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Fragmentation Trajectories as a Review of Existing and Proposed Single-valued Fragmentation Indices

ABSTRACT

Two related single-valued landscape fragmentation indices D and F are proposed, based on patch aggregation, shape complexity, and percent of the focal pixels on the landscape, and are computed using Fragstats metrics on a colonization landscape continuously fragmented over 36 years. The same was done for two existing single-valued fragmentation indices, i.e., the Matheron index based solely on normalized unlike joins, and the Normalized Hypsometric Curve (NHMC) index from GUIDOS Toolbox. All were plotted chronologically, and also against percent non-forest (%nf) of the landscape, and the trajectories were compared for behavior. The NHMC index starts high even if deforestation is low, and continues increasing even further as deforestation continues, while the other three indices all start close to zero and increase gradually. F mimics D very closely, and the Matheron index only behaves differently from F and D at the end of the data range. The deviation may be due to patch aggregation, which the Matheron index does not consider. An accepted single-valued fragmentation index computed from Fragstats landscape metrices could allow for cross-study comparisons relating fragmentation with any other attribute on or of the landscape, hopefully advancing the science of fragmentation in landscape ecology as cross-study generalizations would now be possible.

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INTRODUCTION

How fragmented is a landscape, how does its fragmentation affect the ecological processes and services thereon, and how do those changes impact populations and communities on those landscapes over time and space? These questions have been asked since almost four decades ago, of the relation of pattern to process (*Turner 1989*) and vice-versa. Still what is existing so far is as fragmented a body of knowledge as the science itself.

Most sciences are limited by the technologies of their times, relying on a plethora of tools for their investigations. As the tools and methods develop, so do the sciences advance. Landscape ecology is one such science. Arising from two schools of thought, the earliest landscape ecology approaches first developed in Europe (1938-1972) dealing mostly with built systems, and independently developed in America (1972-1980) where the focus was on natural systems. The two schools of thought “synergized” (1981-1982) and merged into an international field (1983-1987), then grew from there on (*Forman 2015 in Wu 2017*).

Landscape fragmentation is the breaking up of the original land cover into discontiguous patches or fragments. Fragstats is a freeware package now available from different download sites used for analyzing landscape matrix and patch characteristics and configurations from raster data. With its advent in the mid-1990s (*McGarigal and Marks 1995*), the science of landscape ecology had its primary tool for characterizing features on the landscape. However, since there is no single index for landscape fragmentation in Fragstats, landscape fragmentation studies initially used multiple indices as indications of fragmentation. The practice is problematic in three ways. Firstly, the indices used and the sampling extents over which these were measured, or spatial supports in geography, were not standardized across studies, hence the difficulty in forming generalities, a requisite for advancing a science. Secondly, when looking at temporal changes over the same site, the trajectories of different indicators could counter each other, leading to ambiguous results or arbitrariness in the selection of the indices. Lastly, relationships of fragmentation to attributes on the landscape, e.g., biodiversity, endemism, soil fertility, desiccation, etc. would be more definitive given a single value for fragmentation, rather than a multitude of indicative indices.

Fragstats was the first software to facilitate characterization of landscape elements in an easy-to-use package (*McGarigal and Marks 1995; McGarigal et al. 2012*). But the currently available landscape indices still suffer a bias of scale (*Kronert et al. 2001; Wu et al. 2002; Gergel and Turner 2002; Wu 2004; Li and Wu 2004*). The spatial phenomenon being measured may not be homogenous on the landscape, another way of saying the spatial processes may not be stationary within the extents. Stationary spatial processes exhibit the same observed pattern regardless of origin, direction, or distance travelled by the observer on the landscape (*Cressie 2015*).

Since the sampling extents or spatial supports employed to measure them are arbitrary, changing the size (*Riiters et al. 2000; Gergel and Turner 2002; Wu et al. 2002*), shape, or location of these sampling extents could yield different values for a given landscape index (*Frazier and Kedron 2017*), hence precluding generalizations between studies due to non-standard methodology. The inability to generalize between studies of the same topic is holding back the science of landscape ecology (*Vergara 2003*; *Frazier and Kedron 2017*).

Landscape ecology posits process to pattern and pattern to process relationships (*Turner et al. 2001*). But because their primary tool is problematic, relationships drawn are usually within studies. One such phenomenon the literature is replete with is the effects of fragmentation, or the uneven, discontiguous patterns of the dominant landscape cover, on ecosystems and communities. And yet forest fragmentation studies in the last two decades still used multiple landscape indices as indications of fragmentation (*e.g., Hargis et al. 1999; Messina et al. 2006; Jomaa et al. 2008; Liu et al. 2016; Almenar et al. 2019; Xia et al. 2020*). Using different multiple indices as an indication of a phenomenon or spatial process could forego relating the landscape pattern to attributes of the landscape, a requisite for discerning ecosystem processes, given the non-standard choices for indices used. Disagreement as to the indices to be used could cloud the underlying relationships between that phenomenon and other attributes on the landscape (*Bogaert 2003*).

The earliest mention of a single-valued, landscape fragmentation index in literature was by *Matheron* (*1970*), a French mathematician and father of geostatistics (*Agterberg 2004*). It has been used in the remote sensing community (*Imbernon and Branthome 2010*), but hardly for landscape ecology (*Gergel 2007*), probably due to the lack of a tool for its direct computation, or because of a possible deficiency in the index itself.

Bogaert et al. (*2000*) proposed a single-valued landscape fragmentation index $|\varphi|$ that considers normalized values for total habitat area, total habitat perimeter, number of patches, and patch isolation, which were computed directly from either raster or vector data. The relationship is inversed though, higher fragmentation correlates to lower values of $|\varphi|$ on simulated landscapes, and its range is $0 < |\varphi| < 200$.

Butler et al. (*2004*) also proposed a single-valued forest fragmentation index FFI based on normalized values for forested area, percentage edge, and interspersion computed on a per pixel basis. Like $|\varphi|$ above, the components for FFI were computed directly from the data, which were then summed. Normality is not closed under addition (i.e., adding normalized values does not guarantee the sum is also normal) so the average was taken to ensure the index is normal.

There are still other fragmentation indices, a testament to the disagreement in the field (*Bogaert 2003*). Another body of literature considers the Infrastructural Fragmentation Index (IFI) that deals mainly with how transportation infrastructure fragments the landscape (*Mancebo Quintana et al. 2010*). Since it is the aim of this study to explore fragmentation indices to address the questions above that can readily be computed from software provided for that purpose, or those currently in use by the landscape ecology community, Bogaert's $|\varphi|$, the FFI of *Butler et al.* (*2004*), and the rest, will not be considered in this study.

There are new tools and research on fragmentation geared toward conservation efforts (*Vogt and Riiters 2017; Vogt 2018*). Measures of fragmentation, one based on normalized hypsometric curves (NHMC) and another based on entropy, were also implemented by *Vogt* (*2018*). *Bogaert et al.* (*2005*) did show that entropy increases in fragmented habitats. However, their inclusion in

GUIDOS Toolbox comes with a caveat, for historical purposes, and that a different, multi-scale approach is recommended.

Two new landscape fragmentation indices are proposed by the authors in this paper, based on theorized properties of the phenomenon. The first proposed index is theoretically more appropriate, while the second is its analog without its computational limitations on small samples of forest patches.

To observe the behavior of these four landscape indices as percent non-forest (%nf) increases, i.e., the Matheron index, NHMC index from GUIDOS Toolbox, and the two proposed indices, their fragmentation trajectories were derived and plotted over an actual fishbone pattern of forest fragmentation in a colonization frontier in Uruará, Pará, Brazil, from 1986 to 2015 by roughly every three to five years. Their behaviors were then compared, to assess performance.

Landscape Fragmentation and the Edge Effect

Changes in hydrology (*NRC 2008*), erosion, sedimentation, and water quality degradation (*Zeraatpische et al. 2013*), soil structure and chemistry (*Hajabassi et al. 1997*), local weather (*Poore 1993*), desiccation from increased albedo (*Zeng and Yoon 2009*), and if left unchecked, desertification (*Hare 1984; Ehrlich and Ehrlich 1987; Holmes 2008*), are direct abiotic effects due to loss of cover. Loss of habitat from deforestation stresses populations and alters community composition and dynamics, reduces endemicity, and affects biodiversity (*Laurance 1999; Fahrig 2003; Allnutt et al. 2008*).

But more than just loss of forest cover, the patterns that arise from deforestation could greatly compromise the ecological functions of forests (*Saunders et al. 1991; Goldsmith 1998; Primack and Corlett 2005; Laurance and Peres 2006, Sapsford et al. 2019*) and exacerbate the effects of deforestation (F*eirera and Laurance 1997; Goldsmith 1998; Primack and Corlett 2005; Laurance and Peres 2006*).

The effects of deforestation penetrate deep into the remaining forest fragments, in what is known as edge effects (*Laurance 1991; Laurance and Yensen 1991; Murcia 1995*). Fragmentation tends to decrease the core areas of habitats while increasing its edges (*Saunders et al. 1991; Laurance and Yensen 1991*), thus enhancing edge effects (*Laurance 1991; Murcia 1995*). In their monumental review of 32 years of fragmentation research, *Laurance et al.* (*2011*) find that edge effects dominate fragment dynamics as a driver, influencing the microclimate, hydrology (*Kapos 1989*), tree mortality (*Laurance et al. 1997; Mesquita et al. 1999*), carbon storage, biomass collapse (*Laurance et al. 1997; Numata et al. 2009*), but especially floral and faunal population dynamics (*Cramer et al. 2007; Herrerías-Diego et al. 2008*) and attributes (*Hill and Curran 2001; Cagnolo et al. 2006; Arroyo-Rodríguez et al. 2007*). Edge effects, however, are highly contextual to their age, nearby edges, and the vegetative cover in the periphery of the fragments (*Laurance et al. 2011*), and how species therein are affected. *Broadbent et al.* (*2008*) estimate that, due to fragmentation, 6.4% of all forests in the Amazon were within 100 m of a forest edge, a distance where extensive edge impacts occur. *Skole and Tucker* (*1993*) report that in the 1980s, areas fragmented (<100 km^2) or vulnerable to edge effects (<1 km from an edge) in the Amazon forest was one-and-a-half times greater than was actually deforested.

Moreover, aside from such aggregate effects brought about by loss of forest cover (*Laurance 1999*), the pattern of this loss is also important, given the strong link between species behavior and their use of space (*Laurence and Yensen 1991*). The matrix of habitats in fragmented landscapes greatly affects faunal mobility and thus influences their community dynamics (*Laurance 1999*). Different forest species use fragmented spaces differently, and many depend on multiple habitats (*Fahrig 2003*). Large forest bovines, such as the Tamaraw (*Bubalus mindorensis*) of the Philippines, for example, need the forest or tall grasses for cover while it shelters, and open grasslands for grazing (*Boyles et al. 2016*). In contrast, some animal, insect, and plant species require deep forest cover their entire lifetime (*Primack and Corlett 2005*). Endemic and specialist faunal and floral species are specially threatened by habitat loss and fragmentation, and are replaced by other exotic, opportunistic, or generalist species when displaced by uncontrolled deforestation (*Laurance 1999; Fahrig 2003*). Ambulant species likewise suffer when their movement is restricted by lack of corridors (*Primack and Corlett 2005*) or cut off by roads and fences (*Forman and Godron 1986; Forman 1995; Forman et al. 2002*). Research in the Amazon basin has documented how forest fragmentation compromises animal habitats, vegetative regeneration, and biomass (*Laurance et al 1997; Ferreira and Laurance 1997; Aldrich and Hamrick 1998; Benitez-Malvido et al. 1999; Laurance 1998; Scariot 1999; Laurance et al. 2001; Benitez-Malvido and Martinez-Ramos 2003*).

In some highly contextual cases, however, slight forest fragmentation may enhance biodiversity (e.g., Hovel and Lipcius 2001; Tscharntke et al. 2002). Stoll et al. (2020) also report that despite fragmentation, the genetic flow of an endangered desert shrub was not significantly affected, probably due to the actions of a likewise endangered parakeet sps. as its primary seed dispersal agent.

Landscape Indices

Landscapes are made up of mosaics or matrices of patches and corridors of different cover types interacting with each other, facilitating their ecology (*Turner et al. 2001*). As a basis for the study of these relationships, landscape indices are used to measure the composition and configuration characteristics of these features on the landscape. Fragstats (*McGarigal et al. 2012*) offer a multitude of such metrics, on three levels of organization, i.e., on the patches within a class, among different classes of patches, and on the landscape as a whole. To facilitate quantification, the indices are assumed to be homogenous within the extent over which these are measured, which may or may not be entirely true, depending on whether the spatial processes involved are stationary or not. Stationary spatial processes exhibit the same general observable pattern in all directions, regardless of where to start, which direction to take, and how far it can go (*Cressie 2015*), much like the checkered squares of a chessboard is the perfectly stationary spatial process.

Such assumptions of homogeneity of landscape characteristics over the areas for which these are measured leads to an inherent bias of scale in the landscape indices, i.e., changing the size, shape, and/or location of the extent over which the index is measured could change the value of the index computed if the underlying spatial process was not stationary. Since the size, shape, and location of the sampling extents are for now arbitrary, given the varied contexts and objectives of different studies, generality of conclusions across these studies is lost due to non-standardized methodology. The inability to draw general conclusions from the field holds back the advancement of landscape ecology as a science (*Turner et al. 2001*). However, notwithstanding the bias of scale, the use of landscape indices is still very much valid, at least within studies.

Estreguil et al. (*2012*) attempted to standardize methodology for landscape fragmentation to harmonize reporting in the EBONE project of the European Commission - Joint Research Centre, Institute for Environment and Sustainability, Forest Resources and Climate Unit, in Ispra (VA), Italy. *Estreguil et al.* (*2012*) state the common agreement that a single landscape index cannot fully represent the patch arrangement complexity. The efforts underline the dire need for standards in fragmentation studies. Further, *Estreguil et al.* (*2012*) also claim that Bogaert's |φ| failed to render ecological interpretation, as GUIDOS/MSPA (*Vogt et al. 2007*) would.

Wang et al. (*2014*) evaluated landscape indices for fragmentation and argued that indices used for fragmentation studies should incorporate patch aggregation and be able to distinguish between habitat amount and actual fragmentation. *Wang et al.* (*2014*) tested 64 class level indices and strongly recommended nine for fragmentation studies, which include core area, shape, proximity, isolation, contrast, and contagion/interspersion. However, *Wang et al.* (*2014*) are still in the mindset of context dependence in deciding which recommended indices to use, which is essentially still using multiple landscape indices as indicators of fragmentation.

Fragmentation and its Indices

As echoed by others (*Wang et al. 2014; Almenar et al. 2019), Fahrig (2003*) emphasized that cover loss should not be confused for fragmentation, and that the two phenomena should be distinguishable. *Lambin and Ehrlich* (*1997*) modelled fragmentation trajectories using the index by Matheron against percent non-forest (%nf). Generating simulated landscapes, *Lambin and Ehrlich* (*1997*) demonstrated that the trajectories of the different fragmentation patterns produced by different agents would vary only in amplitude. That is, all pattern trajectories would start at zero with complete cover, increase until a maximum is reached, then decrease as the fragments shrink and become fewer, and return to zero at total loss of cover, and thus fragments, forming Kuznets curves (*Agarwal 2018*). Thus, there is no fragmentation when the cover is intact as well as when it is totally gone, as there would be no fragments in both extreme cases.

Given a binary forest and non-forested landscape, the fragmentation index due to Matheron is:

$$\boldsymbol{M} = \frac{(\textit{number of joins between forest and non-forest pixels})}{(\sqrt{number\ of\ forest\ pixels})*(\sqrt{number\ of\ total\ pixels})} \qquad (1)$$

The Matheron index is not popular in the landscape ecology literature probably because it does not provide a tool to count the joins between unlike pixels. *Vergara et al. (2019)* derived it using Fragstat indices adjusted for pixel or cell dimensions of the raster data.

The nature of landscape fragmentation itself has become contextual on the scale and ecological phenomena being observed, and fragmentation is measured differently by different authors (*Fahrig 2003*). For example, *Hargis et al.* (*1999*) quantified landscape fragmentation using patch density, edge density, mean proximity, mean nearest-neighbor distance, and mass fractal dimension to study the influence of fragmentation patterns on martens in forested landscapes in Utah. *Ortega-Huerta* (*2007*) used core area percentage of landscape and aggregation index for biodiversity studies in Mexico. *Messina et al.* (*2006*) used patch area, density, mean patch size, and number of patches to relate land tenure to deforestation patterns in the Ecuadorian Amazon. *Jomaa et al.* (*2008*) instead used number of patches and mean patch area as landscape fragmentation indicators to study the impacts of human and natural stresses on different forest types in Mount Lebanon.

More recently, *Liu et al.* (*2016*) used total core area, normalized total core area, patch density, edge density, and landscape shape index to evaluate the impacts of habitat loss and fragmentation during urbanization. To assess habitat loss, fragmentation, and ecological connectivity for urban green space planning, *Almenar et al.* (*2019*) pre-selected 46 landscape and connectivity indices and narrowed it down to 1. Further, *Almenar et al.* (*2019*) confirmed the non-linear relationship between habitat loss and landscape connectivity. *Xia et al.* (*2020*) used patch density, percentage of landscape, aggregation index, largest patch index, and contagion for the effect of habitat fragmentation on the distribution and declining populations of snub-nosed monkeys in Yunnan Province, China.

Using multiple indices as indicators of fragmentation may forego the opportunity to directly relate biophysical attributes on the landscape with fragmentation. Ideally, landscape fragmentation should be quantified as a single measure (*Bogaert 2000; Abdullah and Nakagoshi 2007*). Done so, it can then be directly related to attributes of the biota such as biodiversity and endemism (*Vergara 1997*), or physical attributes of the land itself, e.g., fertility, moisture, etc. However, currently, there is no single-valued, commonly accepted index that measures landscape fragmentation (*Tischendorf 2001*), although several have been proposed (*Matheron 1970, in Lambin and Ehrlich 1997; Eastman 1996; Vergara 1997 and 2003; Bogaert et al 2000; Butler et al 2004; Vogt 2018*).

There are studies that use unconventional measures of fragmentation to generate synthetic landscapes as the basis for their models (*Zhizia et al. 2018*). *Sapsfor et al.* (*2019*) assume fragmentation by using the disturbance gradient from a roadside into an adjacent forest as their designation of fragmentation. Since the goal of this paper is to propose quantitative indices to relate fragmentation to phenomena or processes on the landscape, qualitative measures of fragmentation will not be considered in this paper.

The Center for Land Use Education and Research of the University of Connecticut developed a Landscape Fragmentation Tool (LFT) that runs from Spatial Analyst of ArcMap (Hurd undated) based on image morphology. Referencing *Vogt et al.* (*2007*), LFT classifies the focal land cover into morphological types of patch, edge, perforated, and core (*Hurd and Civco 2010*). *Vogt et al.* (*2007*) referenced both *Matheron* (*1970*) and B*ogeart et al.* (*2004*) regarding morphological mapping (*Riiters et al. 2009*). The method evolved into Morphological Spatial Pattern Analysis (MSPA), and is now in GUIDOS Tool Box (*Vogt 2018*). MSPA generates more topological morphologies of patches derived from a binary image of a landscape than LFT does, i.e., core, islet, perforation, edge, loop, bridge, and branch. The tool is ideal for conservation work, as resource managers are afforded a landscape with functional elements to model with (*Hernando et al. 2017*). However, the current version exports only the fragmentation classes, and not the actual measures.

A recent paradigm in landscape fragmentation by transportation infrastructure has emerged in the literature, especially by *Mancebo Quintana et al.* (*2010*). It is run as an extension in Spatial Analysis of ArcMap. The model computes how transportation networks fragment a landscape, and so cannot compute it on extents without transportation infrastructure. Thus, neither can it be used to relate fragmentation to any other attribute on the landscape. Hence neither will it be considered in this research.

A new index based on landscape characteristics theorized to cause fragmentation is proposed, as well as an estimate of it for use in smaller spatial supports or sampling tiles. Both are computed with a regular spreadsheet from normalized Fragstats metrics for forest patch density, shape complexity, and aggregation. The behavior of these two proposed indices and two other existing indices, i.e., the Matheron index and the NHMC index from GUIDOS Toolbox, are compared, as graphs of their trajectories over 36 years of deforestation activity in a colonist frontier that produces the fishbone pattern of fragmentation in the Amazon.

MATERIALS AND METHODS

The study site

"Programa da Integração Nacional" (PIN) or the National Integration Program of President Medici of Brazil in 1970, a massive infrastructure development initiative of the federal government (*Smith 1982*), heralded a new paradigm in the management and utilization of the Legal Amazon. To open up its vast resources to the national economy, as well as for geopolitical reasons (*Bratmen 2019*), highways were cut through the vast forest in generally cardinal directions (*Smith 1982*). Along one of the longest, the Trans-Amazon, part of BR-230 official designation, runs almost 3,000 kms, from Maranhão state at the eastern termini, westward to Lábrea, Amazonas in the heart of the Amazon forest (*Walker et al. 2011*). It remains mostly unpaved.

"Projecto Integrado de Colonização" (PIC), or Integrated Colonization Projects for directed colonist settlement projects, were implemented along the main highways constructed (*Moran 1981; Almeida 1992*). Onesuch project was PIC Altamira, where a colonization plan was drawn up by the Instituto Nacional Colonização e Reforma Agrária (INCRA), or National Colonization and Agrarian Reform Institute, along the Trans-Amazon Highway (*Moran 1981*).

The study site for the fishbone pattern is located along an almost 200 km segment of the BR-230 corridor near Uruará, Pará (Lat. 03° 42' 54" S and Long. 53° 44' 24" W), between Altamira and Itaituba, (Walker 2003; Walker et al. 2004), about 180 kms west of Altamira (**Figure 1**). It was opened and initially managed by INCRA as a PIC in the 1970s (*Moran 1981; Walker et al. 2004; Aldrich et al. 2006*). Uruará emancipated as a municipality in the colonization project by the late 1980s (*Arima et al. 2005; Perz et al. 2007a*).

Origins for lateral roads or "travessões" were laid out along the BR-230 or Trans-Amazon at 5 km intervals. They were officially designated as distance west of Altamira and heading from BR-230, e.g., 180 N, 190 S (**Figure 1**). Lots were planned mostly for 100 ha, 2500 m x 400 m lots with the shorter dimension facing the planned lateral road, to be awarded per colonist farming family. Some holdings were larger for agricultural enterprises (*Walker 2003; Walker et al. 2004; Perz et al. 2007a*). Title would be awarded if the farm conformed to the colonization plan (*Perz et al. 2007b*) and is shown to be productive in two years. Forest clearing is done in the summer, for burning biomass as fertilizer, since soil fertility is low (*Moran 1981*). The plot would be good for one to two years, after which fertility drops, and an adjacent plot may be cleared. Depending on the colonist farmer's profile, propensity, and proficiency, the plots

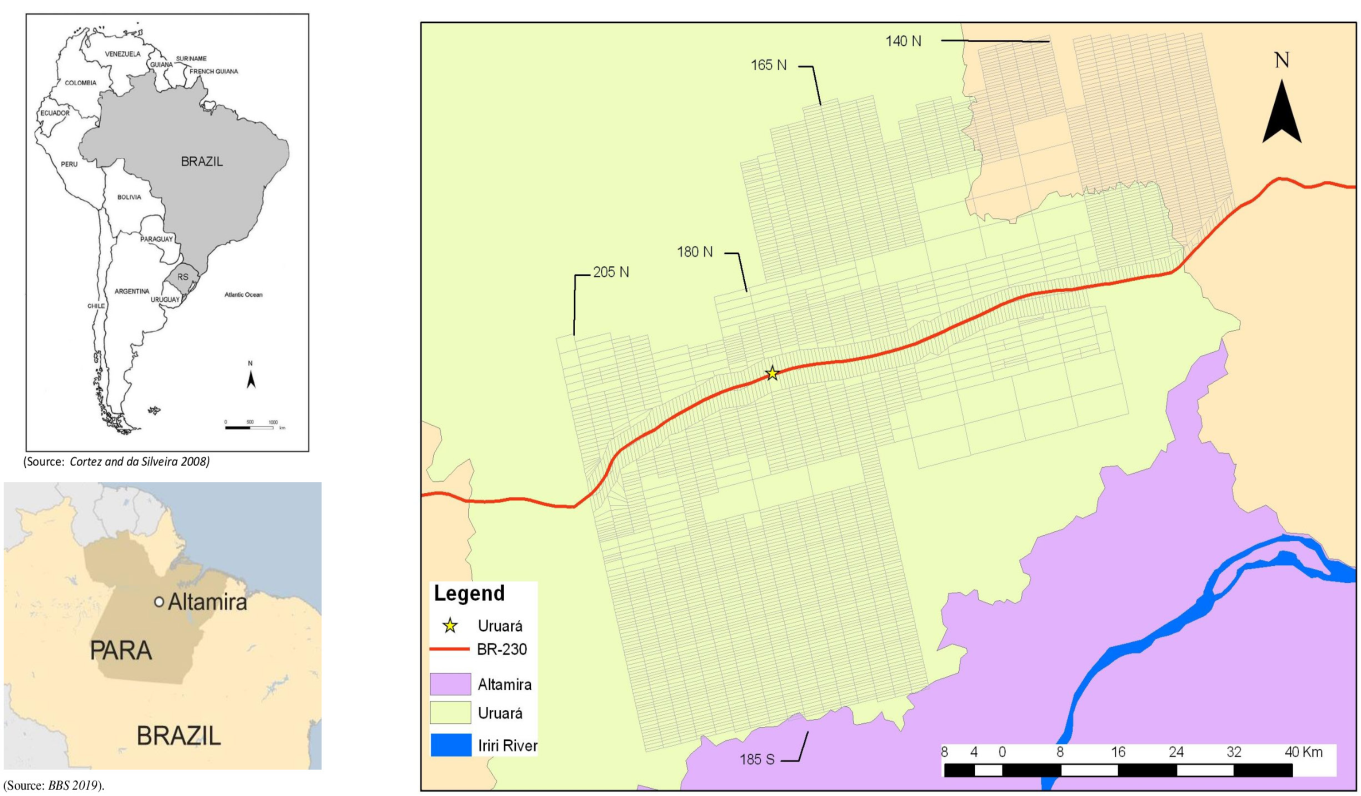

Figure 1. PIC Altamira colonization plan, which gave rise to Uruará, Pará, Brazil.

with degraded fertility could be replanted to perennial crops, converted to pasture for livestock, left to fallow, or any combination thereof over time (*Moran 1981; Deadman et al. 2004; Perz and Walker 2002; Walker et al 2002; Walker et al 2004*). Fallow plots are typically cleared again after enough biomass establishes for an effective burn, around 3 to 6 years for young secondary forests (*Scatena et al 1996*), and the cycle is repeated (*Moran 1981; Moran et al 2000; Perz and Walker 2002; Walker et al 2004*).

Due to clearing and transportation costs, the colonist farmers invariably start clearing from the front of the lot facing the lateral road and work inward (*Deadman et al 2004; Walker et al. 2004*). By statute then (*PR-CC, SAJ 1965 and 2001, which was revoked in 2012*), they can only clear half the property, and the alternating sequence of strips of forest and non-forest induces the fishbone pattern of fragmentation, with the spinal column along the highway, and the retained forest between the lateral roads as the hemal spines. The patterns arise from the collective but autonomous actions of the agents, conforming to the colonization plan in search of title to property.

Four scenes each from Landsat 5, 7, and 8 (Paths 226-227 / Rows 062-063) covering the colonization project in Uruará, Pará, Brazil from 1986 to 2015 in roughly three to five year intervals were acquired. The images were pre-processed for atmospheric effects. Unsupervised ISODATA classification produced 40 classes using a thermal, mid-infrared, near-infrared, and RGB bands. They were later generalized to forest and non-forest pixels by supervised classification based on the spectral signatures of the ISODATA classes. Regrowth retained their non-forest classes until their spectral signature resembled that of forests. The four classified scenes for a particular year were then mosaicked and clipped to the extent used in the study (**Figures 2a** and **b**). Four different single-valued fragmentation indices were computed for each period, then graphed chronologically and by percent non-forest (%nf). The graphs were then compared for behavior of the indices given their trajectories.

To minimize ambiguity, generic nomenclature is used, i.e.,:

M := Matheron index of fragmentation
G:= NHMC index from GUIDOS Toolbox
D:= a proposed new index that utilizes, among others, the fractal dimension of Mandelbrot.
F:= an analogue of D that uses the fractal mean in lieu of the fractal dimension.

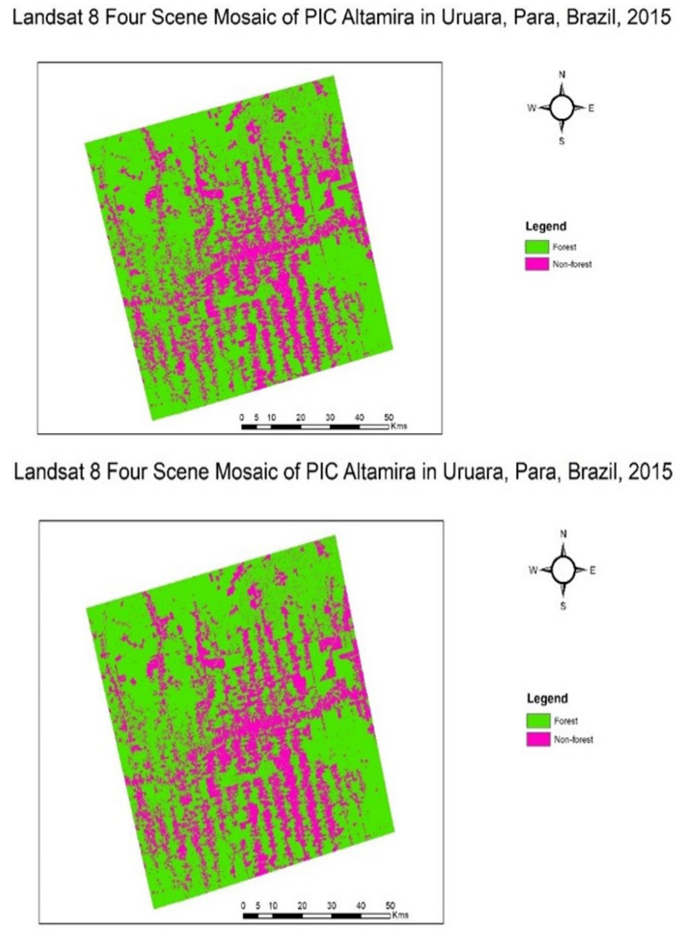

Figure 2. Four scene mosaics of the fishbone pattern of fragmentation in Uruará, Pará, Brazil. (a) Landsat 5, 1986 and (b) Landsat 8 2015.

Given the dimensions of the raster data set pixels, the Matheron index can be computed with class level indices in Fragstats (*Vergara et al. 2019*) on 30 x 30m Landsat data as follows (**Table 1**):

$$M = \frac{(TE/30)}{(SQRT(CA_f * 100/9) * SQRT(CA_{nf} * 100/9))}, \ni 0 \leq D \leq 1. \quad (2)$$

Where:
TE := total edge
CA := class area

G is given in GUIDOS Toolbox, although with a caveat that inclusion was only for historical reasons, and that a different measure based on entropy is recommended (*Vogt and Riiters 2017*). G is computed from normalized hypsometric curves, cumulative distances of a pixel to an edge (*Vogt 2018*).

Table 1. Fragstats metrics used.

Fragstats Metric Definition	Description/Remarks
1. CA $CA = \sum_{j=1}^{n} a_{ij} \left(\frac{1}{10,000} \right)$	Class Area CA equals the sum of the areas (m2) of all patches of the corresponding patch type, divided by 10,000 (to convert to hectares). CA > 0, without limit.
2. PLAND $PLAND = P_i = \frac{\sum_{j=1}^{n} a_{ij}}{A} (100)$	Percent Landscape Pi = proportion of the landscape occupied by patch type (class) i. aij = area (m2) of patch ij. A = total landscape area (m^2). PLAND equals the sum of the areas (m^2) of all patches of the corresponding patch type, divided by total landscape area (m^2), multiplied by 100 (to convert to a percentage); in other words, PLAND equals the percentage the landscape comprised of the corresponding patch type. Note, total landscape area (A) includes any internal background present. 0 < PLAND <= 100.
3. TE $TE = \sum_{k=1}^{m} e_{ik}$	Total Edge Total length (m) of edge in landscape involving patch type (class) i; includes landscape boundary and background segments involving patch type i Total edge at the class level is an absolute measure of total edge length of a particular patch type. TE >= 0, without limit.
4. PAFRAC $AC = \frac{2}{\frac{\left[n_i \sum_{j=1}^{n} (\ln p_{ij} \cdot \ln a_{ij}) \right] - \left[\left(\sum_{j=1}^{n} \ln p_{ij} \right) \left(\sum_{j=1}^{n} \ln a_{ij} \right) \right]}{\left(n_i \sum_{j=1}^{n} \ln p_{ij}^2 \right) - \left(\sum_{j=1}^{n} \ln p_{ij} \right)^2}}$	Perimeter-Area Fractal Dimension aij = area (m^2) of patch ij. pij = perimeter (m) of patch ij. ni = number of patches in the landscape of patch type (class) i. PAFRAC equals 2 divided by the slope of regression line obtained by regressing the logarithm of patch area (m^2) against the logarithm of patch perimeter (m). That is, 2 divided by the coefficient b1 derived from a least squares regression fit to the following equation: ln(area) = b0 + b1*ln(perim). 1 <= PAFRAC <= 2. A fractal dimension greater than 1 for a 2-dimensional landscape mosaic indicates a departure from a Euclidean geometry (i.e., an increase in patch shape complexity). PAFRAC approaches 1 for shapes with very simple perimeters such as squares, and approaches 2 for shapes with highly convoluted, plane-filling perimeters. PAFRAC employs regression techniques and is subject to small sample problems. Specifically, PAFRAC may greatly exceed the theoretical range in values when the number of patches is small (e.g., <10).
5. FRAC_MN	Patch Mean Fractal Dimension An alternative to the regression approach if sufficient data are not available, by taking the mean on the fractal dimensions of each patch. The degree of complexity of a polygon is characterized by the fractal dimension (D), such that the perimeter (P) of a patch is related to the area (A) of the same patch by P ~ √AD (i.e., log P ~ ½D log A).
6. CLUMPY $Given\ G_i = \left(\frac{g_{ii}}{\sum_{k=1}^{m} g_{ik}} \right)$ $CLUMPY = \left[\begin{array}{l} \frac{G_i - P_i}{1 - P_i}\ for\ G_i \geq P_i \\ g \\ \frac{G_i - P_i}{1 - P_i}\ for\ G_i < P_i;\ P_i \geq .5 \\ \frac{P_i - G_i}{-P_i}\ for\ G_i < P_i;\ P_i < .5 \end{array} \right]$	Clumpiness Index gii = number of like adjacencies (joins) between pixels of patch type (class) i based on the double-count method. gik = number of adjacencies (joins) between pixels of patch types (classes) i and k based on the double-count method. Pi = proportion of the landscape occupied by patch type (class) i.

Table 1. Fragstats metrics used. (cont.)

Fragstats Metric Definition	Description/Remarks
	CLUMPY equals the proportional deviation of the proportion of like adjacencies involving the corresponding class from that expected under a spatially random distribution. If the proportion of like adjacencies (Gi) is greater than or equal to the proportion of the landscape comprised of the focal class (Pi), then CLUMPY equals Gi minus Pi, divided by 1 minus Pi. Likewise, if Gi < Pi, and Pi >= 0.5, then CLUMPY equals Gi minus Pi, divided by 1 minus Pi. However, if Gi < Pi, and Pi < 0.5, then CLUMPY equals Pi minus Gi, divided by negative Pi. Cell adjacencies are tallied using the double-count method in which pixel order is preserved, at least for all internal adjacencies (i.e., involving cells on the inside of the landscape). Note, Pi is based on the total landscape area (A) including any internal background present. -1 <= CLUMPY <= 1 Given any Pi , CLUMPY equals -1 when the focal patch type is maximally disaggregated; CLUMPY equals 0 when the focal patch type is distributed randomly, and approaches 1 when the patch type is maximally aggregated. Note, CLUMPY equals 1 only when the landscape consists of a single patch and includes a border comprised of the focal class.

Source: McGarigal KS et al. 2002. FRAGSTATS: Spatial pattern analysis program for categorical maps. Project: Landscape pattern analysis

Proposed fragmentation indices

Saunders et al. (*1991*) maintain that the effects of fragmentation are influenced by the size, shape, location, and isolation of the fragments. For the two proposed indices in this study, fragmentation is modelled theoretically with forest patch density, shape complexity, and spatial arrangement of the patches, improved from earlier attempts (*Vergara 1997 and 2003*) to generate normalized indices. *Wang et al.* (*2014*) include these three factors in the nine they, as well as others, recommend for fragmentation studies. *Bogaert et al.* (*2000*) stressed the importance of using normalized values for the components of fragmentation indices to avoid ambiguities in interpretation.

Initially, as cover is lost, fragmentation increases, hence forest density is a factor. Fragmentation also increases with patch shape complexity. *Imre and Bogaert* (*2003*) conclude that the fractal dimension, to certain extents, can be a measure for habitat quality, being an indicator for the edge effects. Moreover, by the nature of fractals being self-same regardless of scale (*Mandelbrot 1982*), the fractal dimension is not significantly affected by scale, which makes it ideal for use as a component for a fragmentation index. Lastly, as patches cluster or aggregate, the spaces between the clusters increases, isolating populations therein, and thus increasing fragmentation.

The General Linear Models for experimental design in statistics specify the sum of the treatment effects if there are no mixed effects (i.e., the treatments are assumed independent). To allow for mixed or interaction effects, factorial experiments use the product of the treatments effects for their interactions (*NCSS 2020*). The proposed indices are based on three normalized Fragstats metrices that are the least correlated, although there may still be some degree of correlation. As we are only interested with how their interaction contributes to fragmentation, and not the individual contributions per se, following this logic, the proposed indices specify only the product of these three factors, i.e., forest patch density, shape complexity, and aggregation. As normality is closed under multiplication (the product of two positive proper fractions is also a positive proper fraction) the resulting indices are themselves assured to be normal.

Model Specification

Let g(d) := forest density, h(F) := patch shape complexity, j(C) := patch aggregation, be continuous functions between 0 and 1, where:

d := percentage of forest,
F := fractal dimension,
C := patch aggregation index.

Since fragmentation D increases with h(F) and j(C), while D decreases as g(d) increases, this study considered:

$$D = h(F) * j(C)) * g(d)^{-1} \tag{3}$$

For the percentage forest d, g(d) = d / 100. Since this study is dealing with binary images, the density of non-forest is k(d) = (1 – (d / 100)). Increasing k(d) increases fragmentation, thus:

$$D = h(F) * j(C) * k(d) \qquad (4)$$

Using the class level indices in Fragstats (*McGarigal, Cushman, and Ene 2012*), PLAND for forest gives the percentage of forest d. Hence:

$$k(d) = (1 – (PLAND/100))$$

As an estimate of patch shape complexity, PAFRAC goes from 1 for regular polygons to 2 for very complex shapes, or $1 \leq PAFRAC \leq 2$. To keep h(F) normalized, this study used PAFRAC – 1 as a measure of shape complexity. So

$$h(F) = PAFRAC – 1.$$

Lastly, as an estimate of patch aggregation, CLUMPY ranges from -1 for dispersed patches, through 0 for randomly arranged patches, and to 1 for very aggregated patches, or $-1 \leq CLUMPY \leq 1$. To keep j(C) positive, This study considered C = CLUMPY + 1. Since $0 < C < 2$, to keep j(C) normalized, this study considered:

$$j(C) = (CLUMPY + 1) / 2.$$

Thus, $D = h(F) * j(C) * k(d)$

$$= (PAFRAC–1) * ((CLUMPY+1)/2) * (1-(PLAND*100)), \ni 0 \leq D \leq 1…. \qquad (5)$$

Since PAFRAC is the result of a regression, Fragstats will not compute it for less than 10 patches, and so D is problematic for smaller sample extents. To overcome this limitation in estimating D, FRAC_MN, the mean of the fractal dimensions of each patch, which is also normalized between [1, 2], is used in lieu of PAFRAC, with an adjustment for scale. This study considered:

$$F = 10a * (FRAC\_MN-1) * ((CLUMPY+1)/2) * (1-(PLAND*100)), \ni a = 1 \text{ and } 0 \leq F \leq 1. \qquad (6)$$

RESULTS AND DISCUSSION

The evolution of the fishbone pattern of fragmentation illustrated in this study was from 1982 to 2015 (**Figure 3.a** to **Figure 3.c**). The component Fragstats indices and the computed fragmentation indices M, D, and F was from 1986-2015 (**Table 2**).

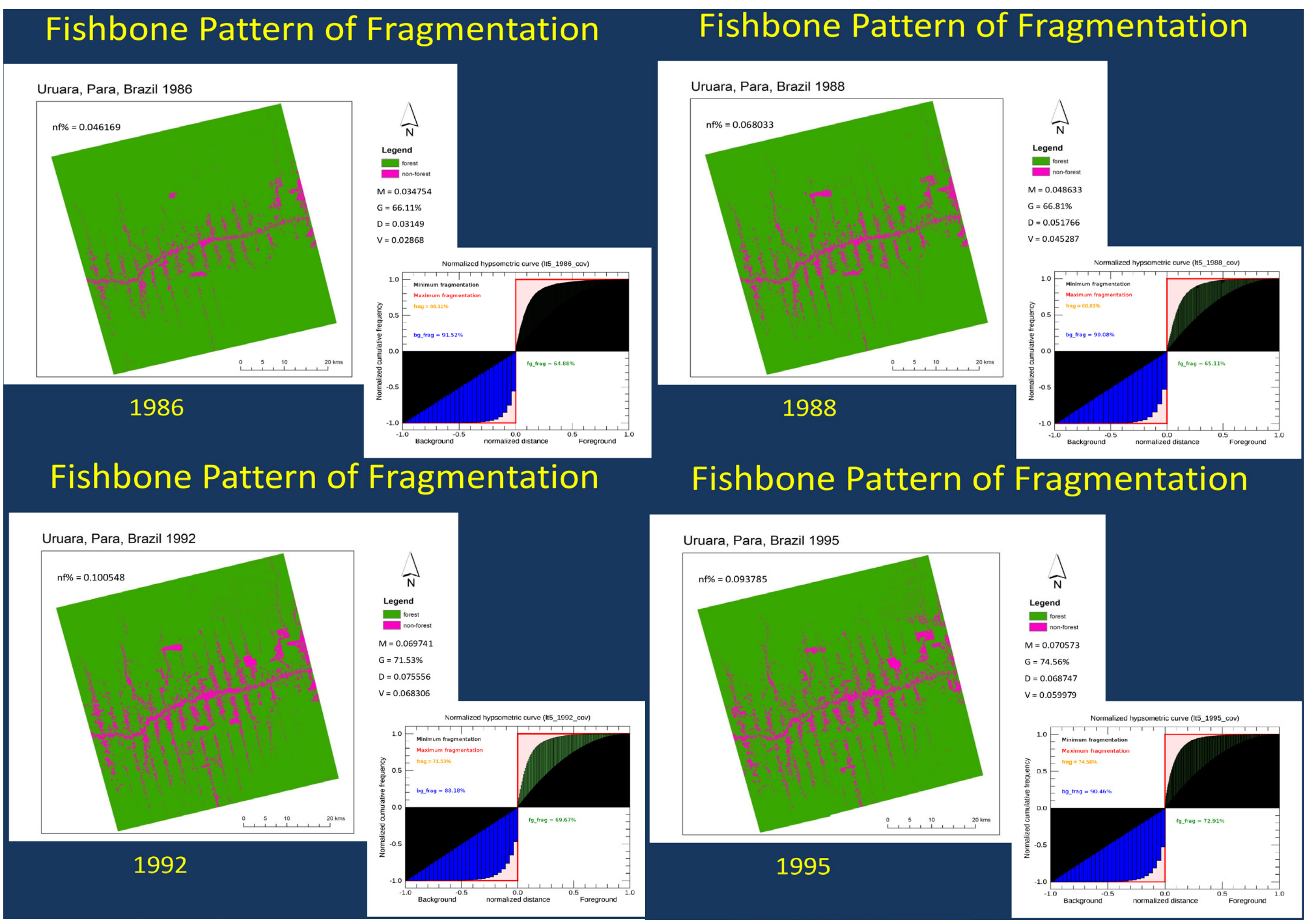

Figure 3.a. Progression of fishbone pattern of fragmentation, Uruara, Pará, Brazil, 1986-1995.

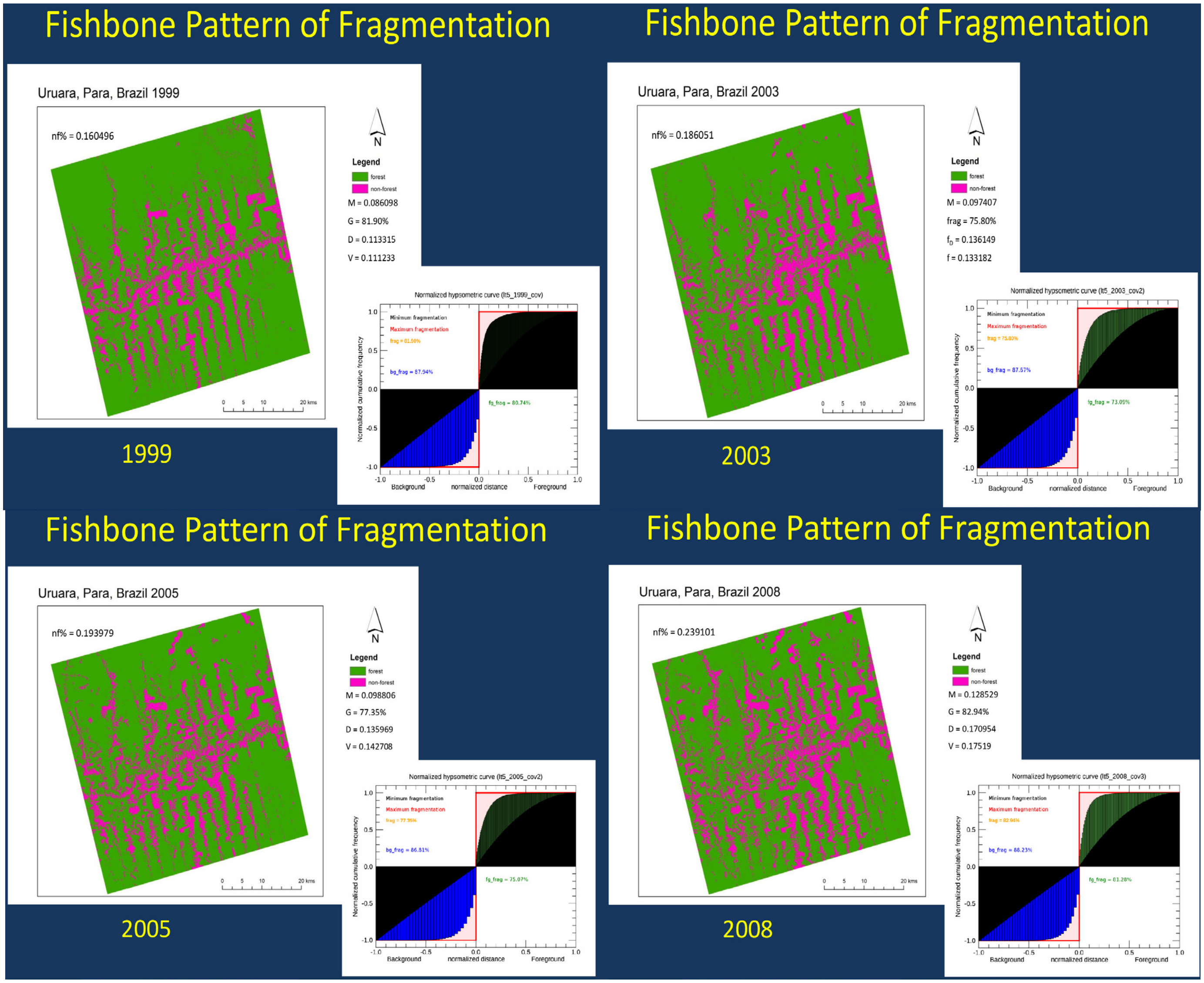

Figure 3.b. Progression of fishbone pattern of fragmentation, Uruara, Pará, Brazil, 1999-2008.

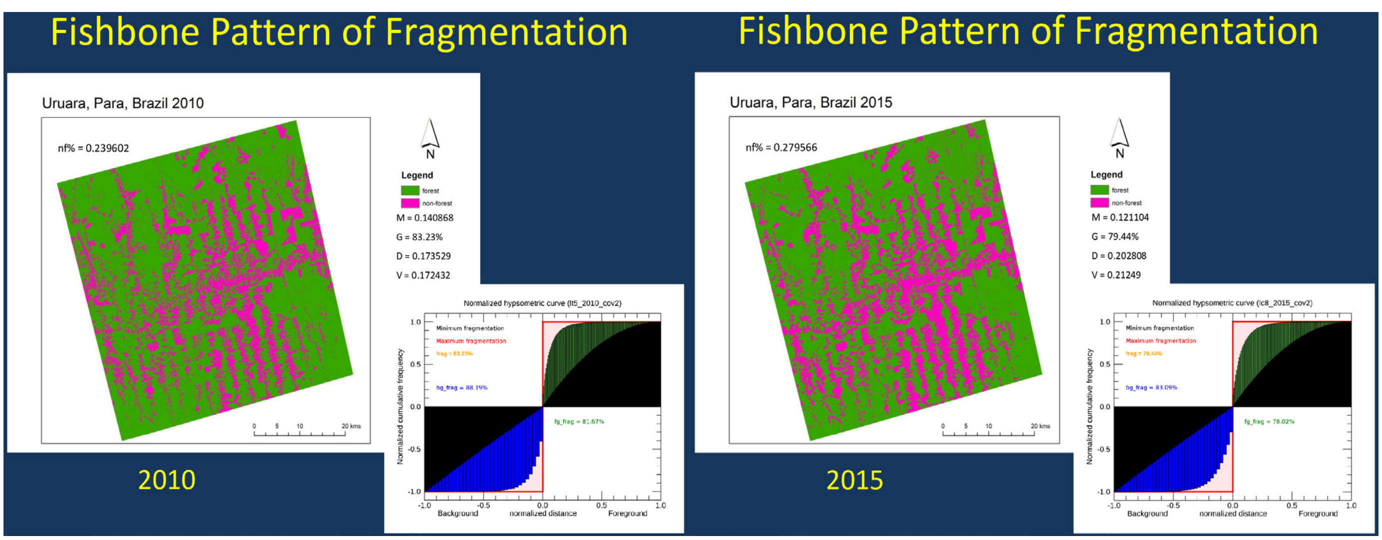

Figure 3.c. Progression of fishbone pattern of fragmentation, Uruara, Pará, Brazil, 2010-2015.

Table 2. Fragstats class metrics and the derived fragmentation indices *M, D, F* and *G**.

Year	CA	PLAND	TE	FRAC_MN	PAFRAC	CLUMPY	%nf	*M*	*D*	*F*	*G*
1986	748834.3	95.3831	8882580	1.0344	1.3777	0.8058	0.046169	0.034754	0.015745	0.01434	0.6611
1988	730608.2	93.1967	12268560	1.0367	1.4195	0.8138	0.068033	0.048633	0.025883	0.022644	0.6681
1992	705263	89.9452	17287260	1.0374	1.4137	0.8164	0.100548	0.069741	0.037778	0.034153	0.7153
1995	704627.6	90.6215	17412450	1.0355	1.4069	0.8015	0.093785	0.070573	0.034374	0.029989	0.7456
1999	652200.1	83.9504	20428620	1.0374	1.381	0.8531	0.160496	0.086098	0.056658	0.055617	0.819
2003	632460.4	81.3949	22761630	1.0386	1.3946	0.8545	0.186051	0.097407	0.068075	0.066591	0.758
2005	626052.1	80.6021	22966770	1.0396	1.3773	0.8578	0.193979	0.098806	0.067985	0.071354	0.7735
2008	591004.7	76.0899	29027400	1.0397	1.3874	0.8456	0.239101	0.128529	0.085477	0.087595	0.8294
2010	590615.3	76.0398	31803510	1.0393	1.3955	0.8312	0.239602	0.140868	0.086765	0.086216	0.8323
2015	559574.3	72.0434	26613270	1.0406	1.3875	0.8721	0.279566	0.121104	0.101404	0.106245	0.7944

*note: *G* is derived from GUIDOS Tool Box

Below are the graphs of the four fragmentation indices, by chronological and %nf order. *G* starts way too high at 66.11% for 0.046 percent non-forest (**Table 2 and Figures 4**) and increases even further over the years (**Figures 4 and 5**). The other three start close to 0 and only gradually increase with deforestation.

Since deforestation increases almost monotonically over time, the same holds true also for fragmentation against increasing %nf (**Figure 5**).

Looking at just *M, D* and *F* against %nf, all three behave similarly (generally increase or decrease between the same periods, albeit at different amplitudes), until just before the very end, where *M* drops sharply while *D* and *F* increases (**Figure 6**). On identifying the two scenes during this period of deviation (2010 and 2015 in **Figure 3**), clustering increased (**Table 2**), which *D* and *F* detected and thus likewise increased, while *M* decreased in the same period. Hence *M* may not respond well to patch clustering or aggregation.

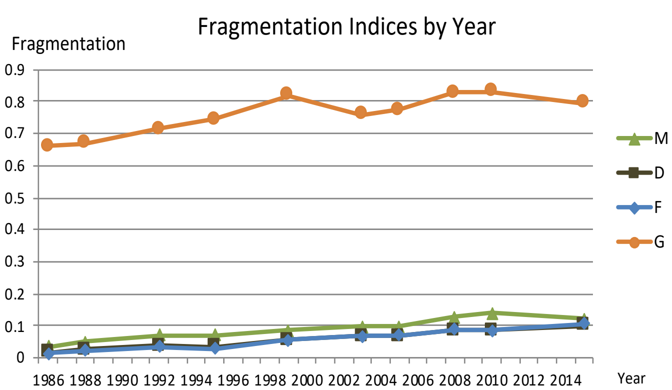

Figure 4. Fragmentation indices arranged chronologically.

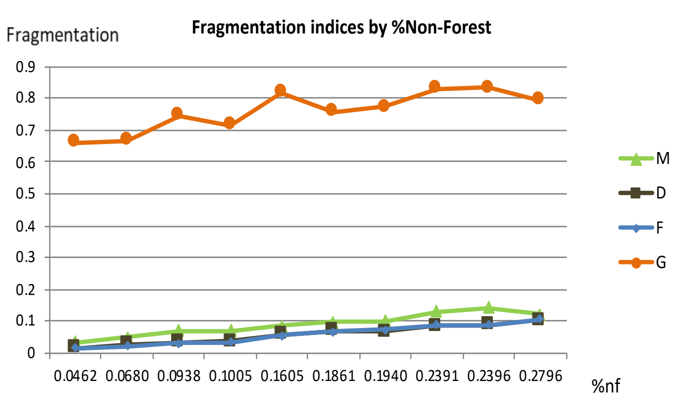

Figure 5. Fragmentation indices by increasing percent non-forest.

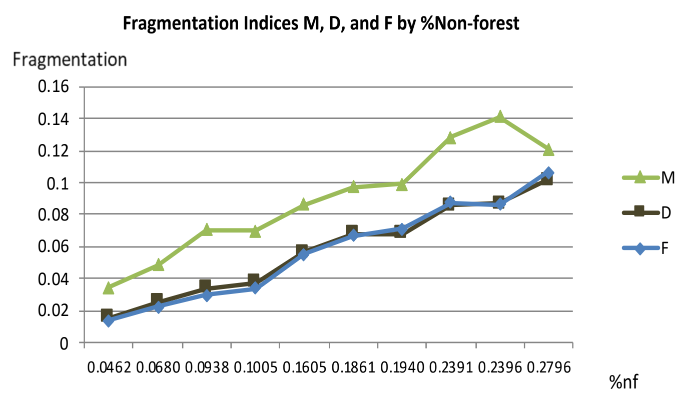

Figure 6. Fragmentation indices M, D, and F for increasing percent non-forest.

CONCLUSIONS

D is modelled on landscape characteristics theorized to affect fragmentation, i.e., how much of the landscape are still fragments, how complex the shapes of these fragments are, and how these fragments are arranged in space. These same landscape indices have been found by others to be robust for fragmentation studies (*Bogaert et al. 2000; Butler et al. 2004; Wang et al. 2014*). *D* is unique from other fragmentation indices in that it takes the product of normalized values that are unweighted so as not to introduce additional bias.

Smaller spatial supports could render *D* problematic. To overcome this limitation for smaller sampling extents, *F*, an analog of *D*, is proposed, that utilizes the fractal mean in lieu of the fractal dimension, with an adjustment for scale. On the small sample size in this study, *F* approximates *D* very closely.

The Matheron index is based solely on normalized dissimilar joins in binary images. So, as long as the patches retain their shapes, sizes, and numbers, changing the spatial arrangement or distribution in space of the patches by clustering them would not change the Matheron index. Island biogeography shows that patch number, sizes, and arrangement affect local colonization, species richness, and extinction (*Lindgren and Cousin 2017; Turner et al. 2001*). Hence patch clustering should be considered in fragmentation studies.

Here, the behavior of *G* was way beyond the theorized Kuznets curve of *Lambin and Ehrlich* (*1997*). Also, the proposed indices are sensitive to aggregation while the Matheron index may not be. Of course, this is but a limited study, too small a sample size for a full review of the behavior of the different fragmentation indices considered. What it does show, however, is insight into the possibility of a fragmentation index that is sensitive to patch density, shape, and arrangement, and that should be able to differentiate between cover loss and fragmentation.

Fragmentation need not be limited to only forest cover studies. Green space fragmentation could be measured for an ideal urban configuration, and then with measures for contiguity, could be used as design standards. Agricultural land, though ideal if unfragmented, would have to be, due to topographic relief, access, ownership, and other parameters that make the landscape heterogenous for agriculture. How fragmented could farmlands be while still attaining acceptable levels of efficiencies to enhance sustainability? Research questions such as these can be addressed by the proposed indices.

Of course, the ultimate motivation for these methods is to seek answers to the question of spatial optimality of fragmented landscapes. Is there a fragmentation configuration that is optimal for all stakeholders, i.e., humans, the biota, and the physical environment? Should farmers clear for long lots or compact farms? *Hof and Flather* (*2007*) outline closed, open, and heuristic approaches for spatial optimality, but conclude that the greatest need for this research topic is still the relevant ecological relationships, the elusive link between landscape pattern and ecosystem processes. Hence, an unambiguous measure for landscape fragmentation is requisite to link its patterns to ecological processes on the landscape.

One might raise the question of why still use Fragstats, with its inherent limitations, when code and scripts are now available for fragmentation studies. Indeed code and Indeed code and plug-ins to ArcMap Spatial Analyst are available for free, the foremost of which is LST ver 2 from the University of Connecticut (*Hurd undated)*, which was reviewed earlier, but which produces fragmentation topologies, rather than an index. *MacLean and Congalton* (*2013*) offer PolyFrag, but it works only for vector data whereas most fragmentation studies use rasters. *Bosch* (*2019*) offers PyLandStats, an open source library of Python codes for landscape metrics, but none specifically for fragmentation. *Jung* (*2016*) offers LecoS, a Python plug-in. Currently, it offers only eight landscape indices, none of which computes forest fragmentation directly. The Fractal Dimension index is included, but no index is offered for patch aggregation, which is essential for a fragmentation index to be developed, based on the theory presented in the paper. Hence, none of the above can produce as yet a single-valued fragmentation index as proposed in this paper.

Bogaert (*2003*) did state that without an accepted fragmentation index, “the correspondence between fragmentation experiments and predicted effects” would blur. Recognition and use by the community of a single-valued fragmentation index could pave the way for cross-study comparisons of the effects of fragmentation on any landscape attribute at the sampling sites. The authors offer the fragmentation index *D*, if all spatial supports used contain 10 or more patches, and *F*, for less than 10. Hopefully cross-study comparisons could lead to generalizations to further theory in landscape science.

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