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Describing the trend of ammonia, particulate matter and nitrogen oxides: the role of livestock activities in Northern Italy during Covid-19 quarantine

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Abstract

Nitrogen oxides (NOx), sulphur oxides (SOx) and ammonia (NH_3) are among the main contributors to the formation of secondary particulate matter ($PM_{2.5}$), which represent a severe risk to human health. Even if important improvements have been achieved worldwide, traffic, industrial activities, and the energy sector are mostly responsible for NOx and SOx release; instead, the agricultural sector is mainly responsible for NH_3 emissions.

Due to the emergency of coronavirus disease, in Italy schools and universities have been locked down from late February 2020, followed in March by almost all production and industrial activities as well as road transport, except for the agricultural ones. This study aims to analyze NH_3, $PM_{2.5}$ and NOx emissions in principal livestock provinces in the Lombardy region (Brescia, Cremona, Lodi, and Mantua) to evaluate if and how air emissions have changed during this quarantine period respect to 2016-2019. For each province, meteorological and air quality data were collected from the database of the Regional Agency for the Protection of the Environment, considering both data stations located in the city and the countryside. In the 2020 selected period, $PM_{2.5}$ reduction was higher compared to the previous years, especially in February and March. Respect to February, $PM_{2.5}$ released in March in the city stations reduced by 19%-32% in 2016-2019 and by 21%-41% in 2020. Similarly, NOx data of 2020 were lower than in the 2016-2019 period (reduction in March respect to February of 22-42% for 2016-2019 and of 43-62% for 2020); in particular, this can be observed in city stations, because of the current reduction in anthropogenic emissions related to traffic and industrial activities. A different trend with no reductions was observed for NH_3 emissions, as agricultural activities have not stopped during the lockdown. Air quality is affected by many variables, for which making conclusions requires a holistic perspective. Therefore, all sectors must play a role to contribute to the reduction of harmful pollutants.

Keywords

Air quality; ammonia; livestock; particulate matter; quarantine

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1 Introduction

Air quality is a big issue in developed countries. The most developed countries and industrialized cities, characterized by important economic exchanges, traffic and highly-density populated commonly have bigger problems than others with air pollution. Several studies are available on these aspects, for example, some Chinese (Xiao et al., 2020; Wang et al., 2015) and European (EEA, 2018; Izquierdo et al., 2020; Koolen and Rothenberg, 2019) cities are recognized as highly polluted.

According to the World Health Organization (WHO) (WHO, 2013), ground-level ozone (O_3), nitrogen dioxide (NO_2) and particulate matter (PM) are the most harmful air pollutants to human health and ecosystems. Among these, PM has been most closely studied due to its adverse health effect. PM identifies fine particles that can have both a primary and secondary origin. In the first case, they are emitted to the atmosphere directly from their sources, such as road traffic and car exhaust gases. Instead, secondary PM precursors are pollutants (e.g. NH_3, NOx, and SO_2) that are partly transformed into particles by photochemical reactions in the atmosphere (Koolen and Rothenberg, 2019). In more detail, air pollution has a direct effect on human health (Boldo et al., 2014), especially for what regards the long exposure to high concentrations of $PM_{2.5}$ (Yang et al., 2019). This is known to cause respiratory and cardiovascular diseases (Dominici et al., 2006) since it can penetrate deeply into the lung and translocate to blood circulation (Li et al., 2018). According to EEA (2019a), in Europe, about 400,000 premature deaths per year are attributable to $PM_{2.5}$ concentrations long-term exposure. Therefore, investigating how to reduce its presence in air is an important issue for society. As regards ecosystems, air pollution is known to cause damages to vegetation and ecosystems, such as eutrophication and soil acidification, which finally lead to biodiversity loss (EEA, 2019b). Moreover, excessive PM concentration causes undoubted problems not only in farms neighboring residents (de Rooij et al., 2017), but also in livestock buildings and on workers' and animals' health (Conti et al., 2020).

On a global scale, the benchmark limit for PM_{10} set at 20 $\mu g/m^3$ and $PM_{2.5}$ set at 10 $\mu g/m^3$ by the WHO is often not respected (EEA, 2019a). It is estimated that around 90% of the global population in 2018 was breathing polluted air (WHO, 2018), in particular, 6-8% of the European population was exposed to $PM_{2.5}$

exceeding limit and 13-19% to PM_{10} exceeding limit (EEA, 2019a). The European Union set the yearly limit for PM_{10} to 50 μg/m^3 and $PM_{2.5}$ to 25 μg/m^3 (Directive 2008/50/EC), but although it is less restrictive than the one by WHO, some countries are not able yet to respect it every day of the year; among these, Italy is an example (Kiesewetter et al., 2015). To date, in Lombardy, one of the most productive Italian regions, the measured annual average concentrations of $PM_{2.5}$ range between 10 and 31 μg/m^3 and those of PM_{10} from about 20 to 38 μg/m^3 (Fattorini and Regoli, 2020). In 2017, among all the Lombardy provincial chief towns, only in Lecco, Sondrio, and Varese the annual average concentrations were lower than the limit value. Instead, Milano, Brescia, Cremona and Mantua were the cities that most exceeded the $PM_{2.5}$ annual limit value (PRIA, 2018).

For this series of reasons, some researchers (Wang et al., 2020a; Wu et al., 2020) have started believing that the global spread of Covid-19 has reinforced in areas characterized by bad air quality.

Focusing on Italy, the Po valley – in the Northern part of the country, where also Lombardy region is located - is a highly industrialized and densely populated area characterized by a large concentration of intensive livestock farms (Arvani et al., 2014). These characteristics are also causing pollution, and in addition to them, also geographical and physical characteristics of the Po valley facilitate the persistence of pollutants (Fattore et al., 2011). In particular, they make Po valley one of the most disadvantaged areas in Europe for air quality. First of all, this area is characterized by the presence of the mountain chains of Alps and Apennines on three sides, which affects the pedo-climatic variables, reduces the air exchanges and negatively affects the local air quality. Secondly, the area is poorly ventilated favoring the stagnation of pollutants and consequently contribute to the accumulation of PM (Carugno et al., 2016). Lombardy is located in the middle of the Po valley and it is Italy's leading industrial and agricultural area (Lovarelli et al., 2020), whose emissions (methane, ammonia, dinitrogen monoxide, etc.) worsen the air quality issue (Rebolledo et al., 2013). To confirm this, Lombardy ranks among the most air polluted areas of Europe (Carugno et al., 2016). Several studies highlighted that the unfavorable geographical context, climate characteristics, and intense anthropogenic activities, such as industry and agriculture, of the Po valley promote a high level of air pollution (Arvani et al., 2014; Diémoz et al., 2019; Thunis et al., 2009). For example, Giannakis et al. (2019) found that the highest $PM_{2.5}$ concentration from the agricultural sector in Europe is present in northern Balkan countries and Northern Italy. Similarly, Thunis et al. (2019) identified as "hotspots" regions for PM emissions the Po valley and Eastern Europe. In addition, by 2050, global ammonia (NH_3) emissions are estimated to further increase because of agricultural intensification (Rebolledo et al., 2013). Emissions of NH_3 are an important aspect to evaluate, in fact it was estimated that 640 kg of PM_{10} can be formed per ton of NH_3 emitted and that 880 kg PM_{10} are formed per ton of nitrogen oxides (NOx) emitted (de Leeuw, 2002).

With the 2020 quarantine caused by the pandemic coronavirus, in Italy, many production and industrial activities have started being locked down from March, except for agricultural ones. The main evidence of this is related to the reduction in traffic: respect to the first part of February 2020, the beginning of March 2020 points out a reduction of about 90% of car traffic and about 50% of heavy vehicles (Buganza et al., 2020). Consequently, it is interesting to identify if and how air emissions are affected by agricultural activities in a polluted area such as Northern Italy, and in particular Lombardy, the Italian region most affected by Covid-19 (Fattorini and Regoli, 2020) and where the lockdown was the most severe (Bontempi, 2020), in a period in which most of the productive activities have been stopped while agricultural-related ones have remained active.

This study aims to analyze the main air emissions related to livestock activities in the cities and provinces of Lombardy that are mostly dedicated to livestock productions. This is carried out to evaluate if and how air emissions have changed in the quarantine period. In particular, from the comparison between the data about air emissions in data stations located in the main cities and those located in small cities or countryside stations, where livestock is the main activity, it is expected to identify a reduction in emissions due to the strong limitation of the industrial sector and transport, but a low reduction of those emissions related to agricultural activities.

1.1 Background

With the lockdown of work activities as a consequence of Covid-19, it can be expected that:

(i) anthropogenic emissions caused by industries, energy-related industries, and traffic deeply reduce,

(ii) emissions caused by agricultural activities should maintain a usual trend.

In Italy, agriculture is known to be responsible for more than 90% of NH_3 emissions (EEA, 2019c), in particular in Lombardy region they account for around 97% of all NH_3 emissions, corresponding to yearly 94,000 tons (INEMAR, 2020). Therefore, it can be assumed NH_3 did not vary in this analyzed quarantine period, except for possible different meteorological aspects. In particular, NH_3 emissions from the agricultural sector derive mainly from manure management and the application of fertilizers (Oenema et al., 2012), among which the superficial slurry spreading. Slurry spreading is generally carried out previous to soil preparation for crops sowing. Because this operation occurs seasonally every year and because agricultural activities have not been stopped during quarantine, emissions of NH_3 should not be subject to variations caused by the quarantine, instead of by the meteorological trend. In fact, NH_3 is a volatile gas whose amount is influenced by temperature, wind speed, and rainfall, other than by livestock housing, storage, and field spreading practices (Anderson et al., 2003; Welch et al., 2005). Given the dependence of agricultural field operations from weather conditions and given the European regulation for organic fertilizers spreading (Nitrate Directive) (Directive 91/676/EEC), the slurry is commonly spread between the end of winter and early spring before soil tillage and summer crops' sowing, as well as in autumn after the harvest of summer crops and before sowing of winter crops. To this, the European Directive that norms organic nitrogen application must be added (Directive 91/676/EEC). In particular, this Directive obliges to avoid the slurry spreading in the months in which no crop requirements for nutrients occur, and consequently in which runoff and leaching occur the most. Moreover, it is also forbidden to spread slurry when rainfall occurs or is expected to occur in the following days. For these reasons, NH_3 emissions can differ on a seasonal and a daily basis. One additional aspect related to NH_3 is that it plays an important role in atmospheric chemical reactions that bring to the formation of secondary PM_{10} and $PM_{2.5}$ (Koolen and Rothenberg, 2019). However, it should be considered that the quantity of secondary particulate matter that is formed in the atmosphere starting from the latter is variable in time and space and depends on non-linear processes and meteorology (Marongiu et al., 2020). PM can be classified as primary or secondary according to its origin and it can be both organic and/or inorganic in nature (Cambra-López et al., 2010). In the atmosphere, NH_3 reacts with atmospheric nitric and sulfuric acids to form particulate sulphate (SO_4^{2-}), nitrate (NO_3^{-}) and ammonium (NH_4^{+}) compounds, which constitute the major fraction of secondary inorganic $PM_{2.5}$ (Behera et al., 2013; Wang et al., 2020a). With the reduction in the release of NOx and sulphur oxides (SOx) from industrial activities, energy-related industries and traffic, $PM_{2.5}$ reduces only partially (EEA, 2018). It can be assumed that if NH_3 is not subject to changes during the Covid-19 quarantine (Marongiu et al., 2020), $PM_{2.5}$ may also reduce only partially, because the decrease in pollutant emissions caused by transport and industrial activities are not sufficient to avoid the chemical reaction (Wu et al., 2016). Regarding the agricultural sector, the main sources of PM emission are buildings housing livestock, in particular those in which are carried out the feed operations, which account for 80%-90% of total PM emissions from the agriculture sector. Pig and poultry livestock farms are the main sources of PM (EEA, 2019b) from the agricultural sector. Particulate matter emissions from pig houses arise also from skin particles, faeces, and bedding, while emissions from poultry housing from feathers and manure (EEA, 2019b). Together with these categories, cattle farming brings to relevant NH_3 and PM emissions, mainly from feed operations (Brown et al., 2018), feedlots (McGinn et al., 2010), barn, and storage tanks. In Lombardy, in 2017, $PM_{2.5}$ and PM_{10} emissions deriving from agriculture accounted for 4% and 6%, respectively (INEMAR, 2020).

2 Materials and methods

2.1 Monitored area

Lombardy houses 38% of the farms specialized in cattle milk production (ISMEA, 2019a), 10% of the farms specialized in cattle meat production (ISMEA, 2019b), and 11% of the pig farms for heavy pig production (ISMEA, 2019c), reaching a total number of reared animals equal to 1,543,639 cattle and 3,984,633 pigs (ISTAT, 2019).

Four provinces have been identified as the most livestock-intensive of the region. They are Brescia, Cremona, Lodi, and Mantua that altogether represent about 77.5% of the total Lombardy livestock production (ERSAF, 2019). Such a livestock intensity makes pressure on the territory, making most of the region susceptible to nitrates leaching, and most of the fields are recognized as Nitrate Vulnerable Zones (NVZ) in the context of the European Directive 91/676/EEC. In NVZ, 170 kg/ha of N from organic origin can be spread on fields. Commonly, farms are equipped with traditional machinery for slurry spreading, which consists of a superficial slurry spreading through slurry tankers equipped with diverter plates (Bacenetti et al., 2016a). This machine undoubtedly contributes to the volatilization of NH_3 during spreading and also in the few subsequent hours. NH_3 volatilization worsens also when optimal meteorological conditions occur (e.g., mild temperatures, low wind speed, and no rainfall) (Brentrup et al., 2000).

2.2 Data collection

Meteorological data and air quality data were collected from the database of the regional agency for environmental protection (ARPA, 2020). Data of ARPA (ARPA, 2020) were downloaded from different ground-based stations per province. The period investigated was January, February, and March of the years 2016-2019 and 2020. Data of years 2016-2019 were averaged to reduce the annual seasonality and were compared with 2020. These months were chosen since Covid-19 started spreading in the Lombardy region from January and almost all production activities have been locked down from February, except for agricultural ones.

The meteorological data used were air temperature (T, °C), relative humidity (RH, %), wind speed (W, m/s), and rainfall (R, mm). The air quality data used were ammonia (NH_3, μg/m³), nitrous oxides (NOx, μg/m³), and secondary particulate matter ($PM_{2.5}$, μg/m³). SOx were not included since these data were not available on the ARPA website. Particulate matter on the order of 10 micrometers (PM_{10}) was investigated only in an initial phase because it is caused by multiple sectors (transport, heating systems, energy sector, etc.) therefore its formation is not only due to agricultural activities and attributing its effect to one single sector may be misleading (Ansari and Pandis, 1998). Instead, since with PM_{10} are intended particles with a diameter equal or less than 10 microns, PM_{10} includes the smaller $PM_{2.5}$, and their trends can be compared. Moreover, because secondary $PM_{2.5}$ is partially formed from NH_3 released from livestock activities and the effect of $PM_{2.5}$ on health and ecosystem is damaging (Wang et al., 2020b), in this study $PM_{2.5}$ was analyzed in more detail instead of PM_{10}.

Figure 1 summarizes the research framework and air quality detection and meteorological stations referred to every province analyzed. In every province (i.e. Brescia, Cremona, Lodi, and Mantua), the stations with the availability of the pollutants NH_3, NOx, and $PM_{2.5}$ were used, except for Brescia where no detection stations for NH_3 emissions were available. All of them were distinguished in stations in the city and the countryside, to investigate the effect of pollutants in the city where most people live, traffic jams may occur and industrial activities are carried out, respect to the countryside, in which the main activities are related to livestock farms. Therefore, with “city” authors refer to those data stations located in a densely populated settlement whose members work primarily on non-agricultural tasks, whereas with “countryside” those located in a rural area mainly used for farming activities (field cultivation and livestock). As mentioned, together with pollutants, the meteorological data (temperature, relative humidity, wind speed, and rainfall) were collected by the same stations for the identified periods. In total, 14 data stations were analyzed for each of the 2 periods (January-March 2016-2019 vs January-March 2020). The final dataset was made of about 6000 data for every emission and meteorological variable. Statistical analyses were conducted using SAS version 9.4 (SAS Institute, Cary, NC, USA) statistics software. The mean and standard deviation of weather parameters for the two periods considered were calculated. Descriptive statistics were calculated using a means procedure in SAS. Principal Components Analysis (PCA), Factor Analysis (FA), and a general linear model (GLM) were used to identify relationships among variables and test the resulting model.

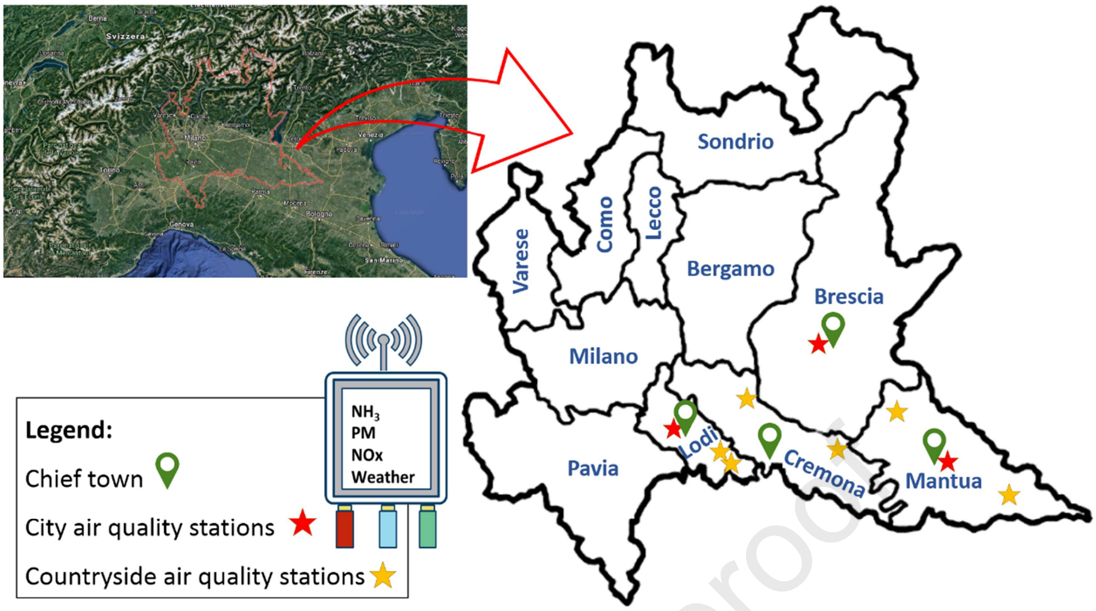

Figure 1. Research framework for data collection

3 Results and discussion

For what concerns the analysis of the meteorological aspects, **Table 1** reports the average data of the 4 provinces for mean, standard deviation (SD) and minimum and maximum values for temperature (T), relative humidity (RH), wind speed (W) and rainfall (R) in the period of January-March for 2016-2019 and 2020. From the results, it emerges that the weather conditions show reduced differences in the selected years 2016-2019 vs 2020. No events of strong wind speed or heavy rainfall were highlighted for these periods. Moreover, no differences can be observed between city and countryside stations in regard to average weather parameters, since in Italy, especially in the Po valley, agricultural areas are closed to cities due to population density.

Table 1 – around here

To support the fact that PM_{10} and $PM_{2.5}$ maintain a similar trend over time, **Figure 2** reports the average daily trend of PM_{10} and $PM_{2.5}$ in the city of Brescia for the period January-March 2019, to which are associated the wind speed and rainfall events of the same period. In particular, it is possible to notice that air pollutants generally reduce when wind and rainfall events occur. Although not reported, a similar trend was observed also for the other 3 provinces considered (Cremona, Lodi, and Mantua) (available in Supplementary Material).

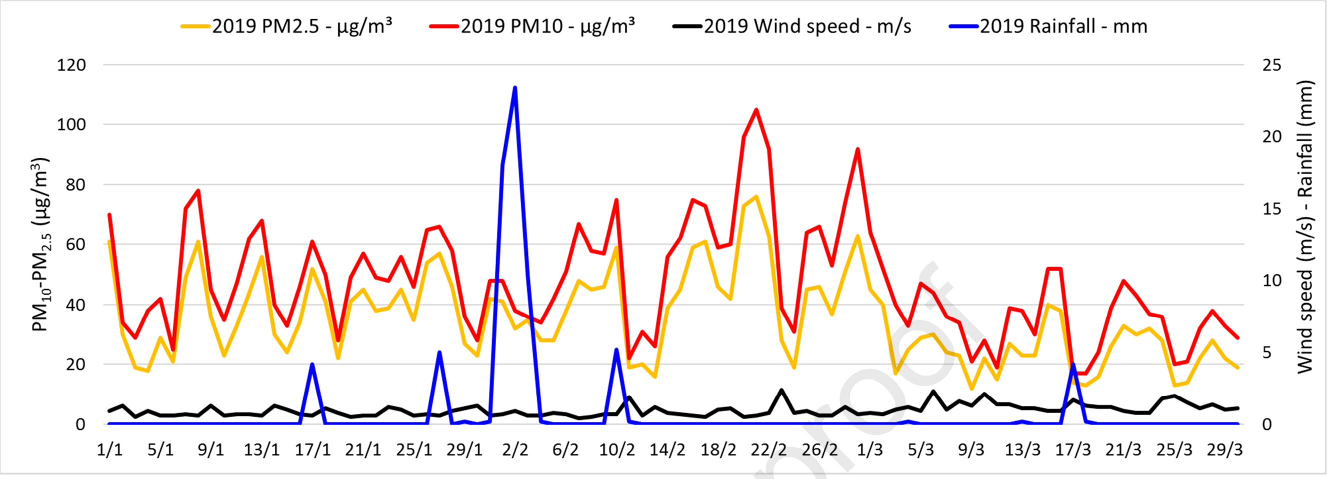

Figure 2. Trend of PM_{10} (red line) and $PM_{2.5}$ (yellow line) in the period January-March 2019 for the city of Brescia. Wind speed (black line) and rainfall events (blue line) are also shown.

Analyzing these data, the average presence of $PM_{2.5}$ in air changes as a consequence of meteorological variables; $PM_{2.5}$ in the air after rainfall events is on average 28.14 (±9.97) µg/m^3, whereas if no rainfall occurred this value was equal to 37.07 (±14.76) µg/m^3. Similarly, although the wind speed is generally low (<1.6 m/s or <6 km/h), when higher wind speed occurred, $PM_{2.5}$ for the studied period of January-March was on average 17.17 (±4.62) µg/m^3, while with low wind speeds it was equal to 40.36 (±13.87) µg/m^3.

In order to analyze the trend of emissions in the two periods considered, the air quality stations were grouped as follows: all data stations were split between stations located in the city ("city") and the countryside ("country"). Average values were calculated with data from Brescia, Cremona, Lodi, and Mantua data stations for each pollutant. Hence, for each pollutant (NOx, NH_3, and $PM_{2.5}$) and each period (2016-2019 and 2020) are available average data for "city" and "country" stations. **Figure 3** reports these results.

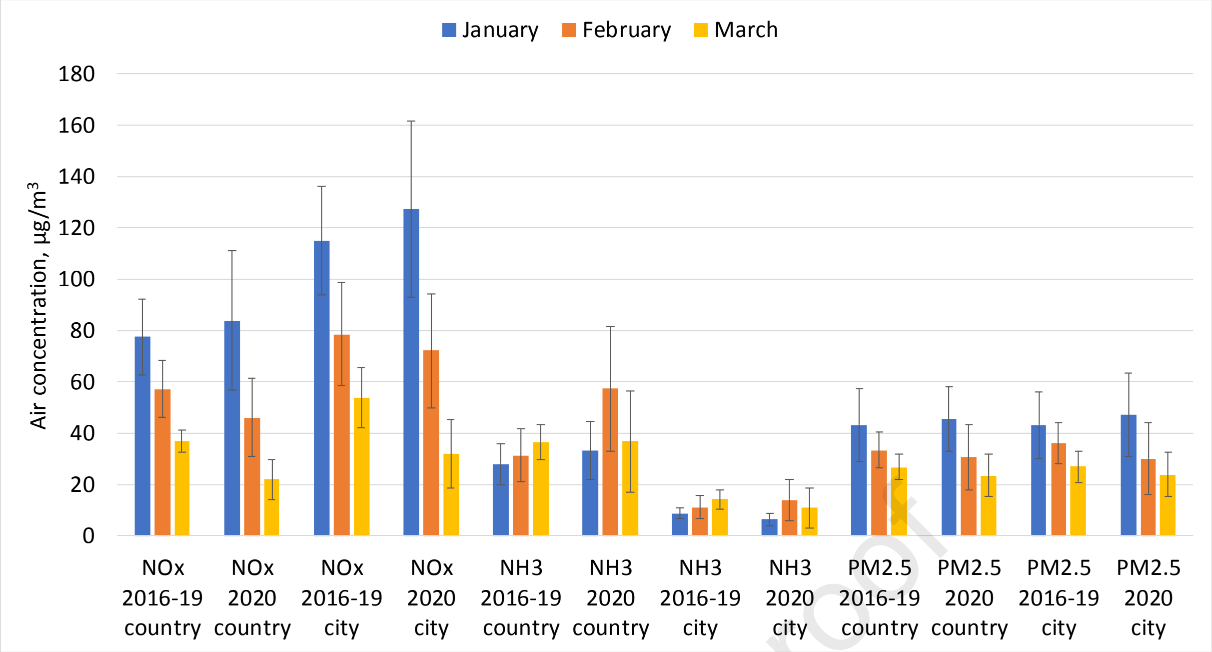

Figure 3. Results of average NOx, NH_3 and $PM_{2.5}$ emission in country and city stations of Lombardy for the period January-March of 2016-2019 and 2020. Bars refer to standard deviation.

From these results emerges that emissions of NOx and $PM_{2.5}$ are in all cases higher in January and follow a common reduction trend in February and March, which is common in all periods analyzed. Moreover, the emission of NOx and $PM_{2.5}$ were higher in January 2020 than in the same month of the previous years, although standard deviations are quite wide. NOx records the highest values in January (the coldest month), with an average of the city stations equal to 115.1 µg/m^3 in 2016-2019 and to 126.0 µg/m^3 in 2020. In the countryside stations, these values were equal to 76.4 µg/m^3 in 2016-2019 and 83.5 µg/m^3 in 2020. Indeed, NOx emissions gradually decrease from January to March in both periods considered, mainly because of the lower use of heating systems with milder temperatures.

In 2020, and in particular, in February and March, the reduction in NOx and $PM_{2.5}$ emissions is bigger than in the previous period and is even bigger in the city stations than in those in the countryside, where are mostly located the livestock and agricultural activities. NOx in March 2020 amounted to 22.03 (±7.72) µg/m^3 in countryside stations and 31.92 (±13.45) µg/m^3 in city stations, whereas in the previous period 2016-2019 they amounted to 36.88 (±4.28) µg/m^3 and 53.84 (±11.85) µg/m^3, respectively in countryside and city stations. This highlight considerable reductions in both city and countryside stations, partially motivated by the mentioned milder temperatures (on average for January-March 2020, 7.46°C) that allowed reducing home heating systems. Therefore, following the adoption of the Ministerial Decree (DPCM 8 March 2020), which introduced measures to limit travels, and following the reduced adoption of home heating system, NOx emissions decreased in March more than in the other months. In particular, the difference between March and February 2016-2019 ranged from 22.5% to 41.9%, while in the same period for 2020, NOx reduced by 43.3% to 61.5%.

Regarding $PM_{2.5}$, the emission in the cities for March ranges between 19% and 32% respect to February for the period 2016-2019 and between 21% and 41% respect to February in 2020, thus with a considerable reduction that characterized firstly the area of Lodi that was the first to be locked down (DPCM 23 February 2020). Respect to January, in some of the evaluated provinces, the reduction of $PM_{2.5}$ even reached 65% in 2020, while in the previous period 2016-2019 it did not exceed 46%. In the countryside stations, the reduction of $PM_{2.5}$ in March respect to February ranged between 16% and 26% in 2016-2019 and between 18% and 43% in 2020. Respect to January 2020, also in the countryside areas the $PM_{2.5}$ reduction in March reached 57%. Such a reduction for $PM_{2.5}$ is due to the fact that some of the pollutants co-participating to the formation of $PM_{2.5}$ have reduced. As reported in Section 1.1, in fact, $PM_{2.5}$ derives from a chemical reaction in which NH_3, NOx, and SOx pollutants can participate. This latter aspect has been highlighted by the study of Collivignarelli et al. (2020), and even if they focused on the area of Milan, they found that $PM_{2.5}$ concentrations were almost halved during the lockdown period probably due to the precursors (e.g. NOx) reduction. However, focusing on March 2020, $PM_{2.5}$ was equal on average to 23.48 (±8.17) $\mu g/m^3$ and 23.99 (±8.59) $\mu g/m^3$, respectively in countryside and city stations, while in the period March 2016-2019, the average values were 26.8 (±4.8) $\mu g/m^3$ and 26.92 (±6.04) $\mu g/m^3$, respectively for countryside and city. For $PM_{2.5}$, therefore, no evident differences emerge between countryside and city stations; however, some small differences emerge between 2020 and the previous years, which can be mainly due to the reduction in car traffic during the lockdown.

As emerges from **Figure 3**, however, the emission of NH_3 did not reduce in 2020 respect to the same period of 2016-2019. The reason is related to the fact that for agricultural activities no restriction was imposed during the quarantine. Therefore, agricultural activities, and in particular slurry spreading on the field, took place (similar to previous years) in the analyzed period.

Values of NH_3 in the countryside are similar between 2016-2019 and 2020 in January (28.0±8.1 and 33.1±11.3 μg/m^3 in 2016-2019 and 2020, respectively) and March (36.5±6.6 and 36.8±19.8 μg/m^3 in 2016-2019 and 2020, respectively), which was expected; however, they were higher in February 2020 respect to the previous years, probably because of the lack of possible slurry spreading events in the previous autumn. This condition was caused by a particularly rainy period that obliged farmers to avoid the slurry spreading in autumn and introduce more spreading interventions in February 2020. In February 2020, in fact, NH_3 emissions are about the double than in the 2016-2019 period (31.4±10.2 μg/m^3 in 2016-2019 and 57.5±24.2 μg/m^3 in 2020). The higher NH_3 emission in the countryside is also reflected by slight increase respect to February 2016-2019 in the stations located in the cities (11.3±4.6 μg/m^3 in 2016-2019 and 14.0±7.8 μg/m^3 in 2020). Respect to the NH_3 emission in the city stations occurred in March, the average values were found equal to 14.2±3.5 μg/m^3 in 2016-2019 and 10.9±7.8 μg/m^3 in 2020, thus with values considerably lower than in the countryside areas. No strong distances can be observed between city and countryside areas, but specifically for NH_3 emission this difference can be relevant.

From the statistical analysis carried out with these data, a Pearson's correlation matrix is reported in **Table 2** and in **Table 3**, where the main relationships among the identified parameters can be highlighted for the studied periods 2016-2019 and 2020. The statistical analyses were carried out separately between 2016-2019 and 2020 in order to better focus on emissions during Covid-19 quarantine. The considered parameters include air quality data of NOx, $NH_{3,}$ and $PM_{2.5}$ from city and countryside stations as well as weather data of temperature, wind speed, relative humidity, and rainfall for city and countryside stations. A good correlation has been considered for values equal to or higher than 0.6. In particular, a good correlation emerges among pollutants, especially between NOx and $PM_{2.5}$ ($r \geq 0.74$), both for city and countryside data. Instead, NH_3 is well correlated only between NH_3 in the countryside station and NH_3 in the city station ($r = 0.88$). Relative humidity and temperature have good correlations with $PM_{2.5}$, NOx and NH_3 ($r \geq 0.60$), while wind speed and rainfall show small correlations. Regarding 2020, the correlations are similar: NH_3 is well correlated with itself (r = 0.86), while $PM_{2.5}$ and NOx are well correlated with each other (r ≥ 0.76). Once more, correlations are obtained with temperature and relative humidity but not interestingly with wind speed and rainfall.

Table 2 – around here

Table 3 – around here

Similar information emerges also from the Principal Components Analysis (PCA) and Factor Analysis (FA). **Figure 4** reports the first graph relating Component 1 and Component 2 of PCA for years 2016-2019 and the year 2020, respectively. These components together explain >60% of the variability.

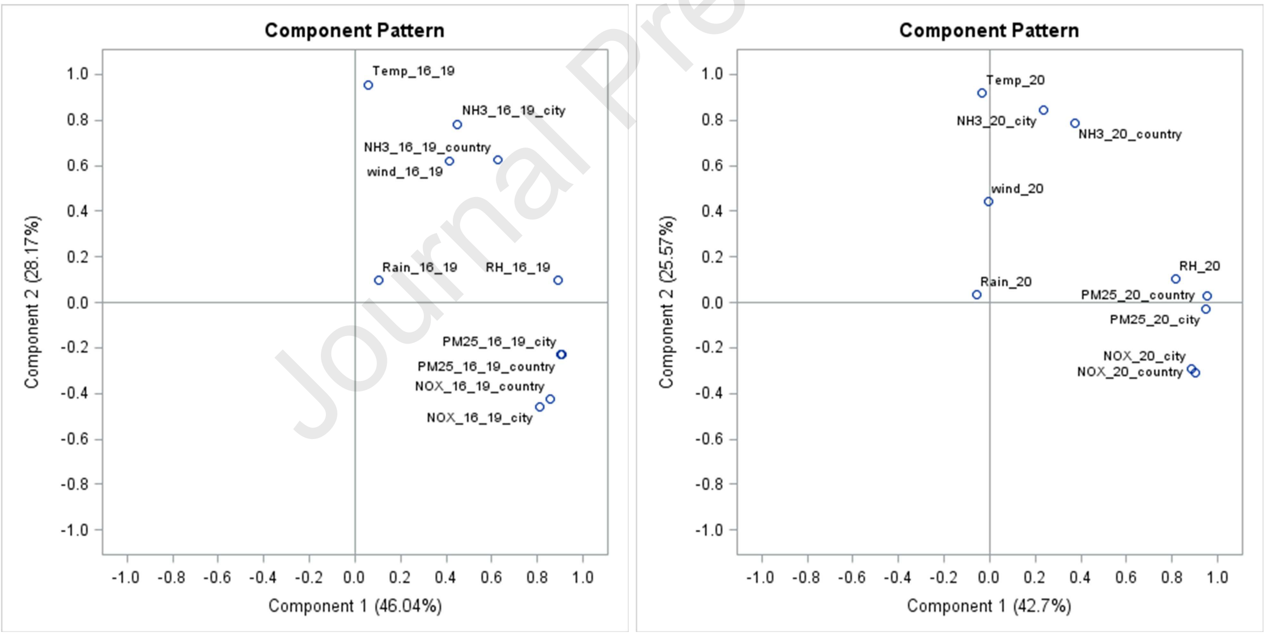

Figure 4. PCA, on the left for the period 2016-2019, on the right for 2020.

In more detail, PCA shows that every pollutant averaged for city and countryside stations is positioned close to each other. NOx and $PM_{2.5}$ are also close, while NH_3 is positioned in the upper quarter. For the 2016-2019 period, wind speed and temperature are quite close to NH_3 emission, while relative humidity and especially rainfall are quite isolated. In 2020, rainfall is isolated, relative humidity is closed to $PM_{2.5}$ emission, while temperature and wind speed are quite far but slightly closer to NH_3.

With FA, a clear distinction emerges between factors. In particular, 3 factors are identified that describe the components. Factor 1 can be entitled as “$PM_{2.5}$, NOx and RH”, Factor 2 as “NH_3, wind and temperature” and Factor 3 as “rain” for the analysis on 2016-2019. In 2020, the factors are much similar, although a small change occurred between Factor 2 that was characterized as “NH_3 and temperature” and Factor 3 that was “wind and rain”. The results of FA are reported in **Table 4** and **Table 5**.

Table 4 – around here

Table 5 – around here

Finally, a General Linear Model (GLM) was carried out with SAS software, from which emerged interesting results. In particular, the model resulted significant, with r^2=0.83.

Table 6 reports the estimates of GLM for NH_3 in 2020 in the city station, from which can be highlighted the effect of NH_3 emitted in the country and of wind and temperature.

Table 6 – around here

The interesting aspect that can be gathered from this study is that among the pollutants, NH_3 is mostly evident in the countryside stations, therefore confirming what reported by EEA (2018) and Pozzer et al. (2017). Moreover, NOx and $PM_{2.5}$ are well correlated, but NH_3 highlights a specific trend, only partially correlated with $PM_{2.5}$. Since NH_3 also contributes with NOx and SOx to $PM_{2.5}$ formation, if NH_3 reduced from the agricultural activities, $PM_{2.5}$ would reduce even more, as confirmed also by Zhao et al. (2017). This would involve additional positive benefits on the environment, ecosystems, and human health. As an example, Zambrano-Monserrate et al. (2020) highlighted as positive effects of Covid-19 quarantine an improvement of air quality associated with $PM_{2.5}$ and NO_2 emissions reduction, improved appearance of beaches and a reduction of the environmental noise level. Moreover, also in the study of Pozzer et al. (2017), it is reported that reducing by 50% the agricultural emissions of NH_3, a reduction of $PM_{2.5}$ equal to 2.4 µg/m^3 could be obtained in the Po valley region. In this way also global mortality and respiratory diseases due to $PM_{2.5}$ could be reduced. This reduction value is important considering that Po valley basin is among the European areas at greatest risk of exceeding the threshold limits of air quality due to its geographic conformation and to the high level of industrialization and anthropization. Regarding the reduction of NOx and partial reduction of $PM_{2.5}$, this effect is more evident in the cities than in the agricultural areas; therefore, their reduction can be partially attributed to the reduction in traffic and interruption of many industrial and energetic activities. As reported by Marongiu et al. (2020), during March 2020 NOx deriving from transport reduced by 60%, and NOx from energy production and industries decreased by 4% and 13%, respectively. Also, Buganza et al. (2020) and Chauhan and Singh (2020) observed a reduction in $PM_{2.5}$ concentrations in the period characterized by the Covid-19 emergency. Improving these aspects of air quality is very important, as reported also by a preliminary study conducted by ARPA and Lombardy region (Buganza et al., 2020). Considering all these emissions, it is important to note that the current strong reduction in traffic and industrial activities has helped reduce $PM_{2.5}$ (Wu et al., 2016), therefore a combined reduction of all air pollutants should be promoted.

Being the agricultural sector responsible for the widest part of NH_3 emissions, this sector should adopt measures for its reduction, providing interventions to improve the agricultural impact on the environment (Zhao et al., 2017). Policymakers and stakeholders should promote policies, incentives and disseminate knowledge to farmers about the need of abating NH_3 emissions with the already widely studied solutions: covering tanks for slurry and digestate storage instead of adopting open tanks (Bacenetti et al. 2016b; Finzi et al., 2019; Guarino et al., 2006), introducing treatment systems for slurry (anaerobic digestion, solid-liquid separation, nitro-denitro, air treatment, additives, etc.) (Dinuccio et al., 2011; Fangueiro et al., 2009; Finzi et al., 2020), removing frequently slurry and manure from the barn (Hoff et al., 2006) and spreading slurry on-field through injection techniques that permit to spread slurry into the soil through anchors and avoiding the superficial spreading with diverter plates (Hansen et al., 2003; Mattila and Joki-Tokola, 2003) that instead favor the conditions for NH_3 volatilization. These just mentioned are all strategies related to manure/slurry management, however, other strategies allow reducing pollutants inside livestock houses, such as biofilters, bioscrubbers (or biotrickling filters), dry filters, water scrubbers, and wet acid scrubbers (Dumont, 2018; Van der Heyden et al., 2015). All these air cleaning systems improve the air quality that animals and farmers breathe daily, with positive effects on animal welfare and thus on-farm performance and profitability, but also assuring a healthier environment for animals and workers. Finally, it could be useful to set a benchmark limit not only for $PM_{2.5}$ and nitrogen application, but also for NH_3, NOx, and SOx. To date, the only limit set by the National Emission Ceilings Directive 2016/2284/EU regards the obligation in European countries to abate NH_3 emission by 6% by 2020 (Directive (EU) 2016/2284).

4 Conclusions

From this preliminary study about air quality variation during the Covid-19 quarantine period, some conclusions can be drawn, and some key aspects can be opened for discussion on the agricultural sector, but also on industrial activities, energy sector, and traffic. The achieved results allowed to confirm what was initially expected. Probably as an effect of the quarantine, some emissions caused by industries, energy production, and traffic deeply reduced (e.g., NOx and $PM_{2.5}$) at least in the cities areas considered, while some emissions caused by the agricultural activities did not change (e.g., NH_3) because no variations occurred for agricultural activities within the quarantine framework. However, further studies focused on agricultural emissions considering more data air quality stations are needed, also over a longer period of time. These could give the opportunity to better monitor the emission of NH_3 on the territory and introduce targeted interventions for its reduction. This study may be considered as a preliminary reference to future evaluations on agricultural emissions.

From some current discussions, it could be concluded that somehow air quality has slightly improved; but this last conclusion cannot be drawn at the time being and relatively to this study. This research aimed to focus on agricultural activities, therefore data stations were selected based on this need. Moreover, air quality is affected by a big series of factors, among which other pollutants such as SOx, local weather conditions, regional air exchanges, traffic, energy-related industry and industrial activities that in this study were not evaluated. For what regards the responsibility of the agricultural sector to $PM_{2.5}$ emission, the need for abating NH_3 emissions is highlighted. Since agricultural NH_3 emissions derive mainly by livestock housing, manure storage and manure spreading, the already studied solutions of covering tanks, introducing additives, removing and treating slurry, using air cleaning systems in barns, and improving the spreading techniques should be promoted by policymakers and stakeholders. This last point can be carried out through the promotion of policies and incentives, and disseminating knowledge to farmers who are the final decision-makers for investing in the improvement of air quality. Moreover, much need to be done to comply with air quality regulations in order to not exceed PM limits and also implement more restrictive rules related to agricultural NH_3 emissions reduction. In any case, to improve the air quality, a combined role of all productive sectors is fundamental because pollutants, and in particular $PM_{2.5}$, derive from the co-presence of multiple pollutants in the air.

Authorship contribution statement

All authors have contributed equally to this work. All authors have read and approved the final manuscript.

References

Anderson, N., Strader, R., Davidson, C., 2003. Airborne reduced nitrogen: ammonia emissions from agriculture and other sources. Environ. Int. 29, 277-286. doi:10.1016/S0160-4120(02)00186-1

Ansari, A.S., Pandis, S.N., 1998. Response of Inorganic PM to Precursor Concentrations. Environ. Sci. Technol. 32, 2706-2714

ARPA, 2020. Agenzia Regionale per la Protezione dell’Ambiente – Lombardia. https://www.arpalombardia.it/Pages/Aria/Richiesta-Dati.aspx (accessed 1 April 2020).

Arvani, B., Teggi, S., Ghermandi, G., Bigi, A., Pierce, R.B., 2014. Air pollution assessment over Po valley (Italy) using satellite data and ground station measurements. XII Simpósio Ítalo-Brasileiro de Engenharia Sanitária a Ambiental – Natal (RN), 19-21 May 2014. doi:10.13140/2.1.3991.4885

Bacenetti, J., Lovarelli, D., Fiala, M., 2016a. Mechanisation of organic fertiliser spreading, choice of fertiliser and crop residue management as solutions for maize environmental impact mitigation. Eur. J. Agron. 79, 107-118. https://doi.org/10.1016/j.eja.2016.05.015

Bacenetti, J., Sala, C., Fusi, A., Fiala, M., 2016b. Agricultural anaerobic digestion plants: What LCA studies pointed out and what can be done to make them more environmentally sustainable. Appl. Energy 179, 669-686. 10.1016/j.apenergy.2016.07.029

Behera, S.N., Sharma, M., Aneja, V.P. Balasubramanian, R., 2013. Ammonia in the atmosphere: A review on emission sources, atmospheric chemistry and deposition on terrestrial bodies. Environ. Sci. Pollut. Res. DOI 10.1007/s11356-013-2051-9

Boldo, E., Linares, C., Aragonés, N., Lumbreras, J., Borge, R., de la Paz, D., Pérez-Gómez, B., Fernández-Navarro, P., García-Pérez, J., Pollán, M., Ramis, R., Moreno, T., Karanasiou, A., López-Abente, G., 2014. Air quality modeling and mortality impact of fine particles reduction policies in Spain. Environ. Res. 128,15-26. https://doi.org/10.1016/j.envres.2013.10.009

Bontempi, E., 2020. First data analysis about possible COVID-19 virus airborne diffusion due to air particulate matter (PM): The case of Lombardy (Italy). Environ. Res. 186, 109639. https://doi.org/10.1016/j.envres.2020.109639

Brentrup, F., Küsters, J., Lammel, J., Kuhlmann, H., 2000. Methods to estimate on-field nitrogen emissions from crop production as an input to LCA studies in the agricultural sector. Int. J. Life Cycle Assess. 5(6), 349–357. http://dx.doi.org/10.1065/lca2000.08.030

Brown, M.S., Cole, N.A., Gruber, S., Kube, J., Teeter, J.S., 2018. Modeling and prediction accuracy of ammonia gas emissions from feedlot cattle. Appl. Anim. Sci. 35, 347–356. https://doi.org/10.15232/aas.2018-01834

Buganza, E., Donzelli, M., Gurrieri, G.L., Lazzarini, M., Scotto Di Marco, E., Bravetti, E., Dal Santo, U., Di Leo, A., Lanzani, G., Cazzuli, O., Minardi, G.P., 2020. Analisi preliminare della qualità dell’aria in Lombardia durante l’emergenza COVID-19.

Cambra-López, M., Aarnink, A.J.A., Zhao, Y., Calvet, S., Torres, A.G., 2010. Airborne particulate matter from livestock production systems: A review of an air pollution problem. Environ. Pollut. 158, 1–17. https://doi.org/10.1016/j.envpol.2009.07.011

Carugno, M., Consonni, D., Randi, G., Catelan, D., Grisotto , L., Bertazzi, P.A., Biggeri, A., Baccini, M., 2016. Air pollution exposure, cause-specific deaths and hospitalizations in a highly polluted Italian region. Environ. Res. 147, 415-424. http://dx.doi.org/10.1016/j.envres.2016.03.003

Chauhan, A., Singh, R.P., 2020. Decline in $PM_{2.5}$ concentrations over major cities around the world associated with COVID-19. Environ. Res. 187, 109634. https://doi.org/10.1016/j.envres.2020.109634

Collivignarelli, M.C., Abbà, A., Bertanza, G., Pedrazzani, R., Ricciardi, P., Miino, M.C., 2020. Lockdown for CoViD-2019 in Milan: What are the effects on air quality?. Sci. Total Environ. 732, 139280. https://doi.org/10.1016/j.scitotenv.2020.139280

Conti, C., Guarino, M., Bacenetti, J., 2020. Measurements techniques and models to assess odor annoyance: A review. Environ. Int. 134, 105261. https://doi.org/10.1016/j.envint.2019.105261

Council Directive of 12 December 1991 concerning the protection of waters against pollution caused by nitrates from agricultural sources (91/676/EEC). OJ L 375, 31.12.1991.

De Leeuw, F.A.A.K., 2002. A set of emission indicators for long-range transboundary air pollution. Environ. Sci. Policy 5, 135–145.

de Rooij, M.M.T., Heederik, D.J.J., Borlée, F., Hoek, G., Wouters, I.M., 2017. Spatial and temporal variation in endotoxin and PM10 concentrations in ambient air in a livestock dense area. Environ. Res. 153, 161-170. https://doi.org/10.1016/j.envres.2016.12.004

Diémoz, H., Gobbi, G.P., Magri, T., Pession, G., Pittavino, S., Tombolato, I.K.F., Campanelli, M., Barnaba, F., 2019. Transport of Po Valley aerosol pollution to the northwestern Alps – Part 2: Long-term impact on air quality. Atmos. Chem. Phys. 19, 10129-10160. https://doi.org/10.5194/acp-19-10129-2019

Dinuccio, E., Berg, W., Balsari, P., 2011. Effects of mechanical separation on GHG and ammonia emissions from cattle slurry under winter conditions. Anim. Feed Sci. Technol. 166-167, 532-538. https://doi.org/10.1016/j.anifeedsci.2011.04.037

Directive 2008/50/EC of the European Parliament and of the Council of 21 May 2008 on ambient air quality and cleaner air for Europe. OJ L 152, 11.6.2008.

Directive (EU) 2016/2284 of the European Parliament and of the Council of 14 December 2016 on the reduction of national emissions of certain atmospheric pollutants, amending Directive 2003/35/EC and repealing Directive 2001/81/EC. OJ L 344/1, 17.12.2016.

Dominici, F., Peng, R.D., Bell, M.L., Pham, L., McDermott, A., Zeger, S.L., Samet, J.M., 2006. Fine Particulate Air Pollution and Hospital Admission for Cardiovascular and Respiratory Diseases. JAMA. 295 (10), 1127-1134.

Dumont, E., 2018. Impact of the treatment of NH_3 emissions from pig farms on greenhouse gas emissions. Quantitative assessment from the literature data. New Biotech. 46, 31–37. https://doi.org/10.1016/j.nbt.2018.06.001

EEA, 2018. European Union emission inventory report 1990-2016 under the UNECE Convention on Long-range Transboundary Air Pollution (LRTAP). No 6/2018. ISSN 1977-8449. doi:10.2800/ 571876.

EEA, 2019a. Air quality in Europe — 2019 report. No 10/2019. ISSN 1977-8449. doi:10.2800/822355.

EEA, 2019b. EMEP/EEA air pollutant emission inventory Guidebook 2019, 3.B Manure management.

EEA, 2019c. National Emission Ceilings Directive emissions data viewer 1990-2017 https://www.eea.europa.eu/data-and-maps/dashboards/necd-directive-data-viewer-2 (accessed 26 June 2020)

ERSAF, 2019. Consistenza dei capi di bestiame al 31.12. bovini e bufalini - Lombardia e province lombarde. https://www.asr-lombardia.it/asrlomb/it/14228consistenza-dei-capi-di-bestiame-al-3112-bovini-e-bufalini-lombardia-e-province-lombarde (accessed 15 March 2020).

Fangueiro, D., Ribeiro, H., Vasconcelos, E., Coutinho, J., Cabral, F., 2009. Treatment by acidification followed by solid–liquid separation affects slurry and slurry fractions composition and their potential of N mineralization. Bioresour. Technol. 100, 4914-4917. doi:10.1016/j.biortech.2009.04.032

Fattore, E., Paiano, V., Borgini, A., Tittarelli, A., Bertoldi, M., Crosignani, P., Fanelli, R., 2011. Human health risk in relation to air quality in two municipalities in an industrialized area of Northern Italy. Environ. Res. 111(8), 1321-1327. https://doi.org/10.1016/j.envres.2011.06.012

Fattorini, D., Regoli, F., 2020. Role of the chronic air pollution levels in the Covid-19 outbreak risk in Italy. Environ. Pollut. 264, 114732. https://doi.org/10.1016/j.envpol.2020.114732

Finzi, A., Mattachini, G., Lovarelli, D., Riva, E., Provolo, G., 2020. Technical, Economic, and Environmental Assessment of a Collective Integrated Treatment System for Energy Recovery and Nutrient Removal from Livestock Manure. Sustainability. 12, 2756. doi:10.3390/su12072756

Finzi, A., Riva, E., Bicoku, A., Guido, V., Shallari, S., Provolo, G., 2019. Comparison of techniques for ammonia emission mitigation during storage of livestock manure and assessment of their effect in the management chain. J. Agr. Eng. 50, 12–19. https://doi.org/10.4081/jae.2019.881

Giannakis, E., Kusha, J., Bruggeman, A., Lelieveld, J., 2019. Costs and benefits of agricultural ammonia emission abatement options for compliance with European air quality regulations. Environ. Sci Eur. 31:93. https://doi.org/10.1186/s12302-019-0275-0

Guarino, M., Fabbri, C., Brambilla, M., Valli, L., Navarotto, P., 2006. Evaluation of simplified covering systems to reduce gaseous emissions from livestock manure storage. Trans. ASABE. 49, 737-747.

Hansen, M.N., Sommer, S.G., Madsen, N.P., 2003. Reduction of Ammonia Emission by Shallow Slurry Injection. J. Environ. Qual. 32, 1099-1104. https://doi.org/10.2134/jeq2003.1099

Hoff, S.J., Bundy, D.S., Nelson, M.A., Zelle, B.C., Jacobson, L.D., Heber, A.J., Ni, J., Zhang, Y., Koziel, J.A., Beasley, D.B., 2006. Emissions of Ammonia, Hydrogen Sulfide, and Odor before, during, and after Slurry Removal from a Deep-Pit Swine Finisher. Japca J. Air Waste Ma., 56, 581-590, DOI: 10.1080/10473289.2006.10464472

INEMAR, 2020. INEMAR - ARPA Lombardia. INEMAR, Inventario Emissioni in Atmosfera: emissioni in Regione Lombardia nell'anno 2017 - revisione pubblica. ARPA Lombardia Settore Monitoraggi Ambientali.

ISMEA, 2019a. Settore lattiero caseario - Scheda di settore. http://www.ismeamercati.it/flex/cm/pages/ServeBLOB.php/L/IT/IDPagina/3521 (accessed 15 March 2020).

ISMEA, 2019b. Allevamento bovino da carne – Scheda di settore http://www.ismeamercati.it/flex/cm/pages/ServeBLOB.php/L/IT/IDPagina/3515 (accessed 15 March 2020).

ISMEA, 2019c. Settore suinicolo – Scheda di settore http://www.ismeamercati.it/flex/cm/pages/ServeBLOB.php/L/IT/IDPagina/3516 (accessed 15 March 2020).

ISTAT, 2019. Consistenze degli allevamenti. Territorio: Lombardia. http://dati.istat.it/Index.aspx?DataSetCode=DCSP_CONSISTENZE (accessed 15 March 2020).

Izquierdo, R., Dos Santos, S.G., Borge, R., de laPaz, D., Sarigiannis, D., Gotti, A., Boldo, E., 2020. Health impact assessment by the implementation of Madrid City air-quality plan in 2020. Environ. Res. 183, 109021. https://doi.org/10.1016/j.envres.2019.109021

Kiesewetter, G., Schoepp, W., Heyes, C., Amann, M., 2015. Modelling $PM_{2.5}$ impact indicators in Europe: Health effects and legal compliance. Environ. Modell. Softw. 74, 201-211. http://dx.doi.org/10.1016/j.envsoft.2015.02.022

Koolen, C.D., Rothenberg, G., 2019. Air pollution in Europe. Chem Sus Chem. 12, 164-172. doi:10.1002/cssc.201802292

Li, T., Hu, R., Chen, Z., Li, Q., Huang, S., Zhu, Z., Zhou, L-F., 2018. Fine particulate matter ($PM_{2.5}$): The culprit for chronic lung diseases in China. Chronic Diseases and Translational Medicine. 4, 176-186.

Lovarelli, D., Finzi, A., Mattacchini, G., Riva, E., 2020. A survey of dairy cattle behavior in different barns in Northern Italy. Animals. 10, 713; doi:10.3390/ani10040713

Marongiu, A., Angelino, E., Fossati, G., Moretti, M., Peroni, E., Pantaleo, A., Malvestiti, G., Abbatista, M., 2020. Stima preliminare delle emissioni in Lombardia durante l’emergenza Covid-19. ARPA Lombardia – Agenzia Regionale per la Protezione dell’Ambiente della Lombardia.

Mattila, P., Joki-Tokola, E., 2003. Effect of treatment and application technique of cattle slurry on its utilization by ley: I. Slurry properties and ammonia volatilization. Nutr. Cycl. Agroecosyst. 65, 221–230. https://doi.org/10.1023/A:1022619304798

McGinn, S.M., Flesch, T.K., Chen, D., Crenna, B., Denmead, O.T., Naylor, T., Rowell, D., 2010. Coarse Particulate Matter Emissions from Cattle Feedlots in Australia. J. Environ. Qual. 39, 791–798. doi:10.2134/jeq2009.0240

Oenema, O., Velthof, G., Klimont, Z., Winiwarter, W., 2012 Emissions from agriculture and their control potentials. IIASA TSAP report 3, Version 2.1

Pozzer, A., Tsimpidi, A.P., Karydis, V.A., de Meij, A., Lelieveld, J., 2017. Impact of agricultural emission reductions on fine-particulate matter and public health. Atmos. Chem. Phys. 17, 12813–12826. https://doi.org/10.5194/acp-17-12813-2017

PRIA, 2018. Piano Regionale degli Interventi per la qualità dell'Aria. Regione Lombardia.

Rebolledo, B., Gil, A., Pallarés, J., 2013. A spatial ammonia emission inventory for pig farming. Atmos. Environ. 64, 125-131. http://dx.doi.org/10.1016/j.atmosenv.2012.10.005

Thunis, P., Triacchini, G., Maffeis, G., Volta, M., 2009. Air pollution and emission reductions over the Po-valley: Air quality modelling and integrated assessment. 18th World IMACS / MODSIM Congress, Cairns, Australia 13-17 July 2009.

Thunis, P., Clappier, A., Tarrason, L., Cuvelier, C., Monteiro, A., Pisoni, E., Wesseling, J., Belis, C.A., Pirovano, G., Janssen, S., Guerreiro, C., Peduzzi E., 2019. Source apportionment to support air quality planning: Strengths and weaknesses of existing approaches. Environ. Int. 130, 104825. https://doi.org/10.1016/j.envint.2019.05.019

Van der Heyden, C., Demeyer, P., Volcke, E.I.P., 2015. Mitigating emissions from pig and poultry housing facilities through air scrubbers and biofilters: State-of-the-art and perspectives. Biosyst. Eng. 134, 74-93. http://dx.doi.org/10.1016/j.biosystemseng.2015.04.002

Welch, D.C., Colls, J.J., Demmers, T.G.M., Wathes, C.M., 2005. A methodology for the measurement of distributed agricultural sources of ammonia outdoors—Part 2: field validation and farm measurements. Atmos. Environ. 39(4), 673-684. https://doi.org/10.1016/j.atmosenv.2004.10.024

WHO, 2013. Health risks of air pollution in Europe – HRAPIE Project: Recommendations for concentration- response functions for cost-benefit analysis of particulate matter, ozone and nitrogen dioxide. Copenhagen, WHO Regional Office for Europe.

WHO, 2018. 9 out of 10 people worldwide breathe polluted air, but more countries are taking action. Geneva, News release. https://www.who.int/news-room/detail/02-05-2018-9-out-of-10-people-worldwide-breathe-polluted-air-but-more-countries-are-taking-action (accessed 15 March 2020).

Wang, P., Chen, K., Zhu, S., Wang, P., Zhang, H., 2020a. Severe air pollution events not avoided by reduced anthropogenic activities during COVID-19 outbreak. Resour. Conserv. Recycl. 158, 104814. https://doi.org/10.1016/j.resconrec.2020.104814

Wang, Y., Niu, B., Ni, J-Q., Xue, W., Zhu, Z., Li, X., Zou, G., 2020b. New insights into concentrations, sources and transformations of NH_3, NOx, SO_2 and PM at a commercial manure-belt layer house. Environ. Pollut. 262, 114355. https://doi.org/10.1016/j.envpol.2020.114355

Wang, S., Nan, J., Shi, C., Fu, Q., Gao, S., Wang, D., Cui, H., Saiz-Lopez, A., Zhou, B., 2015. Atmospheric ammonia and its impacts on regional air quality over the megacity of Shanghai, China. Sci. Rep. 5, 15842. doi:10.1038/srep15842

Wu, Y., Gu, B., Erisman, J.W., Reis, S., Fang, Y., Lu, X., Zhang, X., 2016. PM2.5 pollution is substantially affected by ammonia emissions in China. Environ. Pollut. 218, 86-94. http://dx.doi.org/10.1016/j.envpol.2016.08.027

Wu, X., Nethery, R.C., Sabath, B.M., Braun, D., Dominici, F., 2020. Exposure to air pollution and COVID-19 mortality in the United States. medRxiv preprint. https://doi.org/10.1101/2020.04.05.20054502

Xiao, C., Chang, M., Guo, P., Gu, M., Li, Y., 2020. Analysis of air quality characteristics of Beijing–Tianjin–Hebei and its surrounding air pollution transport channel cities in China. J. Environ. Sci. 87, 213-227. https://doi.org/10.1016/j.jes.2019.05.024

Yang, Y., Ruan, Z., Wang, X., Yang, Y., Mason, T.G., Lin, H., Tian, L., 2019. Short-term and long-term exposures to fine particulate matter constituents and health: A systematic review and meta-analysis. Environ. Pollut. 247, 874-882. https://doi.org/10.1016/j.envpol.2018.12.060

Zambrano-Monserrate, M.A., Ruano, M.A., Sanchez-Alcalde, L., 2020. Indirect effects of COVID-19 on the environment. Sci. Total Environ. 728, 138813. https://doi.org/10.1016/j.scitotenv.2020.138813

Zhao, Z.Q., Bai, Z.H., Winiwarter, W., Kiesewetter, G., Heyes, C., Ma, L., 2017. Mitigating ammonia emission from agriculture reduces PM2.5 pollution in the Hai River Basin in China. Sci. Total Environ. 609, 1152-1160. http://dx.doi.org/10.1016/j.scitotenv.2017.07.240

Table 1. Mean, standard deviation (SD), minimum and maximum values for daily weather parameters in the weather stations of the Lombardy region during the evaluated periods.

Year	**Parameter**	**January**		**February**		**March**	
		mean (SD)	Min-Max	mean (SD)	Min-Max	mean (SD)	Min-Max
2016-2019	W (m/s)	1.26 (0.3)	0.79-2.04	1.47 (0.45)	0.94-3.35	1.58 (0.28)	1.23-2.24
	RH (%)	75.72 (8.4)	60.99-95.6	80.2 (8.25)	64.23-98.63	67.99 (7.8)	55.63-85.48
	T (°C)	3.18 (0.84)	1.58-5.32	6.1 (0.82)	4.5-8.91	10.15 (2.04)	6.3-14.25
	R (mm)	0.53 (0.89)	0-3.99	2.48 (3.56)	0.01-18.65	1.25 (1.59)	0-5.06
2020	W (m/s)	1.29 (0.3)	0.88-2.2	1.69 (0.76)	0.83-3.48	1.75 (0.68)	0.98-3.35
	RH (%)	90.91 (7.61)	66.58-99.53	71.1 (20.5)	31.73-99.63	73.21 (14.64)	40.4-97.5
	T (°C)	4.19 (1.94)	0.35-7.28	8.32 (1.64)	5.45-11.03	9.86 (2.73)	6.05-14.9
	R (mm)	0.68 (2.32)	0-12.7	0.08 (0.18)	0-0.85	1.61 (3.23)	0-10

Notes: W = wind speed; RH = relative humidity; T = temperature; R = rainfall.

Table 2. Pearson's correlations for the period 2016-2019. In bold correlations r ≥ 0.60.

Parameters	**NOx 2016-19 city**	**NH_3 2016-19 country**	**NH_3 2016-19 city**	**$PM_{2.5}$ 2016-19 country**	**$PM_{2.5}$ 2016-19 city**	**Wind 2016-19**	**RH 2016-19**	**T 2016-19**	**Rain 2016-19**
NOx 2016-19 country	**0.97**	0.30	0.05	**0.81**	**0.79**	0.13	**0.68**	-0.35	0.03
NOx 2016-19 city		0.26	0.01	**0.74**	**0.74**	0.10	**0.64**	-0.40	-0.02
NH_3 2016-19 country			**0.88**	0.42	0.42	0.46	0.47	**0.60**	-0.12
NH_3 2016-19 city				0.25	0.26	0.47	0.36	**0.73**	-0.03
$PM_{2.5}$ 2016-19 country					**0.98**	0.14	**0.74**	-0.14	0.00
$PM_{2.5}$ 2016-19 city						0.14	**0.77**	-0.16	0.03
Wind 2016-19							0.56	0.59	0.26
RH 2016-19								0.15	0.35
T 2016-19									0.09

Notes: Wind = wind speed; RH = relative humidity; T = temperature; Rain = rainfall events.

Table 3. Pearson's correlations for the period 2020. In bold correlations r ≥ 0.60.

Parameters	**NOx 2020 city**	**NH$_3$ 2020 country**	**NH$_3$ 2020 city**	**PM$_{2.5}$ 2020 country**	**PM$_{2.5}$ 2020 city**	**Wind 2020**	**RH 2020**	**T 2020**	**Rain 2020**
NOx 2020 country	**0.96**	0.14	-0.04	**0.79**	**0.80**	-0.08	**0.65**	-0.33	-0.09
NOx 2020 city		0.19	-0.05	**0.76**	**0.77**	-0.07	**0.62**	-0.32	-0.09
NH$_3$ 2020 country			**0.86**	0.32	0.28	0.12	0.20	0.57	-0.08
NH$_3$ 2020 city				0.23	0.20	0.03	0.18	**0.63**	-0.01
PM$_{2.5}$ 2020 country					**0.99**	-0.02	**0.76**	0.03	-0.10
PM$_{2.5}$ 2020 city						-0.08	**0.75**	-0.03	-0.08
Wind 2020							0.22	0.54	0.13
RH 2020								0.13	0.21
T 2020									-0.01

Notes: Wind = wind speed; RH = relative humidity; T = temperature; Rain = rainfall events.

Table 4. Factor Analysis for the period 2016-2019.

Parameters	**Factor1**	**Factor2**	**Factor3**
NOx 2016-19 country	**0.86**	-0.42	-0.01
NOx 2016-19 city	**0.81**	-0.46	-0.04
NH$_3$ 2016-19 country	0.63	**0.63**	-0.35
NH$_3$ 2016-19 city	0.45	**0.78**	-0.26
PM$_{2.5}$ 2016-19 country	**0.90**	-0.23	-0.11
PM$_{2.5}$ 2016-19 city	**0.91**	-0.23	-0.08
Wind 2016-19	0.42	**0.62**	0.38
RH 2016-19	**0.89**	0.10	0.33
T 2016-19	0.06	**0.96**	0.00
Rain 2016-19	0.10	0.10	**0.91**

Table 5. Factor Analysis for the period 2020.

Parameters	**Factor1**	**Factor2**	**Factor3**
NOx 2020 country	**0.90**	-0.31	-0.01
NOx 2020 city	**0.89**	-0.29	-0.02
NH$_3$ 2020 country	0.37	**0.78**	-0.31
NH$_3$ 2020 city	0.24	**0.84**	-0.32
PM$_{2.5}$ 2020 country	**0.95**	0.03	-0.03
PM$_{2.5}$ 2020 city	**0.95**	-0.03	-0.03

Wind 2020	-0.01	0.44	**0.69**
RH 2020	**0.81**	0.10	0.40
T 2020	-0.04	**0.92**	0.15
Rain 2020	-0.06	0.03	**0.68**

Table 6. General Linear Model results.

Parameter	**Estimate**	**S.E.**	**t Value**	**Pr > |t|**
Intercept	-0.431	4.823	-0.090	0.929
NOx 2020 country	0.087	0.050	1.760	0.085
NOx 2020 city	-0.096	0.035	-2.730	0.009
NH_3 2020 country	0.249	0.028	9.050	<.0001
$PM_{2.5}$ 2020 country	0.028	0.165	0.170	0.864
$PM_{2.5}$ 2020 city	-0.020	0.141	-0.140	0.889
Wind 2020	-2.371	0.954	-2.490	0.016
RH 2020	0.045	0.043	1.040	0.301
T 2020	0.417	0.271	1.540	0.130
Rain 2020	0.156	0.311	0.500	0.618

Highlights

- $PM_{2.5}$ is formed in the atmosphere by a chemical reaction among NH_3, SOx, and NOx
- Covid-19 outbreak interrupted many productive activities, except agricultural ones
- City and countryside data stations in Lombardy (North Italy) were analyzed
- NOx and $PM_{2.5}$ emissions reduced during quarantine period but not NH_3
- Combined efforts from productive sectors are expected to reduce $PM_{2.5}$ emissions

Declaration of interests

☒ The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

☐The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: