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Anthropogenic Edges and Effective Preserve Size in the Coast Redwood (*Sequoia sempervirens* (D.Don) Endl.) Forest

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Abstract

Ecological edges created through human activities influence both biotic and abiotic factors within forest communities. The extent of this influence within a preserve is informed by the arrangement, location, and abruptness of edges, as well as the nature of the disturbances that created them. The purpose of this study was to evaluate the impacts of anthropogenic edges on soil and vegetation in the *Sequoia sempervirens* (D.Don) Endl. (coast redwood) forest, and to compare two methods for estimating the effective size of forest preserves. We used a combination of field data and remote image analysis collected in six forest preserves in the Santa Cruz Mountains of California. Analysis of field data collected on randomly distributed transects indicated an average depth of influence of 200 meters based on correlations between abiotic and biotic metrics and distance from the forest edge. Abiotic factors including soil temperature and compaction were negatively correlated to distance from the edge in the direction of the forest interior, while soil pH, moisture, and duff depth exhibited positive relationships. Positive correlations were also found for biotic variables including tree canopy cover, the dominance of coast redwood and *Notholithocarpus densiflorus* (Hook. and Arn.) Manos, C.H. Cannon, and S. Oh (tanoak), total understory plant cover, and the cover of coast redwood forest associated plant species. In contrast, the cover and richness of non-native species were highest for samples closest to the edge. To assess the effective size of preserves, high resolution digital areal images were accessed on an ArcGIS platform. Analysis indicated variation in abruptness between types of edges, with the greatest abruptness found on edges associated with urban development and roads and the lowest abruptness associated with agricultural, grazing, and commercial timber uses. Little variation was exhibited in sinuosity between land use types or in the relative influence of edges for exurban versus urban parks. The cumulative result of

edges in the parks studied, based on the depth of influence assessed from field analysis, was a substantial reduction in the operative size of the preserves. Comparison of two methods of estimating this influence indicated a mean affected area of >26% using the "perimeter" method and >64% of the preserved area affected when using an "aggregated" method. These results suggest that including *internal* anthropogenic edges created by roads, developments, and management activities in estimates of effective preserve size is more accurate and significantly reduces the estimated size of the core area. With an increasing level of active management occurring within coast redwood preserves, it is important to consider the cumulative impact of internal anthropogenic edges and the subsequent potential decline in the effective size of preserves.

Keywords

Edge, Core Area, Exurban, Coast Redwood

1. Introduction

An ecological edge is a sharp boundary between two adjacent patches that differ in terms of species composition, successional stage, or management history (Forman, 1995; Fagan et al., 2003; Harper et al., 2005). Edges can be natural or anthropogenic in origin; and are temporary or persistent in expression (Hardt et al., 2013). Urban edges are particularly persistent and can significantly influence adjacent wildlands, affecting both physical and biological features (Bracmort, 2014; Theobald & Romme, 2007; Vallet et al., 2010). The nature of ecological boundaries varies depending upon factors such as sinuosity (curviness) and abruptness (distance from one side of the edge to the other) (Strayer et al., 2003; Fagan et al., 2003). Abrupt linear edges created by clearcutting, roadbuilding, and urban development, are potentially more impactful than gradual sinuous edges (De Chant et al., 2010; Pohlman et al., 2009). Where forest recovery is allowed to proceed along an edge, sinuosity increases with time, allowing for the development of a protective buffer between disturbance and the forest interior and a gradual decline in the edge influence (De Chant et al., 2010). Maintained edges, in contrast, create an enduring area of transition between microclimates, including gradients of light intensity, wind exposure, humidity, and temperature (Gascon et al., 2000; Matlack, 1994). These gradients can be extreme, especially in urban areas where the modified nutrient flow can affect forest ecosystems over the long term (Christie & Hochuli, 2005; Hamberg et al., 2009). As a result, maintained edges have a significant influence on forest composition and structure, tree mortality and reproduction, and the distribution of animals, within the edge environment (Gascon et al., 2000; Hamberg et al., 2008; Hamberg et al., 2009; Malmivaara-Lämsä et al., 2008; Matlack 1994). Soil conditions, including temperature, chemical composition, and structure are also affected by induced

edges as well; influencing nutrient uptake, root growth, and species composition (Vogt et al., 2015; Malmivaara-Lämsä et al., 2008; Paul et al., 2004; Matlack, 1994). Human activities can also cause soil compaction, leading to changes in forest productivity and soil resilience, limiting air and water transportation within the soil, and resulting in oxygen deficiency (Hwang et al., 2020; McBride, 1977). Forests near urban areas tend to have higher foot traffic and a greater degree of road construction which can cause soil compaction and affect forest productivity and soil resilience (Hwang et al., 2020).

Once an artificial edge has been imposed, a forest preserve is divided into two parts, the forest interior (or core area) and the area of edge influence; with the shape of the preserve affecting the core-to-edge ratio (Laurance & Yensen, 1991; Santana et al., 2021). Estimation of the core area is dependent on the size and shape of the preserves, as well as which edges are included in the estimations (Didham & Ewers, 2012). Within the area of edge influence, changes in microclimate and increased exposure to disturbance are apparent (Gascon et al., 2000; Janzen 1983; Chen & Franklin, 1993; Murcia, 1995; Cadenasso et al., 2003; Harper et al., 2005). Solar radiation, wind exposure, soil temperature soil moisture, and reduced soil fertility have all been noted in relation to the proximity of anthropogenic edges (Hardt et al., 2013; Neher et al., 2013). In addition, the influence of the edge can result in reduced stand complexity and more early successional and invasive species, and forest fragments are unable to support the native flora and fauna (Murcia, 1995; Laurance et al., 2002). Species composition can also be affected along edges, increasing the dominance of sun-loving invasive plant species and reducing opportunities for shade adapted forest associates. Reduced understory cover, as a result of a higher density of small trees and increased shrub cover, has also been observed along forest edges (Laurance et al., 2002).

The depth of edge influence varies between forest types and site conditions, yet some generalizations can be made (Chen & Franklin, 1992; Hardt et al., 2013). Harris and Harris (1984) proposed a "three-tree height rule" for edge effects, which has been commonly used to approximate the depth of edge influence. As a result, the effective size of a forest preserve is reduced in direct proportion to the extent that it is affected by artificial edges. Where fragmentation is extensive, an entire preserve may be considered an area of edge influence. In such cases, no forest interior remains and the natural habitat is rendered functionally unable to support the natural forest community (Saunders et al., 1991; Andrén, 1994; Murcia, 1995). The shape of a forest fragment is also an important factor in determining the effective size of preserves (Didham & Ewers, 2012). Preserves that have elongated and irregular boundaries present the highest edge-to-interior ratio and therefore are subject to greater edge influence. Those preserves that are closest to circular in shape present the least edge and are therefore, least susceptible to edge influences. Similarly, edges created within a forest preserve by roads and other developments can serve to increase the ratio

of edge to the preserved area, effectively reducing the core area (Schonewald-Cox & Buechner, 1992).

In coast redwood forests, the floor is generally shaded and covered by a thick layer of duff that creates retains moisture that supports redwood understory associated species and supports an environment for coast redwood seeds to germinate (Burns & Honkala, 1990). As a result, the creation of edge in this forest type is particularly problematic for associated understory species such as *Oxalis oregana* Nutt. (redwood sorrel), *Asarum caudatum* Lindl. (wild ginger), *Trillium ovatum* Pursh (Pacific trillium), *Vaccinium ovatum* Pursh (California huckleberry), and *Polystichum munitum* (Kaulf.) C. Presl (Western sword fern), which is adapted to a cool and moist forest floor, is shade tolerant and is generally sensitive to human disturbance (Hanover & Russell, 2018; Lyons & Lazaneo, 2015). The study of edge effects is also particularly important in coast redwood forests as very little of the original old-growth remains (Harris & Harris, 1984), with the bulk having been converted into managed timber stands and exurban developments what old-growth remains have been fragmented and protected in isolated parks and preserves. All of these preserves are affected to some degree by human activities within or around them (Gascon et al., 2000; Murcia, 1995), including maintenance of permanent roadways (Dangerfield et al., 2021). Understanding the impacts associated with the imposition of artificial edges within coast redwood preserves is essential as plans for active management within forest preserves are currently increasing.

2. Methods and Materials

2.1. Study Area

This study was conducted in the coast redwood forest, which persists in a narrow 720 km long band on the west coast of California and the southern edge of Oregon, in North America (Barbour et al., 2001; Noss, 2000). Within this region, the coast redwood ecosystem covers approximately 647,000 ha of land. Of this area, approximately 95% has been altered by logging resulting in only 5% of the original old-growth coast redwood community remaining (Harris & Harris, 1984). Coast redwoods thrive in a humid region along the Pacific Coast where temperatures are moderate year-round; with wet winters (7˚C - 12˚C) and cool, dry summers (12˚C - 17˚C), and annual precipitation ranges between 640 and 3,100 mm (Brand & George, 2000). The coast redwood's range is determined to a great extent by the distribution of summer fog, rather than the amount of rainfall (Burns & Honkala, 1990; Johnstone & Dawson, 2010; Limm et al., 2012). Marine fog supports forest species by reducing water loss and providing moisture in the summer season. Coast redwoods are poor regulators regarding their water usage, highlighting the importance of summer fog as a water source (Johnstone & Dawson, 2010). Soils and moist and slightly acidic, ranging from 5.0 to 6.5 pH (McBride, 1977; Noss, 2000).

Vegetation in the southern range of the coast redwood forest, where this study

was conducted, includes co-dominant canopy species coast redwood and *Pseudotsuga menziesii* (Mirb.) Franco (Douglas-fir); with a tanoak, *Arbutus menziesii* Pursh (Pacific madrone)*,* and *Umbellularia californica* (Hook. and Arn.) Nutt. (California bay laurel), sub-canopy. The shrub layer is populated by species such as *Rubus ursinus* Cham. And Schltdl. (California blackberry)*,* California Huckleberry*,* Pacific swordfern*,* and *Athyrium filix-Femina* (L.) Roth (lady fern). Redwood associated herbaceous species include a variety of shade loving species including: wild ginger; *Oxalis oregana* Nutt. (redwood sorrel); *Prosartes hookeri* Torr. (Hooker's fairy bells); Trillium; and *Viola sempervirens* Greene (redwood violet) (Russell, 2020). Gap phase succession resulting from windthrow dominates the natural regenerative processes in the coast redwoods (Lorimer et al., 2009), and while fire and flood are common occurrences, they rarely lead to standing replacement (Jacobs et al., 1985).

2.2. Field Data Collection

Collection of field data was conducted on 2830 m of edge created by urban development within the Forest of Nisene Marks State Park in the Santa Cruz Mountains of California. Forty 300 m transects were established randomly across urban edges in the study area. On each transect, five circular ten-meter diameter plots were distributed at 0, 40, 80, 160, and 300 m starting from the forest edge (Sampaio & Scariot, 2011). Within each ten-meter diameter sample plot, three one-meter diameter circular subplots were established to collect data on understory species cover and composition.

At the center of each sample plot, slope and aspect were recorded. Duff depth was measured using a metric ruler (USDI National Park Service, 2003), and soil compaction was estimated within the sample plots after removing the natural mulch. At a depth of 0 - 15 cm and 15 - 30 cm, soil temperature was recorded with a "Rapitest" soil thermometer, pH and moisture were measured using a combination soil pH and moisture meter (Lebron et al., 2012). Canopy cover was estimated using a spherical crown densiometer, and the diameter of all standing trees greater that 1 m in height and 10 cm in diameter was recorded. Trees that were less than one meter in height were recorded as seedlings (Hageseth, 2008). The cover of all shrub and herbaceous species were recorded within each subplot and each species was classified as non-native, native, or native/redwood-associate (Baldwin et al., 2012). Species richness, evenness, and the Shannon diversity index were calculated for each sample (Magurran & McGill, 2010).

2.3. Remote Image Analysis

Six forest preserves were compared within the Santa Cruz Mountains; with three categorized as exurban based on their proximity to urban developments (<20 km from nearest town, and <3/4 km from nearest exurban development), and three categorized as remote (>30 km from nearest town, and >3/4 km distant from nearest exurban development). Henry Cowell Redwoods State Park with an area

of 8.27 km^2, the geographically autonomous Fall Creek Henry Cowell Redwoods Unit of State Park (10.54 km^2), and the Forest of Nisene Marks State Park (40.36 km^2), were all directly adjacent to exurban development including housing developments. Portola Redwoods State Park (11.03 km^2), Big Basin Redwoods State Park (73.74 km^2), and Big Basin Redwoods State Park (18.37 km^2), in contrast were bordered in some areas by grazing and timber lands were not adjacent to exurban developments.

A total of 210 forest edges with a minimum length of 200 m, located within preserve boundaries (or within a 200 me peripheral buffer), were sampled based on criteria developed by Esseen et al. (2006) and De Chant et al. (2010). Edges were characterized using an ArcGIS 10.2 platform applied to ortho-rectified and geo-referenced digital aerial imagery retrieved from the United States Geological Survey taken between 2005-2007 with radiometric resolution in three bands. The source type was continuous with an unsigned integer pixel type, and a pixel depth of eight bits without compression. The spatial reference was NAD 1983, the projection was State Plane California III FIPS 0403 feet, and the spatial resolution was 1 meter or less. Images were compared to ArcGIS base maps, Google Earth 7.1 images, and USGS interactive maps to check for significant deviation from aerial images. Images were mosaicked to form a continuous raster surface and overlaid on GIS boundaries using ArcGIS. State Park boundaries, city boundaries, land use maps, and road maps were sourced form of shape files from a GIS portal maintained by the State of California.

Once identified, individual edges were digitized as polygons using object-based analysis and visual interpretation in order to create shapes. Each edge was classified into one of the following land use groups: Urban—edges created by exurban or urban developments; Roads—edges created by permanent and maintained roads; Agricultural—edges created by permanent agricultural activities; Grazing—edges created by open seasonal grazing lands; Timber—lands actively managed for timber resources.

The area covered by each edge was delineated, and polygons were created for areas of visibly deforested land. Abruptness was determined by measuring the distance from the "canopy drip point" to the lowest canopy point (0 meters) on the forest edge (Esseen et al., 2006). Sinuosity (a measure of edge complexity or convolutions of the edge), was measured using fractal indices in a range of 1 to 5 (De Chant et al., 2010). An intercept was applied lengthwise to the edge polygons, dividing the area into two sub-polygons using a straight line on one side and an irregular line on the other. Sinuosity was determined by dividing the length of the irregular side (path length) by the straight side (linear length). The mean sinuosity of the two sub-polygons was averaged for each edge polygon.

Sinuosity Index (SI) = Path Length (Meters)/Linear Length (Meters)

Two methods of estimating core area, based on the models developed Didham and Ewers (2012), were applied to the six redwood preserves. Both methods were designed to differentiate the forest interior from edge affected areas. With the "pe-

rimeter" method, a negative 200-meter buffer was applied around the boundary of the preserve in all locations where anthropogenic edges occurred, using GIS techniques, in a similar fashion as was conducted in Laurance and Yensen (1991) and Russell and Jones (2001). The second method, "aggregated," was performed by applying the same negative 200-meter buffer along induced boundary edges, with additional buffers applied to all anthropogenic edges (produced by roads, developments, and other significant management activities) occurring within the preserves. The total was calculated by aggregating both perimeter and internal edges.

2.4. Analytical Methods

Statistical analysis was conducted using IBM SPSS Statistics version 25, with a significance level of $\alpha = 0.05$ set for all tests. Spearman's rank correlation was used to analyze relationships between field metrics and distance from the edge. Single-factor ANOVA with post hoc pairwise comparison was used to estimate the minimum depth of influence of metrics that exhibited a strong correlation with distance from the edge. Two tailed t-test for abruptness and sinuosity between exurban and remote parks.

3. Results

Analysis of field data, combined with remote image analysis, indicated significant effects resulting from anthropogenic edges on the effective size of coast redwood forest preserves in the Santa Cruz Mountains of California.

3.1. Field Data Analysis

Evaluation of field metrics across the forest edge indicated correlations with distance from the edge for a variety of metrics related to soil conditions. Spearman's rank correlation indicated a strong negative relationship between distance from the edge and soil temperature at depths of both 0 - 15 cm ($r = -0.425$, $p < 0.001$) and 15 - 30 cm ($r = -0.412$, $p < 0.001$; Figure 1). In addition, significant variation in soil temperature was indicated between 160 and 300 meters distance from the edge using single-factor ANOVA (F = 9.96, $p < 0.001$; F = 9.57, $p < 0.001$) with post hoc pairwise comparison ($t = 4.08$, $p < 0.001$; $t = 3.15$, $p < 0.001$), indicating a minimum depth of influence of 160 meters.

Moderate positive relationships were also recorded for pH at a depth of 0 - 15 cm ($r = 0.254$, $p = 0.011$), and soil moisture at a depth of 15 - 30 cm ($r = 0.319$, $p = 0.001$); while no correlation was found for pH at depth 15 - 30 cm ($r = 0.178$, $p = 0.076$), or for soil moisture at 0-15 cm ($r = 0.082$, $p = 0.418$). A negative correlation ($r = -0.121$, $p = 0.048$) was indicated between soil compaction and distance from the edge, and a strong positive relationship was indicated for duff depth ($r = 0.483$, $p < 0.001$; Figure 2). Significant variation was indicated between 0 and 40 meters, as well as between 160 and 300 meters (F = 9.96, $p < 0.001$; F = 9.57, $p < 0.001$) with post hoc pairwise analysis indicating a minimum depth of influence of 160 m ($t = 4.08$, $p < 0.001$; $t = 3.15$, $p < 0.001$).
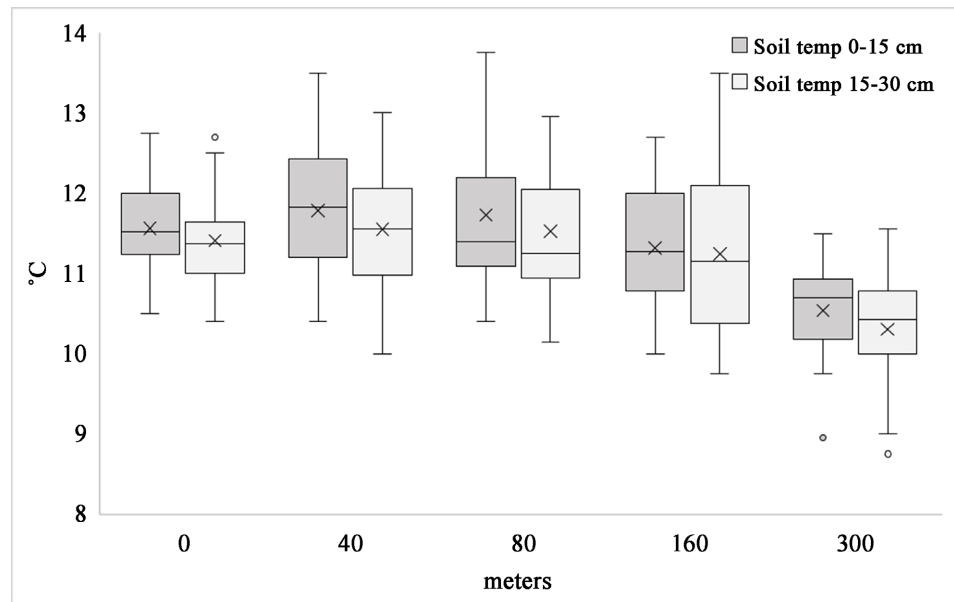
Figure 1. Soil temperatures recorded at depths of 0 - 15 cm and 15 - 30 cm across an edge on the boundary between urban development and a forest preserve.
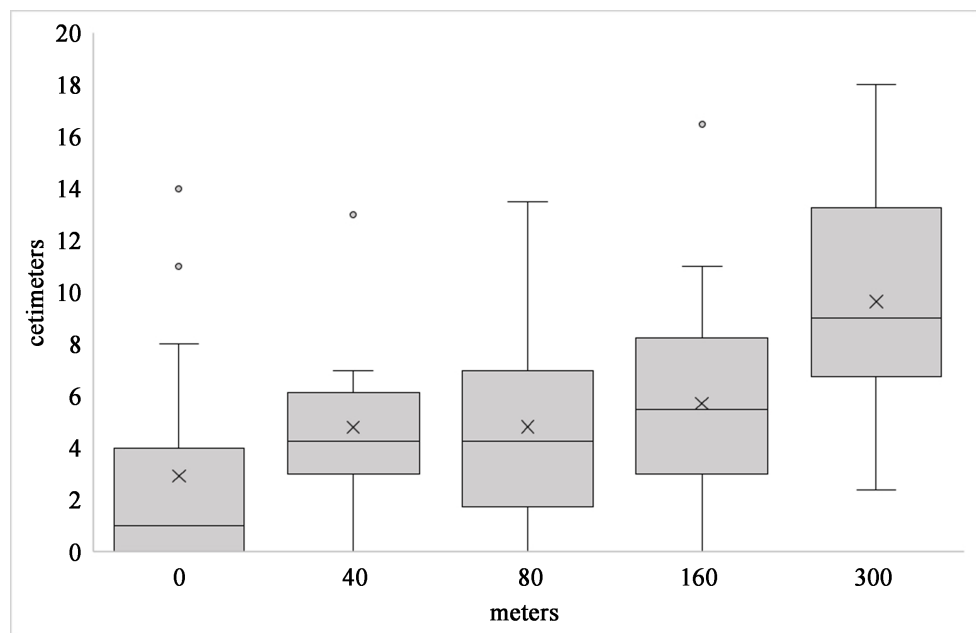
Figure 2. Depth of forest soil duff layer across an urban edge in a coast redwood preserve.

Tree canopy cover, composed of coast redwood, tanoak, Pacific madrone, *Quercus agrifolia* Née (coast live oak), Douglas-fir, *Acer macrophylum* Pursh (big leaf maple), *Alnus rubra* Bong. (red alder), and California bay laurel, was positively correlated with distance from the edge ($r = 0.403$, $p < 0.001$); as was the richness of tree species (r = 0.243, $p = 0.001$). The highest overall tree densities were found for coast redwood, tanoak, and coast live oak, with positive correlations found between distance from the edge and the density of coast redwood ($r = 0.277$, $p = 0.005$) and tanoak ($r = 0.511$, $p < 0.001$), while a negative relationship

was found for coast live oak ($r = -0.557$, $p < 0.001$). Although no relationship was detected between the distance from the edge and the overall basal area ($r = 0.120$, $p = 0.236$), there was a positive correlation indicated for coast redwood ($r = 0.273$, $p = 0.006$) and a negative relationship for coast live oak ($r = -0.376$, $p < 0.001$). In addition, the relative dominance of coast redwood exhibited a positive correlation with distance from the edge ($r = 0.253$, $p = 0.011$; Table 1) as did tanoak ($r = 0.525$, $p < 0.001$), while the correlation with coast live oak dominance was negative ($r = -0.500$, $p < 0.001$). A minimum depth of influence was estimated at 160 to 300 meters for the dominance of coast redwood, tanoak, and coast live oak $p < 0.001$) ($t = 4.08$, $p < 0.001$; $t = 3.15$, $p < 0.001$; $t = 3.15$, $p < 0.001$).

A total of thirty-seven native, and thirteen non-natives, understory species were recorded (Table 2). Though Spearman's rank correlation analysis did not, due to high variation between plots, indicate a significant relationship between distance from the edge and cover or richness of native understory species ($r = 0.193$, $p = 0.063$; $r = 0.189$, $p = 0.060$), a general increase in the mean was noted in the direction of the forest interior (Figure 3). An inverse relationship was found for non-native species, with a negative correlation between distance from the edge and non-native understory species cover and richness ($r = -0.384$, $p < 0.001$; $r = -0.612$, $p < 0.001$). In addition, it was noted that the average richness of non-native species exceeded that of natives at the edge origin (0 meters), while average native richness was greater for all increased distances.

Positive correlations were found for both the cover and richness of coast redwood associated understory species ($r = 0.234$, $p = 0.036$; $r = 0.667$, $p < 0.001$; Figure 4). Significant variation in the richness of coast redwood associated understory species was estimated between 80 and 160 meters based on single-factor ANOVA (F = 9.96, $p < 0.001$; F = 9.57, $p < 0.001$) with post hoc pairwise comparison ($t = 4.08$, $p < 0.001$; $t = 3.15$, $p < 0.001$), indicating a minimum depth of influence of 80 meters.

**Table 1.** Relative basal dominance among eight tree species on a 300 meter edge transect.

	0 m	40 m	80 m	160 m	300 m
	Mean (±SE)	Mean (±SE)	Mean (±SE)	Mean (±SE)	Mean (±SE)
Coast redwood	19.76 (±9.07)	10.00 (±6.88)	5.00 (±5.00)	27.66 (±9.31)	46.35 (±10.86)
Tanoak	0.00 (±0.00)	10.03 (±6.88)	0.00 (±0.00)	10.58 (±6.24)	27.45 (±9.65)
Pacific madrone	0.00 (±0.00)	0.00 (±0.00)	5.00 (±5.00)	0.00 (±0.00)	0.00 (±0.00)
Coast live oak	74.90 (±9.80)	64.89 (±10.86)	69.52 (±10.14)	33.69 (±10.08)	2.58 (±2.37)
Douglas-fir	0.34 (±0.23)	4.97 (±4.86)	14.23 (±7.78)	18.11 (±8.35)	18.59 (±8.56)
Big leaf maple	0.00 (±0.00)	0.00 (±0.00)	0.00 (±0.00)	4.03 (±4.03)	0.00 (±0.00)
Red alder	0.00 (±0.00)	0.00 (±0.00)	0.00 (±0.00)	0.00 (±0.00)	5.00 (±5.00)
California bay laurel	0.00 (±0.00)	0.00 (±0.00)	6.25 (±5.09)	1.59 (±1.59)	0.00 (±0.00)

Table 2. Understory species recorded on edge transects across a coast redwood/urban boundary. Coast redwood associated species are indicated by an asterisk (*).

Adelinia grande (Douglas ex Lehm.) J.I. Cohen (Pacific hound's tongue)	*Rosa gymnocarpa* Nutt. (wood rose)
Adenocaulon bicolor Hook. (American trailplant)	*Rubus ursinus* (pacific blackberry)
Artemisia douglasiana Besser ex Besser (California mugwort)	*Sanicula crassicaulis* Poepp. ex DC. (Pacific sanicle)
Asuram caudatum (wild ginger)	*Satureja douglasii* (Benth.) Kuntze (yerba buena)
Cardamine californica (Nutt.) Greene (milkmaid)	*Solanum Americanum* Mill. (black nightshade)
Claytonia lanceolata Pall. ex Pursh (western springbeauty)	*Stachys bullata* Benth. (California hedgenettle)
Cystopteris fragilis (L.) Bernh. (brittle bladder fern)	*Tellima grandiflora* (Pursh) Dougl. ex Lindl. R.Br. (fringecups)
Dryopteris arguta (Kaulf.) Watt (coastal wood fern)	*Toxicodendron diversilobum* (Torr. and A.Gray) Greene (poison oak)
Fragaria vesca L. (wild strawberry)	*Trillium ovatum* (pacific trillium)
Frangula californica (Eschsch.) A.Gray (coffeeberry)	*Urtica dioica ssp. gracilis* (Aiton) Selander (California nettle)
Galium aparine L. (bedstraw)	*Vaccinium ovatum* (huckleberry)
Heteromeles arbutifolia (Lindl.) M.Roem. (toyon)	*Viola sempervirens* (redwood violet)
Iris douglasiana Herb. (douglas iris)	*Woodwardia fembriata* Sm. (giant chain fern)
Lonicera hispidula (Lindl.) Dougl. ex Torr. and Gray (hairy honeysuckle)	Non-native
Maianthemum racemosum (L.) Link (false solomon seal)	*Ageratina adenophora* (Spreng.) King and H.Rob. (sticky snakeroot)
Nemophila parviflora Dougl. ex Benth. (small-flower nemophila)	*Arum italicum* Mill. (Italian arum)
Osmorhiza berteroi DC. (mountain sweet cicely)	*Crassula ovata* (Miller) Druce (jade plant)
Oxalis oregana (redwood sorrel)	*Hedera helix* L. (common ivy)
Pentagramma triangularis (Kaulf.) Yatsk., Windham and E. Wollenw. (goldback fern)	*Ilex aquifolium* Linnaeus (common holly)
Polypodium californicum Kaulf. (California polypody)	*Mesembryanthemum cordifolium* L.f. (heart-leaf)
Polystichum munitum (western sword fern)	*Myosotis latifolia* Poir. (broadleaf forget me not)
Prosartes hookeri (hooker's fairybells)	*Stellaria media* (L.) Vill. (common chickweed)
Pteridium aquilinum (L.) Kuhn (common bracken fern)	*Tradescantia fluminensis* Vell. (small-leaf spiderwort)
Ribes menziesii Pursh (canyon gooseberry)	*Zantedeschia sp.* Spreng. (calla lily)

3.2. Characterization of Edges and Impacts on Operative Preserve Size

Variation occurred in the abruptness of edges between land uses with the most abrupt edges occurring in proximity to urban developments and roads and the least abrupt edges occurring in proximity to grazing and timber lands (Table 3). In addition, significantly greater mean edge abruptness was found for exurban (9.17 ± 1.67 SE) compared remote (17.25 ± 2.19 SE) preserves based on a two-tailed t-test ($t = -2.78$; $p = 0.003$). In contrast, the sinuosity of edges did not exhibit a pattern of variation between land uses or between exurban and remote
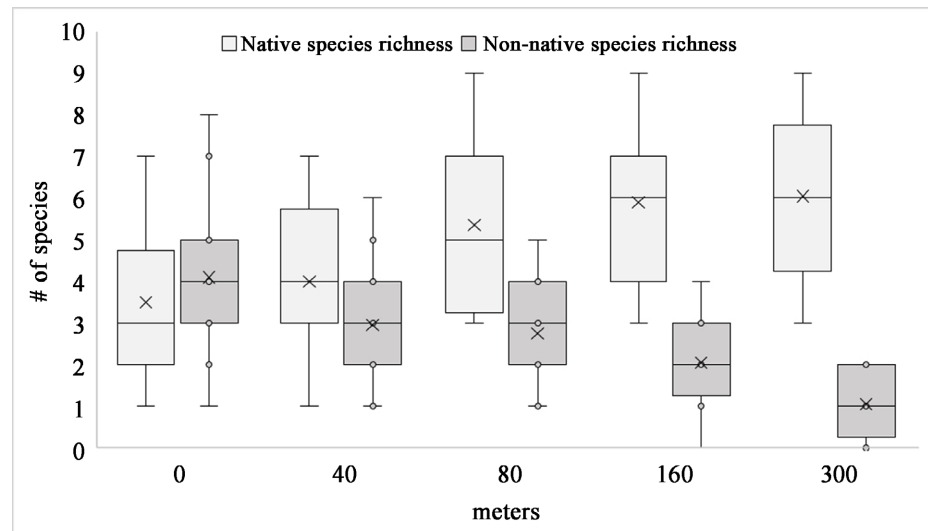
Figure 3. Richness of native and non-native understory species across a 300 meter edge transect in a coast redwood preserve.
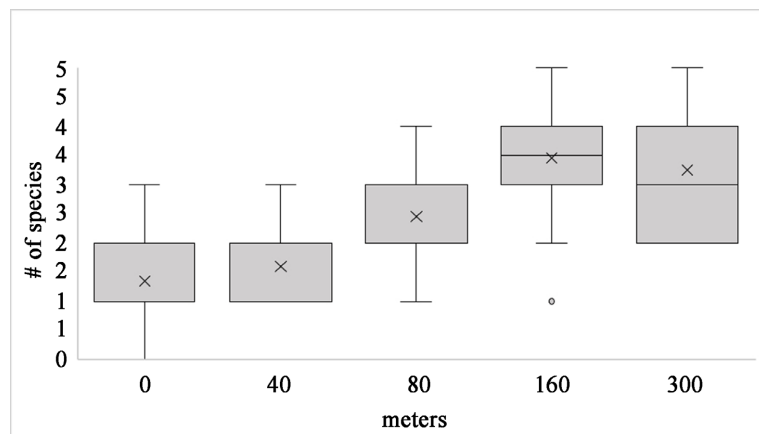
Figure 4. Richness of coast redwood associated understory species richness.

Table 3. Abruptness and sinuosity index measured on edges in six coast redwood forest preserves in the Santa Cruz Mountains, California.

Type of edge	Mean Abruptness (m ± SE)	Mean Sinuosity (SI ± SE)
Urban	2.08 (±1.09)	1.56 (±0.08)
Roads	3.91 (±1.67)	1.45 (±1.22)
Agriculture	4.67 (±2.20)	1.37 (±0.04)
Grazing	11.60 (±2.74)	1.83 (±1.32)
Timber	29.87 (±3.47)	1.48 (±0.12)

locations (1.59 ± 0.62 SE; 1.49 ± 0.65 SE) ($t = 0.13$; $p = 0.447$).

The estimation of the percent area affected, using an average depth of approximately 200 meters, varied appreciably between the two estimation methods with the perimeter method consistently yielding a lower estimate (Table 4; Figures 5-7). However, little variation was found in core area estimates between remote

Table 4. Comparison of % area affected by anthropogenic edges using two methods, perimeter and aggregate, in three remote and three ex-urban redwood forest preserves in the Santa Cruz Mountains of California.

	Perimeter	Aggregate
Exurban		
Nisene Marks	21.03	54.98
Fall Creek	30.77	76.1
Henry Cowell	26.98	64.61
Mean (±SE)	26.26 (±2.83)	65.23 (±6.01)
Remote		
Big Basin	24.08	49.25
Butano	27.47	73.06
Portola	30.07	68.18
Mean	27.21 (±1.73)	63.50 (±7.26)
Combined		
	26.74 (±3.70)	64.37 (±6.64)
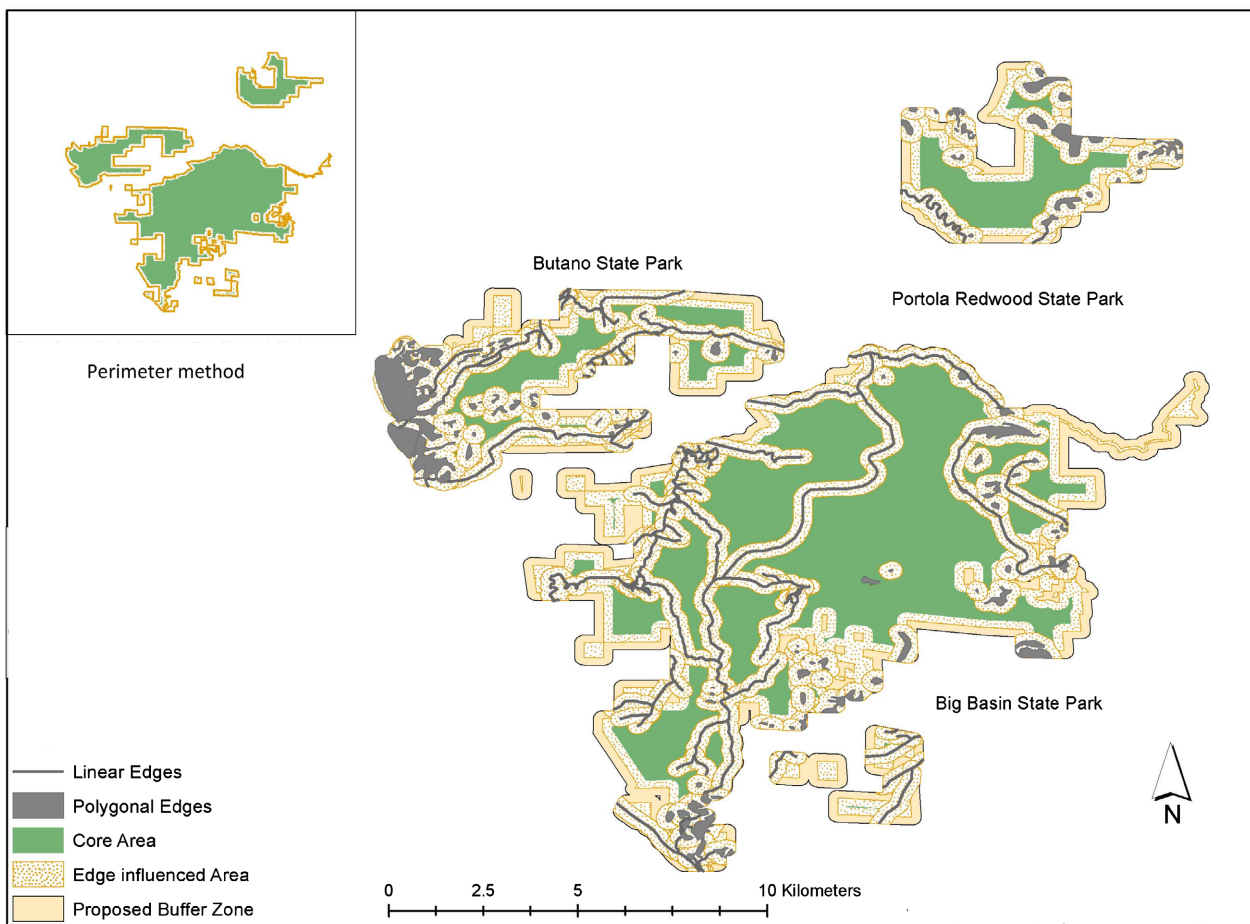
Figure 5. Area of anthropogenic edge influence for three remote coast redwoods preserves (Portola Redwoods State Park, Butano State Park, and Big Basin Redwood State Park) in the Santa Cruz Mountains of California.
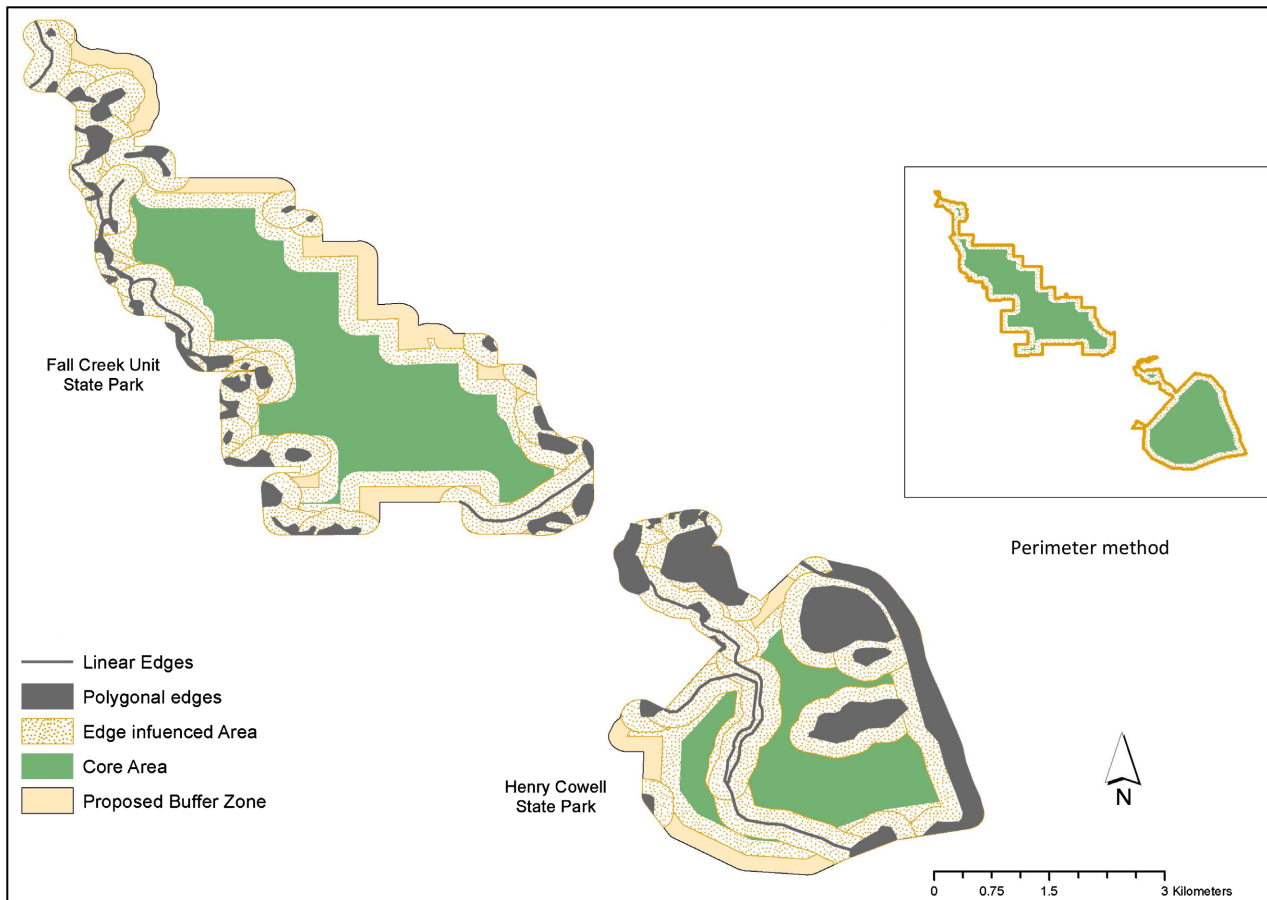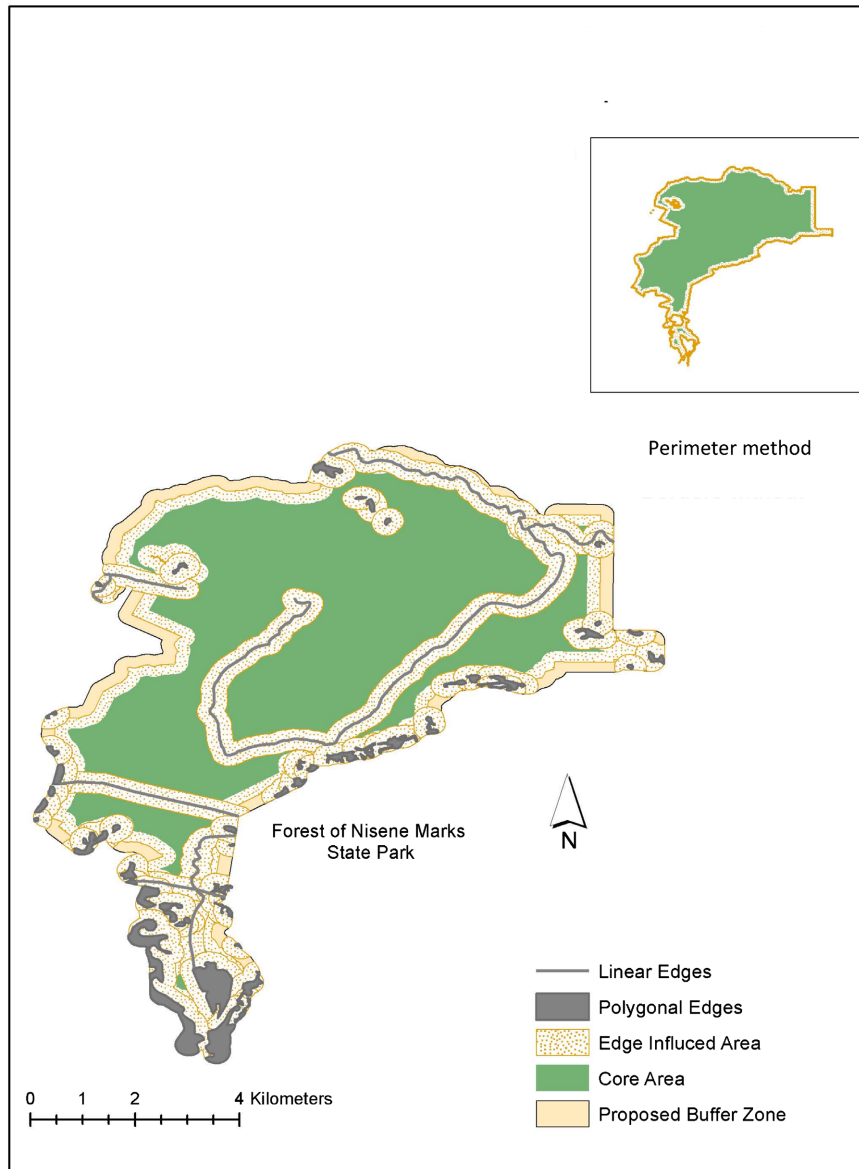
Figure 6. Area of anthropogenic edge influence for two exurban coast redwoods preserves (Henry Cowell Redwood Sate Park and the Fall Creek Unit of Henry Cowell Sate Park) in the Santa Cruz Mountains of California.

and exurban parks using either estimate.

4. Discussion

Anthropogenic edges have profound impacts on the ecology of forest preserves, with the depth of that influence being informed by the longevity, abruptness, and sinuosity of the induced edge (De Chant et al., 2010; Esseen et al., 2006; Harper et al., 2005). In coast redwood forests, where very little primeval forest remains, the impacts of these edges have a significant role in the preservation of parks and preserves (Dangerfield et al., 2021). In the wildland-urban interface (WUI), where exurban development interfaces directly with forest resources, edge influence may be exacerbated (Suarez-Rubio et al., 2013). However, activities *within* parks, including maintenance of park infrastructure such as public roads, as well as management practices such as "restorative" thinning and fuel management, can result in significant impacts on preserves as well (Schonewald-Cox & Buechner, 1992).

The field data that we collected in the coast redwood forest supports previous research suggesting that forest edges influence both physical and floristic elements. Canopy cover measured on our field sites had a strong positive relationship with

Figure 7. Area of anthropogenic edge influence for the Forest of Nisene Marks State Park in the Santa Cruz Mountains of California.

distance from the edge, indicating that forest structure had been altered within the edge environment in a manner similar to that found in other forest types (Didham & Ewers, 2012; Murcia, 1995). The higher air temperatures recorded near the edge of our field sites were a direct result of this reduced canopy cover, with subsequent increased solar radiation. Soil temperatures were influenced in a similar manner, as has been seen in other ecosystem types (de Casenave et al., 1995; Jose et al., 1996; Kapos, 1989). The lower canopy cover, and subsequent reduction in leaf litter and duff, also impacted soil pH on our sites, which supports previous findings (Malmivaara-Lämsä et al., 2008). Not surprisingly, soil moisture was also found to be lower near the edge, supporting previous research that indicated that reduced canopy cover could result in increased solar radia-

tion and wind exposure, causing higher evaporation rates and reduced moisture content in soil and litter (Camargo & Kapos, 1995; Matlack, 1994; Riutta et al., 2012).

Results also indicated that coast redwood and tanoak stand density increased with distance from the edge, while *Quercus wislizeni* A.DC. (interior live oak) stand density decreased, suggesting that stand composition varied between the forest edge and the forest interior as well (Matlack, 1994). Tree species richness, evenness, and diversity were elevated in the vicinity of the edge, echoing the findings of previous studies (Brothers & Spingarn, 1992; Murcia, 1995; Normann et al., 2016). Understory species cover and diversity were higher in proximity to the forest edge due to greater solar exposure and the influence of invasive and horticultural species (Brothers & Spingarn, 1992; Murcia, 1995; Normann et al., 2016), indicated by a negative correlation between non-native species richness and distance from the edge, and a positive correlation for coast redwood associated understory species cover and richness.

Analysis of remote imagery indicated significant differences between the influence of various land use types on edge characteristics such as abruptness and sinuosity, as was found in previous studies (De Chant et al., 2010; Pohlman et al., 2009). Surprisingly however, little difference was found between ex-urban and remote parks in the relative influence of induced edges on core area, regardless of the model used to estimate, possibly as a result of significant rural land use and resource extraction in the area. A sizable difference was found in a total area of edge influence between the results of the "perimeter" and "aggregated" methods, however, suggesting that internal park edges are important factors in the estimation of total edge influence and that using perimeter edges alone may underestimate the total influence of edge. Coast redwood forest parks and preserves have historically been protected from the impacts of commercial resource extraction such as timber harvesting within their boundaries. However, current forest management practices that include commercial harvesting have deviated from this tradition, thereby increasing the fragmentation of key habitats and the potential influence of anthropogenic edges within parks (Hanover & Russell, 2022; Sarr et al., 2004).

5. Conclusions and Recommendations

Anthropogenic edges affect a variety of abiotic and biotic factors within forest ecosystems. In forest preserves, where the implicit desire is to limit anthropogenic influence, the impact of these edges is an important factor to consider when assessing the effectiveness of preserve design. Estimating the "core area," or "effective preserve size," of a park has traditionally been focused on edges located on the boundary of the preserve. However, anthropogenic edges, created by roads, recreational developments, and resource management activities, exist within preserves as well. Excluding these internal edges can result in an underestimation of the total area of edge influence.

While edges vary significantly in terms of the depth and longevity of their influence, both internal and perimeter edges impact factors such as solar radiation, soil temperature, moisture and pH; as well as forest structure, composition, and diversity. The abrupt edges created by maintained roads and recreational developments, for example, can create microclimatic gradients that favor invasive plants and limit opportunities for shade adapted understory species. Temporary edges, created by management activities, such as restorative thinning and fuel treatment, have a more ephemeral impact on any specific location but may create a shifting pattern of edges throughout a preserve that can produce long-term effects on the integrity of the preserve as a whole. In light of our findings, we recommend caution when considering the creation of anthropogenic edges within preserves, as well as additional research on the impact of temporary internal edges.

Acknowledgments

Conflicts of Interest

The authors declare no conflicts of interest regarding the publication of this paper.

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