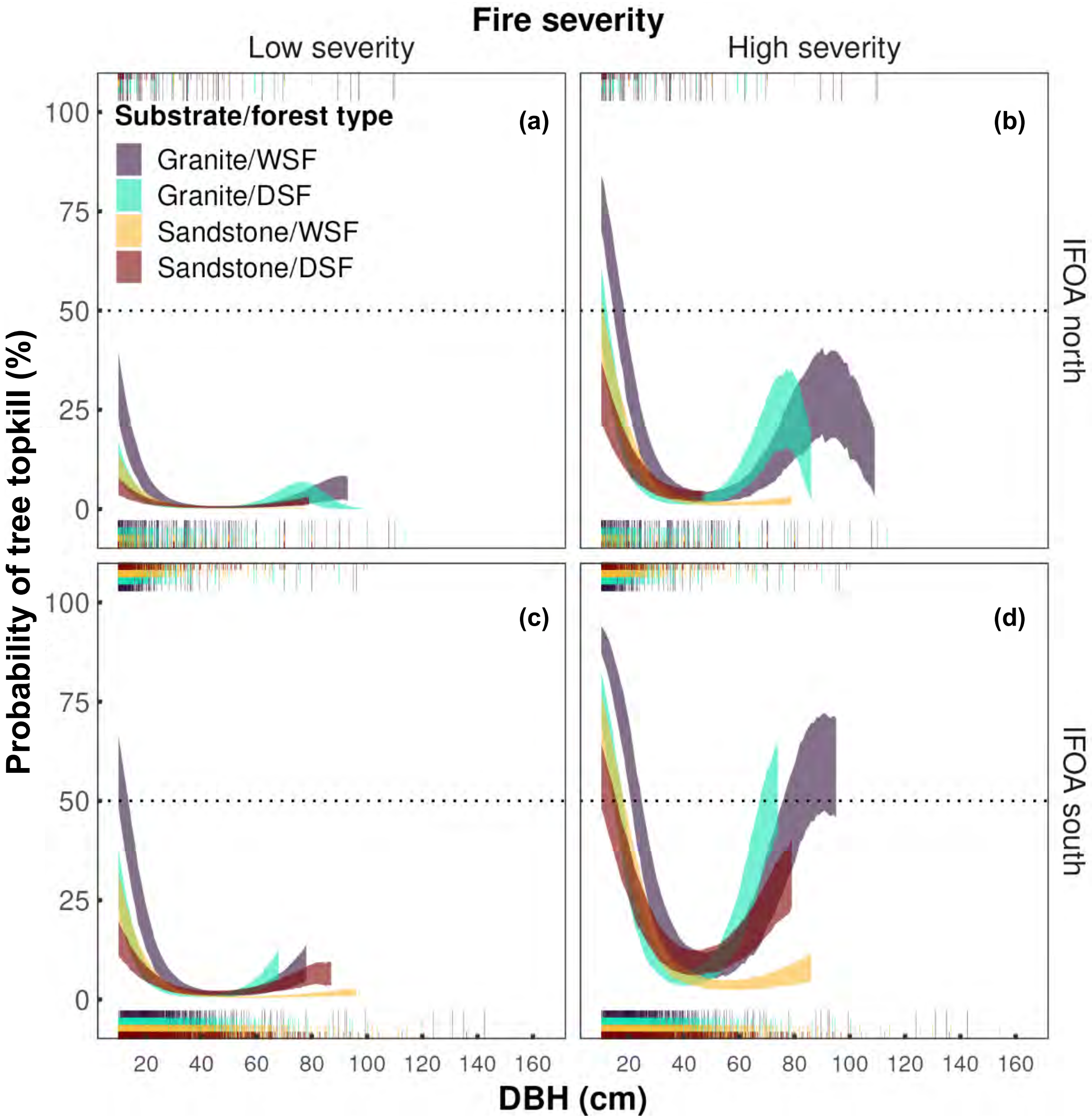

Figure 8. The effect of DBH (x-axis), substrate/forest type combination (coloured ribbons), fire severity presence (left/right panels) and Coastal IFOA region (top/bottom panels) on the probability of tree topkill for trees in forests within the Coastal IFOA region of New South Wales, following combined drought and fire at FCNSW sites. Coloured ribbons represent 50% credible intervals. The observed range of DBH values varied with substrate/forest type, fire severity category and IFOA region. Predictions for each substrate/forest type are constrained to within the 99th percentile of the observed DBH range for each plot window. Rug plots (narrow coloured vertical bars) at top (topkilled trees) and bottom (not topkilled trees) of plot window represent the observation density across the complete range of DBH values used to inform the model. WSF = wet sclerophyll forest; DSF = dry sclerophyll forest.

2.3.4 Relative risk of delayed recovery

The forest/substrate mapping (Table **6**, Fig **9a**) and the relative risk mapping (Table **7**; Fig **9b**) both identified a higher proportion of high and very high risk in the south CIFOA region compared to the north.

For the area of State Forests the Upper North East and Lower North East subregions generally experienced less extensive high severity fire within the analysed forest types than the two southern subregions. Two exceptions to this were: (i) sandstone/WSF in the Upper North

East, which experienced high severity fire across ~30% of its extent, corresponding to ~6% of the total area of the Upper North East; (ii) granite/DSF in the Lower North East, which experienced high severity fire across 38% of its extent, although this corresponded to <1% of the total area of the Lower North East (Table **6**). All forest types in both southern subregions were extensively affected by high severity fire (35-46%), generally corresponding to between 5-9% of the total area of each subregion, although areas of granite forests affected by high severity fire in the Southern subregion represented a smaller total affected area of each subregion (1.3-4.8%, Table **6**).

Table 6. Summary of the percentage of each substrate/forest type affected by fires of different severity (low = FESM classes 1-3; high = 4-5) within each Coastal IFOA subregion. Percentages also given for the totalarea of State Forest affected within each subregion. Bold values indicate >25% of forest type within subregion affected by high severity fire or >5% of total State Forest tenure affected within subregion.

CIFOA sub-region	State Forest total (ha)	Substrate / forest type	2019/2020 fire severity	Affected (ha)	% of forest type affected within sub-region	% of total State Forest affected within sub-region
Upper North East	435,984	Sandstone / DSF	Unburnt	28,201	54	6
			Low	14,115	27	3
			High	9,552	18	2
		Sandstone / WSF	Unburnt	36,567	44	8
			Low	21,609	26	5
			High	24,612	**30**	**6**
		Granite / DSF	Unburnt	12,085	58	3
			Low	5,206	25	1
			High	3,412	16	<1
		Granite / WSF	Unburnt	18,224	39	4
			Low	20,019	42	5
			High	8,975	19	2
Lower North East	491,611	Sandstone / DSF	Unburnt	13,094	38	3
			Low	15,734	46	3
			High	5,418	16	1
		Sandstone / WSF	Unburnt	60,045	68	12
			Low	17,598	20	4
			High	9,958	11	2
		Granite / DSF	Unburnt	247	43	<1
			Low	108	19	<1
			High	218	**38**	<1
		Granite / WSF	Unburnt	8,802	68	2
			Low	2,259	18	<1
			High	1,810	14	<1
Southern	423,613	Sandstone / DSF	Unburnt	16,952	24	4
			Low	26,280	37	6
			High	27,857	**39**	**7**
		Sandstone / WSF	Unburnt	9,954	14	2
			Low	30,597	44	7
			High	29,252	**42**	**7**
		Granite / DSF	Unburnt	6,925	45	2
			Low	3,068	20	<1
			High	5,495	**35**	1
		Granite / WSF	Unburnt	15,513	26	4
			Low	23,406	40	6
			High	20,144	**34**	**5**
Eden	209,911	Sandstone / DSF	Unburnt	9,119	37	4
			Low	5,518	22	3
			High	9,942	**40**	5
		Sandstone / WSF	Unburnt	6,399	20	3
			Low	10,923	34	5
			High	14,780	**46**	**7**
		Granite / DSF	Unburnt	8,700	21	4
			Low	13,867	34	7
			High	18,074	**44**	**9**
		Granite / WSF	Unburnt	9,421	31	4
			Low	8,470	28	4
			High	12,828	**42**	**6**

Table 7. Relative risk of delayed recovery for the different vegetation/substrate/severity complexes, used to classify the vegetation-substrate recovery risk map. Dsf = dry scleophyll forests, wsf = wet sclerophyll forests, unburnt = FESM unburnt class, low = FESM low and moderate classes, high = FESM high and extreme classes.

Pixel Value	Forest type/substrate type/severity class	Relative Risk of Delayed Recovery
1	dsf / sandstone / unburnt	low
2	dsf / sandstone / low	moderate
3	dsf / sandstone / high	high
4	wsf / sandstone / unburnt	low
5	wsf / sandstone / low	moderate
6	wsf / sandstone / high	very high
7	dsf / granite / unburnt	low
8	dsf / granite / low	moderate
9	dsf / granite / high	very high
10	wsf / granite / unburnt	low
11	wsf / granite / low	moderate
12	wsf / granite / high	very high

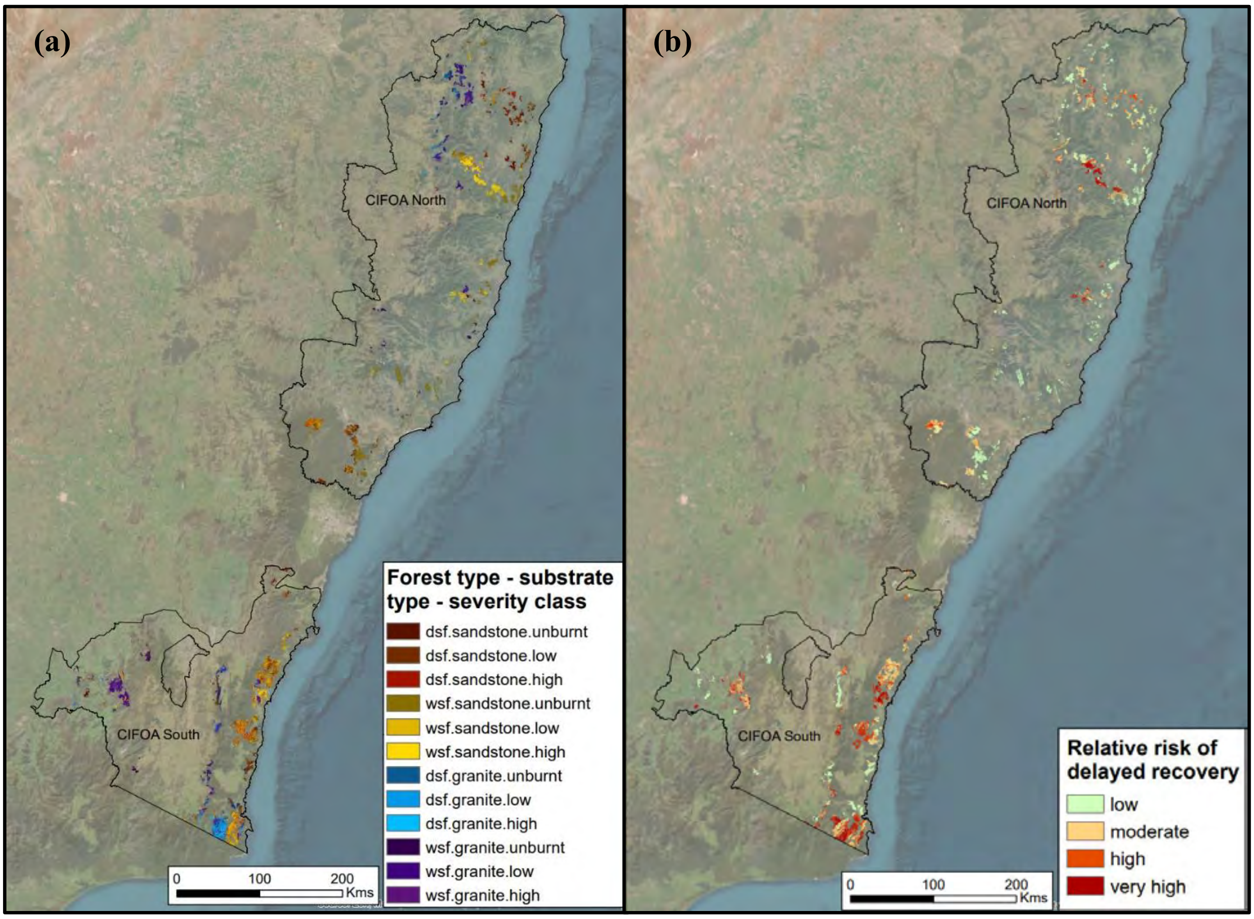

Figure 9. Forest/substrate mapping (panel **a**) and relative risk of delayed recovery (panel **b**), based on relative rates of mortality and topkill in the different categories of forest type, substrate type and severity class. Dsf = dry sclerophyll forests, wsf = wet sclerophyll forests, unburnt = FESM unburnt class, low = FESM low and moderate classes, high = FESM high and extreme classes.

2.3.5 Modelling: Remote sensing assessment against field data

2.3.5.1 Remotely sensed Delayed Recovery Index

In the analysis of the WSU field dataset, sites with a remotely sensed delayed recovery signal also had elevated levels of tree mortality (Fig **10a**) and topkill (Fig **10b**). Wet forests on granite substrate had the highest rate of delayed recovery compared to the other forest/substrate type combinations (Fig **11**). These results closely aligned with the field data results whereby granite soils had higher probability of mortality and topkill compared to sandstone, WSF had higher probability of mortality and topkill than DSF and WSF on granite experienced higher mortality compared to the other substrate/forest type combinations (see **2.3.2**). The analysis of the FCNSW dataset shows similar trends, with elevated levels of mortality (Fig **10c**) and topkill (Fig **10d**) at sites with the delayed recovery signal. There was no significant effect of forest/substrate observed in the FCNSW dataset and there was no difference in any of these relationships between north and south CIFOA regions.

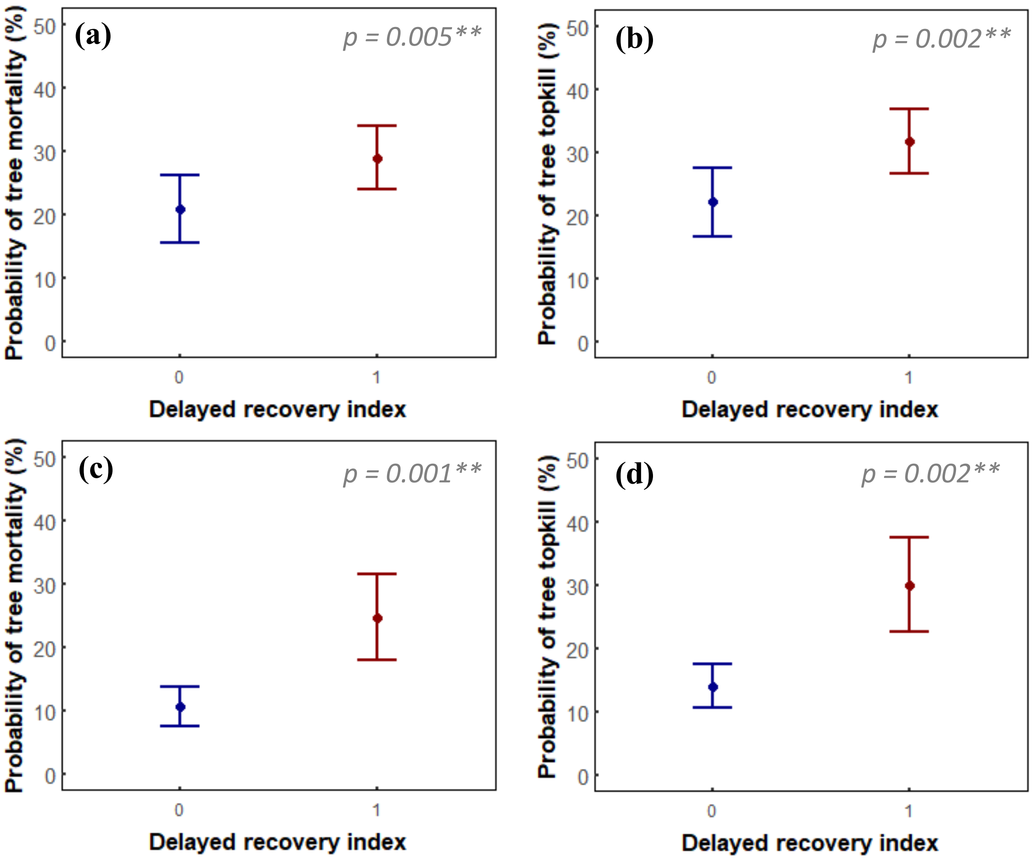

Figure 10. The probability of tree mortality and tree topkill at WSU field sites (panels **a** & **b**), and tree mortality and tree topkill at FCNSW sites (panels **c** & **d**) with delayed recovery indicated from remote sensing in forests within the Coastal IFOA region of New South Wales.

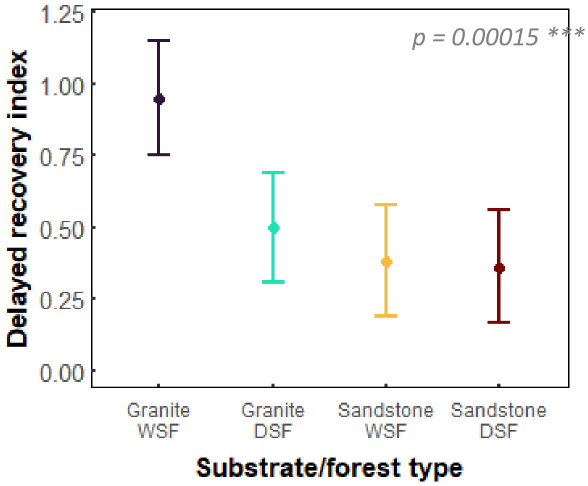

Figure 11. The probability of the limited or delayed recovery indicator from remote sensing in sites across the forest/substrate types (WSF = wet sclerophyll forest, DSF = dry sclerophyll forest) within the Coastal IFOA region of New South Wales.

2.3.5.2 Recovery index

Analysis of the WSU dataset revealed 6 models with strong support, 4 of which had significant effects beyond the $p = 0.05$ level (Table **8**). Variation in the recovery index was significantly influenced by the climate moisture (aridity) index, whereby drier sites had higher values compared to wetter sites (Fig **12a**). Sites where dense *Acacia* regrowth was present (Fig **12b**) or had been burnt at extreme severity (Fig **12c**) had significantly higher values in the recovery index. Granite/WSF sites had the lowest values in the recovery index, while sandstone/DSF sites had the highest (Fig **12d**). The presence of rock at the site did not have a significant influence on the recovery index. These results provide evidence that non-vegetative reflective surfaces, such as rock, do not generate significant commission error in this remote sensing method of monitoring post-fire recovery.

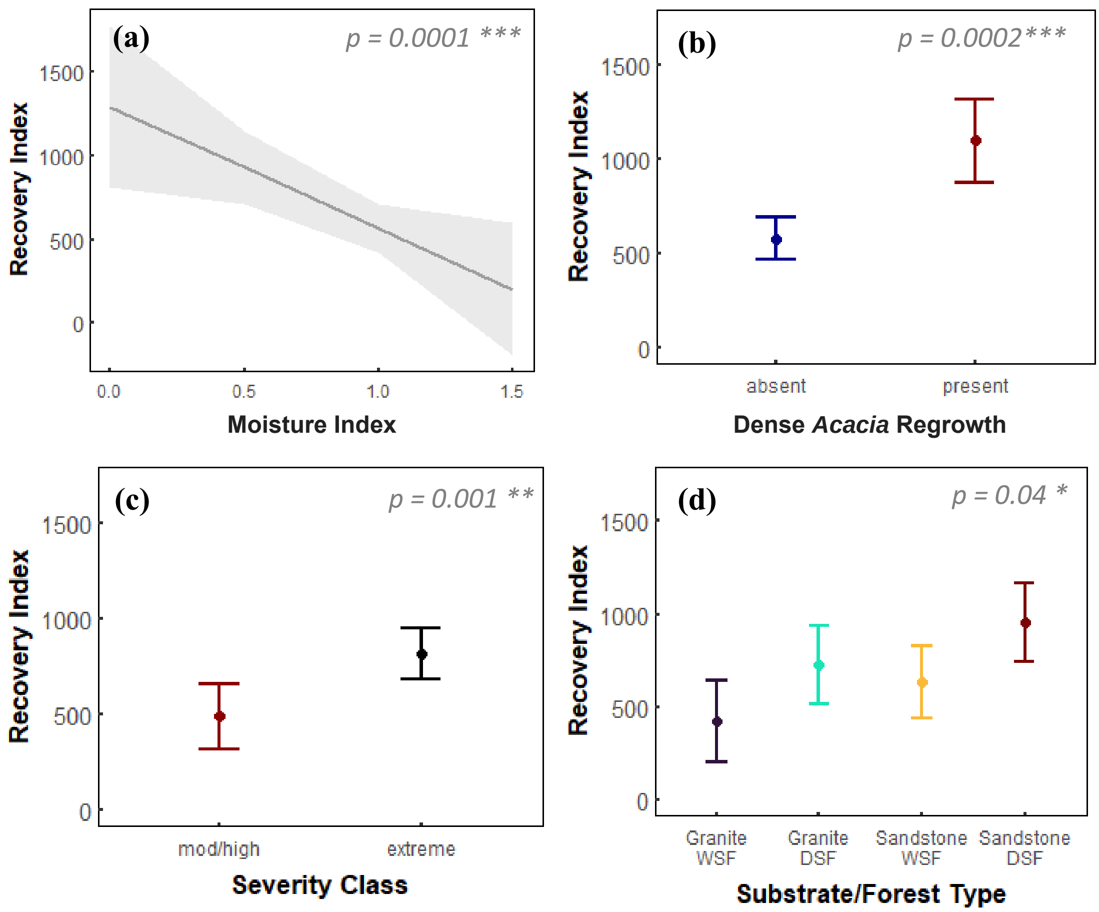

Figure 12. The effect of the climate moisture (aridity) index (panel **a**), presence of dense *Acacia* regrowth (panel **b**), substrate/forest type (panel **c**), and fire severity (panel **d**) on the recovery index (summed across years 1-3) as indicated from remote sensing in forests within the Coastal IFOA region of New South Wales for the WSU dataset.

Table 8. Top models explaining the variation in the remote sensing recovery index for the WSU dataset.

Rank	AICc	dAICc	Model	Estimate	Std.Error	t value	p value	
1	240.64	0.00	glm(sum recovery index ~ aridity)	-2.083	0.516	-4.04	0.0001	***
2	242.14	1.50	glm(sum recovery index ~ dense *Acacia* regrowth)	0.910	0.238	3.821	0.0002	***
3	243.91	3.27	glm(sum recovery index ~ mean canopy cover)	-0.064	0.108	-0.589	0.557	n.s.
4	243.96	3.32	glm(sum recovery index ~ sum % basal area cover**Acacia*)	-0.243	0.2518	-0.966	0.3366	n.s.
5	244.32	3.68	glm(sum recovery index ~ Substrate/Forest Type; Granite/WSF)	-0.591	0.284	-2.081	0.040	*
6	245.51	4.87	glm(sum recovery index ~ severity)	0.673	0.205	3.29	0.001	**

Analysis of the FCNSW dataset revealed two models with strong support, one of which had significance beyond the $p = 0.05$ level (Table **9**). Sites that had been burnt at high/extreme severity had significantly higher values in the recovery index compared to sites burnt at low/moderate severity (Fig **13**).

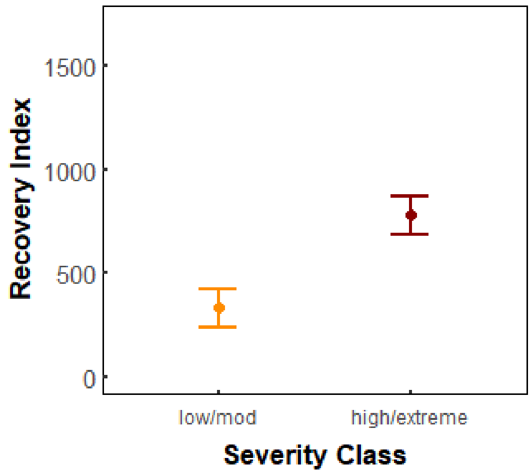

Figure 13. The effect of fire severity on the recovery index (summed across years 1-3) as indicated from remote sensing in forests within the Coastal IFOA region of New South Wales for the FCNSW dataset.

Table 9. Top models explaining the variation in the remote sensing recovery index for the FCNSW dataset.

Rank	AICc	dAICc	Model	Estimate	Std.Error	t value	p value	
1	491.75	0.00	glm(sum recovery index ~ severity)	1.016	0.124	8.170	<0.0001	***
2	493.88	2.13	glm(sum recovery index ~ sum % basal area cover*severity)	0.204	0.143	1.426	0.155	n.s.

2.4 Discussion

2.4.1 Potential demographic changes across forests

The two complementary field-based analyses, along with the remote sensing assessment, indicate clear patterns amongst responses following the Black Summer fires and preceding drought for the main forest types present across the Coastal IFOA region. The WSU results provide a more complete picture of tree responses at the upper extreme of disturbance severity, identifying that fire scars (i.e. pre-existing damage to trees) were a primary contributor to tree mortality and topkill. While the FCNSW data did not differentiate between trees with or without fire scars and likely underestimated mortality and topkill, it did reveal that areas affected by low severity fire were unlikely to have experienced major demographic changes due to the Black Summer fires and preceding drought. Modelling of both datasets revealed congruent pattens of mortality and topkill in relation to tree size, with a 'u'-shaped response curve in some forest types indicating a general pattern of elevated mortality and topkill for small trees (e.g. <20-30 cm DBH) and the largest trees (e.g. >60-70 cm DBH).

Areas affected by high or extreme fire severity experienced the greatest demographic change, likely resulting from attrition of small trees with low resistance to fire and large trees with pre-existing structural damage. These severely affected areas will take the longest to recover structurally, as their demography has been skewed toward average-sized trees (e.g. 30-60 cm DBH), meaning that years could be required to replenish small tree stocks (i.e. dependent on successful recruitment of seedlings and transition to small trees) and decades to centuries required to replenish large tree stocks, although time scales may vary depending on interactions between tree density and competition. Long-term monitoring would be required to confirm whether post-fire seedlings transition to established trees. The rate of recovery will be moderated by rainfall, site productivity, and the number and severity of future disturbances and harvesting operations.

One problematic consequence of the loss of large, old trees is that many could potentially have been hollow-bearing and thus disproportionately important for supporting viable populations of hollow-dependent animals. Evidence from mixed-species eucalypt forests in Victoria suggests that high severity fire reduces total hollow abundance, primarily via loss of larger hollow-bearing trees that contain many hollows (Wagner et al., 2024). However, the number of hollow-bearing trees (and thus hollow occurrence) remained stable due to a shift in the distribution of tree hollows towards smaller hollow-bearing trees, i.e. fire was likely to have accelerated the development of new hollows in smaller trees (Wagner et al., 2024). This could mean that long-term strategies for maintaining suitable numbers of hollow-bearing trees in harvest areas may need to include additional provisions aimed at ensuring adequate recruitment of future hollow-bearing trees from younger tree age-classes. However, the results and any subsequent management choices should be viewed through the lens of substrate and forest type, which have clear implications for the magnitude of demographic changes. Modelling of both datasets revealed that forests growing on higher fertility granite substrates compared to lower fertility sandstone substrates and wet forests compared to dry forests, were more likely to experience mortality and topkill. This means that while dry sclerophyll forests are more likely to remain demographically stable following compound severe drought and fire than wet forests, substrate type will have a moderating effect on demographic change independent of forest type. Potential ecological mechanisms driving the differences between substrates and forest types have been discussed extensively in Bendall et al. (2024). In short, trees growing on low fertility sandstone soils likely have higher resistance to disturbance; that is, they invest more resources into defences than growth, leading to greater population stability when exposed to disturbance. In contrast, forests growing on higher fertility soils likely experience enhanced growth at the expense of defence, leading to higher losses when exposed to disturbance. Collection and analysis of data for data-deficient substrates (e.g. metamorphic rocks) would allow for even broader inferences to be made across the Coastal IFOA region.

2.4.2 Differences across Coastal IFOA region

The WSU results suggested little difference in mortality, topkill, or seedling recruitment rates between northern and southern Coastal IFOA regions. The remote sensing assessment also did not detect any significant differences between Coastal IFOA regions. In contrast, the FCNSW results suggested higher overall mortality and topkill within the southern IFOA region[10]. One explanation for this could be that the WSU data was unable to detect the same pattern as it was far more spatially constrained (see Fig **1**) than the FCNSW dataset in that region and subsequently did not represent as many species or variations of regional forest types. In addition, the extent of extreme severity fire was generally much higher in the southern IFOA region potentially indicating that mortality and topkill were more widespread in the south than in the north.

The Bendall et al. (2024) study and this report are the only works to date that have attempted to quantify levels of mortality, topkill, and post-fire seedling recruitment in eucalypt-dominated forests across both northern and southern parts of the NSW coast and ranges following the Black Summer fires and preceding drought. The analyses presented in section **2.3** can be utilised to estimate structural change in forests and highlight areas of concern across the Coastal IFOA region, in terms of the proportion of geographic area affected by fire and the proportion of burnt area corresponding to the analysed forest types (Table **6**[11]). The four substrate/forest type combinations analysed in this report are highly relevant as they collectively represent 44% of State Forest tenure overall and 46% of State Forest tenure within the Upper North East CIFOA subregion, 28% within the Lower North East CIFOA subregion, 51% within the Southern CIFOA subregion and 61% within the Eden CIFOA subregion, with

[10] See 2.2.2 for definition of Coastal IFOA regions used in the analyses.
[11] An accompanying spatial data layer representing the categories in Table 6 has been provided to the NRC and FCNSW under a data license agreement.

the remainder represented by other forest types (e.g. rainforests, woodlands) and substrates (e.g. metamorphic, clastic).

In general, the area of State Forests affected by high severity fires for each substrate/forest types in Table **6** and Fig **9a,** or with delayed recovery signals by remote sensing (Fig **9b**) can be assumed to have experienced mortality and topkill patterns that resemble those presented in section **2.3,** and have a high and very high risk of delayed recovery (Table **7** and Fig **9**). There are, however, several caveats. Firstly, demographic shifts resulting in 'u'-shaped mortality and topkill response curves will be largely dependent on the proportion of trees within a stand that have prior basal injuries (Fig **4**,**5**,**7**,**8**). While the analyses in this report specifically relate to fire-caused basal injuries, harvesting-related injuries that resemble fire scars could potentially have similar physiological effects to that of fire scars (Bendall et al., 2023; Watson et al., 2020). As a precaution they should be treated similarly until more data becomes available. Secondly, the FCNSW data is limited in its use for assessing absolute levels of mortality or topkill, as it does not contain data for trees <10 cm or downed trees, where a large proportion of stand-level mortality and topkill is concentrated (Table **4**; Figs **4**,**5**; Appendix **1**). While most FCNSW sites were located in previously harvested areas with altered size-class distributions and very few trees exceeding >100 cm DBH (Figs **7**,**8**; Appendix **1**), it is likely that adjacent forest (e.g. Environmentally Significant Areas; ESAs) has not been recently impacted by large-scale harvesting. Hence, the estimates of mortality and topkill provided by WSU dataset, within areas burnt at high or extreme severity, are likely to be broadly applicable to large areas of the Coastal IFOA region as a whole, e.g. at least 55% of State Forest tenure is permanently excluded from harvesting activities and reserved as ESAs or wildlife habitat clumps (FCNSW, 2023; NSW EPA, 2018). Further, ~25% of the Coastal IFOA region is National Park tenure, where logging does not occur or has only occurred in the past. Therefore, to apply the precautionary principle, FESM mapping could be used to identify areas burnt at high or extreme severity that are assumed to be at a relatively high risk of delayed recovery.

2.4.3 Comparison between 2019/2020 fires and previous fires

The Black Summer fires occurred across a much larger area and saw more forest burned at high severity compared to any previous fire season (Boer et al., 2020; Collins et al., 2021). Thus, while tree- or stand-level mortality is comparable to previous events (see sections below), the scale of the fires means it is highly likely that many more trees died or were topkilled across the Coastal IFOA region than in any previous fire season. Another key difference between the

Black Summer fires and previous fire seasons is the manifestation of 'u'-shaped mortality and topkill response curves, which have previously only been reported in one other study in temperate resprouting eucalypt forests, which also experienced combined extreme drought and fire (Bennett et al., 2016). One hypothesis for the mechanism of the 'u'-shaped response is that a rare compound disturbance threshold may have been passed that was beyond what is usually experienced in these forests (see Bendall et al. 2024 for details), resulting in reduced resilience of large, old trees. These trees may have pre-existing basal damage and hollowing that predisposes them to hydraulic failure, stem collapse, and additional risk factors such as insect attack, which is known to increase during drought and can also impact tree hydraulic function (Lawson & Debuse, 2016; Seaton et al., 2015; White, 2014). The 'u'-shaped response curve detected in the WSU and FCNSW datasets generally aligns with predictions from independent modelling conducted by FCNSW (FCNSW, 2016), which was based on long-term average mortality rates (~50 year) and encompasses both fire and non-fire drivers of tree mortality such as insect attack, drought and windthrow, excluding harvesting-related drivers.

All past studies of fire-impacts to tree- and stand-level mortality/topkill relevant to the Coastal IFOA region have been limited in their geographic coverage and therefore limited in their ability to generalise across the broader landscape. For example, results from studies confined to the sandstone areas of the Sydney region are likely to underestimate mortality/topkill if generalised to forests on granite substrates. A key advantage of the WSU and FCNSW datasets is their ability to directly compare forest responses across a range of substrates and forest types, while allowing for confidence in their generalisability and interpretation of remote sensing products of delayed recovery and relative recovery rates, due to the wide geographic coverage of the data and large number of species the data represent (e.g. >100).

This section draws comparisons between the results presented in this report to estimates from other studies following the Black Summer fires and previous fire seasons. Some of the limitations of the studies considered have already been described in section **1.6**. Where possible, comparisons are made between forest types. For simplicity, we used four metrics: 1) overall mortality/topkill, 2) mortality/topkill of trees <20 cm (small trees), 3) mortality/topkill of trees >20 cm (mature trees), and 4) seedling recruitment. The estimates provided here are averaged across fire scar presence (WSU data only; e.g. assumes ~50% of trees within a stand have fire scars) and IFOA region and as a result are very general in nature. Mortality and topkill estimates are generally based on the WSU data as it more accurately represents the level of mortality/topkill. The FCNSW data generally indicates low levels of mortality and topkill in areas exposed to low severity fire, broadly comparable in magnitude to previous studies (see Table **3**). Thus, this section only compares the results for areas exposed to high severity fire. The lower bound of each set of values given represents sandstone substrates and the upper bound represents granite substrates.

2.4.3.1 Wet sclerophyll forests: mortality

Under high severity fire, WSF experienced overall mortality rates of 23-32%, small tree mortality rates of 29-52% and large tree mortality rates of 22-28%. Benyon and Lane (2013) did not divide trees up by size class but reported mortality of ~10-25% following the 2009 fires in comparable forests in Victoria. Bendall et al. (2022b) reported overall mortality of 24.7% in comparable sandstone forests in the Sydney region following fires in 2013. This suggests that the Black Summer fires resulted in comparable overall mortality rates to previous fires in WSF exposed to high severity fire.

2.4.3.1 Wet sclerophyll forests: topkill

Under high severity fire, WSF experienced overall topkill rates of 36-40%, small tree topkill rates of 77-81% and large tree topkill rates of 29-32%. Nolan et al. (2022) also reported high overall topkill rates (47%) in granite/WSF following the Black Summer fires. No other work has quantified topkill for comparable forest types in previous fire seasons.

2.4.3.1 Dry sclerophyll forests: mortality

Under high severity fire, DSF experienced overall mortality rates of 19-21%, small tree mortality rates of 24-33% and large tree mortality rates of 18%. Bendall et al. (2022b) reported overall mortality of 32.8% in comparable sandstone forests in the Sydney region following fires in 2013. Nolan et al. (2020b) reported overall mortality rates of ~10% in comparable sandstone forests in the Sydney region following fires in 2018. Fairman et al. (2019) reported overall mortality of ~10-30% (similar for both small and large trees) in comparable forests in Victoria burnt in 2013. Bennett et al. (2016) reported small tree mortality >80% and large tree mortality between 24-50% (including a ‘u’-shaped response curve) in comparable forests in Victoria burnt during 2009. These comparisons suggests that the Black Summer fires resulted in comparable overall mortality rates to previous fires in DSF exposed to high severity fire, and potentially lower mortality rates for small trees when compared to the Bennett et al. (2016) study.

2.4.3.1 *Dry sclerophyll forests: topkill*

Under high severity fire, DSF experienced overall topkill rates of 33-36%, small tree topkill rates of 69-76% and large tree topkill rates of 21-26%. Volkova et al. (2022) reported overall topkill of 58% and small tree topkill rates between 60-100%. Nolan et al. (2022) reported overall topkill rates of ~65%, primarily due to topkill of trees <20-30 cm DBH in comparable forests burnt during the Black Summer fires. Trouvé et al. (2021a) reported overall topkill up to ~75% at extreme fire intensity in comparable forests in Victoria burnt during 2009. Collins reported overall topkill of ~21% and small tree topkill rates between ~20-40% in comparable forests in Victoria burnt in 2013. Nolan et al (2020) reported overall topkill of 29% in comparable sandstone forests in the Sydney region following fires in 2018, primarily in stems <30cm DBH. This suggests that the Black Summer fires resulted in comparable overall topkill rates to previous fires in DSF exposed to high severity fire.

2.4.3.1 *Seedling recruitment*

Seedling recruitment was high across all forests following the Black Summer fires (median probability of 3600-8300 seedlings ha^{-1}). This is generally comparable to other studies where seedlings have been counted following high severity fire (e.g. 1000-10,000; Table **3**).

3 Recommendations

We have three key recommendations, outlined below.

1. **On-going *in situ* forest monitoring**

The analyses presented here provided a snapshot of forest responses to the 2019/2020 fires. We recommend on-going monitoring to detect whether further mortality continues through time, and to track post-fire recruitment. Vegetative recovery following defoliation events, such as severe fire and drought, are hypothesised to deplete carbohydrate reserves (Fairman et al., 2016; Reed & Hood, 2023; Smith et al., 2018). Repeated short-interval disturbances such as drought, followed by fire, followed by storms and/or waterlogging, and/or timber harvest may result in compounded increases in tree mortality that extend beyond the initial mortality event following fire (Fairman et al., 2016; Nolan et al., 2021a). Additionally, seedling recruitment and survival can be affected by post-fire climatic conditions and biotic factors (Nolan et al., 2021a).

We suggest the following areas be targeted for on-going monitoring:

existing plots where tree mortality/topkill and/or seedling density were measured following the Black Summer fires;

- areas identified in this report as having relatively large structural changes (i.e. wet sclerophyll forests on granite soils);
- areas identified by remote sensing as having delayed recovery, specifically targeting field validation of remote sensing signals.

2. **On-going monitoring of forest recovery by remote sensing and further research and development**

Ground-based surveys of post-fire forest mortality and recovery are resource intensive and are limited in the spatial extent they can cover. Remote sensing offers a good alternative for forest monitoring, however, there are known limitations due to the inability of passive satellite sensors to differentiate between recovery of the understory and recovery of the canopy. As such, remote sensing estimates are based on recovery of the amount and greenness of vegetation cover, but the forest structure and demography may remain substantially different to the pre-fire state (Karna et al., 2019). However, the remote sensing assessment presented here provides evidence of significant relationships between the delayed recovery index and elevated levels of mortality and topkill, indicating changes to structural elements, representing areas that will likely have limited or delayed post-fire recovery for an extended period. While this does not directly quantify tree mortality and recovery of the overstorey, it provides a robust indicator at the statewide scale. Recent advances in remote sensing, including high resolution imagery, LiDAR and GEDI, may improve upon current remote sensing approaches. Research is actively being undertaken and should continue to be invested in by government agencies and research institutions to improve remote sensing of post-fire recovery monitoring. For example, Dixon et al. (2023) developed a method for detecting tree mortality of individual trees from 3m resolution PlanetScope multispectral imagery and lidar. Such an approach has not yet been tested in Australian eucalypt forests.

3. Collection of additional information in NSW Forestry Corporation surveys

As outlined in Section 2.1, there are several key differences between the dataset collected by WSU and by FCNSW. We suggest FCNSW examine the feasibility of including the following additional metrics in their surveys:

- Number and diameter of downed trees (e.g. logs, stumps), and assessment of whether trees were dead pre-fire, based on visual assessment of the degree of charring and decomposition of stems.
- Assessment of number of stems dead or topkilled in the <10 cm DBH size-class. This could be done on a sub-plot and is important for accurately estimating mortality/topkill and assessing recruitment.
- Assessment of whether trees have fire scars or similar injuries related to harvesting.
- Assessment of post-fire recruitment.

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5 Appendix 1

5.1.1 Additional data summaries: WSU dataset

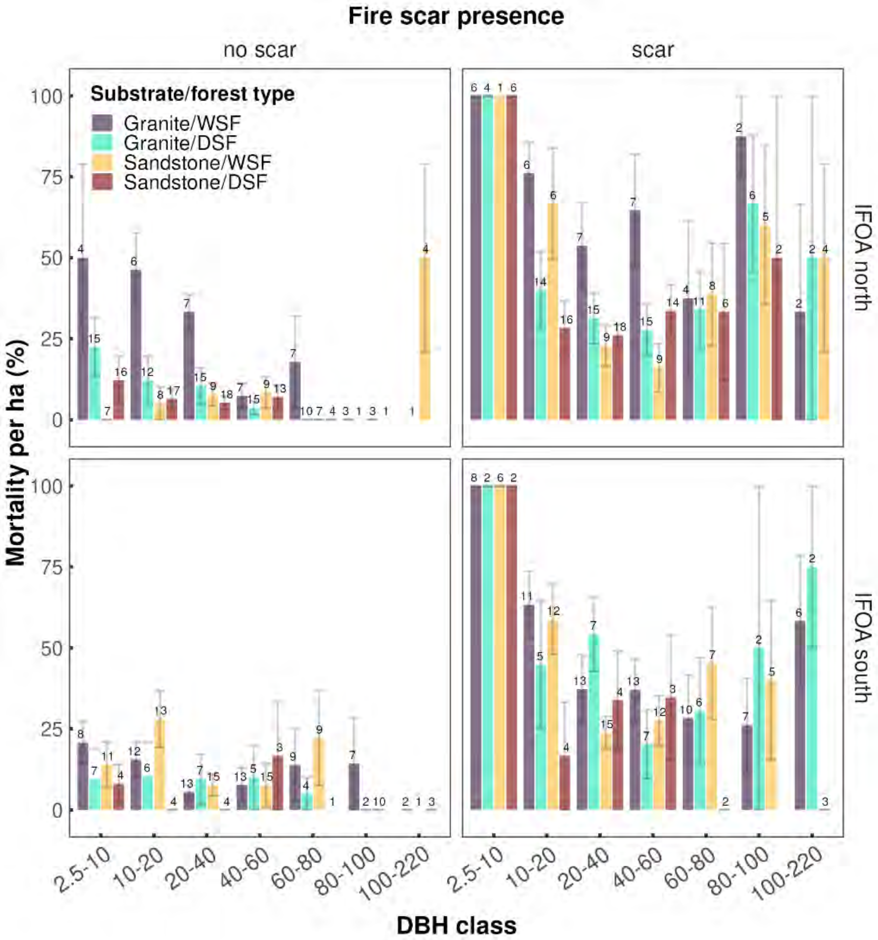

Figure 14. Site-level summary of raw data scaled to one hectare showing mean mortality per hectare in relation to DBH class (x-axis), substrate/forest type combination (coloured bars), fire scar presence (left/right panels) and Coastal IFOA region (top/bottom panels), for WSU sites within or adjacent to the Coastal IFOA region of New South Wales, following combined extreme drought and fire. Top of coloured bars represents mean value, error bars represent standard error of the mean, values above each bar represent the number of sites/samples per bar. WSF = wet sclerophyll forest; DSF = dry sclerophyll forest.

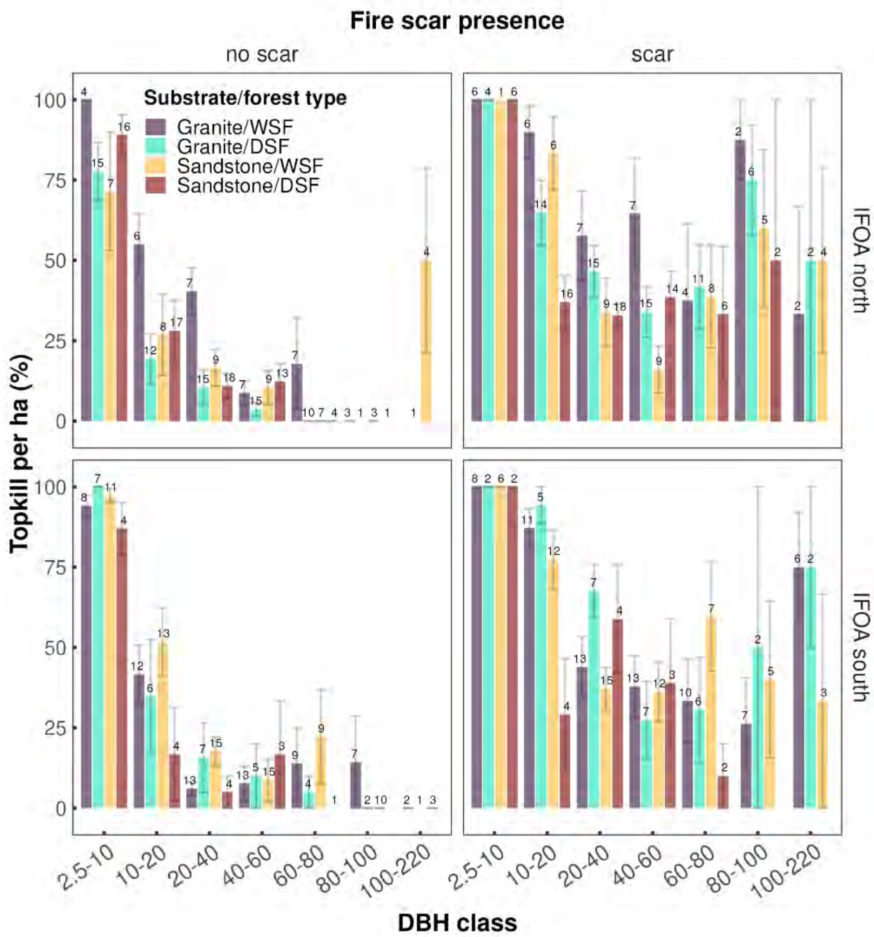

Figure 15. Site-level summary of raw data scaled to one hectare showing mean topkill per hectare in relation to DBH class (x-axis), substrate/forest type combination (coloured bars), fire scar presence (left/right panels) and Coastal IFOA region (top/bottom panels), for WSU sites within or adjacent to the Coastal IFOA region of New South Wales, following combined extreme drought and fire. Top of coloured bars represents mean value, error bars represent standard error of the mean, values above each bar represent the number of sites/samples per bar. WSF = wet sclerophyll forest; DSF = dry sclerophyll forest.

5.1.2 Additional data summaries: FCNSW dataset

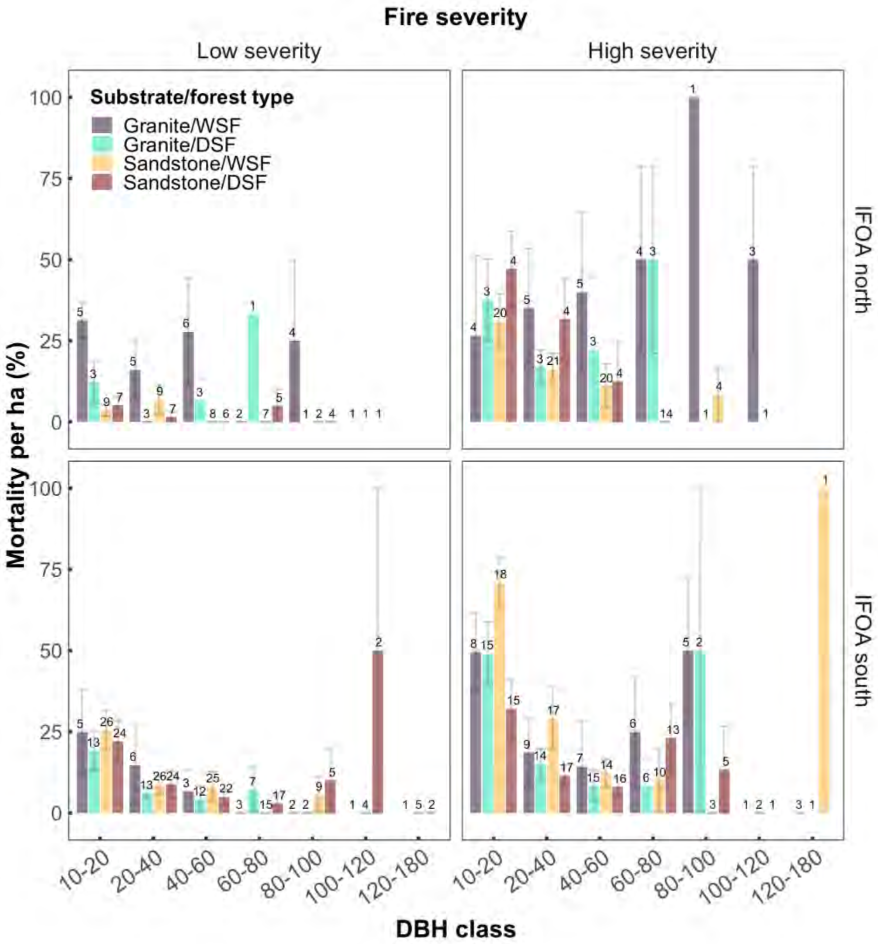

Figure 16. Site-level summary of raw data scaled to one hectare showing mean mortality per hectare in relation to DBH class (x-axis), substrate/forest type combination (coloured bars), fire severity (panels left to right) and Coastal IFOA region (top/bottom panels), for FCNSW sites within the Coastal IFOA region of New South Wales, following combined drought and fire. Top of coloured bars represents mean value, error bars represent standard error of the mean, values above each bar represent the number of sites/samples per bar. WSF = wet sclerophyll forest; DSF = dry sclerophyll forest.

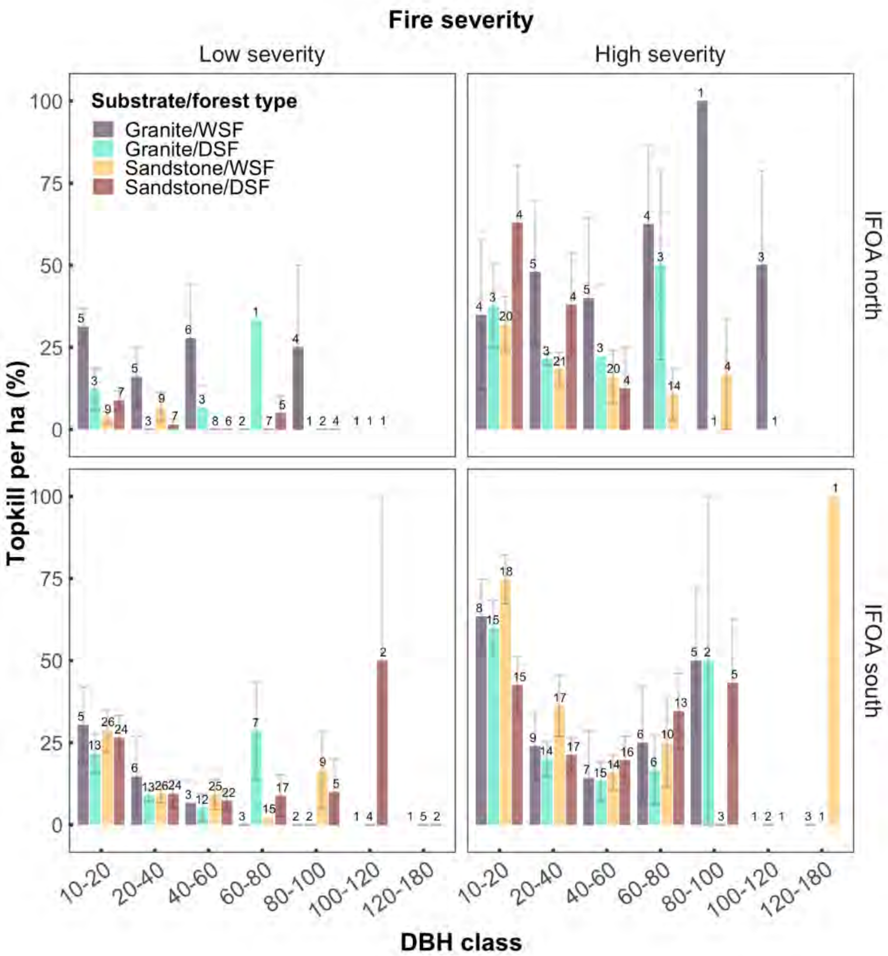

Figure 17. Site-level summary of raw data scaled to one hectare showing mean topkill per hectare in relation to DBH class (x-axis), substrate/forest type combination (coloured bars), fire severity (panels left to right) and Coastal IFOA region (top/bottom panels), for FCNSW sites within the Coastal IFOA region of New South Wales, following combined drought and fire. Top of coloured bars represents mean value, error bars represent standard error of the mean, values above each bar represent the number of sites/samples per bar. WSF = wet sclerophyll forest; DSF = dry sclerophyll forest.

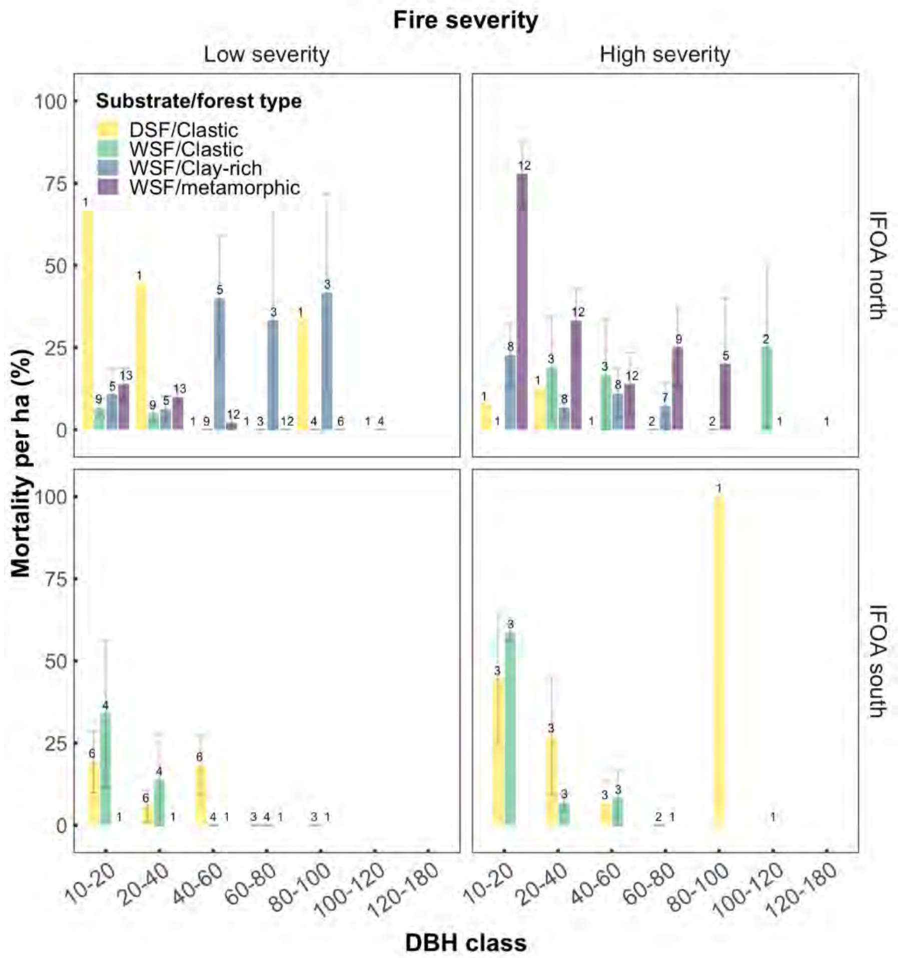

Figure 18. Site-level summary of raw data scaled to one hectare showing mean mortality per hectare in relation to DBH class (x-axis), substrate/forest type combination (coloured bars), fire severity (panels left to right) and Coastal IFOA region (top/bottom panels), for FCNSW sites within the Coastal IFOA region of New South Wales, following combined drought and fire. Top of coloured bars represents mean value, error bars represent standard error of the mean, values above each bar represent the number of sites/samples per bar. WSF = wet sclerophyll forest; DSF = dry sclerophyll forest.

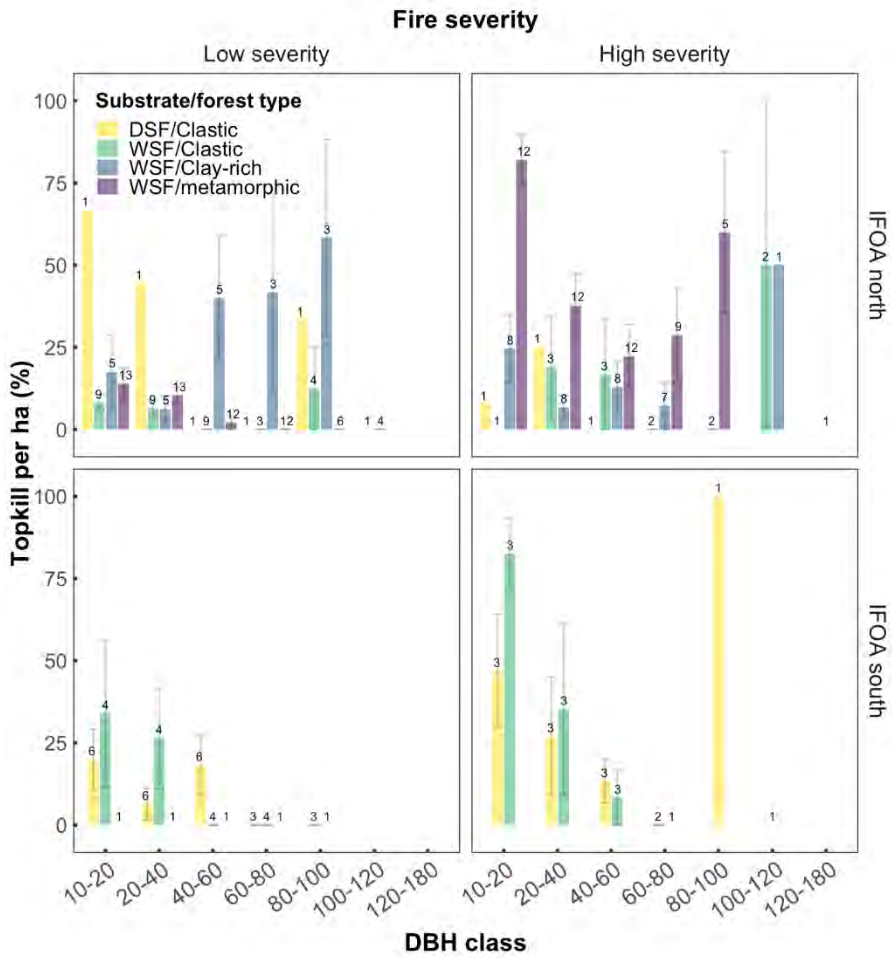

Figure 19. Site-level summary of raw data scaled to one hectare showing mean topkill per hectare in relation to DBH class (x-axis), substrate/forest type combination (coloured bars), fire severity (panels left to right) and Coastal IFOA region (top/bottom panels), for FCNSW sites within the Coastal IFOA region of New South Wales, following combined drought and fire. Top of coloured bars represents mean value, error bars represent standard error of the mean, values above each bar represent the number of sites/samples per bar. WSF = wet sclerophyll forest; DSF = dry sclerophyll forest.

5.1.3 *Additional data summaries: predicted effect of climate moisture index*

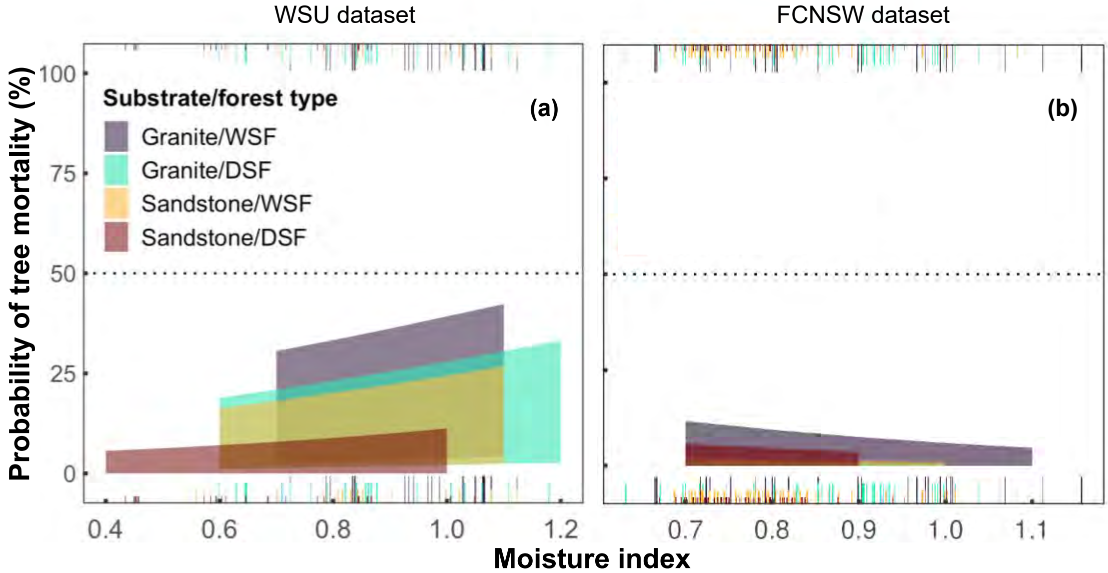

Figure 20. The effect of a moisture index (x-axis) and substrate/forest type combination (coloured ribbons, left/right panels) on the probability of mortality for trees in forests of southeastern Australia exposed drought and fire for two independent datasets (WSU = panel **a**; FCNSW = panel **b**). Coloured ribbons represent 50% credible intervals. The observed range of moisture index varied with substrate/forest type. Rug plots (narrow coloured vertical bars) at top (dead trees) and bottom (live trees) of plot window represent the observation density across the complete range of values used to inform the model. WSF = wet sclerophyll forest; DSF = dry sclerophyll forest.

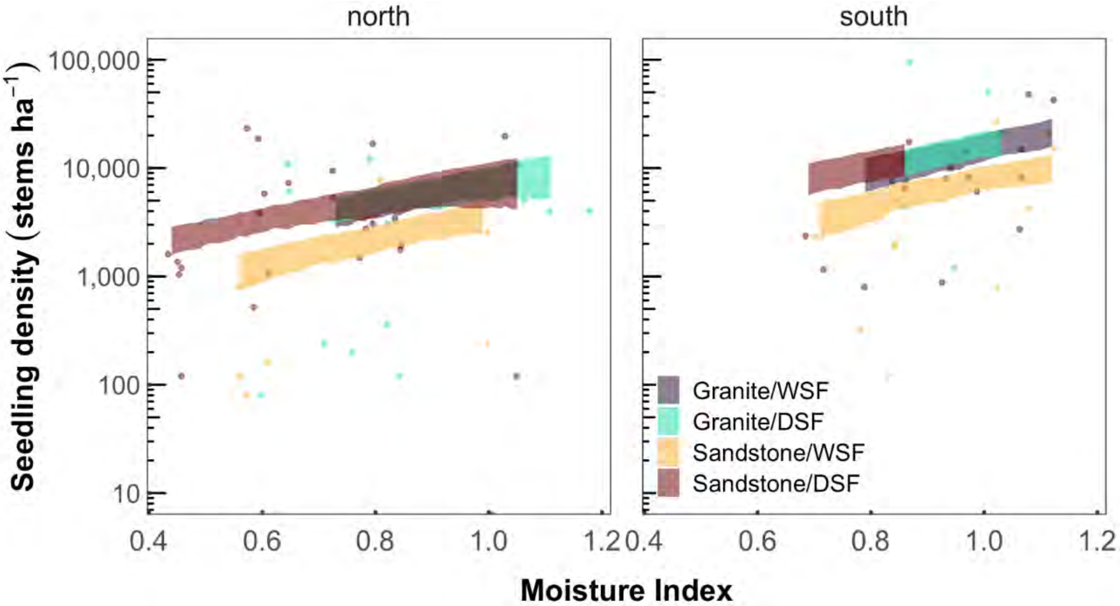

Figure 21. The effect of substrate and forest type (coloured ribbons), a climate moisture index (x-axis) and Coastal IFOA region (panels left/right) on the predicted density of post-fire seedlings per hectare in forests within and adjacent to the Coastal IFOA region of New South Wales, following combined drought and fire. Coloured ribbons represent 50% credible intervals. Coloured points represent raw data (site-level). Y-axis is log scale so that extremely high observations of seedling counts can be displayed. WSF = wet sclerophyll forest; DSF = dry sclerophyll forest.

Table 10. Full list of candidate generalised linear models explaining the variation in the remote sensing recovery index for the WSU dataset, with the Akaike Information Criterion corrected for small sample sizes (AICc), the change in AICc (dAICc) and overall rank of each model.

Rank	AICc	dAICc	Model
1	240.64	0.00	glm(sum recovery index ~ aridity)
2	242.14	1.50	glm(sum recovery index ~ *Acacia*)
3	243.91	3.27	glm(sum recovery index ~ mean canopy cover)
4	243.96	3.32	glm(sum recovery index ~ sum % basal area cover**Acacia*)
5	244.32	3.68	glm(sum recovery index ~ Substrate/Forest Type; Granite/WSF)
6	245.51	4.87	glm(sum recovery index ~ severity)
7	245.85	5.49	glm(sum recovery index ~ vegetation type)
8	246.57	6.22	glm(sum recovery index ~ sum basal area cover %*severity)
9	248.68	8.33	glm(sum recovery index ~ sum basal area cover % * vegetation type)
10	249.23	8.87	glm(sum recovery index ~ live basal area)
11	249.23	8.87	glm(sum recovery index ~ sum basal area cover %)
12	250.81	10.46	glm(sum recovery index ~ sum basal area cover % * substrate/forest type)
13	251.00	10.65	glm(sum recovery index ~ substrate type)
14	251.24	10.88	glm(sum recovery index ~ live basal area canopy resprouting)
15	251.54	11.18	glm(sum recovery index ~ basal area topkill)
16	251.55	11.19	glm(sum recovery index ~ basal area dead)
17	251.80	11.44	glm(sum recovery index ~ basal area basal resprouting %)
18	252.83	12.48	glm(sum recovery index ~ live basal area basal resprouting)
19	253.64	13.29	glm(sum recovery index ~ live basal area stem resprouting)
20	254.00	13.64	glm(sum recovery index ~ presence of rock)
21	254.64	14.28	glm(sum recovery index ~ basal area dead %)
22	254.64	14.28	glm(sum recovery index ~ sum basal area cover)
23	254.68	14.33	glm(sum recovery index ~ sum basal area cover % * presence of rock)
24	255.08	14.73	glm(sum recovery index ~ basal area topkill %)
25	255.54	15.19	glm(sum recovery index ~ basal area canopy resprouting)
26	255.66	15.30	glm(sum recovery index ~ basal area stem resprouting)

Table 11. Full list of candidate generalised linear models explaining the variation in the remote sensing recovery index for the FCNSW dataset, with the Akaike Information Criterion corrected for small sample sizes (AICc), the change in AICc (dAICc) and overall rank of each model.

Rank	AICc	dAICc	Model
1	491.75	0.00	glm(sum recovery index ~ severity)
2	493.88	2.13	glm(sum recovery index ~ sum % basal area cover* severity)
3	540.36	48.60	glm(sum recovery index ~ basal area topkill %)
4	540.43	48.68	glm(sum recovery index ~ basal area epicormic resprouting %)
5	543.53	51.78	glm(sum recovery index ~ basal area basal resprouting %)
6	544.40	52.65	glm(sum recovery index ~ basal area dead %)
7	544.40	52.65	glm(sum recovery index ~ sum % basal area cover)
8	546.77	55.01	glm(sum recovery index ~ sum % basal area cover*forest type)
9	546.81	55.06	glm(sum recovery index ~ sum % basal area cover*substrate type)
10	547.07	55.32	glm(sum recovery index ~ sum % basal area cover*ifoa)
11	547.24	55.49	glm(sum recovery index ~ substrate/forest type)
12	547.74	55.99	glm(sum recovery index ~ aridity)
13	547.77	56.02	glm(sum recovery index ~ forest type)
14	547.84	56.09	glm(sum recovery index ~ sum % basal area cover*substrate/forest type)
15	549.08	57.32	glm(sum recovery index ~ substrate type)
16	549.26	57.50	glm(sum recovery index ~ ifoa)

6 Appendix 2 – Basal injuries

Figure 22. Images showing basal injuries due to fire. Images: Eli. R. Bendall.