ASSESSING BURNT AREA SEVERITY IN THE CRITICAL ZONE MONITORING SITE OF A PHILIPPINE NATURAL PARK

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ABSTRACT

Wildfires are a global phenomenon shaping ecosystems and influencing biodiversity. However, knowledge gaps remain regarding fire severity and ecological recovery in tropical protected areas, particularly those dominated by invasive and pioneer grass species. Hence, this regional case study aimed to create the first burn severity map of the Core Zone Monitoring area in Mts. Iglit-Baco Natural Park, Philippines, using Landsat satellite imagery from 2020-2021 to evaluate fire impacts on grassland dynamics. The analysis employed pre- and post-fire satellite data and burn indices, revealing that 44.39 % of the landscape remained unburned, while the rest experienced varying burn severities. Remarkably, 45.64 % of the burned areas showed enhanced regrowth within 11 months, demonstrating significant recovery potential of the area. These findings highlight the interplay between fire disturbances and ecological resilience with a geomatic approach that provides a replicable framework for fire severity assessments and offers valuable insights for conservation planning globally.

Keywords; geospatial analysis, landscape management mapping, normalized burn ratio, plant regrowth, pyroecology

INTRODUCTION

Fire is a significant ecological disturbance worldwide, affecting biodiversity, ecosystem processes, and human livelihoods. While naturally occurring wildfires have historically shaped ecosystems, human-induced fires, such as agricultural burning and land management practices, have drastically altered fire regimes (Andersen *et al*., 2005; Kelly *et al*., 2020; Kirchhoff *et al*., 2021). These changes, compounded by climate change-induced droughts, have led to more frequent and intense fires in many regions (Bowman *et al*., 2011; Littell *et al*., 2016). Concurrently, the growing recognition of fire's ecological role has shifted research paradigms towards understanding fire as a critical process supporting ecosystem structure and function (He *et al*., 2019; Schmerbeck & Seeland, 2007). Studies increasingly emphasize fire's potential to enhance species richness and abundance, as well as to facilitate habitat restoration, especially in fire-prone landscapes (Bohlman *et al*., 2016; Jung *et al*., 2010; Kovář *et al*., 2011). However, significant knowledge gaps persist regarding fire dynamics, burn severity, and ecosystem recovery in tropical protected areas.

Grassland ecosystems, often dominated by pioneer and invasive species, are particularly vulnerable to fire disturbances. Research shows that such species tend to form dense, monotypic stands, reducing biodiversity and altering habitat structure (Pandey *et al*., 2015; Petermann & Buzhdygan, 2021; Dhakal *et al*., 2024). These dynamics are evident in Mts. Iglit-Baco Natural Park (MIBNP), Philippines, where grasslands have expanded due to anthropogenic disturbances, including fire regimes and cattle grazing (Gonzalez *et al*., 1999). Historical records suggest these grasslands replaced forests, creating novel ecosystems characterized by invasive dominance and reduced native biodiversity (Merritt, 1908; Gonzalez *et al*., 1998). Despite this understanding, there remains a lack of detailed, spatially explicit data on fire extent and severity in this region, leaving critical gaps in fire ecology and conservation planning.

This study aims to address these gaps by developing the first burn severity map of the Core Zone Monitoring area in the park, using Landsat satellite imagery and geospatial analyses to assess the impact of fire disturbances during the 2020-2021 period. Specifically, the study hypothesizes that: (1) the current fire regime significantly influences the spatial distribution of grasslands in MIBNP, (2) burn severity varies across the landscape, with implications for ecological recovery, and (3) fire management practices can be optimized to promote better conservation goals for MIBNP. By bridging knowledge gaps on fire dynamics in a sample area of tropical protected areas, this research provides now critical insights for putative policy-driven conservation and a replicable framework for fire monitoring not only at a regional scale but can be applied globally.

METHODOLOGY

Study Site

Mindoro, being the seventh-largest island in the Philippines, is bounded by two provinces: Oriental Mindoro, and Occidental Mindoro (Fig. 1). Due to its distinctive vegetation and wildlife, specific areas within the province are protected under the National Integrated Protected Areas System (NIPAS) Act of 1992 as a means to safeguard the unique biological species thriving in their respective habitats (Carreon-Lagoc, 1994). Mounts Iglit-Baco Natural Park (MIBNP) (N12°54', E121°13') is one such area, located at the central spine of the island. It was designated as a protected area (PA) in 1970 as declared under Republic Act No. 6148. Moreover, it is an ancestral domain to the Tau-buid, an indigenous tribe, and an ASEAN Heritage Park. It stretches through four municipalities in Occidental Mindoro, namely Sablayan, Calintaan, Rizal, and San Jose, and five municipalities in Oriental Mindoro, those being Pinamalayan, Gloria, Bansud, Bungabong, and Mansalay. Critically endangered and found only on the island of Mindoro, the tamaraw (*Bubalus mindorensis* Heude, 1888) is a unique wild buffalo species (Boyles *et al*., 2016) which is also known to thrive in MIBNP. Subsequently, the Core Zone Monitoring (CZM) area was established within MIBNP primarily because it is an area with limited disturbance due to continual and sustained monitoring from forest rangers as deployed by the Department of Environment and Natural Resources – Tamaraw Conservation Program (DENR-TCP). Moreover, the no-hunting agreement between the Tau-buid communities residing near the area and TCP rangers has reduced human pressure, thus creating a more conducive environment not only for tamaraws but for various wildlife as well.

Fig. 1: Map of Mindoro showing the range of Mts. Iglit-Baco Natural Park (dark green) and the Core Zone Monitoring area within (red) the protected area.

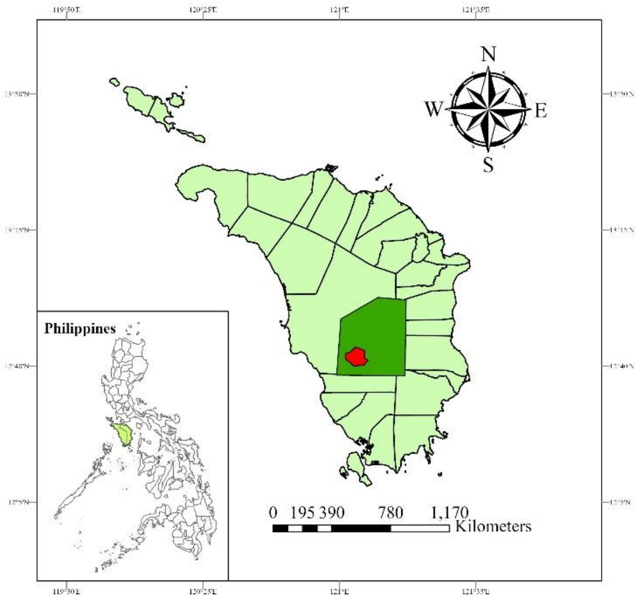


Table 1: Landsat™ 8 OLI/ TIRS C1 Level Imagery details

Satellite and Sensor	Acquisition Date	Land Cloud Cover	Scene Cloud Cover	Landsat Product Identifier
Landsat 8 OLI_TRS	02/08/2020	19.23%	13.52%	LC08_L1TP_116051_20200208_20200823_02_T1
Landsat 8 OLI_TRS	03/27/2020	21.42%	15.83%	LC08_L1TP_116051_20200327_20200822_02_T1
Landsat 8 OLI_TRS	02/26/2021	7.12%	2.52%	LC08_L1TP_116051_20210226_20210304_02_T1

Calculation of Burnt Area Severity

As a means to assess the burned area severity in the study site, remotely sensed imageries were obtained from the Landsat Collection 1-Level 1. This included Landsat 8 Operational Land Imager (OLI)/Thermal Infrared Sensor (TIRS) C1 Level 1 data, which were accessed through the United States Geological Survey's (USGS) EarthExplorer image library.

Several important factors were considered when selecting satellite imagery to guarantee the highest level of accuracy and dependability. These requirements included the need for minimum cloud cover, the fact that both scene and cloud must be less than 25 % (see Table 1), and the dates to account for pre-fire and post-fire. The dates between March 1 and March 21, 2020, were chosen as a temporal filter, as this is considered the timeframe from which annual prescribed burning occurs within MIBNP; thus, March 22 until the last week of April 2020 is considered to be the optimal dates to acquire post-fire satellite images. On the other hand, the temporal filter for post-fire ran between January and the last week of February 2021, considering that these are the months where vegetation regrowth can be observed.

Equation (1) by García & Caselles (1991) presents the Normalized Burned Ratio index (NBR), which was designed to aid in the quantification of areas affected by fire while assessing the damage to those areas at the same time. The NBR is employed to establish and classify the outermost limits of burned areas, utilizing reflectance data from Landsat OLI sensor bands 5 (near infrared, NIR) and 7 (short-wave infrared, SWIR). To quantify fire-induced changes in the landscape, Differenced Normalized Burn Ratios (dNBRs) were employed. This index, derived from Equation 2 (Key & Benson, 2006; dos Santos *et al.*, 2020), provides a scaled measure of the environmental changes caused by fire within a specific area.

$$NBR = \frac{(\text{NIR}-\text{SWIR})}{(\text{NIR}+\text{SWIR})} \quad (1)$$

$$dNBRs = (\text{NBRpre} - \text{NBRpost}) \quad (2)$$

$$dNBR = (\text{NBRpost} - \text{NBRregrowth}) \quad (3)$$

The dNBR (Equation 3) was calculated by extracting the difference of bitemporal values from the preprocessed NBR images derived from bands 5 and 7 (Adagbasa *et al.*, 2018). A study by Teobaldo & Baptista (2016) centered on the measurement of fire severity and regrowth capacity was conducted using the dNBR, which puts emphasis on the presence of fire disturbance by means of enhancing the changes between NBR scenes. This was performed by taking the postfire images and subtracting them from other dates succeeding the fire event; this is referred to as the regrowth (dos Santos *et al.*, 2020).

The severity and regrowth classification used for this study were adopted from Key & Benson's (2020) severity levels. In this study, seven classifications were used to categorize the pixel values of the maps generated: namely enhanced regrowth (high), enhanced regrowth (low), unburned, low severity, low to moderate severity, moderate to high severity, and high severity (Table 2). The intervals of dNBR values potentially varies amongst paired scenarios. There are also instances that values may potentially be greater than +1.300 or less than -0.500. If this is the case, the script will not process the values into a distinct classification. Rather, the map will not include these values attributed to inadequate recording, such as cloud cover or other circumstances unrelated to actual variations in land cover. Additionally, this classification method means that “unburned” classification in the post-fire scenario does not necessarily indicate areas that have never been burnt in the past; rather, it may include areas that have undergone regrowth after previous burning incidences.

Table 2: dNBR range value and their corresponding severity levels

Designated Color	Severity Level	dNBR Range Values
	Enhanced Regrowth (High)	-0.500 – -0.251
	Enhanced Regrowth (Low)	-0.250 – -0.101
	Unburned	-0.100 – 0.099
	Low Severity	0.100 – 0.269
	Low – Moderate Severity	0.270 – 0.439
	Moderate – High Severity	0.440 – 0.659
	High Severity	0.660 – 1.3

RESULTS

Burnt Area Severity

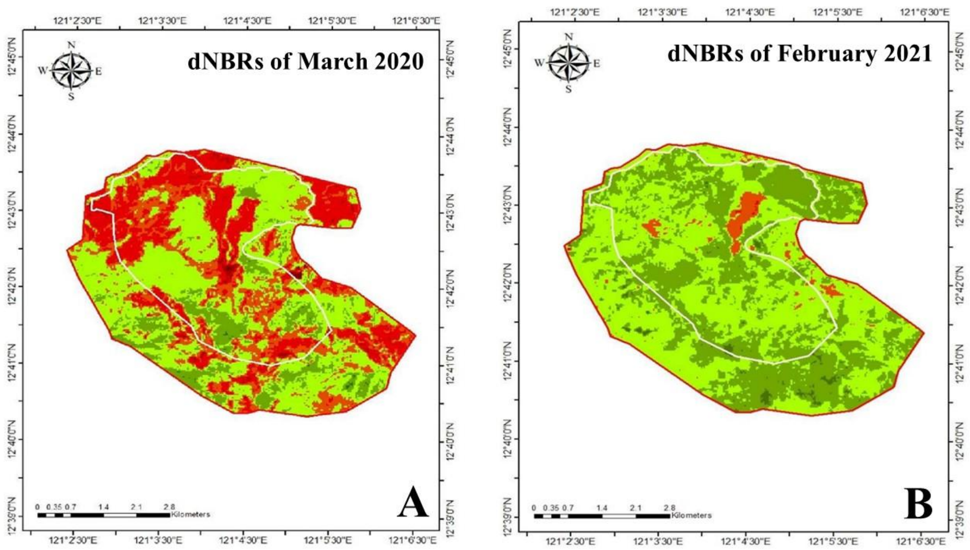
Table 3 exhibits the measure of central tendency from the post-fire and regrowth dataset. Data from the postfire map indicates the average index value is 0.104, which falls under the low severity category, whereas the data from the regrowth map signify that majority of CZM is unburnt with an average index value of -0.077, which falls under the low enhanced regrowth category.

In quantifying the spatial extent of various burnt severity levels, it was calculated that merely 0.32 % of the CZM exhibited high levels of enhanced regrowth, covering approximately 6.830 ha. In comparison, 6.43 % displayed lower levels of enhanced regrowth, totaling 137.688 ha. A substantial portion of the landscape, accounting for 950.410 ha or 44.39 %, remained unburned, suggesting that controlled burning practices employing strategic burning techniques were employed. Additionally, areas experiencing varying degrees of burn severity were identified, with 22.31 % classified under low and low to moderate (90.152 ha) and moderate to high (4.211 ha) severity burns. Furthermore, while high-severity burns occurred on a smaller scale, comprising approximately half a hectare (0.535 ha) or 0.025 %, their ecological significance warrants careful consideration. Eleven months post-fire and one month preceding the subsequent burning event, shifts in the spatial extent of burning were observed. High levels of regrowth expanded across 90.133 ha or 4.21 % of the area, with lower levels covering 886.838 ha or 41.43 %. Substantially, 54.16 % (1,158.904 ha) of the landscape remained unburned, facilitating the natural recovery. 68.190 ha or 3.19 % exhibited low severity regrowth, while 0.342 ha or 0.016 % displayed low to moderate severity regeneration. On the other hand, categories denoting "Moderate to High Severity" and "High Severity" showed complete recovery, with no area falling under these severity levels. This directly conveys the positive outlook suggested by the results.

Table 3: Central Tendency of Post-Fire Burnt Severity (March 2020) and Regrowth Burnt Severity (February 2021) dNBR

	Minimum	Maximum	Mean	Standard Deviation
Post-Fire	-0.303	1.029	0.104	0.171
Regrowth	-0.427	0.376	-0.077	0.088

Fig. 2: The map showing the Burnt Area Severity post-fire (A) and eleven months after the fire burning (B) of the Core Zone Monitoring area.



DISCUSSION

Comparisons of post-fire and regrowth burnt area severity for the past decades have increasingly relied on the dNBR derived from Landsat satellite data, as it is one of the most commonly used indices for quantifying burned areas (Parker *et al.*, 2015). This is due to the capacity of the index to provide spectral separation between NIR and SWIR wavelengths, where NIR reflectance increases with healthy vegetation, while SWIR reflectance rises in areas with exposed soil or fire-scarred vegetation (Schepers *et al*., 2014; Soverel *et al*., 2010). In relation to this, two spatial distribution maps exhibiting burn severity using dNBR were generated using ArcGIS as seen in Figure 2A and 2B. These maps provide baseline information on the quantified extent of post-fire burned areas and regrowth in the landscape. The calculation for dNBR was derived by subtracting the pre-fire image from the post-fire image. In the case of this study, the dNBR of February 2020, a month prior to the prescribed burning practice, was subtracted from the NBR of March 2020, the period from which the authorities conduct annual fire-based monitoring strategy for tamaraws. Higher values of > +0.100 entail a more significant modification in terms of spectral radiance, correlated with more severe burns that result in increased barren land from loss of vegetation cover and possibly higher soil temperatures (Miller & Thode, 2007; Murphy & Reynolds, 2008; Soverel *et al*., 2010). Contrary to this, regions in the map with low dNBR values (< -0.500) have undergone less severe burns or are undergoing regrowth subsequent to the fire event. Areas with low dNBR values are characterized to have the potential for plant regeneration (Veraverbeke *et al*., 2012; Santos *et al*., 2021).

Furthermore, in analyzing the dNBR values derived from the two burned area maps, a clear distinction in fire severity is evident. The March 2020 map, captured closer to the fire event, exhibits significantly higher dNBR values compared to the regrowth map (Fig. 2B). This observation aligns with the expectation that dNBR values would be highest immediately following a fire due to the presence of burned materials, as indicated by the presence of ashes in the ground surface, and minimal vegetation recovery (Vergara *et al*., 2024). The regrowth map, taken 11 months after the fire disturbance, shows a degree of vegetation recovery, reflected in the lower dNBR values. However, it is important to note that these values likely do not represent a complete restoration of the pre-fire ecosystem.

Clearly, the deliberate utilization of fire, usually through prescribed burning and controlled wildfire, is seen as a management strategy that can be mutually beneficial to the environment, the people, and the animals (Fernandes *et al*., 2013; Eales *et al*., 2018; Valkó & Deák, 2021; Tomchencko *et al*., 2023). This is because such strategies are deduced to reduce severe damage caused by natural wildfire while furthering the progress of native biota conservation. However, the adoption of fire-prescribed burning practices worldwide is not one and the same, considering that socio-political factors are at play (Calkin *et al*., 2015; Buizer & Kurz, 2016; Weir *et al*., 2020). In fire-prone regions worldwide, various approaches are made in terms of prescribed fire management. For instance, in savanna ecosystems in Africa, grazing, fire, and selective tree-cutting are widely used to increase diversity (Savadogo *et al*., 2008). Whereas in North America, the historical fire regime concept is quite prevalent as it serves as a guide for biodiversity conservation and restoration, considering that the approach's primary objective is the replication of historical fire patterns caused by natural forces such as lightning (Freeman *et al*., 2017). Contrary to this, an approach centered on contemporary management, which emphasizes hazard reduction, is used in Australia (Mccaw *et al*., 2005; Penman *et al*., 2012). These approaches have in common that they aim to address issues and support fire monitoring practices that enable the implementation of intensive and holistic research. Furthermore, these approaches develop precise and distinguishable fire management policies that work best in their respective regions. It also involves strategic planning to include traditional Aboriginal burning in modern prescribed fire practices. The continuous refinement of these strategies is greatly needed given that the ecological impacts of various burning practices remain dynamic, especially considering that these ecological processes depend heavily on the ever-changing environment.

Whether deliberate or unintentional, human interference with regards to landscape fires has exerted a profound ecological influence on the conservation of flora and fauna communities, the provision of ecosystem services, and the likelihood of fires that may potentially cause economic disruption (Bowman *et al*., 2011). However, there is still much discussion and dispute amongst fire managers, ecologists, and conservation biologists in relation to achieving both environmentally and economically sustainable management of fire practices. This continuous discourse results from the myriad, sometimes incompatible goals for managing fires that are influenced by varying social and political standards of different countries worldwide, as well as the intrinsic complexity and unpredictability of fire ecology, itself (Jones & Tingley, 2022). In relation to this, the notion that pyrodiversity gives rise to biodiversity, which postulates that modifying the spatial and temporal characteristics of fire regimes can promote biodiversity using human intervention, serves as an example of such heated discussions (Parr & Andersen, 2006; Bowman *et al*., 2011).

Moreover, it is worth noting that the implementation of prescribed fire-burning practices necessitates a cautious and evidence-based approach. While prescribed fires can be valuable land management tools, their utilization should not be undertaken lightly. Fire regimes are immensely dynamic; it involves a continuous evolution according to general socio-economic patterns and weather conditions which affect fire seasonality and regimes (Galizia *et al*., 2023). This continuous evolution is further intensified by the effects of global warming, making the process of plotting the fire seasonality harder to accomplish due to an increase in its unpredictability. With this in mind, if prescribed fires were to be used in a patch of land with a high occurrence of small-scale fire, the effects would do the ecosystem more harm than good. For one, dead trees that arise from fires – when used as nesting sites by young birds and mammals – radiate extreme temperatures that greatly impact their incubation and survival (Wiebe, 2001). Furthermore, forest fires lead to a tremendous loss of fauna and flora as well as a negative effect on the diversity of the land (Nasi *et al*., 2002; Nitschke & Innes, 2007). Wildlife habitat becomes simple and poor, and food sources become scarce, forcing wildlife to re-locate and find another habitat with adequate vegetation, a case seemingly already transparent in the protected areas of Mindoro, hence the establishment of CZM.

CONCLUSION

The map generated for this study provides a well-developed grasp of fire and landscape ecology. As understanding spatiotemporal dynamics of the regrowth pattern among plant communities in a frequently disturbed protected landscape sheds light on the role of effective and sustainable land management strategies, this invites further research as there is much more to be tackled. Knowledge gaps still remain, for instance, in a spatial context that in this study, with respect to the Indigenous sovereignty and cultural protocols implemented by Indigenous communities, particular zones within the study area were restricted because of specific locales within MIBNP that are known to be sacred to the IPs. Considering that the study has laid the foundational groundwork for the investigation of the species composition in the northern, central, and southern regions of the CZM, future research that will establish vegetation plots on the eastern and western sectors of the CZM may be beneficial to fully understand the entirety of the landscape by addressing questions centered on the potentiality of vegetation zonation caused by the fire disturbance. Additionally, the methodologies outlined here may be applied globally in similar protected zones to inform management schemes when conducting purposive fire burning practices.

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CONFLICT OF INTEREST

The authors state that they have no conflicts of interest.

REFERENCES

Adagbasa, G. E., Adelabu, S. A., & Okello, T. W. (2018). Spatio-temporal assessment of fire severity in a protected and mountainous ecosystem. In *IGARSS 2018-2018 IEEE International Geoscience and Remote Sensing Symposium* (pp. 6572-6575). IEEE. https://doi.org/10.1109/IGARSS.2018.8518268

Andersen, A. N., Cook, G. D., Corbett, L. K., Douglas, M. M., Eager, R. W., Russell-Smith, J., Setterfield, S.A., Williams, R.J., & Woinarski, J. C. (2005). Fire frequency and biodiversity conservation in Australian tropical savannas: implications from the Kapalga fire experiment. *Austral ecology, 30*(2), 155-167. https://doi.org/10.1111/j.1442-9993.2005.01441.x

Bohlman, G. N., North, M., & Safford, H. D. (2016). Shrub removal in reforested post-fire areas increases native plant species richness. *Forest Ecology and Management, 374*, 195-210. https://doi.org/10.1016/j.foreco.2016.05.008

Bowman, D. M., Balch, J., Artaxo, P., Bond, W. J., Cochrane, M. A., D'antonio, C. M., Defries, R., Johnston, F.H., Keeley, J.E., Krawchuk, M.A., Kull, C.A., Mack, M., Moritz, M.A., Pyne, S., Roos, C.I., Scott, A.C., Sodhi, N.S., & Swetnam, T. W. (2011). The human dimension of fire regimes on Earth. *Journal of biogeography, 38*(12), 2223- 2236. https://doi.org/10.1111/j.1365-2699.2011.02595.x

Boyles, R., Schutz, E., & de Leon, J. (2016). *Bubalus mindorensis*. The IUCN Red List of Threatened Species, 2016, e. T3127A50737640.

Buizer, M., & Kurz, T. (2016). Too hot to handle: Depoliticisation and the discourse of ecological modernisation in fire management debates. *Geoforum*, *68*, 48-56. https://doi.org/10.1016/j.geoforum.2015.11.011

Calkin, D. E., Thompson, M. P., & Finney, M. A. (2015). Negative consequences of positive feedbacks in US wildfire management. *Forest Ecosystems, 2*, 1-10. https://doi.org/10.1186 /s40663- 015-0033-8

Carreon-Lagoc, J. (1994). The NIPAS Act of 1992. *Aqua Farm News*, *12*(3), 8-9.

Dhakal, S., Shrestha, B. B., Sharma, K. P., Paudel, S., & Siwakoti, M. (2024). Grasslands are more vulnerable to plant invasions than forests in south-central Nepal. *Environmental Challenges*, *15*, 100929. https://doi.org/10.1016/j.envc.2024.100929

Eales, J., Haddaway, N. R., Bernes, C., Cooke, S. J., Jonsson, B. G., Kouki, J., Petrokofsky, G., & Taylor, J. J. (2018). What is the effect of prescribed burning in temperate and boreal forest on biodiversity, beyond pyrophilous and saproxylic species? A systematic review. *Environmental Evidence, 7*, 1-33. https://doi.org/10.1186/s13750-018-0131-5

Fernandes, P. M., Davies, G. M., Ascoli, D., Fernández, C., Moreira, F., Rigolot, E., Stoof, C.R., Vega, J.A., & Molina, D. (2013). Prescribed burning in southern Europe: developing fire management in a dynamic landscape. *Frontiers in Ecology and the Environment, 11*(1), 4-14. https://doi.org/10.1890/120298

Freeman, J., Kobziar, L., Rose, E. W., & Cropper, W. (2017). A critique of the historical-fire-regime concept in conservation. *Conservation Biology, 31*(5), 976-985. https://doi.org/10.1111/cobi.12942

Galizia, L. F., Barbero, R., Rodrigues, M., Ruffault, J., Pimont, F., & Curt, T. (2023). Global warming reshapes European pyroregions. *Earth's Future*, *11*(5), e2022EF003182. https://doi.org/10.1029/2022EF003182

García, M. L., & Caselles, V. (1991). Mapping burns and natural reforestation using Thematic Mapper data. *Geocarto International*, *6*(1), 31-37. https://doi.org/10.1080/10106049109354290

Gonzalez, J. C. T., & Dans, A. T. L. (1998). Birds and mammals of the fragmented forests along the Anahawin River, Mt. Iglit-Baco National Park, Mindoro Island, Philippines. *Sylvatrop: the technical journal of Philippine Ecosystems and Natural Resources, 8*(1-2).

Gonzalez, J.C.T., Dans, A.T.L. and Afuang, L.E. (1999) Rapid Island-Wide Survey of Terrestrial Fauna and Flora on Mindoro Island, Philippines. *Mindoro Biodiversity Conservation Programme.*

He, T., Lamont, B. B., & Pausas, J. G. (2019). Fire as a key driver of Earth's biodiversity. *Biological Reviews, 94*(6), 1983-2010. https://doi.org/10.1111/brv.12544

Jones, G. M., & Tingley, M. W. (2022). Pyrodiversity and biodiversity: A history, synthesis, and outlook. *Diversity and Distributions, 28*(3), 386-403. https://doi.org/10.1111/ddi.13280

Jung, C., Kim, J. W., Marquardt, T., & Kaczmarek, S. (2010). Species richness of soil gamasid mites (Acari: Mesostigmata) in fire-damaged mountain sites. *Journal of Asia-Pacific Entomology, 13*(3), 233-237. https://doi.org/10.1016/j.aspen.2010.04.001

Kelly, L. T., Giljohann, K. M., Duane, A., Aquilué, N., Archibald, S., Batllori, E., Bennett, A.F., Buckland, S.T., Canelles, Q., Clarke, M.F., Fortin, M.J., Hermoso, V., Herrando, S., Keane, R.E., Lake, F.K., McCarthy, M.A., Morán-Ordóñez, A., Parr, C.L., Pausas, J.G., Penman, T.D., Regos, A., Rumpff, L., Santos, J.L., Smith, A.L., Syphard, A.D., Tingley, M.W., & Brotons, L. (2020). Fire and biodiversity in the Anthropocene. *Science, 370*(6519), eabb0355. https://doi.org/10.1126/science.abb0355

Key, C. H., & Benson, N. C. (2006). Landscape assessment (LA). *FIREMON: Fire effects monitoring and inventory system*, *164*, LA-1.

Kirchhoff, C., Callaghan, C. T., Keith, D. A., Indiarto, D., Taseski, G., Ooi, M. K., Le Breton, T.D., Mesaglio, T., Kingsford, R.T., & Cornwell, W. K. (2021). Rapidly mapping fire effects on biodiversity at a large-scale using citizen science. *Science of the Total environment, 755*, 142348. https://doi.org/10.1016/j.scitotenv.2020.142348

Kovář, P., Štefánek M., and J. Mrázek (2011). "Responses of vegetation stages with woody dominants to stress and disturbance during succession on abandoned tailings in cultural landscape." *Journal of Landscape Ecology 4* (2), 35-48. https://doi.org/10.2478/v10285-012-0037-9

Littell, J. S., Peterson, D. L., Riley, K. L., Liu, Y., & Luce, C. H. (2016). A review of the relationships between drought and forest fire in the United States. *Global change biology,* *22*(7), 2353-2369. https://doi.org/10.1111/gcb.13275

McCaw, L., Hamilton, T., & Rumley, C. (2005). Application of fire history records to contemporary management issues in south-west Australian forests. In *6th National Conference of the Australian Forest History Society Inc* (pp. 555-564). Rotterdam, The Netherlands: Millpress Science Publishers.

Merritt, M. L. (1908). *The forests of Mindoro* (No. 8). Bureau of Printing.

Miller, J. D., & Thode, A. E. (2007). Quantifying burn severity in a heterogeneous landscape with a relative version of the delta Normalized Burn Ratio (dNBR). *Remote sensing of Environment*, *109*(1), 66-80. https://doi.org/10.1016/j.rse.2006.12.006

Murphy, K. A., Reynolds, J. H., & Koltun, J. M. (2008). Evaluating the ability of the differenced Normalized Burn Ratio (dNBR) to predict ecologically significant burn severity in Alaskan boreal forests. *International Journal of Wildland Fire*, *17*(4), 490-499. https://doi.org/10.1071/WF08050

Nasi, R., Dennis, R., Meijaard, E., Applegate, G., & Moore, P. (2002). Forest fire and biological diversity. *UNASYLVA-FAO-*, 36-40.

Nitschke, C. R., & Innes, J. L. (2007). Interactions between fire, climate change and forest biodiversity. *CABI Reviews*, (2006), 9-pp. https://doi.org/10.1079/PAVSNNR2006106

Pandey, V.C., Bajpai, O., Pandey, D.N., Singh, N. (2015). *Saccharum spontaneum*: an underutilized tall grass for revegetation and restoration programs. *Genetic Resources and Crop Evolution, 62*(3), 443-450. https://doi.org/10.1007/s10722-014-0208-0

Parker, B. M., Lewis, T., & Srivastava, S. K. (2015). Estimation and evaluation of multi-decadal fire severity patterns using Landsat sensors. *Remote sensing of Environment*, *170*, 340-349. https://doi.org/10.1016/j.rse.2015.09.014

Parr, C. L., & Andersen, A. N. (2006). Patch mosaic burning for biodiversity conservation: a critique of the pyrodiversity paradigm. *Conservation biology, 20*(6), 1610-1619. https://doi.org/10.1111/j.1523-1739.2006.00492.x

Petermann, J. S., & Buzhdygan, O. Y. (2021). Grassland biodiversity. *Current Biology*, *31*(19), R1195-R1201. https://doi.org/10.1016/j.cub.2021.06.060

Penman, T. D., Bradstock, R. A., & Price, O. (2012). Modelling the determinants of ignition in the Sydney Basin, Australia: implications for future management. *International Journal of Wildland Fire*, *22*(4), 469-478. https://doi.org/10.1071/WF12027\

Santos, F. M., Terra, G., Piotto, D., & Chaer, G. M. (2021). Recovering ecosystem functions through the management of regenerating community in agroforestry and plantations with *Khaya* spp. in the Atlantic Forest, Brazil. *Forest Ecology and Management*, *482*, 118854. https://doi.org/10.1016/j.foreco.2020.118854

Santos, S. M. B. D., Bento-Gonçalves, A., Franca-Rocha, W., & Baptista, G. (2020). Assessment of burned forest area severity and postfire regrowth in chapada diamantina national park (Bahia, Brazil) using dnbr and rdnbr spectral indices. *Geosciences*, *10*(3), 106. https://doi.org/10.3390/geosciences10030106

Savadogo, P., Tiveau, D., Sawadogo, L., & Tigabu, M. (2008). Herbaceous species responses to long-term effects of prescribed fire, grazing and selective tree cutting in the savanna-woodlands of West Africa. *Perspectives in Plant Ecology, Evolution and Systematics, 10*(3), 179-195. https://doi.org/10.1016/j.ppees.2008.03.002

Schepers, L., Haest, B., Veraverbeke, S., Spanhove, T., Borre, J. V., & Goossens, R. (2014). Burned area detection and burn severity assessment of a heathland fire in Belgium using airborne imaging spectroscopy (APEX). *Remote Sensing*, *6*(3), 1803-1826. https://doi.org/10.3390/rs6031803

Schmerbeck, J., & Seeland, K. (2007). Fire supported forest utilisation of a degraded dry forest as a means of sustainable local forest management in Tamil Nadu/South India. *Land Use Policy, 24*(1), 62- 71. https://doi.org/10.1016/j.landusepol.2006.01.001

Soverel, N. O., Perrakis, D. D., & Coops, N. C. (2010). Estimating burn severity from Landsat dNBR and RdNBR indices across western Canada. *Remote Sensing of Environment, 114*(9), 1896-1909. https://doi.org/10.1016/j.rse.2010.03.013

Stephens, S. L., Adams, M. A., Handmer, J., Kearns, F. R., Leicester, B., Leonard, J., & Moritz, M. A. (2009). Urban–wildland fires: how California and other regions of the US can learn from Australia. *Environmental Research Letters, 4*(1), 014010. https://doi.org/10.1088/1748-9326/4/1/014010

Teobaldo, D., & Baptista, G. M. D. E. (2016). Measurement of severity of fires and loss of carbon forest sink in the conservation units at Distrito Federal. *Revista Brasileira de Geografia 9*, 250-264.

Tomchenko, O. V., Khyzhniak, A. V., Sheviakina, N. A., Zahorodnia, S. A., Yelistratova, L. A., Yakovenko, M. I., & Stakhiv, I. R. (2023). Assessment and monitoring of fires caused by the War in Ukraine on Landscape scale. *Journal of Landscape Ecology*, *16*(2), 76-97. https://doi.org/10.2478/jlecol-2023-0011

Valkó, O., & Deák, B. (2021). Increasing the potential of prescribed burning for the biodiversity conservation of European grasslands. *Current Opinion in Environmental Science & Health*, 22, 100268. https://doi.org/10.1016/j.coesh.2021.100268

Veraverbeke, S., Somers, B., Gitas, I., Katagis, T., Polychronaki, A., & Goossens, R. (2012). Spectral mixture analysis to assess post-fire vegetation regeneration using Landsat Thematic Mapper imagery: Accounting for soil brightness variation. *International Journal of Applied Earth Observation and Geoinformation*, *14*(1), 1-11. https://doi.org/10.1016/j.jag .2011.08.004

Vergara, D. C. D. M., Canlas, C. P. I., & Blanco, A. C. (2024). Mapping and assessment of burned areas in Rizal, Palawan using SAR burned and vegetation indices. *Proceedings of SPIE, Eighth Geoinformation Science Symposium 2023: Geoinformation Science for Sustainable Planet,* 12977. https://doi.org/10.1117/12.3009673

Weir, J. K., Sutton, S., & Catt, G. (2020). The theory/practice of disaster justice: Learning from indigenous peoples' fire management. *Natural hazards and disaster justice: Challenges for Australia and its neighbours*, 299-317. https://doi.org/10.1007/9

Wiebe, K. L. (2001). Microclimate of tree cavity nests: is it important for reproductive success in Northern Flickers? *The Auk*, *118*(2), 412-421. https://doi.org/10.1093/ auk/118.2.412