Crop burning in north India: How has the practice changed over previous years and contributed to air pollution in the region? And why?

Éléonore Karen Monette

Capstone Thesis

BSc Global Responsibility & Leadership
University of Groningen

June 2021

Abstract:

Considered to be a global hotspot of elevated aerosol loading, the densely populated north Indian region is one of the few to experience enhancements in fine particulate matter ($PM_{2.5}$, small particulate matter of 2.5 microns or less in diameter). The World Health Organization and the Indian Central Control Board daily threshold for unhealthy air being often exceeded, $PM_{2.5}$ poses significant health risks, reduces visibility and has strong climate impacts. Due to the crop residue burning practice, $PM_{2.5}$ emission spikes are observed in the post-monsoon season, during the months of October and November. Using the Fire INventory of the National Center for Atmospheric Research (NCAR) version 1.0 (FINNv1), and through plotting a time series of daily fire emission values as well as analysing FINN maps of north-India, this paper develops an understanding of the evolution of fires. Along the Indo-Gangetic Plain, emissions tend to increase after 2009, year marking the promulgation of the Punjab and Haryana groundwater acts of which unintended consequences strengthened air pollution enhancements. Additionally, emission peaks have shifted to the first fortnight of November, a period during which conditions lead to atmospheric stability and thus hinder air pollution dispersion.

Keywords:

Crop residue burning

Indo-Gangetic Plain

Biomass burning

Air pollution

Agriculture

Groundwater Acts

TABLE OF CONTENTS :

Chapter 1: Introduction

Regarded as a global hotspot of elevated aerosol loading, the highly populated Indo-Gangetic Plain (IGP) experiences air pollution spikes following the cessation of the monsoon rains during the autumn season (Balwinder-Singh et al., 2019; Ojha et al., 2020). The region is one of the few to face $PM_{2.5}$ – small particulate matter of 2.5 microns or less in diameter – enhancements (Ojha et al., 2020). Fine particulate matter has climate and health impacts and significantly reduces visibility (Wiedinmyer et al., 2010; Ojha et al., 2020). Indeed, Delhi's daily mean levels of surface $PM_{2.5}$ often exceed both the World Health Organization and the Indian Central Pollution Control Board daily threshold for unhealthy air (Kulkarni et al., 2020; Tiwari et al., 2013, as cited in Cusworth et al., 2018). As such, compared to the national average, research has shown that residents of Delhi suffer from air pollution diseases at a rate twelve times higher (Kandlikar & Ramachandran, 2000, as cited in Cusworth et al., 2018).

As a major agrarian country, India has an estimated annual crop production of 627.96 Mt/year, of which sugarcane and rice paddy production represent the largest part (Jain et al., 2014). Consequently, Indian agriculture is also one of the major in terms of agricultural waste, with 620.43 Mt/year of dry residue generated, of which an estimated 16% was burnt in-situ during the 2008-2009 period (Jain et al., 2014). Despite banned in 2010 by the National Green Tribunal Act (Nain Gill, 2010 as cited in Cusworth et al., 2018), crop residue burning remains a widespread practice as it allows farmers to efficiently and cheaply proceed to the next crop unhindered by previous crop residues (Balwinder-Singh et al., 2019; Cusworth et al., 2018; Jain et al., 2014). Agriculture in the states of Punjab and Haryana is based on two growing seasons: a winter wheat crop, harvested from April-May and a summer rice crop, harvested from October-November (Vadrevu et al., 2011, as cited in Cusworth et al., 2018). Consequently, considerable air pollution spikes have been observed during the post-monsoon season (Balwinder-Singh et al., 2019; Cusworth et al., 2018; Roozitalab et al., 2021). In fact, countryside burning in Punjab and Haryana – known as the breadbasket of India – is known to strongly impact urban air quality (Bikkina et al., 2019; Cusworth et al., 2018). As such, rice crop burning in both states have contributed from 25% to 70% of the $PM_{2.5}$ in New Delhi, during late October and early November (Balwinder-Singh et al., 2019). Indeed, air quality degradation led to a ~60% increase in Delhi mortality between 2000 and 2010 (Cusworth et al, 2018). Additionally, air pollution is responsible for an overall life expectancy reduction of six years in the New Delhi capital region (Balwinder-Singh et al., 2019). Importantly, the unfavourable topography of the Himalaya extending from northwest to southeast tends to favour the accumulation of aerosols along the IGP (Mogno et al., 2021; Ojha et al., 2020). It should also be note that biomass burning

(BB) mainly emits CO_2 as well as other pollutants and Green House Gases such as CO, NHMC, NMVOC and $PM_{2.5}$ (Jain et al., 2014). Importantly, fire or smoke-emitted PM comprises black carbon (BC) and organic aerosol (OA) known as carbonaceous aerosol (Akagi et al., 2011; Bond et al., 2013, as cited in Carter et al., 2020).

In order to explore the evolution of these burnings, this paper will address the following Research Question: "Crop burning in north India: How has the practice changed over previous years and contributed to air pollution in the region? And why?". Following a description of the issue at hand, on a global and regional scale, the analysis will be centred on the Fire INventory of the National Center for Atmospheric Research (NCAR) version 1.0 (FINNv1) which produces 1km resolution, daily global estimates of emissions from open BB (NCAR, 2021; Wiedinmyer et al., 2010). For developing a better understanding of the evolution of the practice, the paper will look at FINN daily fire emissions maps for the October to November period for the years 2014 to 2020. Additionally, intense fires during this period will be closely examined. Secondly, a time series of daily FINN emission values will be plotted for the years 2002 to 2019, followed by an analysis of the emissions peak occurrence (NCAR, 2021). Further, limitations and uncertainties are explored, followed by a conclusion addressing the main results, providing potential solutions and drawing similarities with other case studies.

Chapter 2: Methodology

Firstly, a literature review was conducted in order to provide sufficient background information regarding the issue of air pollution, globally and for north India, what it entails as well as its causes and consequences. As such, key terms such as 'crop residue burning', 'Indo-Gangetic Plain', 'air pollution', 'post monsoon air quality degradation, 'fires, 'particulate matter' and 'agriculture' were searched using Google Scholar. During this process, it was attempted to include recent as well as both international and/or Indian academic papers.

Then, the National Center for Atmospheric Research (NCAR) provides near-real time fire emissions from the Fire INventory of NCAR version 1.0 (FINNv1) (Wiedinmyer et al., 2010). As such, FINN is based on Moderate Resolution Imaging Spectroradiometer (MODIS) Rapid Response fire counts (Earth Data NASA, 2020; NCAR, 2021). As detailed by Wiedinmyer and colleagues (2010), FINN produces 1km resolution, daily global estimates of emissions from open BB. Consisting of agricultural fires, prescribed burnings and wildfire, open BB does no comprise biofuel use and trash burning (Wiedinmyer et al., 2010). NCAR provides online daily fire emission maps of various regions from the 01.05.2012 to now (NCAR, 2021). As such, daily maps of Asia were selected for the months of October to November, for the years 2014 to 2020 (figure 1). These images were further cropped in a way to solely focus onto north-India. Additionally, maps of the 10.10.2014, the 15.11.2014, the 09.10.2015, the 07.10.2016, the 13.10.2017 and 12.10.2018 were unavailable and thus could not be used. Also, as maps were shown in a 6x10 grid, maps of the 31.10 of years 2019 and 2020 were removed. Additionally, deriving from figure 1, table 2 was developed using the daily intense fire emissions in the IGP during the October-November period of years 2014 to 2020. Days identified in dark red, as shown by the scale of figure 1, indicate the presence of various intense fire emissions of 1.e+13 molecules/cm^2/s (NCAR, 2021).

Further, daily FINN emission values were used to plot a time series of October and November for the years 2002 to 2019 (figure 2). It is important to note that data was unavailable for the year 2011. Data was plotted representing the emission values and every October and November period following one another. This allowed for a clear comparison of emissions of October-November of the 18 years. Additionally, emission peaks of every October-November period were also plotted separately in order to analyse when to compare exactly occurred (figure 3).

Chapter 3: Background and literature review

The problem of air pollution: globally

According to the World Health Organization (WHO), outdoor air pollution accounted for 4.2 million premature deaths globally in 2016 (WHO, 2018). Even though this represents a worldwide issue, residents of low- and middle-income countries have disproportionately been affected by air pollution: an estimated 91% of these premature deaths is located in these countries. The WHO Western Pacific and South-East Asia regions are among the worst areas in this regard (WHO, 2018). As table 1 indicates, the mean annual exposure to $PM_{2.5}$ – small particulate matter of 2.5 microns or less in diameter – air pollution (micrograms per cubic meter) is found to be the highest in South Asia, in the Middle East and in Central Africa. As such, India is ranked fourth, after Nepal, Niger and Qatar (WHO, 2018; World Bank, 2017). Additionally, it has been found in 2016 that merely 9% of that world population was living in places where the WHO air quality guidelines were met (WHO, 2018). Additionally, the WHO has identified major sources of outdoor air pollution: fuel combustion from motor vehicles, as for example cars and heavy-dusty vehicles, heat and power (oil and coal power plants and boilers) as well as industrial facilities, such as manufacturing factories, oil refineries and mines (WHO, n.d.). Additionally, agricultural and municipal waste sites and waste incineration/burning, as well as residential cooking, lighting and heating with polluting fuels have been listed (WHO, n.d.). Six main pollutants composing air pollution have been identified: ground-level Ozone, Carbon Monoxide (CO), Nitrogen Oxides (NO_2), lead, Particulate Matter (PM) as well as Sulphur Oxides (SO_2) (United States Environmental Protection Agency, 2021). The 4.2 million premature deaths per year worldwide are caused by exposure to $PM_{2.5}$, resulting in cancers as well as respiratory and cardiovascular diseases (Carter et al., 2020; Ojha et al., 2020). Harmful to the human respiratory system (Balakrishnan et al., 2019, as cited Takigawa et al., 2020), aerosols' small size and related ability to penetrate and deposit in lungs can cause respiratory infections, lung cancer and asthma (Brook et al., 2010; Pope III and Dockery, 2006, as cited in Carter et al., 2020), particularly the high levels of PM from fire events (Liu et al., 2015; Reid et al., 2016; Williamson et al., 2016, as cited in Carter et al., 2020). Further, research indicates that there is no threshold below which particle exposure has no dangerous impacts (Chen et al., 2016).

Air pollution in India

As identified above, the problem of air pollution in India is critical: it has caused an estimated 1.09 million deaths in 2015, costing the economy 3% of the Indian GDP (Landrigan et al., 2017 as cited in Balwinder-Singh et al., 2019). Indian aerosol emissions include industrial, residential energy usage, transportation and biomass burning (Guttikunda et al. 2014; Sharma et al. 2018; Saraswat et al. 2013, as cited in Takigawa et al., 2020). Importantly, South Asian megacities – such as New Delhi – are global hotspots for poor air quality, and spikes in air pollution are occurring prevalently in the Indo-Gangetic Plain (IGP) following the monsoon rains during the autumn season (Balwinder-Singh et al., 2019). Acknowledged as the second-highest risk factor in India (Shyamsundar et al., 2019), polluted air poses serious risks to the densely populated IGP, which accounts for a seventh of the world's total population (Ojha et al., 2020). The region, known as a global hotspot of elevated aerosol loading, is one of the few to observe $PM_{2.5}$ enhancements (Ojha et al., 2020). Studies have shown that residents of Delhi – which has a population of ~16.5 million (Cusworth et al., 2018) – have suffered from air pollution related diseases at a rate twelve times higher than compared to the national average (Kandlikar & Ramachandran, 2000, as cited in Cusworth et al., 2018). Researchers have estimated an overall reduction of six years in terms of life expectancy in the New Delhi capital region (Balwinder-Singh et al., 2019). Additionally, 16,000 premature deaths are caused yearly due to air pollution in the region (Balwinder-Singh et al., 2019). Finally, daily mean levels of surface $PM_{2.5}$ in Delhi often exceed the WHO and the Indian Central Pollution Control Board daily threshold for unhealthy air of 25 $\mu g\ m^{-3}$ and of 60 $\mu g\ m^{-3}$ respectively, as daily mean levels of surface $PM_{2.5}$ reach more than 100 $\mu g\ m^{-3}$ (Kulkarni et al., 2020; Tiwari et al., 2013, as cited in Cusworth et al., 2018).

Caused by residential energy consumption and emissions from land transportation – particularly in cities – anthropogenic emissions show a continuous pattern, as similar $PM_{2.5}$ levels have been observed during all seasons (Mogno et al., 2021), compared to pyrogenic emissions. Observed as episodic, these ensue in localized contributions over upper and central regions of the IGP during all non-monsoonal seasons, with the largest effect during the post-monsoon seasons (Mogno et al., 2021). This pattern corresponds to the post-harvest season in the agricultural calendar. Wiedinmyer and colleagues suggest that open BB emissions can contribute to climate forcings and air quality problems locally, regionally, and globally (Crutzen & Andreae, 1990, as cited in Wiedinmyer et al., 2010). These emissions caused 33% of global CO emissions, 62% of global primary particulate OC and 27% of global particulate BC emissions (Emmons et al., 2010, as cited in Wiedinmyer et al., 2010).

Crop residue burning in North India

As an important agrarian country, India has an estimated annual crop production of 627.96 Mt/year, including a sugarcane production of 285.03 Mt/year and a rice paddy production of 153.35 Mt/year (Jain et al., 2014). Estimated at 500 Mt/year according to Bhuvaneshwari and colleagues, and 620.43 Mt/year as indicated by Jain and colleagues, dry residue generated makes Indian agriculture one of the major ones in terms of agricultural waste (Bhuvaneshwari et al., 2019; Jain et al., 2014). Serving as animal feed, thatching for rural homes, residential cooking fuels and industrial fuel, yet, an important part of crop residues is left on the fields (Jain et al., 2014). In order to clear the field rapidly and inexpensively, farmers opt for in-situ burning, allowing the next crop to make a start unhindered by previous crop residues (Jain et al., 2014). Additionally, alternative practices such as integrating crop residue into the soil being costly and time consuming, only enhances the burning incentives (Balwinder-Singh et al., 2019). As such, it is indicated that around 16% of the total amount of residue generated in 2008-2009 was burnt in-situ (Jain et al., 2014). As suggested by Cusworth and colleagues, farmers burn crop residue that is abundantly left by a mechanized combine harvesting compared to traditional methods. In fact, since the 1980s farmers have switched to such technique as it enhances efficiency of their agricultural activity (Balwinder-Singh et al., 2019; Cusworth et al., 2018) as well as allows for a decrease in costs. To exemplify such process, more than ¾ of rice is harvested using such a combiner harvester in Punjab (Kumar et al., 2015, as cited in Cusworth et al., 2018). Even though banned in 2010 by the National Green Tribunal Act (Nain Gill, 2010 as cited in Cusworth et al., 2018), such cheap agricultural burning practice is widely used by farmers after harvest (Cusworth et al., 2018).

Estimated at 98.49Mt/year using coefficients of Jain and colleagues and at 131.86 Mt/year using IPCC coefficients for the 2008-2009 period, agricultural residue burned on farm is principally located in the states of Uttar Pradesh, Punjab and Haryana (Jain et al., 2014). These states accounted for 23%, 22% and 9 % of in-situ burnt residues, respectively (Jain et al., 2014). Between 2008 and 2009, crop residues burnt in Punjab have been estimated to reach 13.30Mt/year (based on IPCC default coefficients) and 21.32 Mt/year (based on coefficients of Jain and colleagues). An estimated 6.86 Mt/year and 9.18 Mt/year of crop residues has been burned in the province of Haryana, using IPCC default coefficients for the former, and the coefficients of Jain and colleagues for the latter (Jain et al., 2014). Additionally, during the 2008-2009 period, burnings in Uttar Pradesh reached 22.25 Mt/year (coefficients of Jain and colleagues) to 22.38 Mt/year (IPCC default coefficients) (Jain et al., 2014). Balwinder-Singh and colleagues indicate that an estimated 23 million tonnes of rice residues are burned in Punjab and Haryana on a yearly basis (Ahmad et al., 2015, as cited in Balwinder-Singh et al., 2019).

Agriculture in the states of Punjab and Haryana is based on two growing seasons: a winter wheat crop and summer rice crop, harvested from April-May, and from October-November respectively (Vadrevu et al., 2011, as cited in Cusworth et al., 2018). Strong biomass-burning activities have been observed over the northwest IGP during October, and lower ones during December (Ohja et al., 2020). As such, it is important to note the increasingly common dramatic air pollution spikes during the post-monsoon season, from October to November (Balwinder-Singh et al., 2019; Cusworth et al., 2018; Roozitalab et al., 2021). These strong increases are more considerably observed in Delhi than in Kanpur and Varansi with mean relative effects from biomass burnings of estimated 30.2%, 19.6% and 9.4% respectively (Ojha et al., 2020). From late October to early November, 25%-70% of New Delhi's $PM_{2.5}$ pollution derives from rice crop residue burning from states of Punjab and Haryana, known as the 'breadbasket' of India (Balwinder-Singh et al., 2019; Cusworth et al., 2018). Thus, crop residue burning emissions are a major contributor to $PM_{2.5}$ aerosols during the months of October and November. In fact, contributions of biomass burning emissions to nitrate (NO_3^-), ammonium (NH_4^+), organic matter (OM) and elemental carbon (EC) are seen to drastically decrease in December. In fact, the IGP is then impacted by 90-100% by anthropogenic emissions (Ojha et al., 2020).

Biomass Burning emissions

Comprising agricultural and other prescribes burnings as well as wildfires, biomass burning (BB) is known to produce a range of emissions: carbon dioxide and oxides of nitrogen, as well as volatile organic compounds (VOCs) and PM (Akagi et al., 2011, as cited in Carter et al., 2020). Importantly, fire or smoke-emitted PM includes black carbon (BC) and organic aerosol (OA) known as carbonaceous aerosol (Akagi et al., 2011; Bond et al., 2013, as cited in Carter et al., 2020). As such, burnings result in the emission of greenhouse gasses, air pollutants, particulate matter and smoke (Jain et al., 2015), contributing to global warming and causing health issues (Bhuvaneshwari et al., 2019). Accounting for 91.6% of total emissions (149240.68 Gg/yr of CO_2), CO_2 is the main air pollutant emitted due to crop residue burning in India for the year 2008–09 (Jain et al., 2014). The remaining 8.4% include 13 other pollutants and GHGs, namely CO, NHMC, NMVOC and $PM_{2.5}$ (Jain et al., 2014). As mentioned, fine particulate matter has climatic impacts, significantly reduces visibility and strongly affects human health (Ojha et al., 2020; Wiedinmyer et al., 2010). Also, biomass burning aerosol (BBA) can influence the climate system through absorbing and scattering radiation (Bond et al., 2013, as cited in Carter et al., 2020). It is important to further note that countryside burning practices strongly impact urban air quality (Bikkina et al., 2019). In fact, Cusworth and colleagues have found a ~60% increase in Delhi mortality due to the degradation of air quality between 2000 and 2010 (Cusworth et al., 2018). Balwinder-Singh and colleagues (2019) indicate that rice crop residue burning in Punjab and Haryana have contributed from 25 to 70% of the $PM_{2.5}$ in New Delhi, during late October and early November (Balwinder-Singh et al., 2019). As such, agricultural burning in states of Punjab and Haryana strongly contributed to such degradation (Cusworth et al., 2018). Further, in order to comprehend the time evolution of Delhi's high pollution episodes in 2019, Takigawa and colleagues (2020) have looked at the mean transit time of particles emitted from BB, and the potential pathways, especially from Punjab (Takigawa et al., 2020). As such, following their release, the mean transit time to Delhi was calculated to be 1 to 2 days (Takigawa et al., 2020). Additionally, during the dry monsoon season, particles released from Punjab fires are expected to reach the IGP within a week (Takigawa et al., 2020). Besides air quality degradation, biomass burning results in the loss of nutrients. Due the in-situ burning of rice straw, wheat straw and sugarcane trash, nutrient loss has been estimated to reach 1.43Mt/year (Jain et al., 2014). These include Nitrogen (N) loss of an estimated 0.394Mt/year, Phosphorus (P) loss of 0.014Mt/year and 0.295Mt/year of Potassium (K) loss (Jain et al., 2014).

Chapter 4: Results and discussion

As Wiedinwyer and colleagues indicate, FINNv1 provides daily high-resolution spatial and temporal estimates of open BB emissions, globally (Wiedinmyer et al., 2010). Estimated by Emmons et al. (2010) for the 2008 global totals, FINNv1 emissions made up 27% of global particulate BC emissions, 33% of global CO emissions and 62% of global primary particulate OC emissions (Emmons et al., 2010, as cited in Wiedinmyer et al., 2010). This paper will analyse FINN daily satellite pictures, ranging from the 1st of October to the 30th of November, between 2014 and 2020 in order to study to intensity and evolution of post-monsoon burning. As such, figure 1 presents the FINN fire emissions for the IGP regarding the October-November period as a 6x10 grid for the seven years studied.
From figure 1, the highest fire intensities in the IGP during the October-November period were highlighted for the seven years of data (2014-2020) as provided by table 2. These days were selected based on the presence of various fires identified in dark red on the maps, indicating fires of 1.e+13 molecules/cm^2/s. Regarding data from 2014 and 2015, intense burnings start at a similar period (around the 21.10/22.10) and end on the 18th of November. Data from 2016 show the intense fires start slightly earlier: on the 17.10. Further, two intense burning periods are identified in 2017. The first one occurs from the 16.10 until the 06.11, and then resumes from the 19.11 to the 24.11. The start of the intense fires stats at a similar time in the year compared to data from 2014 and 2015. However, it ends on the 24.11, similarly to data from the previous year. The 2019 data show a more scattered period. Besides a period from the 19.11 to the 30.11, intense fires occur at more random times: from the 4th to the 6th of November, on the 9th, 18th, 20th and 25th of November. Finally, the last year observed has a shorter period compared to other years (namely from the 19th to the 11th of November) with two intense fires identified: on the 13th and on the 20th of November. From this analysis, it can be concluded that the results appear quite scattered. As such, no clear shift in terms of burning intensity can be deduced from the data. Intense fires start slightly earlier in October 2016 and 2017. Intense burnings end slightly later in November 2017 and 2018. Also, data from 2017, 2019 and 2020, and especially the one from 2019, appear more scattered than data from other years.

Secondly, FINN emission values were plotted for the period ranging from the 1st of October to the 30th of November for the years 2002 to 2019 (figure 2). The highest daily emission values were found on the 3rd of November 2013. As such, the 2013 emissions peaks has more than tripled since 2002 and has doubled compared to the 2010 peak. Overall, daily emission values have been found to increase, with a peak in 2013, a small decrease in 2017 followed by an increase in 2019. Deriving from figure 2, peak occurrences of the October-November period of years 2002 to 2019 were plotted, as shown in figure 3. As such, a clear trend regarding peaks in daily emission values can be observed as peaks in emissions are occurring later year after year. The peak has shifted from the 20th of October to the 10th of November of the years of 2002 and 2016, respectively.

In order to make sense of the results of the plotted FINN data, it is important to look at the groundwater conservation laws and their consequences on air pollution. As Balwinder-Singh and colleagues (2019) indicate, groundwater depletion due to irrigating rice presents a strong threat to national food security. Due to rice-favouring subsidies, attempts at replacing rice with less water-depending crops have failed. However, coercing farmers to adapt agronomic practices has been more successful in that regard. It has to be noted that rice tends to be transplanted before the onset of monsoon (Balwinder-Singh et al., 2019). As such, the Haryana Preservation of Subsoil Water Act and the Punjab Preservation of Subsoil Water Act, known as groundwater acts and promulgated in March 2009, have banned transplanting before the 10th of June, and later altered to the 20th of June. Consequently, the bill has successfully delayed the transplanting, as less than 40% of the total rice area has been planted on or before the 28th of June since 2009. Indeed, rice harvest has shifted accordingly: in Punjab, an estimated 40% of the rice crop was harvested by the 26th of October, and further decreased to 14% (Balwinder-Singh et al., 2019).

However, following the groundwater acts, residue burnings have shifted as well as increased. Before the implementation of the acts, emissions peak occurred on the 24th of October, with daily 490 fires, and shifted to the 4th of November at 681 fires per day. Indeed, these acts resulted in a concentration of crop residue burning within a smaller time window, later in the season, leading to a 39% higher peak intensity (Balwinder-Singh et al., 2019). As mentioned, the burnings' maximum occurrence thus shifted to the first fortnight of November, a period during which winds are weaker and temperatures in New Delhi are 3°C lower than compared to the previous burning period, the second fortnight of October. As such, these conditions lead to atmospheric stability and thus hinder air pollution dispersion. Indeed, it can be noted that average daily $PM_{2.5}$ concentrations in November increased of 29% after the implementation of the acts (Balwinder-Singh et al., 2019). Additionally, following the acts, the total rice area and aggregate rice production noted a 10% and 11% increase, respectively. Consequently, these increases strongly enhance the crop residue burnings in the area (Balwinder-Singh et al., 2019).

Consequences of groundwater acts as discussed above allow for an improved understanding of the obtained results. Firstly, emissions peaks are seen to occur later in the post-monsoon season: a 21-day shift has been identified between emission peaks in 2002 and 2016. Following 2012, it can be noted that the peak occurrence has not shifted according to the previous trend.

As figure 3 shows, the 2015 and 2017 peak occurred slightly earlier than expected. Also, figure 2 further indicates an increase in emission values after 2009, as Balwinder-Singh and colleagues (2019) suggest. Even though an increase in values since 2002 can be observed, the impacts of the 2009 groundwater acts only strengthened the mentioned trend. Indeed, a 122% and a 177% increase in peak emission values can be noted between 2009 and 2012 as well as between 2009 and 2013, respectively. As mentioned earlier, 2013 emission values have tripled since 2002.

Limitations of FINN

Regarding FINN, the authors point out that emission from open BB present important regional variability (Wiedinmyer et al., 2010). As such, the non-methane organic compounds (NMOC) emissions, for example, are proven to vary daily, enhancing the need for high temporal resolution (Wiedinmyer et al., 2010). Compared to other inventories, one of the advantages of FINNv1 emission estimates concerns its high spatial and temporal resolution as well as rapid availability (Wiedinmyer et al., 2010). FINNv1 allows to easily incorporate adaptations in order to target regions of interest more precisely (Wiedinmyer et al., 2010). However, several limitations and uncertainties are explored. Firstly, estimates are identified to be very uncertain. Additionally, due to global fires being small, the largest uncertainties derive from missed fires, resulting in the underestimation of the number of fires as well as the overestimating of the small fires' size detected. Authors note that these errors tend to cancel (Wiedinmyer et al., 2010). As remote sensing thermal anomaly products are not able to detect most fires less than ~100 ha (Hawbaker et al., 2008, as cited in Wiedinmyer et al., 2010), an important source of emissions to the atmosphere may not be included into the dataset (Wiedinmyer et al., 2010). Moreover, further uncertainty could occur based on inaccurate fuel loading, parameterizations of combustion completeness, misidentification of the land cover, as well as uncertainty and natural variation in the emission factors (Akagi et al., 2010, as cited in Wiedinmyer et al., 2010). Besides, cloud cover and satellite overpass timing may hinder the accurate detection of fires (Wiedinmyer et al., 2010). Further, ecosystem type determination, which differs from one land cover data product to another, may further lead to uncertainty. In fact, the study by Wiedinmyer and colleagues (2006) found 26% differences in annual emission estimates based on three different land use/land cover (LULC) datasets in line with regional fire emissions model for Central and North America (Wiedinmyer et al., 2006, as cited in Wiedinmyer et al., 2010). Besides the uncertainties related to the land cover classifications, biomass loading, the assumed area burned, fire detections, the amount of fuel burned and emission factors are considerable uncertainties which need to be kept in mind while referring the FINNv1 estimates (Wiedinmyer et al., 2010). Nevertheless, updates aimed at decreasing such uncertainties will be included in future versions of FINN (Wiedinmyer et al., 2010).

Uncertainties associated with fire emissions

Uncertainties need to be further addressed in light of the results obtained. According to Carter and colleagues, the properties and abundance of carbonaceous aerosol (including black carbon and organic carbon) emitted by biomass burning (BB) are insufficiently constrained and remain uncertain (Carter et al., 2020). As such, the emissions uncertainty largely hinders researchers' capability of assessing, understanding and modelling air quality and the impacts of fires onto climate (Carter et al., 2020). Conduced using different inventories, the study by Carter and colleagues, shows the large differences observed between those. Indeed, important differences emerge according to the inventory utilized, referring to the "population-weighted annual fire $PM_{2.5}$ exposure" (Carter et al., 2020). Additionally, magnitude and spatial extent of BBA-only daily and annual surface concentrations are further included in these differences (Carter et al., 2020).

According to simulations, OA and BC are either overestimated or underestimated. Thus, as Carter and colleagues suggest, emissions uncertainty plays a crucial role and thus challenges researchers' capacity to model both air quality and climate consequences of fires (Carter et al., 2020). In fact, this uncertainty may largely influence our understanding of these (Carter et al., 2020). Other than the main uncertainty in emissions, identified as the spread across inventories, other factors may also enhance the actual uncertainty (Carter et al., 2020).
Carter and colleagues suggest that our understanding of secondary organic aerosol (SOA) remains incomplete. The magnitude of SOA from fires greatly ranges between field and laboratory studies (Carter et al., 2020). The former ones often do not find secondary aerosol formation (Hodskire et al., 2019, as cited in Carter et al., 2020), while the latter always indicate important SOA formation from fires (Grieshop et al., 2009; Hennigan et al., 2011; Ortega et al., 2013; Tkacik et al., 2017; Lim et al., 2019, as cited in Carter et al., 2020). The authors point out that these differences are not well understood (Shrivastava et al., 2017; Hodshire et al., 2019, as cited in Carter et al., 2020) and suggest them to become central aspects of future studies (Carter et al., 2020).
Becoming a growing problem – wildfires are increasingly occurring in specific regions – it is crucial to develop reliable and accurate models and emission inventories which are able to rightly identify consequences of fires and emitted aerosols onto the climate, human health and

environment (Carter et al., 2020). Indeed, emissions uncertainties play a key role in researchers' ability to understand both climate impacts of fires and air quality. Thus, it is highly recommended to encompass these into modelling studies (Carter et al., 2020). As such, a further assessment of satellite-based fire emission inventories would greatly improve current insight into the discussed uncertainties (Carter et al., 2020). Finally, the authors further recommend conducting additional observations at various scales: surface, satellite and aloft, for a better fire emissions understanding (Carter et al., 2020). These refer to further investigations of uncertainties in fire aerosol processing and aging (Carter et al., 2020).

Chapter 5: Conclusion

Through exploring the case study of crop residue burning in the IGP, this paper has provided an understanding regarding the problem, its significance and its evolution. It is important to note that this issue remains a huge challenge as it has climate, air quality and health consequences for the densely populated region. Following the analysis of results, this paper can draw several conclusions. Firstly, the FINN fire emissions maps appear quite scattered, and no clear shift regarding intense fire emissions (1.e+13 molecules/cm^2/s) in the IGP can be identified. Further, the FINN time series (figure 2) show a strong increase in emission values since 2002, intensified after 2009. As discussed earlier, consequences of groundwater acts promulgated that year can be strongly observed in figure 2. Emissions values have risen of 122% and 177% between 2009 and 2012, as well as between 2009 and 2013, respectively. Overall, it can also be noted that 2013 emission values have tripled since 2002, while the latest data available (2019) indicates a 260% increase since 2002. Additionally, impacts of these acts can be also be seen in figure 3 representing the peak occurrence for every October-November period in 2002 to 2019. As such, the graph shows peaks are occurring later year after year, from the second fortnight to the first fortnight of November. In fact, figure 3 indicates a 21-day shift between emission peaks in 2002 and 2016.

Other regions, such Eastern China for example, appear to have similar experiences (Chen et al., 2016). The study by Cheng and colleagues, in which air pollution was monitored in the cities of Shanghai, Nanjing, Hangzhou, Suzhou and Ningbo located at the Yangtze River Delta (YRD), suggests a ten days heavy haze episode (Cheng et al., 2014, as cited in Chen et al., 2017). The study further indicates a visibility of 2.9-9.8 km during the period from 28 May to 6 June 2011 (Chen et al., 2016). During this episode, daily $PM_{2.5}$ concentrations have risen to 82 μg/m^3 (average values) to 144 μg/m_3 (maximum values) (Chen et al., 2016). Regarding North China, it

has been found that straw burning pollution in Beijing in June was mainly caused by winter wheat area of the North China Plain, resulting in southerly transport paths (Li et al., 2008, as cited in Chen et al., 2016).
Other examples from China indicate similar issues as the ones explored in regards to the IGP. Firstly, crop residue is known as the most bulk biomass open-burned in China (Chen et al., 2016). As such, researchers report the threat to Chinese air quality of the substantial amounts of combustion products from BB emitted to the atmosphere (Chen et al., 2016). As such, due to open burning of crop residue strongly impacting the environment and public health, researchers urge for the prohibition of the practice (Zhang and Cao, 2015; Gustafsson et al., 2009, as cited in Chen et al., 2016).

Regarding recommendations, Zhou et al. (2017) address the need for high-temporal hourly resolution for effective control as well as numerical simulation of BB pollution research (Zhou et al., 2017). Further, Chen et al. (2016) developed several recommendations regarding the issue and consequences of BB in China. Firstly, field campaigns provide an opportunity for further investigating and understanding air quality systematically, regionally and on a large-scale. Conducted in different regions, different examples are explored: the *CARE-Beijing* campaigns (Campaign of Air Quality Research in Beijing and Surrounding Region), *PRIDE-PRD* campaigns (Program of Regional Integrated Experiments of Air Quality over the Pearl River Delta) and the *PEACE-YRB 2015* (Program of Extensive Air Quality Research Campaign over the Yangtze River Basin in 2015) (Chen et al., 2016). Besides playing a role in establishing and enforcing environmental policies, these campaigns further address pollution conditions, atmospheric chemistry, sources as well as physiochemical profiles (Chen et al., 2016). Moreover, numerical model simulation and further studies were further supported thanks the database resulting from these campaigns (Chen et al., 2016). Then, a second recommendation refers to investigating the optical properties of smoke particles during aging (Chen et al., 2015; Peng et al., 2016, as cited in Chen et al., 2016). Further studying mixing state with urban pollutants, toxicological organics and morphology is also advised (Chen et al., 2016). Thirdly, Chen and colleagues (2016) point out the strong impacts BB smoke particulates have unto health and the climate, and thus highly advise the international research community to prioritize studies dedicated to the climate of effects of Black Carbon and Brown Carbon, as a major part of the Earth's BB is emitted in China (Chen et al., 2016).

Regarding the IGP, Bikkina and colleagues suggest that the current urban efforts need to be complemented with countryside mitigation (Bikkina et al., 2019). Additionally, reductions in biomass combustion need to be accompanied by decreases in anthropogenic emissions. In fact, as explained above, biomass-burning reductions will drastically enhance the IGP region's air quality during the post-monsoon period, with lower positive results in December. As such, anthropogenic emissions strongly need to decrease, especially as the dynamics of the region and meteorology foster stagnant atmospheric winter conditions (Ojha et al., 2020). It is further important to acknowledge the unfavourable topography of the Himalaya extending from northwest to southeast favouring the accumulation of aerosols along the IGP (Mogno et al., 2021; Ojha et al., 2020). Furthermore, studies suggest enhancing the development of cohesive, organized and awareness-driven systems of organized networks among farmers (Jain et al., 2014; Bhuvaneshwari et al., 2015). Indeed, it is crucial to emphasize and raise awareness regarding the negative impact of such combustion practices as well as crop residues incorporation in soil (Jain et al., 2014) through empowering farmer stakeholders and other stakeholders (Bhuvaneshwari et al., 2015). Additionally, the aforementioned residue can be used in incorporation as well as regarding bio-energy, if collected and managed suitably (Jain et al., 2014). Further, other sustainable management practices regarding crop residue include composting, biochar, as well as in-situ management through mechanical intensification (Bhuvaneshwari et al., 2015). Finally, the Happy Seeder practice is also described as an alternative to the current technique as it allows for reduced GHG emissions and lower social costs regarding air pollution (Bhuvaneshwari et al., 2015). Balwinder-Singh and colleagues (2019) provide a similar analysis: adopting agronomic technologies, namely the Happy Seeder, as mentioned above, would consequently allow crop establishment into residue without depending on burning (Balwinder-Singh et al., 2019). Shyamsundar and colleagues (2019) indicate a larger profitability compared to other practices. Indeed, Happy Seeder-based systems are around 10% more profitable than the most-profitable burning option, with zero-till seeders. Additionally, the most common burn system with conventional seeders is around 20% less profitable than such Happy Seeder-based systems (Shyamsundar et al., 2019). Machinery costs and absence of knowledge regarding no-burn alternatives and external impacts of burning need to be taken into account (Shyamsundar et al., 2019).

Balwinder-Singh and colleagues (2019) further promote the cultivation of shorter-duration rice varieties for the burning patterns to be more dispersed and less concentrated in November (Balwinder-Singh et al., 2019). Additionally, off-farm uses for crop residues, such as energy production, are further recognized as potential solutions (Balwinder-Singh et al., 2019). Regarding water conservation, Balwinder-Singh et al. (2019) indicate that incentives such as the full pricing of energy could allow for reducing the policy imperatives to ban earlier rice establishment (Balwinder-Singh et al., 2019). Also, in order to deter the use of agricultural machinery innovations for burning practices, the Indian Government has launched a US$157m (October 2018 exchange rate) initiative. Besides these promising solutions, more structural changes need to be ensured as current policies strengthen cereals productivity maximization, and thus high levels of residue production. As such, research (Balwinder-Singh et al., 2019) suggests to develop a sustainable intensification in other areas such as the Eastern Gangetic Plain, known for its abundant water resources and tighter crop-livestock systems coupling which could thus provide a range of end uses for crop residues (Balwinder-Singh et al., 2019).

Sources

Balwinder-Singh, McDonald, A. J., Srivastava, A. K., & Gerard, B. (2019). Tradeoffs between groundwater conservation and air pollution from agricultural fires in northwest India. *Nature Sustainability*, *2*(7), 580-583.

Bhuvaneshwari, S., Hettiarachchi, H., & Meegoda, J. N. (2019). Crop residue burning in India: Policy challenges and potential solutions. *International journal of environmental research and public health*, *16*(5), 832.

Bikkina, S., Andersson, A., Kirillova, E. N., Holmstrand, H., Tiwari, S., Srivastava, A. K., Bisht, D.S., & Gustafsson, Ö. (2019). Air quality in megacity Delhi affected by countryside biomass burning. *Nature Sustainability*, *2*(3), 200-205.

Carter, T. S., Heald, C. L., Jimenez, J. L., Campuzano-Jost, P., Kondo, Y., Moteki, N., ... & Kaiser, J. W. (2020). How emissions uncertainty influences the distribution and radiative impacts of smoke from fires in North America. *Atmospheric Chemistry and Physics*, *20*(4), 2073-2097.

Chen, J., Li, C., Ristovski, Z., Milic, A., Gu, Y., Islam, M. S., ... & Dumka, U. C. (2017). A review of biomass burning: Emissions and impacts on air quality, health and climate in China. *Science of the Total Environment*, *579*, 1000-1034.

Cusworth, D. H., Mickley, L. J., Sulprizio, M. P., Liu, T., Marlier, M. E., DeFries, R. S., Guttikunda, S. K., & Gupta, P. (2018). Quantifying the influence of agricultural fires in northwest India on urban air pollution in Delhi, India. *Environmental Research Letters*, *13*(4), 044018.

Earth Data NASA. (2020). Active Fire Data. Retrieved April 2021, from https://earthdata.nasa.gov/earth-observation-data/near-real-time/firms/active-fire-data

Jain, N., Bhatia, A., & Pathak, H. (2014). Emission of air pollutants from crop residue burning in India. *Aerosol and Air Quality Research*, *14*(1), 422-430.

Mogno, C., Palmer, P. I., Knote, C., Yao, F., & Wallington, T. J. (2021). Seasonal distribution and drivers of surface fine particulate matterand organic aerosol over the Indo-Gangetic Plain. *Atmospheric Chemistry and Physics Discussions*, 1-39.

National Center for Atmospheric Research. (2021). *FINNV1 FIRE EMISSIONS*. Retrieved April 2021, from https://www.acom.ucar.edu/acresp/forecast/fire-emissions.shtml

Ojha, N., Sharma, A., Kumar, M., Girach, I., Ansari, T. U., Sharma, S. K., Singh, N., Pozzer, A. & Gunthe, S. S. (2020). On the widespread enhancement in fine particulate matter across the Indo-Gangetic Plain towards winter. *Scientific reports*, *10*(1), 1-9.

Roozitalab, B., Carmichael, G. R., & Guttikunda, S. K. (2021). Improving regional air quality predictions in the Indo-Gangetic Plain–case study of an intensive pollution episode in November 2017. *Atmospheric Chemistry and Physics*, *21*(4), 2837-2860.

Shyamsundar et al. (2019). Fields on fire: Alternatives to crop residue burning in India. *Science*, *365*(6453), 536-538.

Takigawa, M., Patra, P. K., Matsumi, Y., Dhaka, S. K., Nakayama, T., Yamaji, K., Kajino, M. & Hayashida, S. (2020). Can Delhi's pollution be affected by crop fires in the Punjab region?. *SOLA*.

United States Environmental Protection Agency. (n.d.). *Criteria Air Pollutants*. Retrieved May 2021, from https://www.epa.gov/criteria-air-pollutants

Wiedinmyer, C., Akagi, S. K., Yokelson, R. J., Emmons, L. K., Al-Saadi, J. A., Orlando, J. J., & Soja, A. J. (2011). The Fire INventory from NCAR (FINN): A high resolution global model to estimate the emissions from open burning. *Geoscientific Model Development*, *4*(3), 625-641.

World Bank, World Development Indicators. *PM2.5 air pollution, mean annual exposure (micrograms per cubic meter)*. https://data.worldbank.org/indicator/EN.ATM.PM25.MC.M3?most_recent_value_desc=true&type=shaded&view=map

World Health Organization. (2018). *Ambient (outdoor) air pollution*. Retrieved March 2021, from https://www.who.int/en/news-room/fact-sheets/detail/ambient-(outdoor)-air-quality-and-health

World Health Organization. (n.d.). *Ambient Air Pollution: Pollutants*. Retrieved March 2021, from https://www.who.int/airpollution/ambient/pollutants/en/

Zhou, Y., Xing, X., Lang, J., Chen, D., Cheng, S., Wei, L., ... & Liu, C. (2017). A comprehensive biomass burning emission inventory with high spatial and temporal resolution in China. *Atmospheric Chemistry and Physics*, *17*(4), 2839-2864.

Appendices

Table 1. $PM_{2.5}$ air pollution, mean annual exposure (micrograms per cubic meter), World Bank, 2017.

Country	**2017 value**
Nepal	100
Niger	94
Qatar	91
India	***91***
Saudi Arabia	88
Egypt, Arab Republic	87
Cameroon	73
Nigeria	72
Bahrain	71
Chad	66

Figure 1: FINN Fire emissions along north India, for years 2014-2020, months of October and November. See figure 1a, 1b, 1c, 1d, 1e, 1f and 1g

Figure 1a : FINN Fire emissions, October and November 2014

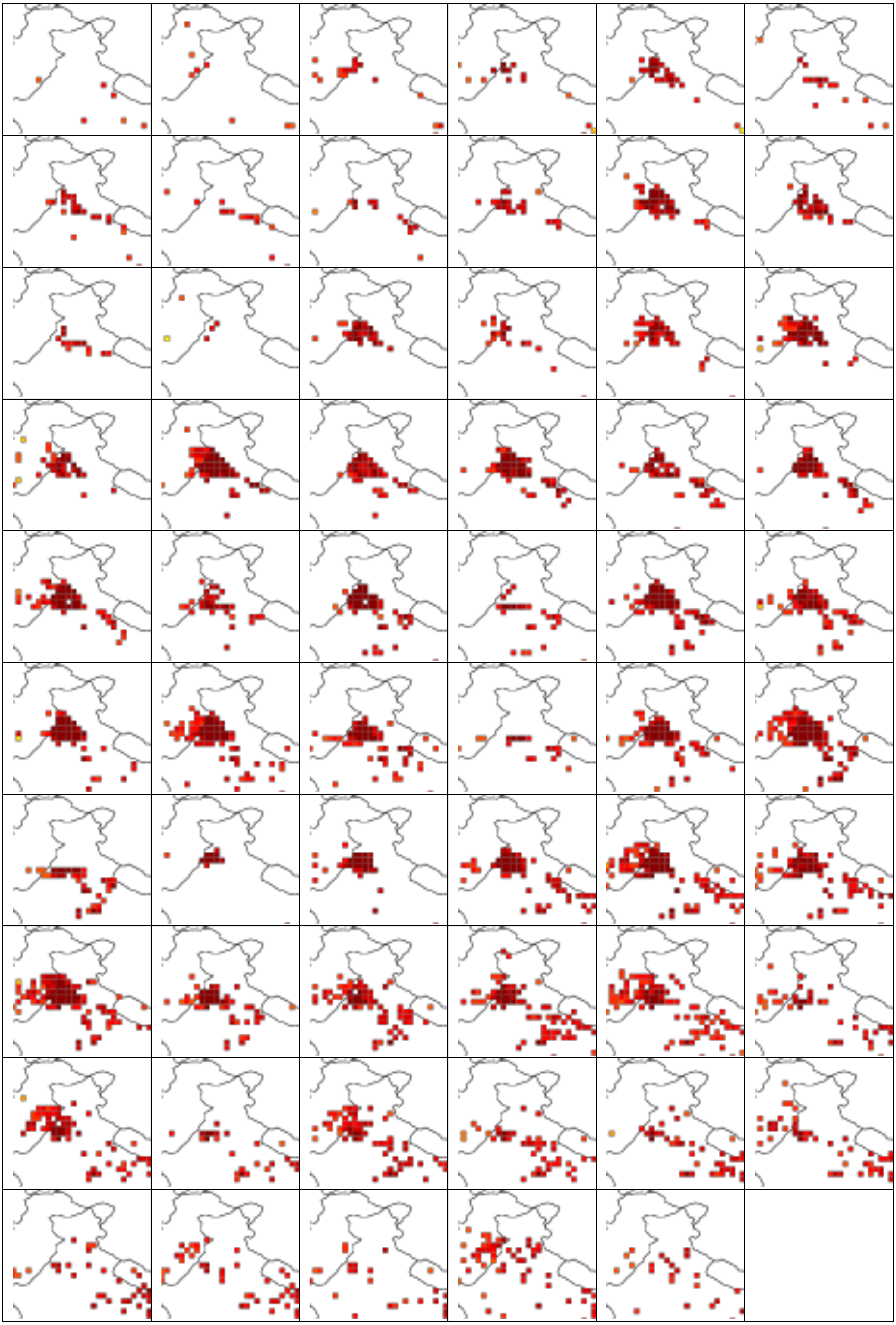


Figure 1b : FINN Fire emissions, October and November 2015

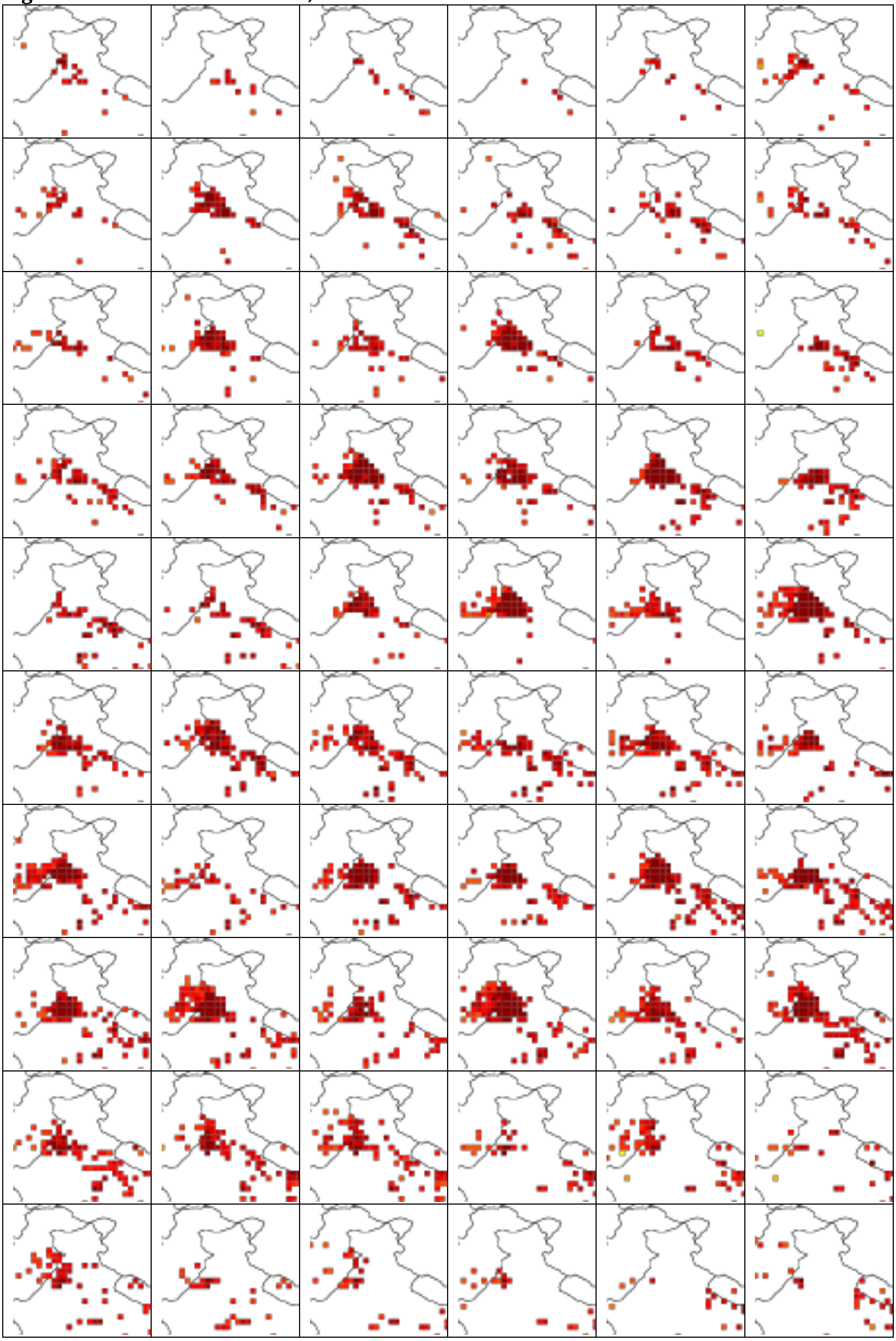


Figure 1c : FINN Fire emissions, October and November 2016

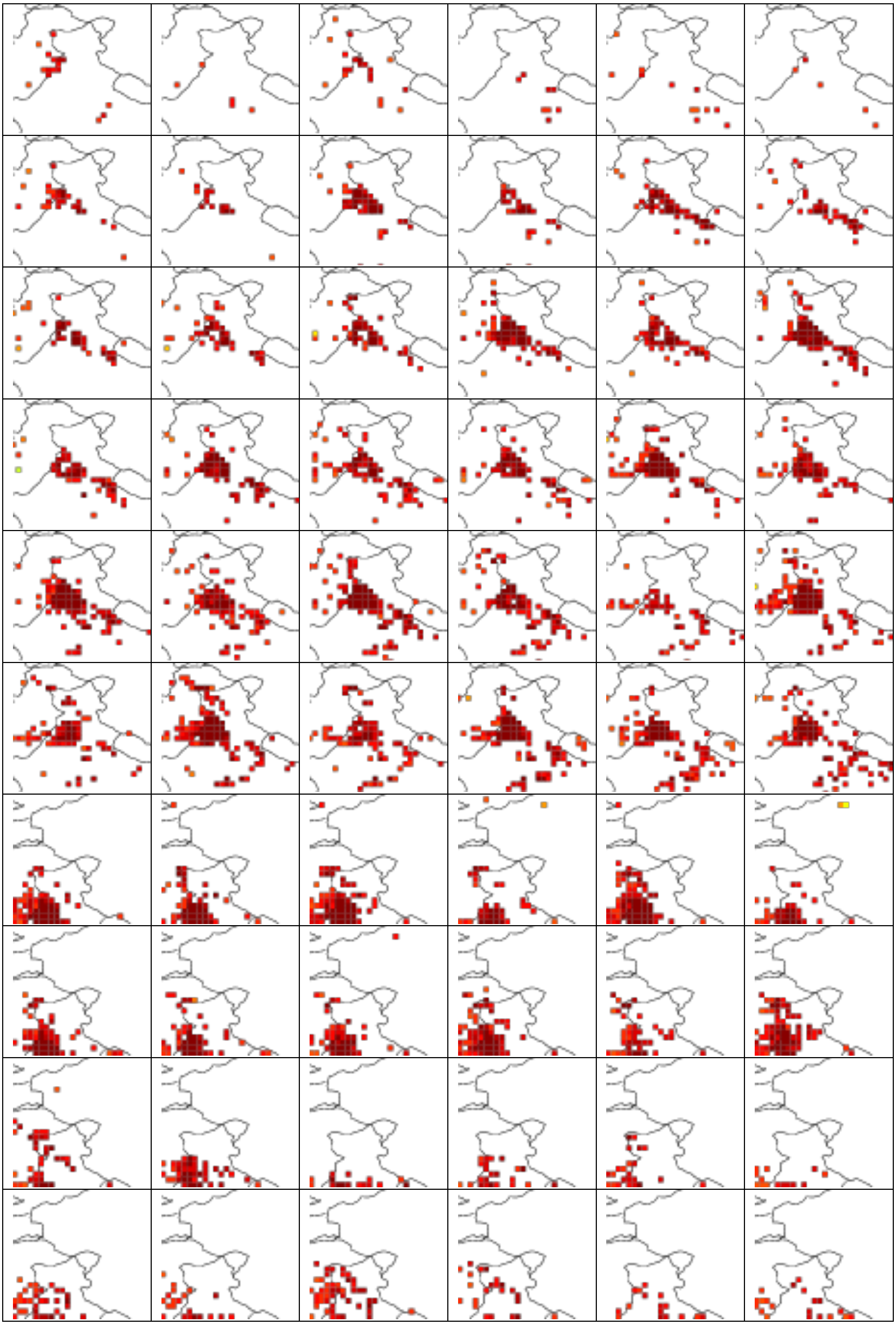


Figure 1d : FINN Fire emissions, October and November 2017

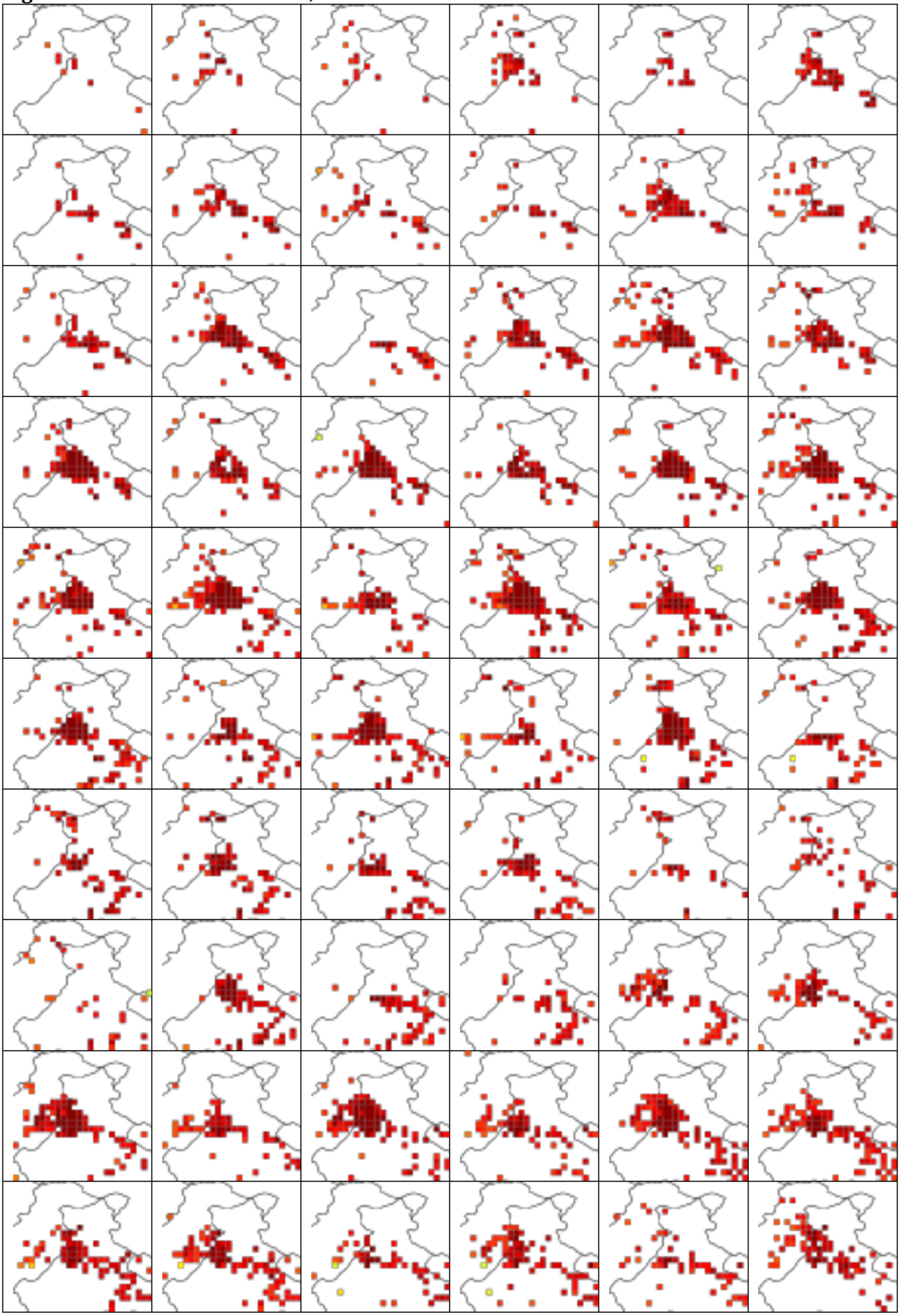


Figure 1e: FINN Fire emissions, October and November 2018

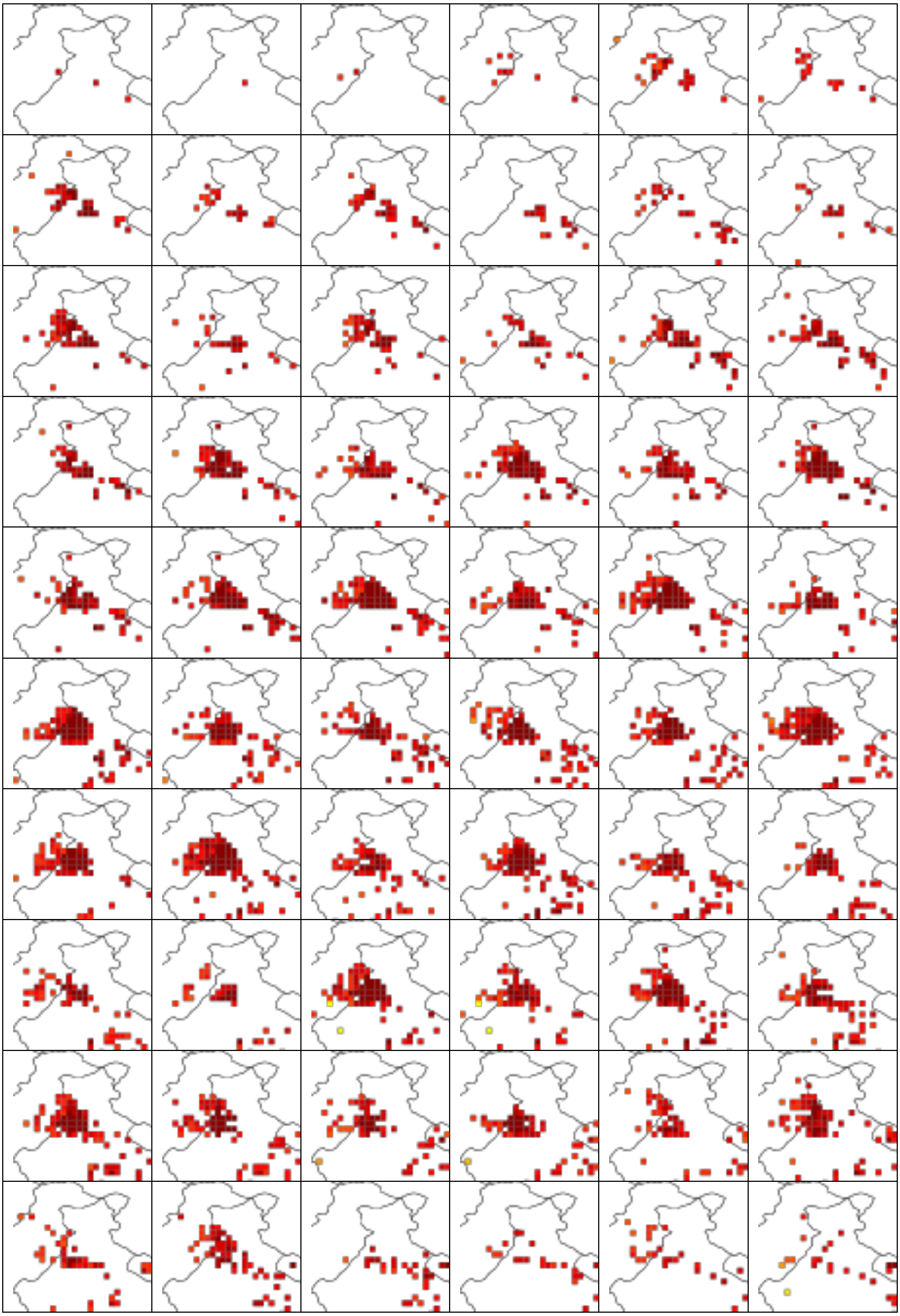


Figure 1f : FINN Fire emissions, October and November 2019

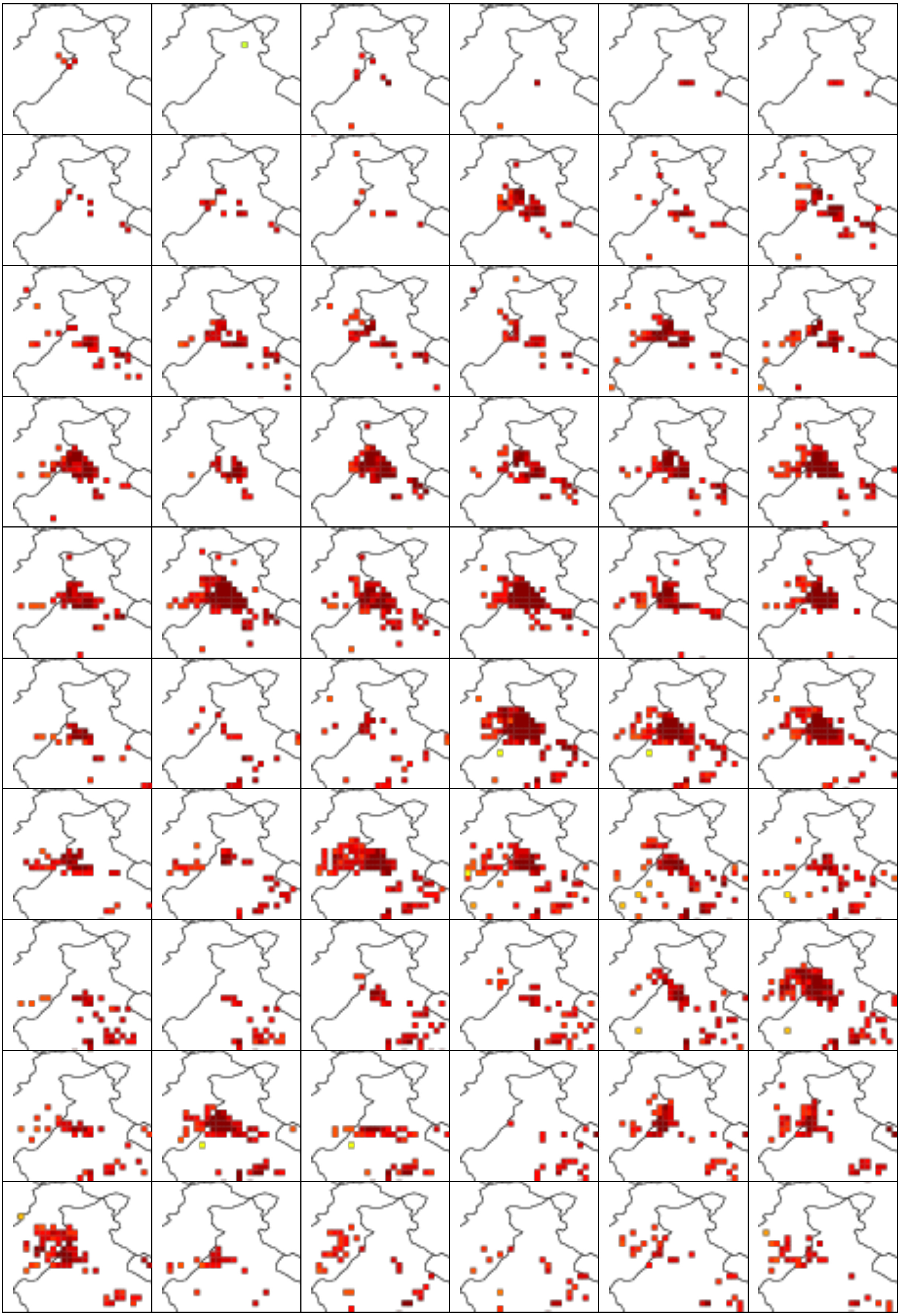
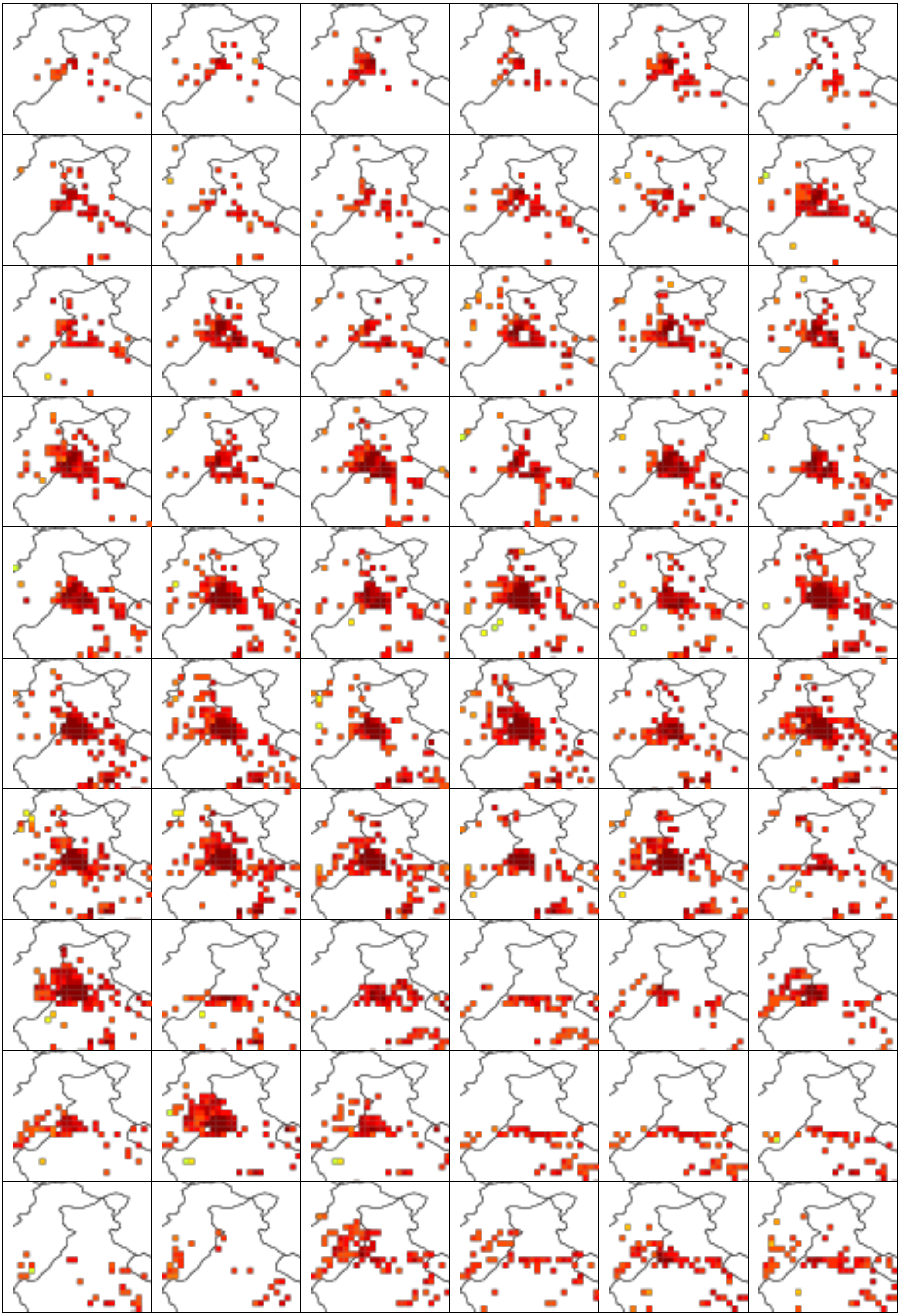


Figure 1g : FINN Fire emissions, October and November 2020



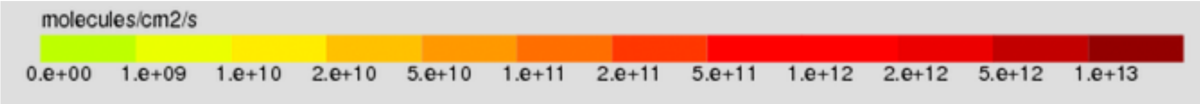

molecules/cm2/s
0.e+00
1.e+09
1.e+10
2.e+10
5.e+10
1.e+11
2.e+11
5.e+11
1.e+12
2.e+12
5.e+12
1.e+13

Table 2. Presence of various intense fire emissions (1.e+13 molecules/cm^2/s) in the IGP as indicated in dark red during the October-November period of years 2014 to 2020. See scale of figure 1 for additional information.

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30
2014																																																													
2015																																																													
2016																																																													
2017																																																													
2018																																																													
2019																																																													
2020																																																													

Figure 2: FINN daily emission values time series, 2002 to 2019, months October-November. The x-axis refers to the October to November periods of years 2002 to 2019, such as that every post-monsoon period follows each other.

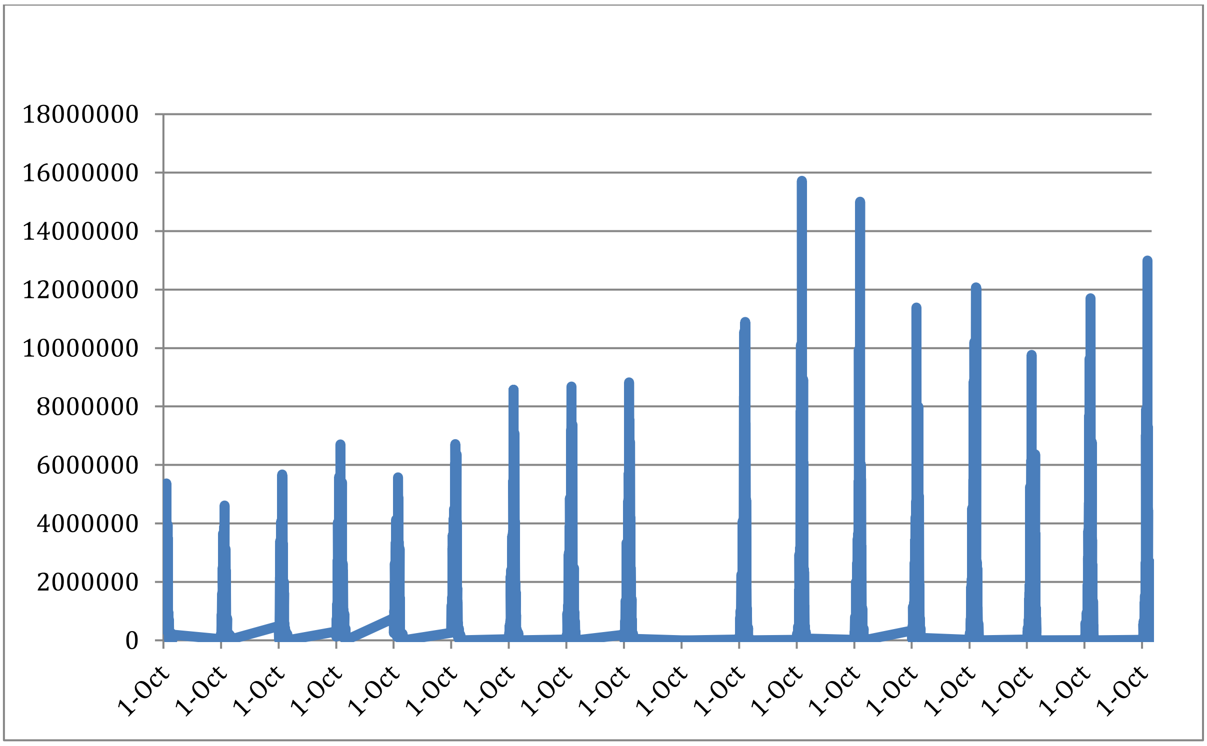


Figure 3. Peak occurrence of FINN emission values for years 2002 to 2019 during the October to November period.