Ozone Deposition on Free-Running Indoor Materials and the Corresponding Volatile Organic Compound Emissions: Implications for Ventilation Requirements

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Abstract: Recently, tropospheric ozone has become a public health concern worldwide, along with the continuous battle against ambient fine particulate matter in countries like China. In this study, we investigate the impact of indoor ozone pollution using seven materials categorized as either wood-based panels or synthetic fibers, which were freely-stored in an office/lab environment. Most materials were considered as used and aged more than 1–2 years. An experimental apparatus was used to study ozone deposition and detect volatile organic compound (VOC) emissions from the specimens when exposed to ozone at three concentration levels: <10, 100, and 300 ppb. A simplified model is proposed to discuss ventilation requirements based on a standard room. We found that the mean ozone deposition velocities from the seven materials ranged from 0.005 to 0.062 cm s⁻¹. Both the engineering wood and some of the synthetic fibers were, moreover, prone to ozone deposition. Second, 15 VOCs were found in the sampling air from a 24-VOC target list after ozone exposure. The emission rates of the VOCs from all seven materials were then determined. Third, when the ozone concentration in the outdoor air is not severely high, it is possible to use ventilation to maintain acceptable indoor air quality.

Keywords: environmental chamber; wood-based panels; engineering wood; synthetic fibers; ventilation rate

1. Introduction

Tropospheric ozone (O₃), namely ground-level O₃, is a key component of photochemical air pollution and is widely considered a threat to vegetation, infrastructure, and human health (e.g., nonaccidental, cardiovascular, and respiratory mortality) [1-6]. Compared to air pollution of fine particulate matter, O₃ pollution is much less visible, making it a smaller target for public concern. Nevertheless, the overall picture is alarming. For example, over the last three decades, the emission of chemical precursors to O₃, such as, nitrogen oxides and volatile organic compounds (VOCs), has increased sharply in China, surpassing that of North America and Europe [7]. Consequently, the annual average daily maximum 8 h O₃ concentration in China ranged from 74 to 201 μg·m⁻³ (138 ± 24.7 μg·m⁻³) in 2016. Approximately 30% of the total population was exposed to an O₃ concentration of more than 160 μg·m⁻³, which is the Chinese national ambient air quality standard [8], and 67.2% of the urban population was exposed to concentrations above the WHO (World Health Organization) recommended concentration of 100 μg·m⁻³ [9]. In April 2019, the 90th percentile of the average daily

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maximum 8 h O₃ concentration reached or exceeded 100 and 160 µg·m⁻³, in 164 and 44 of the 168 reported cities, respectively [10].

In the indoor environment, ozone concentrations vary as a function of both the outdoor contributions and indoor conditions [11–13] and are generally lower than the outdoor concentrations [14]. Early studies indicate that the ratios of indoor to outdoor ozone concentrations are mainly in the range of 0.2–0.7 [12]. However, indoor exposure can constitute up to 60% of a person’s total exposure to ozone [15,16]. Copiers, laser printers, electronic air cleaners, and ozone generators can act as ozone sources, with emission rates ranging from 0.1 to 100 mg·h⁻¹, which are comparable to outdoor air as a source of ozone on a highly polluted day [11]. In the absence of indoor sources, the indoor ozone concentration depends on the outdoor ozone concentration, air change rate (ACH or ventilation rate), and removal rate of indoor surfaces, including human hair, skin and clothes, and ozone-initiated reactions with organic gases [17–20]. Among the various mechanisms, surface removal and ventilation play major roles in changing indoor ozone concentrations because the gas-phase reactions initiated by ozone are slow for most VOCs compared to ACH and the surface removal rate under typical indoor conditions [12,17,21,22]. The effects of surface removal and ventilation on indoor ozone concentrations are coupled for the following reasons. On the one hand, although surface removal is a prevalent and effective way to eliminate indoor ozone, it is sometimes accompanied by harmful or irritating byproducts, including C1-C13 carbonyls, dicarbonyls, hydroxycarbonyls, and ultrafine particles [23–25], which may require higher ventilation rates to dilute [26]. On the other hand, it is obvious that outdoor ozone can be brought indoors by ventilation. Therefore, the indoor air concentrations of ozone and its byproducts may not be improved by increasing the ventilation rate. One possible solution is to adopt mass accretion on material surfaces, which may passively remove indoor ozone while producing no byproducts or only benign byproducts [27–29]. At present, this potential improvement measure has not attracted enough attention in China.

Throughout the available literature, the relationships between ozone and indoor materials has been extensively studied, mostly focusing on carefully-controlled, newly-obtained materials in developed countries. However, very few studies were conducted under Chinese scenarios using aged materials on a large scale [17]. The effects of potentially high indoor ozone concentrations that partially stem from outdoor pollution on everyday materials in Chinese indoor environments are still unclear. Limited studies on the long-term emissions from aged or free-running materials show that the emission mechanisms of pollutants may change over time due to the physical and chemical changes in the materials (e.g., diffusion coefficients), and/or the emission process may be associated with primary and secondary emissions based on chemical reactions that are triggered by changes in environmental conditions (e.g., ozone-initiated reactions and hydrolysis due to the presence of water) during the application of a material [30,31]. In addition, there are major differences in the average residences in the U.S. and China. Carpets are commonly used in the U.S. (approximately 70% of the floors in American residences are covered with carpets) [32]. By comparison, only approximately 10% of floors are covered with carpets in Chinese buildings [33]. Generally speaking, the reaction probabilities of carpets are one or two orders of magnitude higher than those of finished floors [25]. As a result, the portion of indoor ozone consumed by carpets, as well as the corresponding byproduct yields, would be less in a typical residential setting in China. Moreover, VOC emissions from indoor building materials are more strictly regulated in developed countries [34] due to the implementation of various label schemes, such as AgBB in Germany [35]. Thus, the effects of ozone on both the primary and secondary emissions from aged or free-running indoor materials in Chinese markets could be different from the results reported in previous controlled chamber studies. This research gap still needs to be addressed.

In this study, seven indoor materials (three types of wood-based panels and four types of synthetic fibers), which can all be commonly found indoors and were purchased from local stores or online retailers in China, were selected as the testing materials. Because wood-based panels may be applied indoors over the long term, and because emissions more or less reach a quasi-steady state after 3–6 months [36], materials that were free-run for 1–2 years, which can be considered “used”, were selected. The estimated ages for the synthetic fibers before testing were also approximately 1–2
years. The ozone deposition velocities on the seven materials, as well as the emission rates of the 24 VOCs (including formaldehyde), were measured at three different ozone concentration levels using two 50 L chambers. The reaction probabilities between the ozone and the materials were excluded due to technical limitations. Furthermore, the effects of ozone pollution on indoor ventilation requirements were analyzed. We also discuss the key chemical substances that can be used as ventilation determinants when considering ozone exposure.

2. Materials and Methods

2.1. Material Selection and Preparation

Detailed information for the specimens of the selected materials is summarized in Tables 1 and 2.

As shown in Tables 1 and 2, the seven materials were subjected to testing for the following reasons. First, according to the China National Forest Products Industry Association (CNFPIA), both the total production (by approximately 6400 domestic manufacturers) and total consumption of wood-based panels exceeded 290 million m$^2$ in 2017 in China [37]. Since the thickness of wood-based panels is usually in the range of 10–20 mm, the total exposed area of potential emissions and ozone deposition in Chinese markets annually is approximately more than ten billion m$^2$. In this study, three of the most abundant types of wood-based panels (oriented strand board (OSB) with green-core moisture-resistant additives, lauan plywood, and paint-free pine) were purchased from a local market in Shanghai on March 8th, 2018. Upon purchase, the preceding storage time for each of the materials was approximately 1–2 years. Although we expected to detect high concentrations of formaldehyde released from both the OSB and plywood, due to the additive of urea-formaldehyde resin, it remained unknown to what extent ozone would change the corresponding ventilation requirements.

| Table 1. Material information on the specimens of the three wood-based panels. |
|-----------------------------|-------------------------------|-------------------------------|-------------------------------|
| Item                        | OSB with a Green-Core, i.e., | Lauan Plywood                | Paint-Free Pine               |
| Estimated age               | >1 year                       | >1–2 years                    | >1–2 years                    |
| Pictures                    |                               |                               |                               |
| Exposed area, m$^2$         | 0.061                         | 0.054                         | 0.075                         |
| Thickness, mm               | 18.97                         | 7.65                          | 16.55                         |
| Weight, g                   | >500.000$^{1}$                | 234.083                       | 623.731                       |
| Density, kg·m$^{-3}$        | >432.1                        | 566.6                         | 502.5                         |
| Loading ratio, m$^2$·m$^{-3}$| 1.24                          | 1.10                          | 1.52                          |

$^{1}$ The weight was out of the range of the first balance. The specimen of pine was weighed by a second balance (similarly thereafter).

Second, synthetic fibers, e.g., synthetic leather, started to gain popularity in the 1980s in China as a substitute for leather and fur, which most people could not afford. Today, synthetic fibers are everywhere, partially due to the economic situation decades ago, as people would like to own what they could not afford when they were young. In addition to carpet, three types of commonly-found synthetic fibers (i.e., double-sided flannel, weaved leather, and microfiber lining) that are heavily-used indoors for sofas and bedding were subjected to testing. The carpet was purchased from a local
market in Shanghai on 8th March 2018. Although carpets are an unpopular product in China, they are still worth testing for the sake of comparison. The rest of the materials, which were less-frequently tested for either VOC emissions or ozone deposition, especially for building applications in a free-run manner, were purchased online on 22nd November 2018, and were received within one week. During testing, the edges of the materials were wrapped and covered with aluminum foil. The estimated ages of the selected fibers were 1–2 years before testing.

| Table 2. Material information on the specimens of the four synthetic fibers. |
|------------------------|------------------|------------------------|------------------|------------------|
| Item                   | Flannel          | Weaved Leather         | Microfiber Lining| Carpet           |
| Estimated age          | <1 year          | <1 year                | <1 year          | 1–2 years        |
| Exposed area, m²       | 0.091            | 0.088                  | 0.080            | 0.064            |
| Thickness, mm          | 0.90             | 0.59                   | 0.80             | 4.83             |
| Weight, g              | 18.327           | 70.514                 | 30.727           | 200.065          |
| Density, kg-m⁻³        | 233.8            | 1358.1                 | 480.1            | 647.2            |
| Loading ratio, m²-m⁻³  | 1.62             | 1.79                   | 1.84             | 1.30             |

As previous studies have suggested [26,38], many wood-based materials exhibit homogeneous properties at a large scale, so inherent differences can be pivotal when physically or chemically comparing small samples. Therefore, two specimens of OSB, lauan plywood and paint-free pine, were tested separately (each specimen was tested at least twice), while only one specimen from each synthetic fiber was cut from the main product and subjected to testing two to four times throughout the experiment. According to the pre-experiments, the specimens of synthetic fibers were much more homogeneous compared to the wood-based ones. Before testing, all the specimens were kept at room temperature and freely stored in an office/lab room for more than a month without further control measures. The dimensions and weights of the samples were measured by a Vernier caliper and an electronic balance (with a precision of 0.001 g), respectively. Moreover, because the physical and chemical properties of the materials could be slightly different after exposure to ozone, a minimal gap of five days for material recovery was ensured between two consecutive tests.

2.2. Environmental Chamber Set-Up

Both the ozone deposition velocities and VOC emissions were measured in a laboratory chamber apparatus, as shown in Figure 1. A schematic of the apparatus is shown in Figure 2, including two 50 L stainless-steel emission chambers (No. 1 and No. 2; the inner dimensions for each chamber were 40 cm × 39 cm × 31.7 cm).

Each of the 50 L chambers was equipped with 18 on/off ports, nine on each side, as shown in Figure 1b, as air inlets or outlets for exhaust, sampling, etc., and the leakage rate of each chamber was measured at 5%–10%. All the tubes were made of polytetrafluoroethylene (PTFE). Before each round of testing, the inner surfaces of the emission chambers were scavenged orderly by medicinal alcohol, deionized water, heating, and clean air to minimize the sink and source effects. Chamber No. 1 was dedicated to investigating the effects of ozone, e.g., the deposition velocities and the primary and secondary VOC emissions, at either 100 ppb (i.e., a fairly-polluted level, approximately 214.3 μg-m⁻³ under standard conditions) or 300 ppb (i.e., an extremely-polluted scenario, equivalent to 642.9 μg-m⁻³ under standard conditions) for the materials, while No. 2 was used to measure emissions from the material with low background ozone concentrations (<10 ppb or <21.4 μg-m⁻³, i.e., below the detection limit) for comparisons. The same specimen was placed into the two chambers separately for designated testing.
Figure 1. The laboratory chamber apparatus. (a) Environmental chamber. (b) Two emission chambers with multiple ports.

Figure 2. Schematic diagram of the laboratory apparatus for measuring ozone deposition and volatile organic compound (VOC) emissions rates.

Two channels of clean air were supplied separately at approximately 1.25 ± 0.1 L·min⁻¹ (to avoid suck-back when sampling) to the two emission chambers (the ACH was approximately 1.52 h⁻¹) using an electromagnetic air compressor (Hailea ACO-009E, 160 W, Guangdong Hailea Group Co., Ltd., Chaozhou, China), a zero-air generator (BCHP ZA-1000, Beijing BCHP Analytical Technology Institute, Beijing, China), and two mass flow controllers (Aera FC-7700CD, 0–1000 sccm, Advanced Energy Industries, Inc., Fort Collins, CO, USA) in parallel. The clean-gas cylinder was used in the beginning and was replaced by the air compressor and zero-air generator combination due to technical issues. A primary airflow calibrator (MesaLabs Bios Defender 520-M, 50–5000 mL·min⁻¹, Mesa Laboratories, Inc., Lakewood, CO, USA) was used to calibrate the flow rate.

Both emission chambers were stored inside a 1 m³ environmental chamber to maintain a constant temperature of approximately 23 °C, which was also measured using high-precision sensors (Vaisala HMP 110, Vaisala Oyj, Vantaa, Finland) at a 1 min interval. In addition, since the air flowrate in the two channels was slow, the air was pre-conditioned to the same temperature as the emission chamber, and the relative humidity (RH) of the air was approximately 20%. All the data on temperature, RH, and flowrate were recorded by a graphical programming platform called ForceControl (V7.0, Beijing ForceCon Yuantong Technology, Beijing, China).
2.3. Testing Procedures

2.3.1. Ozone Deposition Velocities

The tests on ozone deposition velocity and VOC emissions were performed separately. To conduct ozone deposition on materials, the following methods and procedures were employed.

First, two ten-time series of experiments—one using 100 ppb of ozone and another using 300 ppb—were conducted on the empty chamber to obtain the two average deposition velocities of ozone on the chamber wall. Clean air was pumped through the channel connected to the chamber into an ozone generator (2B 306, 2B Technology, Inc., Boulder, CO, USA, the flow rates were 1.0–1.3 L·min⁻¹ during the experiments). Then, air was introduced into the chamber with a constant concentration of ozone. The ozone concentration was measured and recorded by an ozone analyzer (2B 106-L/205, 2B Technology, Inc., Boulder, CO, USA, with a flowrate ranging from 0.8 to 1.1 L·min⁻¹ when sampling) at a 1 min interval using one of the outlet ports.

As shown in Equation (1), a mass balance model of ozone concentration in the chamber was used to determine the ozone deposition on the chamber surfaces:

\[
\frac{dy_{\text{ozone}}}{dt} = NV(y_{\text{ozone,in}} - y_{\text{ozone}}) - 36v_{d,s}A_{d,s}y_{\text{ozone}} - 36v_{d,w}(A_{d,w} - A_{d,s})y_{\text{ozone}},
\]

where \( V \) is the volume of the chamber, m³; \( y_{\text{ozone}} \) is the indoor ozone concentration in the chamber (or in a room), µg·m⁻³; \( N \) is the air change rate, h⁻¹; \( y_{\text{ozone,in}} \) is the ozone concentration in the supplied air, µg·m⁻³; \( v_{d,s} \) and \( v_{d,w} \) are the area-averaged ozone deposition velocity on the testing specimen and the chamber wall, respectively, cm·s⁻¹; and \( A_{d,s} \) and \( A_{d,w} \) are the exposed area for ozone deposition on the specimen and chamber wall, respectively, in m².

The zone deposition velocity on the material specimen under a steady state can be calculated via Equation (2):

\[
v_{d,s} = \frac{NV}{A_{d,s}} \left( \frac{y_{\text{ozone,in}}}{y_{\text{ozone}}} - 1 \right) - v_{d,w} \left( \frac{A_{d,w}}{A_{d,s}} - 1 \right),
\]

and \( v_{d,w} \) can be determined by Equation (3) when no sample is placed in the chamber:

\[
v_{d,w} = \frac{NV}{A_{d,w}} \left( \frac{y_{\text{ozone,in}}}{y_{\text{ozone}}} - 1 \right).
\]

We tested the ozone deposition in an empty chamber by purging the ozone at two different concentrations ten times. In this way, \( v_{d,w} \) s were both determined to be \( 1.25 \times 10^{-3} \pm 0.28 \times 10^{-3} \text{ cm·s}^{-1} \), which is similar to the results reported in [39]. After determining \( v_{d,w} \) using the empty-chamber tests, \( v_{d,s} \) was determined by conducting ozone deposition tests on different materials. Each time, one specimen from each of the materials was placed in the chamber, and the steady-state \( v_{d,s} \) value was determined using Equation (2) along with the procedure described in this section. Since each test lasted for approximately 1–6 h, at least two tests were repeated on the same specimen, and two specimens were prepared for all the wood-based materials.

A summary of the testing procedure for ozone deposition velocities is given in Table 3.

**Table 3.** The testing procedure for ozone deposition velocities.

| Specimen | \( y_{\text{ozone,in}} \) Number of Tests | \( y_{\text{ozone}} \) | \( N \) | \( V \) | \( A_{d,s} \) | \( A_{d,w} \) | \( v_{d,s} \) | \( v_{d,w} \) |
|----------|--------------------------------|-----------------|------|------|------|------|------|------|
| None     | 100 ppm 10 | 300 ppm 10 | | | | | | |
| One at a time | 100 ppm Measured 2–4 | 300 ppm Measured 2–4 | | | | | | |
2.3.2. Primary and Secondary VOC Emissions

The primary and secondary VOC emissions from the materials before and after exposure to ozone were analyzed using the following procedure. After putting the material in the chamber, with or without exposure to ozone, the air was extracted after 4 h from the chamber for chemical analysis. Concentrations of the 24 target pollutants were sampled (GilAir Plus, Sensidyne, St. Petersburg, FL, USA) in Tenax TA tubes at 0.1 L·min⁻¹ for 30 min. The selected pollutants included aromatic (e.g., benzene and toluene), aliphatic (e.g., undecane), and ester compounds (e.g., ethyl acetate). The gas chromatography-mass spectrometer (Finnigan TRACE DSQ, Thermo Electron Corporation, Waltham, MA, USA) method was used to analyze the mass of the targeted compounds in the Tenax TA tubes [40], which were kept at room temperature after sampling. Despite the advantages of using the Chinese 40 L desiccator method to measure the formaldehyde from wooden materials [41], the method outlined in [42] was adopted to measure aldehydes (formaldehyde, acetaldehyde, etc.) and ketones (e.g., acetone) using the same procedure for all seven materials. Aldehydes and ketones were sampled by DNPH cartridges with ozone filters (Sep-Pak, Waters, Milford, MA, USA) at 0.5 L·min⁻¹ for 30 min. The samples were stored at <4 °C after being eluted from the DNPH cartridges and were analyzed via the ultra-performance liquid chromatography (Acquity UPLC H-Class Plus System, Waters, Milford, MA, USA) method with a tunable UV detector (Waters, Milford, MA, USA). In terms of the possible high concentrations of air pollutants, the DNPH cartridge and Tenax TA tube were connected in series to avoid breakthrough. The background VOC concentrations in the empty chambers exposed to ozone at a designated concentration were also obtained following the procedure before each round of testing.

Assuming all the free-run materials were exposed to the indoor environment long enough to reach a quasi-steady state, the emission factor of the i-th compound from the testing specimen, \( F_i \), was determined using Equation (4) [29]:

\[
F_i = \frac{nv(y_i - y_{i,in}) - nv(y_{i,e} - y_{i,in})}{A_{ds}} \left(1 - \frac{A_{ds}}{A_{dw}}\right),
\]

where \( y_i \) is the concentration of the i-th compound in the chamber containing a material specimen, \( \mu g \cdot m^{-3} \) and \( y_{i,in} \) and \( y_{i,e} \) are the concentrations of the i-th compound in the supplied air and the empty chamber, respectively, in \( \mu g \cdot m^{-3} \).

Notably, the gas-phase and surface reactions involving the i-th compound were not independently measured. Therefore, in Equation (4), \( F_i \), was grouped together considering the total molecules consumed by reacting with ozone and/or the possible production yield as one of the outcomes of the reactions. Nevertheless, methods to independently measure the gas-phase consumption and area-averaged yield of related compounds by ozone deposition are recommended.

A summary of the testing procedure for detecting primary and secondary VOC emissions is given in Table 4. Due to the limited resources and time, one specimen from each material was subjected to testing of the changes in VOC emission rates, while two parallel samplings were conducted, and the averaged concentration data were used to determine the \( F_i \) for each compound.

| Specimen | Chemicals | \( y_{ozone,in} \) | \( y_{i,in} \) | \( y_{i,e} \) | \( y_i \) | \( F_i \) |
|----------|-----------|-----------------|--------------|-------------|--------|--------|
| One specimen from each material | Aromatic, aliphatic, and ester compounds | First 0, then 100 ppm | Measured following [40] | Determined by Equation (4) |
| | Aldehyde and ketone compounds | First 0, then 100 ppm | Assumed at 0 |

2.4. The Effects of Ozone on Ventilation Requirements

Because both the ozone and VOCs emitted from indoor materials are indoor air pollutants, the discussions on ventilation requirements should balance inducing ozone indoors with diluting VOCs that originate from materials and surface deposition. However, it can be difficult to take such a
holistic approach, as many details of the indoor chemistry of this study remain unknown. Instead, the implications of exposing indoor materials to ozone were determined using a hypothetical standard room from the following two perspectives:

(1) The effects of ventilation requirements on indoor ozone concentrations after deposition.
(2) The ventilation requirements for diluting the VOC emissions from the selected free-running indoor materials by considering ozone-related reactions.

2.4.1. Simplified Methods to Discuss Ventilation Requirements

A general steady-state mass-balance model was revised based on the model in [26]. The model accounted for the ozone-initiated reaction in both the gas- and material- phases. As shown in Equation (5), the indoor gas-phase ozone concentration can be affected by the outdoor ozone concentration, ventilation rate, consumption (with an assumed second-order reaction in the gas phase [21]), and consumption by surface deposition. The following assumptions have been made:

(1) The room air was well-mixed.
(2) No indoor air cleaners or any other means were used to purify the indoor air.
(3) No occupants were present, as occupants are both emission sources and sinks.
(4) The VOC concentrations in the outdoor air were ignored.
(5) The production of any compound from the gas-phase ozone reactions was not considered.
(6) If one reaction produced by-products that could be reactants in another reaction, the effects of the by-products on the ventilation rate requirements were ignored.

\[
\frac{d y_{\text{ozone}}}{d t} = NV(y_{\text{ozone}}^0 - y_{\text{ozone}}) - \sum r_i \cdot MW_{\text{ozone}} V - \sum v_d A_{d,s} y_{\text{ozone}}
\]  

(5)

where \( y_{\text{ozone}} \) is the indoor ozone concentration, \( \mu g \cdot m^{-3} \); \( y_{\text{ozone}}^0 \) is the outdoor ozone concentration, \( \mu g \cdot m^{-3} \); \( MW_{\text{ozone}} \) is the molecular weight of the ozone, \( \mu g \cdot mol^{-1} \); and \( r_i \) is the second-order reaction rate between the \( i \)-th compound and ozone, mol\( \cdot m^{-3} \cdot h^{-1} \), which can be determined by Equation (6):

\[
r_i = k_i \cdot \frac{y_{\text{ozone}}}{MW_{\text{ozone}}} \cdot \frac{y_i}{MW_i}
\]  

(6)

where \( MW_i \) is the molecular weight of the \( i \)-th compound, \( \mu g \cdot mol^{-1} \), and \( k_i \) is the rate constant for the reaction between the \( i \)-th compound and ozone, cm\(^3\)-molecule\(^{-1} \cdot s^{-1} \). As recommend in [21,22], the \(-\log_{10}(k_i)\) value and unit for \( m \)-xylene, \( p \)-xylene, toluene, and formaldehyde were taken as 21.2, 21.4, 21.8, and 23.7 cm\(^3\)-molecule\(^{-1} \cdot s^{-1} \), respectively. \( k_i \) was regarded as zero for any unreactive chemicals with ozone, as well as for all the chemicals that were not in the original target list but may be reactive to ozone.

Alternatively, a mass-balance model for the \( i \)-th compound can be coupled with Equation (5) to obtain the steady-state concentrations of both ozone and the chemical compound. However, since the gas-phase consumption and surface yield of the compound were not independently recorded during the experiments, the lumped emission factor, \( F_i \), was used to predict the gas-phase concentration of the \( i \)-th compound, \( y_i \), in hypothetical scenarios using Equation (7), and then \( y_i \) was substituted into Equation (6):

\[
\frac{d y_i}{d t} = F_i A_{d,s} - NV y_i.
\]  

(7)

Based on Equations (5)–(7), under a steady state, the indoor ozone concentration considering chemical depletion can be assessed by Equation (8), while the parameters in Equation (8) can be either measured or pre-determined:

\[
y'_{\text{ozone}} = \frac{NV y_{\text{ozone}}}{\sum (k_i \cdot \frac{A_{d,s}}{MW_i}) + v_d A_{d,s} + NV'}
\]  

(8)

where \( y'_{\text{ozone}} \) is the estimated indoor ozone concentration under a steady state through consumption in the gas phase, considering ozone deposition on the indoor surfaces and removal by ventilation, in \( \mu g \cdot m^{-3} \). Further, based on Equation (7), assuming the emission rate, \( F_i \), is a constant, a corresponding
ventilation requirement under a steady state can be obtained by setting an upper indoor concentration level \( C_i \) for the \( i \)-th compound. Then, assuming the material emitting the compounds was placed in an indoor environment, the ventilation requirements for the indoor environment can be determined by identifying the maximum individual ventilation rate requirements, as given by Equation (9):

\[
N = \max \left\{ \frac{F_i A_{ds}}{C_i V} \right\},
\]

where \( C_i \) is the indoor concentration limit for the \( i \)-th compound, in \( \mu g \cdot m^{-3} \).

At present, the current national standard on indoor air quality in China (GB/T 18883-2002) [43] recommends an indoor concentration limit, \( C_i \), for formaldehyde (100 \( \mu g \cdot m^{-3} \)), benzene (110 \( \mu g \cdot m^{-3} \)), toluene (200 \( \mu g \cdot m^{-3} \)), and xylene (200 \( \mu g \cdot m^{-3} \)) from the target list of VOCs, as well as ozone (160 \( \mu g \cdot m^{-3} \), equivalent to 74.7 ppb in standard conditions). The above concentration thresholds were thus adopted when using Equations (8) and (9). In contrast, because the rest of the VOCs were not regulated in [43], they were not used to determine the ventilation requirements in this study.

2.4.2. Standard Room Configurations

To maximize the impact of the emissions of different materials and the ozone-related reactions on the ventilation requirements, a standard office room was used [44]. The major configurations of the standard room are summarized in Table 5.

| Parameters of the standard office | Value |
|-----------------------------------|-------|
| Volume (\( m^3 \))                | 65.2  |
| Emission loading ratio (\( m^2 \cdot m^{-3} \)) | 1.2   |
| Ceiling/flooring surface area (\( m^2 \)) | 23.8  |
| Wall surface area (\( m^2 \))      | 53.1  |

Using a standard room or reference room allows for a comparison between emission sources at the same time. This method can also greatly overestimate the concentrations of pollutants (e.g., formaldehyde) in indoor areas when diverse sources are simply added together [45]. Therefore, to avoid overestimating the indoor pollution levels by adding multiple materials into one scenario, each material from the group of either wood-based panels or synthetic fibers was individually subjected to ventilation rate requirement studies. The ozone deposition area, \( A_{ds} \), was assumed to be equal to the VOC emission area, and the loading ratio was set to 1.2 \( m^2 \cdot m^{-3} \) (i.e., \( \frac{A_{ds}}{V} = 1.2 \)), which may not be necessarily true in practice. Ozone interactions with the rest of the indoor surface were ignored.

The purpose of using a standard room was to more precisely compare the disparities in the ventilation requirements for various materials exposed to ozone. Absolute ventilation requirements should be used with caution.

3. Results and Discussion

3.1. Ozone Deposition Velocities

3.1.1. On Wood-Based Materials

Ozone deposition on the specimens comprising three types of wood-based panels was conducted with 100 ppb and 300 ppb of ozone. The ozone deposition velocities on the materials \( v_{ds} \) were calculated using Equations (1)–(3), as shown in Figure 3.
It can be observed from Figure 3 that, among the selected materials, the deposition velocities were the highest on the green-core OSB and the lowest on the paint-free pine. When the inlet ozone concentration, $y_{\text{oz},\text{in}}$, was controlled at 100 ppb, it took only 30 min for ozone deposition to reach a steady state on the pine, while approximately 2.5 h was needed for ozone deposition to reach a steady state on the OSB. The outlet concentrations of ozone, $y_{\text{oz}}$, which were assumed to be equal to the concentration of well-mixed ozone in the chamber, were the lowest when testing with OSB and the highest with pine. This is reasonable since both OSB and plywood are engineered wood types manufactured by binding or fixing strands or veneers of wood together using adhesives or other methods of fixation. Compounds that may be reactive to ozone were far more densely present on or in the OSB and plywood than on or in the pine, which, in this case, was the paint-free pine. By comparison, when $y_{\text{oz},\text{in}}$ was at 300 ppb, a similar trend was observed between the three specimens of the panels. It took approximately 1 h for ozone deposition to reach a steady state on both the OSB and the plywood. However, it nearly took 6 h for the outlet concentration of ozone to reach a steady state when a piece of pine was in the chamber. Moreover, when elevating $y_{\text{oz},\text{in}}$, the $v_{d,s}$ values decreased for both the plywood and pine but increased for OSB.

3.1.2. On Synthetic Fibers

The $v_{d,s}$ values on the four types of synthetic fibers were determined after two series (i.e., $y_{\text{oz},\text{in}} = 100$ ppb or $y_{\text{oz},\text{in}} = 300$ ppb) of experiments and are summarized in Figure 4.
As shown in Figure 4, the ozone deposition velocities on various synthetic fibers showed large discrepancies. The fastest velocities, which surpassed 0.055 cm·s⁻¹ by two times, were found for the interactions with flannel, for which it took almost 3 h for ozone deposition to reach a steady state after introducing ozone at a concentration of either 100 ppb or 300 ppb. The most rapid deposition was found on the carpet, which was covered by a surface of napped velvet. The steady-state ozone concentrations were around 46 ppb (after 1.5 h, and the concentration was equivalent to 98.6 μg·m⁻³) and 121 ppb (less than 3 h, equivalent to 259.3 μg·m⁻³). The slowest velocities (smaller than 0.022 cm·s⁻¹ at both ozone concentrations during the experiments) were found for the microfiber lining coated with a smooth layer of polyurethane. The texture of the lining was similar to that of leather, albeit the ozone deposition progress was observed at a lower ozone concentration of 100 ppb. Notably, the deposition velocities were very close at a higher ozone concentration of 300 ppb for the leather and the lining, at about 0.016 cm·s⁻¹. When purging a higher concentration of ozone into the chamber, the ozone deposition velocities decreased on all the fiber specimens, except for the carpet, on which the mean deposition velocities increased from 0.024 to 0.034 cm·s⁻¹.

3.1.3. Comparison with Data in the Literature

Ozone deposition velocities measured in this study were compared with data from the literature, and the results are shown in Table 6.

Table 6 shows that carpets, a ubiquitous indoor material in a Western context, have been more extensively studied for ozone deposition than wood-based materials, such as OSBs, plywood, and pine. New materials (including carpets and wood-based materials), with exceptions, were more prone to ozone deposition than used or aged ones. The highest and lowest ozone deposition velocities reported in Table 6 (which were at 0.553 ± 0.157 and 0.005 cm·s⁻¹, respectively) were both found on a specimen of pine. Second, the testing temperatures in all the listed references were in a small range, while the RH values varied between 0 to 90%. The results indicate that RH has an impact on ozone deposition. As stated in [25], in a low RH range (e.g., from 0 to 50%), an increase in the RH may not necessarily lead to higher ozone deposition, while with an increase in the RH at a higher range (>50%), the ozone deposition velocity may be increased. It should be noted that the RH in this study was controlled under dry conditions. Therefore, the reported deposition velocities may increase when tested in a more humid environment. Third, ozone deposition tests on indoor materials that are commonly found in non-Western contexts remain rare. Furthermore, the reported ozone deposition velocities were somewhat different from the values of similar materials reported in the literature. More studies could help shed new light on air quality control in non-Western indoor environments.
| Material               | Description                        | Age            | Chamber | ACH (h⁻¹) | Temp. (°C) | RH (%) | Ozone Conc. (ppm) | Deposition Velocity (cm⁻¹) | Source          |
|-----------------------|------------------------------------|----------------|---------|-----------|-----------|--------|------------------|---------------------------|----------------|
| OSB                   | Green-core                         | >1 year        | 50 L    | 1.52      | 23        | 20     | 100, 300         | 0.040, 0.051               | This study       |
|                       | No data                            |                |         |           |           |        |                  |                           | [46]            |
| Plywood               | Lauan                              | >1–2 years     | 50 L    | 1.52      | 23        | 20     | 100, 300         | 0.035, 0.026               | This study       |
|                       | One-side varnished                 | No data        |         |           |           |        |                  |                           | [47]            |
|                       | Paint-free                         | >1–2 years     | 50 L    | 1.52      | 23        | 20     | 100, 300         | 0.009, 0.005               | This study       |
| Pine                  | Raw, untreated, without knots      | New and unused | 17 L    | 12        | 23 ± 2    |        |                  |                           | [48]            |
|                       | Paint-free                         | 1–2 years      | 50 L    | 1.52      | 23        | 20     | 100, 300         | 0.024, 0.034               | This study       |
| Synthetic             |                                    |                |         |           |           |        |                  |                           | [49]            |
| Wool                  |                                    |                |         |           |           |        |                  |                           | [50]            |
| Carpet                |                                    |                |         |           |           |        |                  |                           | [51]            |
|                       | Nylon, GlasBac backing             | Recycled       | 10 L    | 12        | 25        | 50     | 150–200          | 0.068                     | [50]            |
|                       | Nylon, SoftBac platinum backing    |                |         |           |           |        |                  |                           | [50]            |
|                       | Nylon fibers and latex backing     | >1 year        | 35 mL   | -1543     | 21±2      | 50 ± 5 | 50 ± 1           | 0.7 ± 0.4 (sample 1) 0.032 ± 0.004 (sample 2) | [51]            |
|                       | Rubber backing                     | New and unused | 17 L    | 12        | 23±2      |        |                  |                           | [48]            |
|                       | Textile backing                    |                |         |           |           |        |                  |                           | [48]            |
|                       | Bitumen backing                    |                |         |           |           |        |                  |                           | [48]            |
|                       | PVC backing                        |                |         |           |           |        |                  |                           | [48]            |
|                       | 100% BCF (Bulked continuous filament) | Unused       | 52 L    | 3         | 21 ± 1    | 50 ± 2 | 120 ± 2         | 0.147                     | [52]            |

Table 6. Comparison of ozone depositions measured in this study and the data from the literature.
| Material Type                                      | New          | Used         | 50 ± 5       | 100–150      | 0.14–0.38   | [53] |
|---------------------------------------------------|--------------|--------------|--------------|--------------|-------------|-----|
| Eight samples                                     | 10.5 L, 17   | 23 ± 1       | 10 ± 1       | 105 ± 5      | 0.111       |     |
| Two samples                                       | Used         |              |              |              | 0.18        |     |
| Nylon cut pile                                   | Used         |              |              |              | 0.040–0.150 | [54]|
| Wool rugs                                         | 4.25 L, 28   | Room temp.   | 50 ± 5       | 100–150      | 0.125       |     |
| Polypropylene                                     | Used         |              |              |              | 0.090, 0.250|     |
| Nylon loop pile                                  | No data      | 48 L, 2      | 50 ± 2       | 147 ± 10     | 0.053–0.083 | [29]|
| 100% BCF synthetic fiber with PVC backing        | New 10.7 L, 10| 22, 28       | 25, 50, 75   | 60–62        | 0.110–0.140 | [55]|
3.2. Primary and Secondary VOC Emissions

3.2.1. From Wood-Based Materials

The concentrations of the 15 detected VOCs (out of the 24 in the target list), which fell into groups of aromatics, esters, aldehydes, ketones, and aliphatic compounds, were quantified after exposing the wood-based materials to a certain level of ozone, i.e., <10 ppb, 100 ppb, or 300 ppb. Then, the emission factors, \( F_i \), were determined by Equation (4) for all the detected compounds and are summarized in Figure 5a,b. Full data on emission factors of detected VOCs from wood-based panels can be found in Table S1. Background emissions were removed.

![Graph](image)

**Figure 5.** Primary and secondary emissions between ozone and wood-based panels. (a) Aromatics, esters, and aliphatic compounds. (b) Aldehydes and ketones.

Although no clear trends can be directly summarized from Figure 5, a few conclusions can be drawn as follows. In general, more chemical compounds are emitted in larger quantities from wood-based materials when ozone concentrations are increased. However, the emission results for each material varied.

For the green-core OSB, when no detectable concentration of ozone was present, formaldehyde was emitted at about 111.7 \( \mu g \cdot m^{-2} \cdot h^{-1} \), along with benzene and toluene (both < 10 \( \mu g \cdot m^{-2} \cdot h^{-1} \)). By increasing \( y_{ozone,in} \) to 100 or 300 ppb, the \( F_i \) for formaldehyde also increased to 120.4 and 176.2 \( \mu g \cdot m^{-2} \cdot h^{-1} \), respectively. Additional aldehydes were detected at emission factors ranging from 1.2–32.2 \( \mu g \cdot m^{-2} \cdot h^{-1} \). For aromatics, esters, and aliphatic compounds, butyl acetate (<10 \( \mu g \cdot m^{-2} \cdot h^{-1} \)), ethylbenzene (<10 \( \mu g \cdot m^{-2} \cdot h^{-1} \)), and xylene (10.1–26 \( \mu g \cdot m^{-2} \cdot h^{-1} \)) were detected when \( y_{ozone,in} \) was set at 100 ppb. Compared to the compounds emitted at 100 ppb of ozone exposure, additional compounds, such as toluene (93.3 \( \mu g \cdot m^{-2} \cdot h^{-1} \)) and benzene, were detected under 300 ppb of ozone.
Although the other emissions were small, the emissions of butyl acetate were as high as 22.8 μg·m⁻²·h⁻¹.

As expected, a considerable amount of formaldehyde was detected from the lauan plywood during each round of testing, with emissions ranging from 43.6 to 99.2 μg·m⁻²·h⁻¹. For the other target VOCs, when $y_{ozone,in}$ was less than 10 ppb, no other compounds were detected, except for methyl benzaldehyde at 55.7 μg·m⁻²·h⁻¹. When blowing ozone into the chamber at 100 ppb, very small quantities of acetaldehyde (6.5 μg·m⁻²·h⁻¹) and acetone (1.8 μg·m⁻²·h⁻¹) were obtained, while xylene was identified at 23.7–33.8 μg·m⁻²·h⁻¹ and styrene at a smaller rate of 9.6 μg·m⁻²·h⁻¹. When the $y_{ozone,in}$ concentration was as high as 300 ppb, a diverse group of compounds, including hexanal and methyl benzaldehyde, was observed in the sampled air, and the $F_i$ values were determined at 8.9–41.0 μg·m⁻²·h⁻¹.

For the paint-free pine, relatively smaller and stable emissions, ranging from 30.3 to 35.8 μg·m⁻²·h⁻¹ for formaldehyde, were found in all three rounds of experiments. By introducing ozone at 100 ppb or 300 ppb, few additional aldehydes and ketones were observed at high emission rates, i.e., all the detected compounds were close to or less than 10 μg·m⁻²·h⁻¹. For aromatics, esters, and aliphatic compounds, toluene was frequently observed in the samples, albeit at different emission rates (8.5–35.6 μg·m⁻²·h⁻¹). Modest amounts of benzene (6–11.4 μg·m⁻²·h⁻¹) and xylene (3.6–17.9 μg·m⁻²·h⁻¹) were also reported at various background ozone concentrations.

3.2.2. From Synthetic Fibers

The $F_i$ values of the detected target VOCs are summarized in Figure 6a,b for the synthetic fibers. Full data on emission factors of detected VOCs from synthetic fibers can also be found in Table S2.

According to Figure 6a,b the emission behaviors of VOCs from the specimens of synthetic fibers under ozone exposure were visibly varied. On the one hand, the emissions of aldehydes and ketones were all small from weaved leather, microfiber lining, and carpet. However, formaldehyde emissions were significantly high from flannel, with $F_i$ the lowest at 55.4 μg·m⁻²·h⁻¹ when $y_{ozone,in}$ was at 100 ppb, and highest at 150.8 μg·m⁻²·h⁻¹ when no detectable ozone was introduced, indicating that formaldehyde-contained adhesive was probably used in the specimen of the flannel. When the given concentration of ozone was as high as 300 ppb, formaldehyde emission was approximately 125.5 μg·m⁻²·h⁻¹. One hypothesis was that formaldehyde was both the reactant and product of the chemical reaction(s) in the chamber, which depended on the reaction rate(s) and yield(s) from the ozone-initiated reaction(s).

On the other hand, when $y_{ozone,in}$ was less than 10 ppb, no targeted aromatics, esters, and aliphatic compounds were observed at notable amounts from the leather and the lining. However, when $y_{ozone,in}$ was as high as 100 ppb or 300 ppb, a large number of compounds, mostly benzene and toluene, were emitted from all the synthetic materials. Even though the carpet was not as popular in Chinese societies as in Western countries, an alarm was raised when high emissions (35.6–91.5 μg·m⁻²·h⁻¹) of toluene were observed from the carpet specimen when exposed to high concentrations of ozone.
3.2.3. Comparison with the Emission Data from the NRC Database

Numerous studies reporting VOC emissions from various materials can be found in the literature, whereas comprehensive databases, such as the NRC (the National Research Council Canada) database, developed based on the measurements of emissions from a broad range of indoor materials in chambers, are still limited. For example, the VOC emission rates (assumed to be at a steady state) determined in this study for the wood-based panels and the carpet, with or without ozone exposure, were compared, and the ranges of the life-time-averaged emission rates were determined using the NRC database [56], as shown in Table 7. In the NRC database, ozone was not intentionally induced into the emission chamber, and the samples for OSB, plywood, pine, and carpet were six, one, three, and seven, respectively. The eight chemicals listed in Table 7 were common compounds measured during both studies.

Table 7. Comparison of VOC emission rates determined in this study and from the National Research Council Canada (NRC) database.

| Sources | This Study | NRC Database |
|---------|------------|--------------|
|         | OSB | Plywood | Pine | Carpet | OSB | Plywood | Pine | Carpet |
| Chemicals | Emission rates with no intentional ozone exposure, μg m⁻² h⁻¹ | (Emission rates with exposure to ozone at concentrations around 100 ppm, μg m⁻² h⁻¹) |
| Formaldehyde | 111.6 (120.4) | 58.0 (43.6) | 30.3 (34.6) | 0 | 5.6–10.4 | 0 | 0 | 1.5–2.2 |
| Acetaldehyde | 0 | 0 | 0 | 0 | 34.3–66.1 | 0 | 0 | 0.7–3.7 |
Overall, the following observations can be made based on Table 7. First, among the listed chemicals, formaldehyde is very important for wood-based materials in China, though it may be less important in the NRC database. The emission rates of toluene were also high based on the measurements in this study. Although few specimens of OSBs in the NRC database exhibited high emission rates of toluene, in most cases, the emission rates of toluene were low. Second, by conducting a cross-material comparison, the VOC emission rates of similar materials purchased from different markets can be diverse (i.e., with or without ozone exposure), suggesting that empirical emission rates in the literature based on similar materials may not be reliable. Testing is still needed when the emission rate of a specific material is of interest, no matter where the material is from.

3.3. Ventilation Rate Determinants

3.3.1. Ventilation Rates vs. Indoor Ozone Concentrations

The effects of ventilation rates on indoor ozone concentrations were investigated for each of the wood-based panels and synthetic fibers based on a standard room using Equation (8) and are summarized in Figure 7.
Figure 7. The effect of ventilation rates on indoor ozone concentrations by considering ozone deposition on indoor materials. (a) On wood-based panels; outdoor ozone concentration = 100 ppb. (b) On wood-based panels; outdoor ozone concentration = 300 ppb. (c) On synthetic fibers; outdoor ozone concentration = 100 ppb. (d) On synthetic fibers; outdoor ozone concentration = 300 ppb.

As shown in Figure 7a,b, the effect of ventilation rates on indoor ozone concentrations varies from material to material. Based on Equation (8), both the deposition velocity and the VOC emissions play a role in affecting the indoor ozone concentration, although the former was much more crucial than the latter. For wood-based panels, the deposition velocities of green-core OSB and lauan plywood were similar, while those of paint-free pine were significantly lower. As a result, the indoor environment was more tolerant toward (outdoor) ozone pollution when OSB or plywood was used, as an ACH of less than approximately 5.1 or 4.4 h⁻¹, respectively, resulted in an indoor ozone concentration below 74.7 ppb (160 μg·m⁻³) (see [43]) when the outdoor ozone concentration was 100 ppb. As a comparison, when pine is used, the ACH should be less than 1.1 h⁻¹ to maintain a relatively low level of indoor ozone. When outdoor ozone is extremely high at 300 ppb, little to no ventilation
should be provided if the pine covers a considerable area of indoor surfaces. However, to keep the indoor ozone concentration at a “safe” level, the maximum ACH would still be reasonably close to 0.5 h⁻¹, which is the common minimum ventilation rate requirement for residential buildings worldwide [57–59] when either OSB or plywood is the dominant indoor building material.

A similar trend for ventilation rates on indoor ozone concentrations was observed for the scenarios using synthetic fibers, as shown in Figure 7c,d. When the ozone was at 100 ppb, all the cases resulted in an ozone concentration below 74.7 ppb (160 μg·m⁻³) when the ventilation rates were less than approximately 2.7 h⁻¹. Flannel and weaved leather are more prone to ozone deposition than micro lining and carpet. Accordingly, the indoor environment has a superior capacity for depleting outdoor ozone via ventilation when flannel or leather is present in large quantities. Even when the outdoor ozone concentration is as high as 300 ppb, by ventilating at around 0.5 h⁻¹, the indoor environment of the standard room should still be able to endure ozone pollution without further eliminating measures.

3.3.2. Ventilation Rates vs. VOC Emissions

The ventilation requirements to dilute indoor VOC emissions from materials exposed to ozone at various concentration levels were predicted using Equation (9) and are shown in Figure 8.

Two major findings can be drawn from Figure 8. First, the ventilation requirements were heavily dependent on the air volume required to dilute the formaldehyde concentrations. In other words, formaldehyde was the ventilation rate determinant for all the cases using wood-based panels, as well as the flannel, since all the above materials emitted and/or produced a considerable amount of formaldehyde at almost all three ozone concentration levels. Only when formaldehyde was absent or low were the ventilation requirements determined by other pollutants, such as xylene. Notably, many pollutants that may end up as determinants were not measured during the VOC analysis. Previous studies, such as the study in [60], showed that formaldehyde can be a serious problem in indoor environments in China, especially in newly-built/decorated environments. This paper provides further evidence that formaldehyde can be frequently detected at concentrations that require high ventilation rates on materials sold in Chinese markets.

Second, by combining Figures 7 and 8, the results indicate that when ozone concentration in the outdoor air is not severely high, it is possible to use ventilation to dilute indoor VOC emissions while maintaining an acceptable indoor ozone concentration. However, when the outdoor ozone concentration is too high, directly using ventilation as a means to dilute indoor VOC emissions can be counter-productive because indoor ozone pollution would still be considered harmful to the occupant’s health. Lower ventilation requirements for VOC dilution (such as those of leather and lining) seem to better reduce ventilation to cope with outdoor ozone pollution.

![Ventilation Requirement vs. VOC Concentration](image-url)
Figure 8. The ventilation requirements to dilute indoor VOC emissions with or without exposure to ozone. (a) Using wood-based panels as the indoor material. (b) Using synthetic fibers as the indoor material.

4. Conclusions

At present, studies on ozone deposition on indoor materials from a non-Western perspective are still limited. In this work, experiments were carried out to study ozone deposition on seven common indoor materials from Chinese markets. We obtained the ozone deposition velocity and VOC emissions from each material when exposed to various ozone concentrations. A simplified model was used to discuss ventilation requirements based on a standard room by considering the indoor concentrations of both ozone and VOCs. The major conclusions are outlined below.

First, ozone deposition velocities varied between materials, ranging from 0.005 to 0.062 cm·s⁻¹. In general, engineering wood (i.e., green-core OSB and lauan plywood) and some synthetic fibers, such as flannel, are prone to ozone deposition.

Second, 15 out of the 24 targeted VOCs were detected after exposing the materials to ozone, and the corresponding area-averaged emission rates were obtained. Overall, the higher the ozone concentration, the more plentiful the VOCs. Among others, formaldehyde, toluene, and xylene were frequently observed, often in high concentrations. The materials that emit and/or produce the most formaldehyde were also the ones most prone to ozone deposition, suggesting that although some of the gas-phase formaldehyde is directly linked to the ingredients of the materials, such as the adhesives used during manufacturing the engineering wood, some could result from ozone-initiated reactions.

Third, both the deposition velocity and the VOC emissions play a role in affecting the indoor ozone concentration, although the former is more crucial than the latter. The simulations indicate that when the ozone concentration in the outdoor air is not severely high, it is possible to use ventilation to maintain acceptable indoor air quality, considering the concentrations of both ozone and major VOCs.

The major limitations of this work are summarized as follows. First, although the selected materials were freely-stored in an office/lab environment for enough time before the experimental procedure, only one specimen of each material was subjected to testing. A follow-up study should be done to examine at the changes in ozone deposition and the corresponding VOC emissions, as well as the changes in the physical and chemical properties of free-running materials responding to ozone deposition over the long term. Second, the proposed model to investigate ventilation requirements is based on many assumptions that do not represent applications in practice. For example, the product yield on material surfaces was not measured, and the emissions of VOCs were thus grouped together. In addition, the concept of a standard room is not rigorous, so interpretations should be tentative.
Supplementary Materials: The following are available online at www.mdpi.com/2076-3417/10/12/4146/s1. Table S1. Emission factors of detected VOCs from wood-based panels. Table S2. Emission factors of detected VOCs from synthetic fibers.

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