Air Pollutant Emissions and Possible Health Effects Associated with Electronic Air Cleaners

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Lead Principal Investigator:
Christopher Cappa, PhD
Department Chair and Professor
Department of Civil and Environmental Engineering
University of California, Davis

Report Authors:
Theresa Pistochini, MS, PE, UC Davis Western Cooling Efficiency Center
Christopher Cappa, PhD, UC Davis Dept. of Civil and Environmental Engineering
Deborah Bennett, PhD, UC Davis Dept. of Public Health
Francis Offerman, PE, CIH, Indoor Environmental Engineering

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Executive Summary

Consideration of human exposure to indoor air contaminants is important as people spend an average of 87% of their time indoors [1]. The COVID-19 pandemic and worsening air quality from wildfires has increased interest in indoor air quality and indoor air cleaners. Mechanical air cleaners (particle filters and gas-phase adsorption-based filters) remove contaminants from the air and collect them on filters that must be periodically replaced. Electronic air cleaners use other electricity-driven processes to remove particles, break down volatile organic compounds, and/or inactivate biological contaminants. However, use of electronic air cleaners may alter indoor air chemistry. Electronic air cleaners may directly emit reactive compounds or promote the formation of chemical byproducts through interaction with the environment, or both. There is concern that these emissions and associated byproducts may have adverse health effects.

This whitepaper describes the working principles for the following types of electronic air cleaners: ion generators, electrostatic precipitators, photocatalytic oxidation devices, ultraviolet germicidal irradiation devices, hypochlorous acid generators, and nano-confined catalytic oxidation devices. Electronic air cleaners may combine more than one type of electronic air cleaning mechanism and may also combine electronic air cleaning with mechanical filtration methods.

Comprehensive assessment and quantification of risks and byproduct formation from the use of electronic air cleaners in indoor environments is challenging given that many of the byproducts formed often depend on the initial presence of reactive compounds in the air, especially volatile organic compounds, and the abundance of such compounds will vary among buildings and over time in a given space. Moreover, the number of distinct byproducts formed from the reaction of even a single compound can number in the hundreds, which complicates the determination of potential health impacts of these byproducts. Our understanding of the risks of electronic air cleaner use is limited to the available literature on byproduct compounds produced, both in terms of identification and understanding of health impacts relative to the initial compounds from which they are formed.

Despite lack of comprehensive understanding of electronic air cleaner emissions and resulting indoor chemistry, compounds of clear concern are ozone, formaldehyde, and ultrafine particles, which were widely observed across studies from use of electronic air cleaners. Other compounds identified in electronic air cleaner studies that may have negative health effects from inhalation were acetaldehyde, acetone, acrylonitrile, propanediol, acetic acid, ethanol, and isopropanol. Skin and eye exposure to ultraviolet radiation used in certain devices was also identified health concern.

While California already requires electronic air cleaners have ozone emissions less than 50 ppb, we recommend California further reduce ozone emissions from electronic air cleaners by requiring compliance with UL2998, a more stringent ozone emission standard of 5 ppb. This would reduce the allowable indoor ozone emissions by an order of magnitude which would provide a direct health benefit and subsequently reduce secondary formaldehyde and ultrafine particle formation that is driven by ozone chemistry. Furthermore, ASTM-WK81750 (under development) aims to provide a standard method of test for evaluating emissions of byproducts of greatest health concern (ozone, formaldehyde, and ultrafine particles) created by portable air cleaners when indoor-relevant levels of challenge compounds are present. This standard, if finalized and widely adopted, promises to provide an important source of data source for consumers and could serve as the basis for regulation of formaldehyde and ultrafine particle emissions from electronic air cleaners.

Although this whitepaper solely discusses the risks of byproduct formation, a well-informed decision to implement (or not implement) electronic air cleaning technology requires an assessment of the potential risks versus the potential benefits. In addition to research to quantify the risks of electronic air cleaners, additional research is needed to quantify performance in terms of contaminant removal under realistic and reproducible conditions so that the risks versus benefits can be considered. Such contaminants might include airborne pathogens, volatile organic compounds, or particulate matter.

In addition to these nearer-term actions designed to protect consumers from compounds of clear concern, we recommend CARB continue to further scientific understanding of electronic air cleaners, and indoor air chemistry and human exposure to indoor air compounds in general, by working to address gaps in research identified in this whitepaper. Finally, continued effort to educate the public and institutional procurement staff on the risks associated with electronic air cleaners is helpful to build awareness and inform purchasing decisions.

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1. Introduction

1.1. Purpose

Electronic air cleaners, such as ion generators, electrostatic precipitators, photocatalytic oxidation devices, and ultraviolet lamps may directly emit reactive compounds or promote the formation of chemical byproducts through interaction with the environment, or both. There is concern that these emissions and associated byproducts may have adverse health effects. The purpose of this whitepaper is to summarize current knowledge of:

- reactive compounds and byproducts emitted or formed by operation of electronic air cleaners;
- potential human exposure pathways and potential adverse health effects of these compounds and byproducts;
- test standards and regulations regarding emissions from electronic air cleaners; and
- gaps in understanding of the risks of electronic air cleaners that could be addressed through future research.

This whitepaper does not address the performance or efficacy of electronic air cleaners for contaminant removal or inactivation of biological contaminants. A complete assessment of an electronic air cleaner would need to consider risk of emissions and byproducts, contaminant removal performance, energy efficiency, and noise.

1.2. Background

Consideration of human exposure to indoor air contaminants is important as people spend an average of 87% of their time indoors (with 80% of that time spent in their private residence) [1]. The COVID-19 pandemic and worsening air quality from wildfires has increased interest in indoor air quality and indoor air cleaners. Mechanical air cleaners (particle filters and gas-phase adsorption-based filters) remove contaminants from the air and collect them on filters that must be periodically replaced. Electronic air cleaners use other electricity-driven processes to remove particles, break down volatile organic compounds, and/or inactivate biological contaminants. However, use of electronic air cleaners may alter indoor air chemistry.

The highly varied nature of the indoor environment presents a challenge for characterizing the potential impacts of electronic air cleaners. The concentration of an air contaminant in an indoor space is impacted by:

- the emission or production rate of a contaminant;
- the air exchange rate of outdoor air with indoor air through mechanical ventilation, natural ventilation, and infiltration;
- the outdoor air concentration of the contaminant;
- the indoor concentration of other chemicals that may react with or form the contaminant;
- types and area of surfaces that the contaminant may react with; and
- temperature and humidity conditions.

Key pollutants of concern in indoor environments include particulate matter, carbon monoxide, ozone, nitrogen oxides, and the class of molecules known as volatile organic compounds (VOCs). VOCs are chemicals that have a low boiling point and volatilize to the air in typical indoor conditions. Examples of VOCs commonly found in indoor environments include formaldehyde, benzene and toluene, siloxanes, and terpenes (which are often components of fragrances). VOCs can react with atmospheric oxidants such as ozone (O_3), hydroxyl radicals ($^\bullet OH$), and nitrate radicals ($^\bullet NO_3$). The reactivity of a given VOC depends on what specific chemical it is. Upon reaction, VOCs tend to form reaction products that have an oxygen-containing functional group, such as: carbonyls (carbon double-bonded to oxygen), aldehydes (carbonyl bonded to at least one hydrogen atom), or alcohol (oxygen bonded to hydrogen). Certain reactions will cause the VOCs to decompose to become two smaller molecules; a common compound formed is formaldehyde (a known respiratory irritant and carcinogen).

Unsaturated VOCs contain double or triple bonds which makes them more reactive. Within the category of unsaturated VOCs, compounds with non-aromatic carbon-carbon double bonds (such as terpenes) are more reactive, compared to aromatic compounds that contain a ring of six carbon atoms with double bonds (such as benzene). These unsaturated molecules are highly reactive and therefore have relatively short chemical lifetimes. However, even less reactive aromatic compounds or alkanes, also found in indoor environments, can still be transformed through chemical reactions. While the formation of reaction products and transformation of VOCs can occur indoors generally, these processes may be enhanced by electronic air cleaners.

A major concern associated with the use of electronic air cleaners is their potential emission of reactive compounds (e.g., ozone, hydrogen peroxide, hypochlorous acid, hydroxyl radicals). These chemicals may be directly harmful to human health and may also chemically transform and/or decompose VOCs existing in the indoor environment. The reaction products of VOCs, referred to as byproducts, could potentially pose greater health risks than the original compounds. An additional concern is possible formation of ultrafine particles that adversely affect human health when inhaled. Finally, potential human exposure to ultraviolet (UV) radiation from lamps used in some electronic air cleaners must be considered.

1.3. Challenges and Limitations of Characterizing Electronic Air Cleaner Emissions

When an electronic air cleaner is used and emits reactive compounds, the resulting byproducts will vary since every indoor environment has a unique chemical composition. Also, emissions from an electronic air cleaner may change over time. Therefore, laboratory testing of new electronic air cleaners may not reflect the risks when such devices are installed and used in real-world environments. Additionally, real-world tests may produce a range of results in different environments, since the baseline composition of indoor air varies. This makes it difficult to know how representative a given study is of general conditions.

Studies of byproduct production and emissions are limited by the analytical tools used for analysis, which can be very specific [3]. For example, most studies cited in this whitepaper targeted the detection of specific byproduct compounds (e.g., ozone or formaldehyde, or more generally small carbonyl compounds) using analytical methods that are not well suited to the characterization and quantification of other possible byproducts. As such, knowledge of chemical byproducts thus far is predominately limited to those compounds that researchers have chosen to look for.

1.4. Whitepaper Approach

With an understanding of these challenges in characterizing the byproducts from electronic air cleaners, this paper aims to provide an overview of electronic air cleaner types and their basic operating principles, along with an overview of the most likely indoor air chemistry that may be initiated from their use. Then, a summary of findings from literature is described for different byproducts of concern generated by electronic air cleaners under tests in laboratory and real-world environments. The literature describing health effects of these byproducts is also discussed. Finally, our knowledge of the development of test standards for electronic air cleaners is summarized.

2. Air Cleaner Types and Basic Operating Principles

The working principles of each known type of electronic air cleaning technology are described, although manufacturers may use different names to describe these technologies (or combination of technologies). Electronic air cleaners may combine more than one type of electronic air cleaning mechanism and may also combine electronic air cleaning with mechanical filtration methods, such as high efficiency particle filtration (HEPA), or the use of carbon filters to remove VOCs.

2.1. Ion Generator

An ion generator may be described by several names, including unipolar or bipolar needlepoint ionization, corona discharge, or plasma generator. All these devices apply the same operating principle to generate charged gas molecules, or ions, which may then impart charge to particles in the air to promote their agglomeration for easier removal by deposition onto room surfaces and mechanical filtration. A high voltage is applied to an electrode which has a sharp metal tip (Figure 1). When the electric field surrounding the electrode exceeds the dielectric strength of air, the air becomes electrically conductive around the tip. Free electrons are accelerated and collide with neutral molecules, creating ions and freeing more electrons, causing a chain reaction called an electron avalanche [4]. A positive electrode generates positive ions, and a negative electrode generates negative ions. The electrons move toward a positive electrode or away from a negative electrode, creating a current on the scale of microamps. This current is called a corona discharge, and air containing the increased concentration of ions is called plasma. Generation of only negative or positive ions is called unipolar ionization, and generation of both positive and negative ions with two electrodes is called bipolar ionization [5].
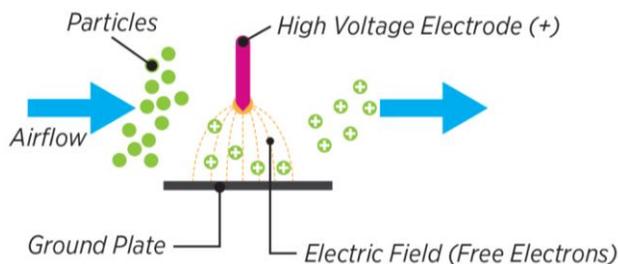
Figure 1: Ion generator with a single positive electrode schematic. The high voltage electrode creates an electric field that exceeds the dielectric strength of air, creating a corona discharge current.

An ion generator is a source of energy that can initiate chemical reactions among the existing constituents in indoor air. For example, if the energy is high enough, the bond in oxygen (O_2) can be broken, freeing oxygen atoms to combine with other existing O_2 to form ozone (O_3), which is a lung irritant and highly reactive, unstable gas molecule [6]. Design of ion generators can vary using different

electrode materials (e.g. aluminum, copper, silver, tungsten, carbon), electrode geometry, and working current and voltage [2]. Carbon fiber brush electrodes, which consist of several hundred electrodes in a brush configuration, are designed to produce ions with reduced ozone production [5, 6].

2.2. Electrostatic Precipitator (ESP)

An electrostatic precipitator (ESP) is designed to charge and collect particles via a working principle similar to that of an ion generator. The high voltage electrodes are generally in the form of wires that create the corona discharge. Plates of the opposite charge from the electrode are used to collect the charged particles downstream of the electrodes (Figure 2). A single-stage electrostatic precipitator has electrodes of one charge sandwiched between two oppositely charged collection plates. A two-stage electrostatic precipitator has the electrodes upstream of stacked plates of alternating charge; the particles collect on the plates of the opposite charge from the electrodes [7]. Like an ion generator, an electrostatic precipitator is a source of activation energy that may initiate chemical reactions. For example, ozone production rate is affected by electrode and collection plate design [8]. Although electrostatic precipitators can be designed to reduce ozone production potential, particles collecting on the discharge wires and collection plates can alter the device performance and increase ozone production [9].
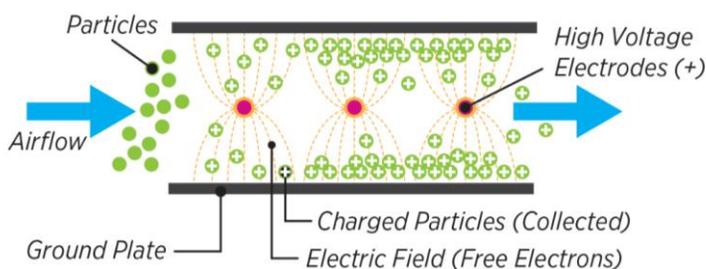
Figure 2: Single-stage electrostatic precipitator schematic. The high voltage electrodes create an electric field that exceeds the dielectric strength of air, creating a corona discharge current. Charged particles collect on the oppositely charged collection plates.

2.3. Photocatalytic Oxidation (PCO)

A photocatalytic oxidation (PCO) air cleaning device uses an ultraviolet radiation source to irradiate a photocatalyst, which is a metal oxide semiconductor, also referred to as the reactor (e.g. TiO_2, ZnO, CdS, Fe(III)-doped TiO_2) (Figure 3) [10]. PCO is designed to break down VOCs (i.e. hydrocarbons) to final compounds of carbon dioxide and water, a process referred to as mineralization [11].

UV radiation of sufficient energy on the surface of the reactor creates an electron-hole pair, promoting an electron from its valance band to the conduction band. The electron reacts with O_2 adsorbed to the surface to produce O_2^- while the electron hole reacts with surface-adsorbed H_2O, which, together, ultimately produce surface-bound $^\bullet OH$ through a series of intermediate reactions. The intermediate reactions include formation of hydrogen peroxide (H_2O_2), leading to these air cleaners to also be described as "dry hydrogen peroxide technology". The surface-bound $^\bullet OH$ react with and break down VOCs; if the reaction goes to completion, the only products are CO_2 and H_2O [11].

Achieving a complete reaction in PCO is challenging. When reactions do not go to completion, intermediate VOC oxidation products (i.e., byproducts) are produced. In an optimally designed PCO system, the VOC destruction rate is limited by mass transfer of the VOC to the surface of the reactor. In

practice, such optimal designs are difficult to achieve. The photocatalyst degrades over time and requires regeneration or replacement to maintain performance [11]. In addition to byproducts generated by incomplete reactions, the UV radiation source lamp can cause photolysis reactions, which are the decomposition of molecules by the absorption of light. Of primary concern is the photolysis reaction that produces ozone, discussed further in the next section.
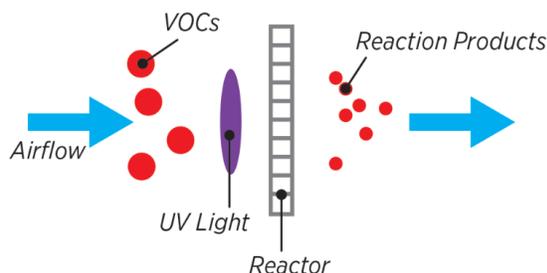

Figure 3: Photocatalytic oxidation schematic with UV light irradiating a photocatalyst reactor designed to decompose VOCs.

2.4. Ultraviolet Germicidal Irradiation (UVGI)

UVGI air cleaning devices expose room air to radiation which has the potential to inactivate biological contaminants by damaging their nucleic acids and proteins. Use of UVC radiation, which is a subset of UV with wavelengths 100 – 280 nm, is most common. While the topic of this paper is air cleaning devices, UVGI devices may also be designed and marketed for surface disinfection (or combination surface and air disinfection). A concern of UVGI is the potential to damage human cells (i.e. skin and eyes) from direct exposure. Research is ongoing to evaluate the safety of human exposure, which varies by wavelength and dose [12]. Concerns of direct human exposure can be mitigated by designing air cleaning devices that irradiate air while protecting occupants from direct exposure (e.g. upper room UVGI, in-duct devices, and enclosed devices that draw air through a fan). The challenge is designing a system with no direct human exposure that delivers a sufficient UV dose (a function of power density and contact time) to the air to inactivate biological contaminants. UV lamps vary in their spectral distribution (Figure 4) depending on the lamp technology:

- Low pressure mercury vapor glass lamps filled with a combination of an inert gas and mercury vapor produce peaks of UV radiation at 254 and 185 nm wavelengths. These lamps may use a doped glass to block transmission of the 185 nm wavelength to prevent ozone production [13, 14].
- Light emitting diodes (LED) tailored to produce peak radiation wavelengths mostly commonly in the range 255 to 280 nm [14].
- Pulse xenon arc lamps (PXL) produce broad spectrum UVC radiation (as well as light in the visible spectrum), with a spike near 230 nm [14].
- Excimer glass lamps produce one peak UV radiation wavelength that is dependent on the inert gases applied. Excimer lamps of greatest interest for UVGI applications contain krypton-bromine (Kr-Br) with peak wavelength of 207 nm or krypton-chlorine (Kr-Cl) with peak wavelength of 222 nm [15]. Bandpass filters may be applied to filter out undesired wavelengths.

In addition to the risks of direct human exposure, generation of ozone and other byproducts from photolysis reactions are a concern. Ozone generation from UV radiation is correlated with wavelength,

where wavelengths below 240 nm are more readily absorbed by oxygen and therefore more efficient at generating ozone (Figure 4). Maximum efficiency for ozone generation occurs at a wavelength of approximately 160 nm [16]. Therefore, lamp design and filtering of wavelengths outside the disinfection target is critical for limiting ozone production. VOCs containing carbonyl groups may also photolyze and initiate other chemical reactions.
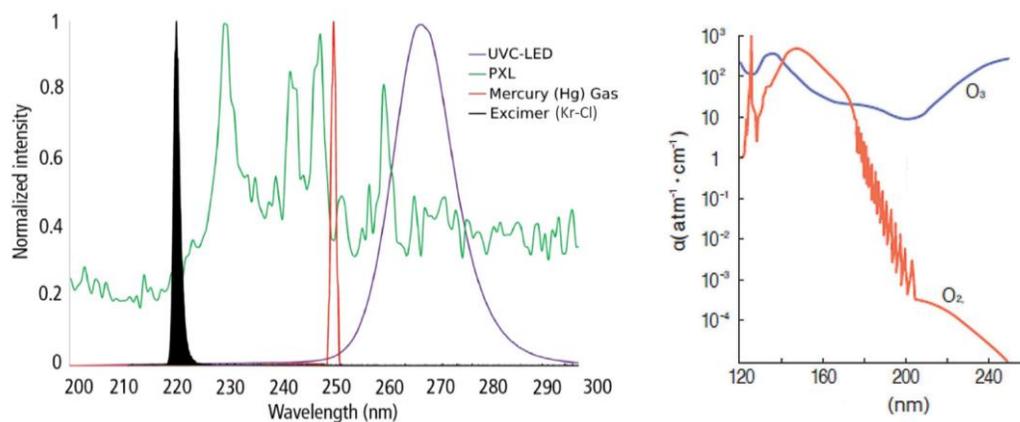
Figure 4: Spectral distribution of several types of UV lamps (left) [14]. Excimer lamp shown is representative of Kr-Cl gas. Mercury vapor lamps also have a spike at 185 nm unless doped glass is used to block transmission. Absorption of UV radiation for oxygen (O$_2$) and ozone (O$_3$) (right) [16]. Maximum ozone production occurs around 160 nm where oxygen absorption is high and ozone absorption is low.

2.5. Other Electronic Air Cleaning Devices

Here, two other types of electronic air cleaning devices are briefly described: hypochlorous acid (HOCl) and nano-confined catalytic oxidation (NCCO). At least one company is marketing a device that applies electrolysis to a tank filled with water and table salt and claims to generate HOCl and distribute it to the room air. However, no peer-reviewed literature was found that describes testing of an air cleaner with this design, so very little is known about this air cleaning mechanism or possible HOCl concentrations. For completeness, we provide a limited device description here and consider possible health effects from HOCl exposure in Section 3.5.

NCCO devices employ an ozone generator upstream of a catalyst comprised of a microporous, crystalline aluminosilicate framework, also known as zeolite. The operating principle is that ozone reacts with VOCs adsorbed in the catalyst to decompose both the VOCs and ozone, where the final decomposition products are carbon dioxide, water, and oxygen [17]. Like PCO, in practice complete mineralization (i.e. decomposition to CO$_2$ and H$_2$O) may be difficult to achieve, and partially decomposed VOCs are possible byproducts. Since NCCO devices generate ozone by design, ozone emissions from incomplete reactions pose another potential risk. Performance and byproduct generation may be impacted by ozone generation rate, air flow rate, temperature, humidity, initial VOC concentrations, and catalyst composition and length [17, 18].

2.6. Mechanical Air Cleaning Methods

Mechanical air cleaners (i.e. filters) are often integrated into electronic air cleaners and so are described here for completeness. HEPA filters can remove at least 99.97% of all airborne particles in a single pass [19]. Other particle removal filters that have a minimum efficiency reporting value (MERV)

will remove some fraction of particles (not as effectively as HEPA), where MERV ratings of 13 or greater are required to remove a substantial fraction of all particle sizes [19]. Both HEPA and MERV rated filters need to be periodically replaced as particles collect on the filter and restrict airflow. They are ineffective at removing gas-phase air contaminants. Gas-phase adsorption-based filters, which generally consist of activated carbon or alumina with sodium permanganate, can be effective at removing some gas-phase compounds without byproduct formation, although the media has to be replaced periodically [20]. It is important to consider that inclusion of HEPA, MERV, and/or activated carbon filters in electronic air cleaning devices may contribute to their performance and remove byproducts, making it difficult to isolate the impact of the electronic air cleaning technology alone. The emissions of the electronic air cleaner may change over time if the particle or gas-phase filtration components are not replaced regularly.

3. Potential Emissions, Toxicological, and Epidemiological Data

People are primarily exposed to emissions and byproducts of electronic air cleaners through inhalation of indoor air and through dermatological UV exposure. Below we discuss what is known from the literature about potential health effects from exposure to emissions and byproducts from electronic air cleaners, focusing on ozone and hydroxyl radicals (Section 3.1), formaldehyde (Section 3.2), other volatile organic compounds (Section 3.3), ultrafine particles (Section 3.4), hypochlorous acid (Section 3.5), ions (Section 3.6), and direct UV exposure (Section 3.7).

3.1. Ozone and Reactive Oxygen Species

3.1.1. Ozone Introduction

Ozone is an inorganic molecule that is readily formed outdoors by photolysis of O_2 from UV radiation. Absent direct production from indoor sources, the main source of ozone indoors is ozone in outdoor air that enters the building through mechanical outdoor ventilation, open windows, and infiltration through leaks in the building envelope. Outdoor ozone is regulated by the National Ambient Air Quality Standards (NAAQS) with an attainment metric for 8-hour ozone concentration of 70 ppb [21]. Based on a review of epidemiology studies exploring the relationship between outdoor ozone concentrations and non-accidental mortality and respiratory mortality, the World Health Organization air quality guidelines recommend a maximum 8-hour ozone concentration of 51 ppb for one-day exposure and 31 ppb for average peak season exposure [22].

3.1.2. Ozone Direct Health Effects

Ozone is associated with a wide range of health impacts, including inflammation of the respiratory tract, coughing, throat irritation, increased frequency of asthma attacks, and can aggravate asthma, chronic bronchitis, and emphysema [23]. A recent literature review determined increased exposure of 5 ppb was associated with a relative risk (RR) of 1.008 for asthma-related emergency room visits and hospital admissions, demonstrating that even small changes in ozone concentration can have health impacts [24]. Indoor ozone concentrations are generally in the range of 20% to 70% of the outdoor concentration when there are no sources of indoor ozone [25].

3.1.3. Ozone Reaction Products and Health Effects of Ozone Reaction Byproducts

In addition to the direct health risk posed by exposure to ozone, ozone is a reactive molecule that is a primary driver of indoor air chemistry. This results in an additional health risk from exposure to byproducts produced from ozone reactions; epidemiology studies that assess risk between outdoor ozone concentration and health effects are evaluating the combination of exposure to ozone and ozone reaction byproducts. The reactions resulting from ozone and VOCs are complex, involving numerous intermediate radicals, and are dependent on the particular VOCs present in indoor air [26]. Ozone readily reacts with alkenes at the carbon-carbon double bond, ultimately producing very reactive compounds. Nonetheless, the stable products of these reactions can be generalized as having added oxygen-containing functional groups, primarily aldehydes and ketones, but also peroxides [25]. Some reactions result in the breakdown of the original compounds to produce two smaller compounds, while others result in oxygenated compounds of higher molecular weight. When ozone reacts with alkenes, $^{\bullet}$OH are produced which are highly reactive towards VOCs and further drive indoor air chemistry.

The health effects of some ozone-reaction products, such as formaldehyde and acetaldehyde, have been well studied, and are discussed in Sections 3.2 and 3.3. There is additional literature related to adverse health effects of simultaneous exposure to all ozone-reaction products. The US EPA Building Assessment and Survey Evaluation (BASE) found increased building-related symptoms with higher outdoor ozone levels [27]. These symptoms included composite measures of individual symptoms including cough, dry eyes, dry skin, and headache as well as aggregate categories of lower and upper respiratory symptoms and neurological symptoms (including fatigue or difficulty concentrating). There is a hypothesis that reactions with terpenes (from scented products like candles air fresheners, and cleaning products) are primarily driving these findings. There have been two recent reviews of the health impacts of terpene oxidation products [28, 29].

As reviewed by Rohr, there is considerable evidence that when terpene and ozone react, the reaction products cause sensory irritation and airflow limitation at high concentrations in a mouse bioassay [28]. The effects at lower, environmentally relevant concentrations in humans are not clear. Rohr reviewed the controlled human exposure studies conducted prior to 2011, which had mixed results. In controlled human exposure studies, participants are exposed to low levels of the compound being studied and the participants report on symptoms or have blood samples taken to study changes that result from the exposure. Two studies highlighted in the review found no changes in symptoms following exposure to either a mix of VOCs or to a mix of VOCs mixed with ozone. However, other studies did find impacts from exposure to terpene-ozone reaction products. Specifically, in a study of proteasome activity, Kipen et al. found significantly decreased white blood cell proteasome activity for participants exposed to the reaction products of limonene (a terpene) and ozone, specifically secondary organic aerosol with concentration of 194 $\mu g/m^3$ (higher than generally observed in indoor environments) and associated gases [30]. Two additional studies highlighted in the review found increased blink frequency, a marker for eye irritation, with one study reporting the lowest observed levels at 92 ppb limonene and 101 ppb ozone, which are higher concentrations than generally observed in indoor environments [28]. Rohr discussed one hypothesis that health effects from terpene and ozone reactions are a result from the formation of reactive oxygen species, such as hydroperoxides, which may interact with biological tissues and cause changes in pulmonary cytology and cytokine production in response to co-exposure of ultrafine particles and hydrogen peroxide, suggesting that the particles may

carry the reactive species into the lungs. However, studies are lacking at environmentally relevant concentrations.

As reviewed by Wolkoff, in-vitro studies with human lung cells "have not convincingly shown ozone-initiated terpene solid organic aerosol to cause inflammation at indoor relevant levels" [29]. Results from more recent controlled human experiments were also reviewed by Wolkoff, finding either very weak sensory irritation in the eyes or no observed effects in studies that used realistic levels of exposure for two to four hours in climate chambers. There may be greater impacts in low humidity scenarios because water vapor in the air is also involved in indoor chemistry and causes a slight shift in the distribution of reaction products in the air [31]. The increased impacts at low relative humidity may be relevant in some portions of California.

Mouse exposure studies were conducted by Wolkoff to determine reference levels for additional compounds formed from ozone-terpene reactions, specifically 4-AMCH, IPOH, 6-methyl-5-heptene-2-one (6-MHO), dihydrocarvone, and 4-OPA [29]. Relatively low exposures caused airflow limitation (bronchoconstriction) for 4-OPA (123 $\mu g/m^3$, 30 ppb) and sensory irritation for IPOH (1100 $\mu g/m^3$, 160 ppb). However, the author noted that these compounds are generally produced at lower concentrations than formaldehyde yet require higher levels than formaldehyde to cause irritation, and are not likely to be the primary cause of irritation, bur rather could be a factor in increasing irritation following ozone-terpene reactions.

3.1.4. Ozone Emissions from Electronic Air Cleaners

Ozone can be generated by electronic air cleaners via either a photochemical mechanism (UV light of wavelength ~240 nm or less, used in PCO and UVGI air cleaners) or corona discharge mechanism (used in ion generators and electrostatic precipitators) [2]. Existing California Air Resources Board (CARB) regulations require that portable and in-duct electronic air cleaners be tested for ozone safety to Underwriters Laboratory (UL) Standard 867 or Canadian Standard Association (CSA) C22.2 no. 187-20 [32]. UL 867, which is most commonly applied in the US, requires testing the device in a chamber of volume 26.9 to 31.1 m^3 with a total loss rate from air exchange and surface deposition of 1.33 hr^{-1}. After a 48-hour run in period, the location of the peak ozone emission from the device is determined from measurements made across the air stream discharge at 2 inches from the air discharge grill. The ozone concentration is then measured over an 8 to 24 hour period at the location of the peak ozone emission and the device must not produce an ozone concentration that exceeds 50 ppb. A box model was used to calculate that a device ozone emission rate must be under 4.0 mg/hr to achieve a final concentration of 50 ppb concentration or less when the largest chamber allowable is used in the scenario where the chamber is well-mixed (see Supplementary Information for box model calculation). Since the UL 867 standard measures ozone at the location with maximum concentration, the measured concentration may be higher than the average test chamber concentration required to calculate the emission rate. Thus, 4 mg/hr is a theoretical upper bound on emission rate for a passing device.

In a 2019 literature review, Guo et al. reported ozone emissions summarized from studies on 21 portable air cleaners and four in-duct air cleaners [2]. The results from all studies, which are often reported in terms of changes in ozone concentration, were used to calculate an ozone emission rate for the tested devices using a series of models enabling direct comparisons between studies. The 21 portable air cleaners included ion generators, ESPs, PCO devices, and combined technologies that included UV. Ozone emissions ranged from 0.056 to 30.5 mg/hr (M = 4.6 mg/hr). Ozone emissions from

the four in-duct air cleaners were much higher, ranging from 8.4 to 347 mg/hr (M = 62.8 mg/hr). These in-duct air cleaners were UV/ESP, UV/PCO/ESP, and ESP devices, and the highest emissions were from a UV-containing air cleaner. In a recent pre-print, Peng et al. evaluated eight different 222 nm UV lamps from five manufacturers and reported ozone production rates of 0.45 to 0.76 mg/hr [33]. While the literature review generally covered testing of air cleaners in new condition, there is evidence that electrodes contaminated with particles in ion generators and electrostatic precipitators can modify the corona discharge and increase ozone generation [9].

3.1.5. Other Reactive Oxygen Species

Reactive oxygen species describe an array of highly reactive compounds that are derived from oxygen. In addition to ozone, •OH and hydrogen peroxide are other reactive oxygen species that may be byproducts of electronic air cleaners. All reactive oxygen species are expected to accelerate indoor chemistry reactions and increase the production of both reactive intermediates and final products. While measurements of ozone concentration are commonly reported in electronic air cleaner studies, •OH concentrations are generally not reported due to the short lifetimes which make measurement and quantification extremely difficult [34]. Indoor air chemistry models indicate that

Figure 5: Ozone emission rates reported by Guo et al. in literature review [2].

•OH concentrations indoors (on the order of 10^6 molecules per cm^3) are typically similar to outdoor concentrations due to their high reactivity [35]. Because the half-life of •OH is very short and they coexist with other reaction products (some of which have known health effects), there is no practical way to study the health impacts of •OH in isolation. Hydrogen peroxide concentrations, while more feasible to measure, are also rarely reported in electronic air cleaner studies. Zeng et al. estimated hydrogen peroxide concentrations of 10 and 36 ppb in two tests of a bipolar ion generator, which equated to emissions rates of 5 mg/hr and 17 mg/hr, respectively [36]. These concentrations are well below indoor concentrations of approximately 600 ppb observed when cleaning with hydrogen-peroxide based cleaning solutions and the 1 ppm 8-hr exposure limit published by the National Institute for Occupational Safety and Health (NIOSH) [37] [38].

3.2. Formaldehyde

3.2.1. Formaldehyde Introduction

Formaldehyde (HCHO) is a VOC that is commonly found in indoor environments at concentrations elevated above those found in outdoor air [39]. A primary reason for this is the somewhat ubiquitous use of particle board in construction and furniture. Historically, particle board often used a urea formaldehyde resin as an adhesive, which led to the emission of formaldehyde over time. Other indoor materials (e.g., insulation, laminates, surface coatings) and some products used indoors (e.g., cleaning products, disinfectants, cosmetics) can also release formaldehyde [39]. Various regulations, driven by concerns over the health risks of exposure to formaldehyde, have led to decreases in formaldehyde emissions from materials over time [40]. Nonetheless, formaldehyde is commonly found at elevated
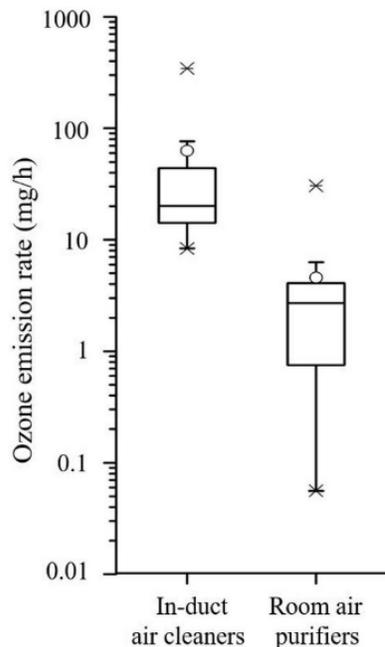
levels indoors [41]. Current building standards require new homes be built with mechanical ventilation (i.e. exhaust and/or supply ventilation fans) to increase the air exchange rate with outdoors and reduce indoor VOC concentrations. Singer et al. found that California homes built between 2011 to 2017, which were compliant with mechanical ventilation code requirements, had an average formaldehyde level of 19.8 ppb and a 10th to 90th percentile range of 13 to 28 ppb [42]. For comparison, California homes built between 2002 and 2005 (prior to mechanical ventilation code requirements) had an average formaldehyde level of 35.0 ppb, with a 10th to 90th percentile range of 11 to 70 ppb.

3.2.2. Formaldehyde Health Effects

Formaldehyde is of concern for both cancer and non-cancer endpoints. California's Office of Environmental Health Hazard Assessment (OEHHA) has established Reference Exposure Levels (REL) for formaldehyde [43]. Air at levels lower than the reference level are considered safe. Specifically, both the 8-hour inhalation REL (the safe level for an 8-hour workplace exposure) and the chronic REL (the safe level if exposure occurs all the time) for non-cancer endpoints are 9 $\mu g/m^3$ (7.3 ppb at standard temperature and pressure (STP)). For cancer, the inhalation unit risk, the risk resulting from a 1 $\mu g/m^3$ concentration, is 6.0 x 10^{-6} $(\mu g/m^3)^{-1}$. To achieve a one in a million risk of cancer, assuming constant exposure, the indoor air concentration would need to be 0.17 $\mu g/m^3$ (0.14 ppb).

The US EPA's Integrated Risk Information System (IRIS) states that formaldehyde is carcinogenic to humans and released a draft toxicological review in April 2022 that sets the reference concentration (RfC) (based on sensory irritation, pulmonary function, allergy-related conditions, and degree of asthma control/prevalence) at 7 $\mu g/m^3$ (5.7 ppb) [44]. The prior value established in 1991 was 200 $\mu g/m^3$ (163 ppb). The unit risk estimate is proposed as 6.4 x 10^{-6} $(\mu g/m^3)^{-1}$ to protect against nasopharyngeal cancers, which is very similar to the current level in California. The new value is based on a human occupational cohort and is considerably lower than the prior level of 1.3 x 10^{-5} $(\mu g/m^3)^{-1}$ which was based on studies carried out on rats.

3.2.3. Formaldehyde Emissions from Electronic Air Cleaners

Formaldehyde is also a ubiquitous intermediate product formed during the chemical degradation of many VOCs [26]. As such, ambient outdoor formaldehyde concentrations tend to be highest in regions where biogenic emissions of VOCs are high due to vegetation (e.g., the southeast US, the Amazon [45, 46]). General understanding and expectations of potential formaldehyde byproduct formation from electronic air cleaners can therefore be derived from general understanding of VOC oxidation chemistry. When indoor VOC concentrations are low, relatively low production of formaldehyde from electronic air cleaners would be expected. Correspondingly, when VOC concentrations are high, comparably higher formaldehyde production would likely occur. However, an important caveat when considering byproduct formation is that this assumes degradation of precursor VOCs; if the device is ineffective at reducing VOCs, then byproduct formation may be correspondingly small. Moreover, it is important to keep in mind that formaldehyde is itself reactive, and thus during testing of an electronic air cleaner formaldehyde may simultaneously produced and lost, with the observations indicating the net result of these competing processes. With this in mind, we turn to consider in detail results from specific studies that considered formaldehyde production from electronic air cleaners; production of other byproduct VOCs is considered in subsequent sections.

Waring and Siegel characterized steady-state formaldehyde concentrations in a 14.75 m^3 stainless steel chamber with and without operation of a portable ion generator referred to as a "common tower model" [47]. They observed a small increase in the formaldehyde concentration over background (10%; from 17.6 to 19.3 ppb) when the ion generator was operated in the clean chamber. The formaldehyde concentration increased substantially with the addition of a plug-in air freshener without the ion generator on, and there was an additional small increase in formaldehyde (7%; from 45.9 to 49.3 ppb) with the ion generator on. The statistical significance of these increases was not clear. Measurements by Waring and Siegel in a subsequent study in an unoccupied, unventilated 27 m^3 furnished room in a residential home indicated formaldehyde concentrations increased by 9-14% (5.4 to 8.5 ppb increase) when a portable ion generator was operated, for various conditions (carpeted versus non-carpeted; air freshener vs. none) [48]. The differences observed are within the measurement uncertainty, however the consistency between the various test conditions suggests that the increase was credible, albeit relatively small.

Crawford et al. measured formaldehyde concentrations in an unoccupied high school classroom with and without an ion generator operating, and with active ventilation at two different rates (425 m^3/hr and 765 m^3/hr) [49]. The formaldehyde concentrations were higher during the periods when the ion generator was on (3.05 and 3.74 ppb) than when it was off (2.42 and 1.87 ppb). Although the increase in concentration was clear, it could not be unambiguously established that this increase was not driven by variability in the ambient (background) formaldehyde concentration, although there was a concomitant and substantial increase in ozone suggesting that the formaldehyde increase resulted from ozone and VOC chemistry.

Gunschera et al. investigated byproduct production within two test rooms of differing size (24 m^3 and 48 m^3) from four commercially available portable PCO air cleaners using longer wavelength UVA radiation (315–400 nm wavelengths). The lamp type and specific wavelengths were not reported [50]. During the experiments the rooms were ventilated with an air exchange rate of 0.4 hr^{-1}. They challenged the systems with individual compounds and mixtures of compounds that are commonly emitted from cleaning products and building materials in indoor environments; the challenge compounds included decane, formaldehyde, toluene, α-pinene, heptanal, and 1,2-dichlorobenzene. The effectiveness of the air cleaner at removing formaldehyde was determined by challenging the device with only formaldehyde. Additional experiments that challenged the air cleaners with a mixture of compounds showed byproduct formation, including formaldehyde, from three of the four cleaners. This was inferred to be net formaldehyde production (the result of simultaneous generation and removal processes). The cleaner from which no byproduct formation was observed was also equipped with an activated carbon filter placed after the PCO module. The authors considered newly purchased air cleaners, and thus it is unclear how effective the activated carbon filter would be with continued operation over long time periods. Comprehensive data were not presented across all mixtures nor across all devices, making it difficult to generalize the results beyond documentation of formaldehyde observed as a byproduct. In the one single-compound experiment (toluene), results showed the formaldehyde conversion efficiency (ppb-formaldehyde formed per ppb-toluene consumed) was 20% for an initial toluene concentration of 850 ppb. The starting toluene concentration in this experiment was higher than is typically observed in indoor environments by about a factor of 10-100 [51]. It is possible that the reaction pathways involved in the degradation of toluene could be different at lower concentrations, leading to different distributions of products. In this same study they also observed production of formaldehyde with clean

air when the device was turned on, with a steady state level of ~4.5 ppb. This indicates production even in the absence of added VOCs.

Ye et al. characterized byproduct production from four commercially available oxidation-based air cleaners in the presence of various VOCs testing conducted at initial concentrations of 8 to 17 ppb (unless otherwise noted) [52]. The air cleaners included a PCO-based system that had an added HEPA filter and activated carbon, a negative ion UVC system with carbon filtration, a plasma-based technology with a HEPA and carbon filter, and a photoelectrochemical oxidation system, which upon further investigation is a PCO device with carbon filter. Type and wavelength of UV lamps was not reported. Passive off-gassing of the devices while turned off, meaning emissions from the air cleaners' construction materials and not their operation, included formaldehyde and other VOCs and was substantial for the plasma-based system (Figure 6). An increase in the formaldehyde concentration occurred for all systems when they were turned on in clean air, with the increase ranging from about 1 ppb to 4 ppb over 60-120 minutes. This indicates that formaldehyde emissions from the air cleaner materials occur even in the absence of a separate VOC source when the devices are turned on. These are direct emissions related to the operation of the air cleaners, independent of other indoor air conditions. Notably, the direct emissions increased substantially as the relative humidity (RH) of the test air increased; further consideration of the influence of RH is needed. Ye et al. also observed that the addition of limonene, a common component of fragrances used in air fresheners and cleaning agents, initiated additional formaldehyde production, with formaldehyde concentrations ranging from about 10% to 70% of the initial limonene introduced. For the photoelectrochemical oxidation system, the formaldehyde production after limonene addition (150 ppb) decreased dramatically, by about 10 times, when the UV lights in the system were covered. The addition of an aromatic hydrocarbon, toluene (200 ppb), resulted in little formaldehyde production.

Sleiman et al. investigated byproduct production from a custom PCO device when challenged with toluene (20 to 400 ppb) [53]. Notably, they found that toluene was mineralized to CO_2 with high efficiency (~95%) under dry conditions (0% RH), but under more realistic conditions, the mineralization efficiency was much lower (~50-70%). Flow rate and inlet toluene concentration had slight effect on toluene conversion efficiency to formaldehyde and negligible effect on mineralization efficiency. The conversion of toluene (120 ppb) to formaldehyde was limited, only about 1% on a ppb basis and 0.2% on a per carbon mass basis. Mo et al. also considered byproduct formation from a custom PCO device when challenged by toluene [54]. They too found some production of formaldehyde but with low conversion efficiency.

Farhanian and Haghighat examined byproduct formation from a PCO device that used a UVC lamp with irradiance of 2.85 mW/cm^2 at 254 nm, and then also with a lamp with irradiances of 3.11 mW/cm^2 and 0.88 mW/cm^2 at 254 and 185 nm, respectively [55]. Theirs was a custom, non-commercial system that was much larger (3.56 m length, 1.2 m height) than commercial systems designed for a general consumer market. They characterized byproduct formation for the air cleaner with each lamp in the presence of the following challenge compounds tested individually: 1-butanol (495 ppb), n-hexane (510 ppb), octane (450 ppb), methyl-ethyl ketone (450 ppb), acetone (550 ppb), toluene (475 ppb), and p-xylene (500 ppb). In their default tests they observed formaldehyde conversion efficiencies (ppb formaldehyde per ppb challenge compound) ranging from 8% (acetone) to 140% (n-hexane) for the 254 nm UVC lamp, and from 11% (acetone) to 63% (octane) for the 185/254 nm UVC lamp; conversion efficiencies greater than 100% are possible for challenge compounds having more than 1 carbon. For the

non-aromatic compounds, the conversion efficiency nominally increased with the number of carbons comprising the challenge compounds. The conversion efficiencies for the aromatic compounds ranged from 22% to 80%. Farhanian and Haghighat observed that the conversion efficiency tended to vary with the concentration of the challenge compound, as did the removal efficiency, but it was difficult to discern a general relationship between concentration and conversion efficiency across all compounds or system conditions. The formaldehyde conversion efficiencies observed here for the aromatic compounds are notably larger than those observed by Sleiman et al. and Ye et al.

Hodgson et al. considered byproduct formation from a prototype PCO device installed in a well-ventilated room and challenged with various VOC mixtures representative of building scenarios at low, medium, and high concentrations [56]. They observed formaldehyde production that scaled with the initial concentration of their "office building" VOC mixture (dominated by ethanol, isopropanol, toluene, and various alkanes). They similarly observed formaldehyde production from their "cleaning product" mixture (dominated by isopropanol, butoxyethanol, and various terpene compounds), with greater concentrations observed at lower room ventilation rates, as expected. Specific tests with formaldehyde indicated that the PCO degraded formaldehyde, indicating that there was net production in the mixture tests. For the office building mixture, the net formaldehyde conversion efficiency was ~10% by ppb, while for the cleaning product mixture the conversion efficiency ranged from 19-42% by ppb.

Zeng et al. considered byproduct formation from operation of a needlepoint bipolar ionizer located in the air supply duct for two environments: (i) an aluminum test chamber that was filled with various furnishings and materials and supplied with charcoal-fiber filtered air, and (ii) a small, furnished office building that was occupied during the measurements and supplied with a mix of outside and return air that passed through a MERV 8 filter [5]. In the test chamber, they observed a net decrease of 19% in the total concentration of select identified compounds with the ionizer on, mainly owing to decreases in m- and p-xylenes and ethylbenzene, but with no corresponding increase in formaldehyde. No results for formaldehyde were presented for the in-field test.

Zeng et al. separately characterized the impact of an in-duct bipolar ionizer that also generated hydrogen peroxide on byproduct formation when tested in an aluminum test chamber and an unoccupied laboratory [36]. They considered both standard and perturbed conditions wherein one or more VOCs were added to the air stream. For the standard conditions, there was some indication of formaldehyde production, although the increases over the background levels were small (<10%) and only notable at especially low air flow rates through the device. For the perturbation experiments using limonene, they observed no formaldehyde production over the background, but also did not observe enhanced loss of limonene when the ionizer was on.

Taken all together, there seems to be a greater propensity for electronic air cleaners to produce formaldehyde when challenged with non-aromatic compounds compared to aromatic compounds. The conversion efficiencies to formaldehyde can be quite large (>50%), although the net production rate of formaldehyde is constrained by the concentration of VOCs present in a room and the efficiency with which the electronic air cleaner degrades that compound. However, some electronic air cleaners emit formaldehyde from the materials that they are made of (separate from the air cleaner operation). This is not unique to electronic air cleaners; many products in indoor environments off-gas VOCs, including formaldehyde. Since background formaldehyde concentrations in California homes already exceed the

REL and the 1 in million cancer risk level, any additional formaldehyde production that occurs from electronic air cleaner operation exacerbates the health concern.

3.3. Other VOC Byproducts

3.3.1. Other VOC Introduction

Many VOC byproducts are produced as VOCs react and fragment into smaller compounds; only complete mineralization of a VOC to CO_2 and H_2O with eliminate the presence of VOC byproducts. The number of possible byproducts will generally increase as the size (number of carbon atoms) of an initial VOC increases [57]. As noted above, the compounds detected in a study depends explicitly on the analysis method(s) used. Consequently, comprehensive understanding of byproduct formation is lacking. Nonetheless, the literature indicates that some generalization may be possible.

3.3.2. Other VOC Emissions from Electronic Air Cleaners

Acetaldehyde (C_2H_4O), the two-carbon analogue of formaldehyde, is commonly observed as a co-byproduct with formaldehyde. For example, Hodgson et al. found that the acetaldehyde/formaldehyde production ratio at the outlet of the PCO air cleaner tested varied from 0.5-0.7 when challenged with a VOC mixture representative of indoor air from of an office building, but only 0.15-0.3 when challenged with a VOC mixture representative of a cleaning product mixture (primarily terpenes) [56]. The office building mixture comprised a wide range of compounds including small alcohols, aromatics, alkanes, and chlorinated compounds whereas the cleaning mixture was primarily composed of terpenes. Farhanian and Haghighat observed acetaldehyde/formaldehyde ratios ranging from about 0.4 to 1.1 for their various test compounds [55]. Zeng et al. observed acetaldehyde/formaldehyde ratios close to unity [36]. Crawford et al., in a field test in a school with added limonene, observed acetaldehyde increased by a factor of 1.7 over background levels when the ionizer was on [58]. Gunschera et al. observed acetaldehyde production from PCO in the presence of various challenge compounds in a test chamber and reported the acetaldehyde/formaldehyde production ratio ranged 0.12 to 0.56 [50]. And Mo et al., in a review, reported that many studies observed acetaldehyde along with other small aldehydes [59]. Besides acetaldehyde, numerous studies have identified acetone as a common byproduct along with other low-molecular weight aldehydes [49, 50, 56, 60]. Most studies focused on characterization of aldehydes and small ketones (owing in part to measurement limitations), and thus there is a bias towards detection of such compounds. Sleiman et al. also noted the production of benzaldehyde from PCO when challenged with toluene, which is consistent with the summary results presented in Mo et al. [53, 59]

Probably the most comprehensive study on byproduct formation from electronic air cleaners, from a product identification perspective, comes from Ye et al. [52]. They used a combination of methods that collectively enable characterization of a large proportion of possible byproducts according to their molecular weight. Where possible, we indicate the potential identify of the compound based on our knowledge of indoor chemistry. One of their more interesting findings is that the total concentration of VOCs increased when the electronic air cleaners were turned on even in the absence of added VOCs (Figure 6). This indicates that the operation of the air cleaners leads to emissions on top of passive emissions from the materials when the air cleaners were off. This could have resulted from internal heating of components during operation, leading to increased passive emissions coupled with oxidation of the emitted compounds. They observed hundreds of individual compounds produced with widely

different emissions among air cleaners tested. For example, for one system (P4, Figure 6) the byproducts were dominated (>90% of the total concentration) by just three compounds: C_3H_6O (likely acetone), formaldehyde, and $C_3H_8O_2$ (potentially propanediol). In contrast, for another system, no single compound made up more than ~10% of the total, with formaldehyde, $C_2H_6O_3$, and $C_2H_4O_2$ (likely acetic acid) most abundant, and with most compounds contributing <1% (P1, Figure 6). Interestingly, for the system with the highest VOC concentrations observed before it was turned on (P3, Figure 6), the main compounds observed were C_8H_8 and C_7H_8 (potentially styrene and toluene, respectively). These compounds cannot be produced through oxidation chemistry; they must be directly emitted, and operation of the device increased emissions (Graeffe et al., also observed increases in similar non-oxygenated compounds with UVGI [60]). Notably, for the P3 system a nitrogen containing compound was observed, C_3H_4N (likely acrylonitrile); such nitrogen containing compounds are often harmful to human health as discussed below. The results mentioned above were all made at relatively low RH (<10%). Ye et al. investigated byproduct emission for one of the systems at elevated RH (75%) and found that the emissions were increased. They attributed this to decreased effectiveness of the charcoal filter inside this system resulting from water adsorption; this supposition was consistent with a decreased efficiency of the system towards limonene removal at elevated RH versus dry conditions.
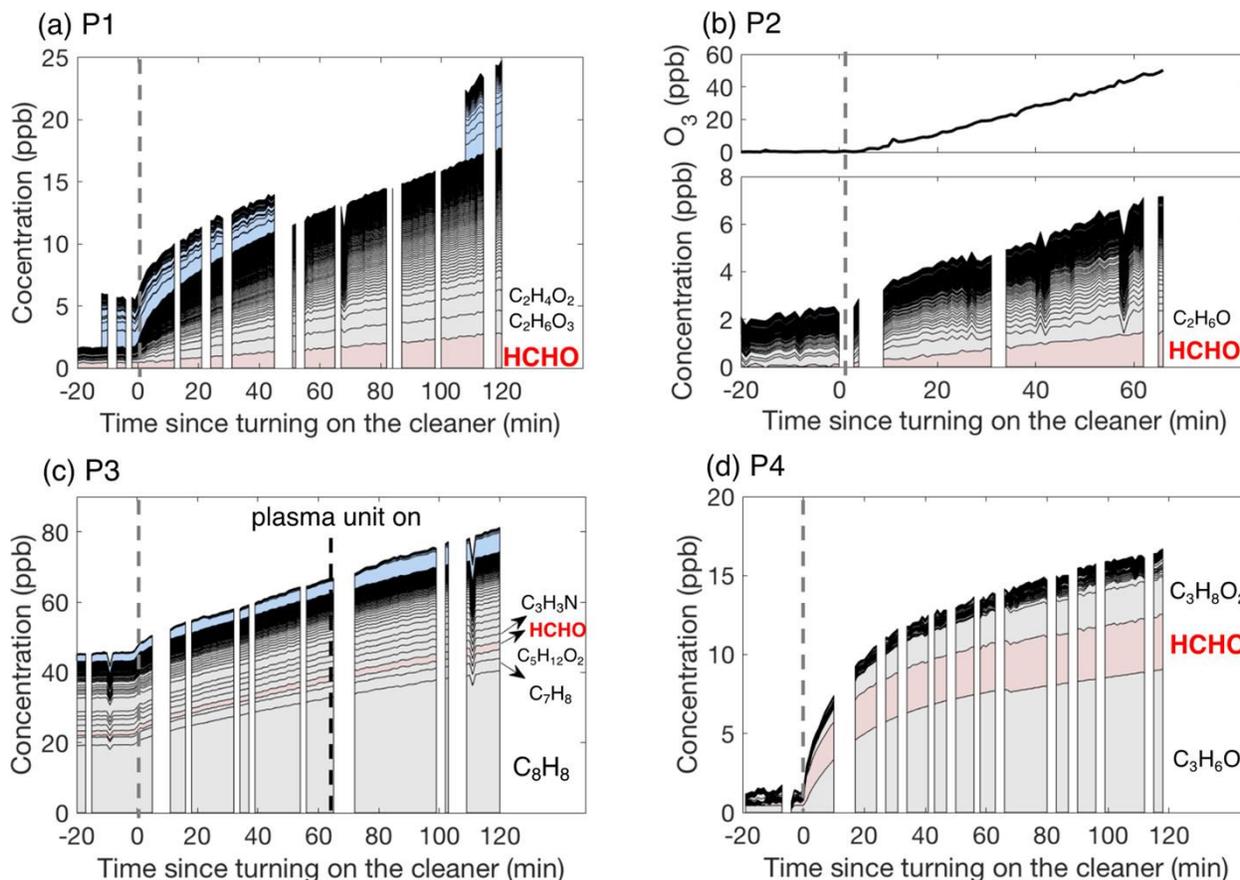

Figure 6: VOCs directly emitted from four air cleaners tested by Ye et al. (no challenge compounds). All cleaner components were turned on at t=0, except P3 where the air cleaner was turned on at t=0 and the plasma component was turned on at t=60. See Ye et al. for details [52].

Ye et al. also measured byproducts resulting from two of the systems challenged with limonene (system P2 challenged at 15 ppb and system P4 challenged at 150 ppb) and for one system challenged with toluene (system P4 challenged at 200 ppb) [44]. For one system (P2), the introduction of limonene increased the overall concentration of oxygenated species, indicating conversion of limonene to byproducts. These byproducts persisted over time, indicating limited subsequent loss or conversion. The total byproduct concentration (in ppb) was on the order of the amount of limonene introduced. P2 was also challenged with toluene but no removal was measured and therefore byproducts were not reported. For the other system (P4), byproduct formation, primarily formaldehyde, was observed upon the introduction of limonene. The concentrations of the byproducts peaked around 20 minutes and then decayed towards zero, and the peak concentration was only about 10% of the limonene that was introduced (on a ppb basis). For P4, the limited byproduct production likely resulted from physical uptake of both the limonene and the byproducts on an internal carbon filter. Ye et al. also challenged P4 with toluene, which was removed by the air cleaner with very limited byproduct formation. Again, this likely indicates the utility of an internal carbon filter to limit byproduct formation. At the same time, the results from this study indicated that the electronic air cleaner was only slightly more efficient at VOC removal when the active component (PCO) was on versus off, thus calling into question the utility of the PCO relative to the carbon filter.

3.3.3. Acetaldehyde Health Effects

Acetaldehyde, a byproduct reported in many electronic air cleaning studies described above, is of concern for both cancer and non-cancer endpoints, including respiratory symptoms and eye irritation. California's OEHHA 8-hour inhalation REL is 300 $\mu g/m^3$ (166.5 ppb at STP) and the chronic REL for non-cancer endpoints is 140 $\mu g/m^3$ (77.7 ppb) [61]. For cancer, the inhalation unit risk is 2.7×10^{-6} $(\mu g/m^3)^{-1}$. To achieve a one in a million risk for cancer, assuming constant exposure, the concentration would need to be 0.37 $\mu g/m^3$ (0.2 ppb). The IRIS RfC is 9 $\mu g/m^3$ (5 ppb), based on impacts to the nervous and respiratory systems [62]. For cancer, the inhalation unit risk is 2.2×10^{-6} $(\mu g/m^3)^{-1}$, which is similar to California's risk assessment. In general, the OEHHA and IRIS have higher allowable concentration limits for acetaldehyde exposure compared to formaldehyde (except for the IRIS RfC, which is slightly lower on a volume concentration basis for acetaldehyde).

3.3.4. Acetone Health Effects

Acetone, a byproduct reported in many electronic air cleaning studies described above, has been evaluated by the EPA through their IRIS program, but no IRIS reference concentration for inhalation was determined. It was also determined that the data are inadequate for assessment of human carcinogenic potential. Acetone has been reported to cause irritation of eyes, nose, and throat in human studies at levels ranging from 250 to 1000 ppm, higher than occurring from air cleaners (which is generally in the ppb range) [63, 64].

3.3.5. Acrylonitrile Health Effects

Acrylonitrile has been evaluated by the EPA through their IRIS program with an inhalation RfC of 2 $\mu g/m^3$ (0.9 ppb) based on "Degeneration and inflammation of nasal respiratory epithelium; hyperplasia of mucous secreting cells" [65]. IRIS classifies acrylonitrile as a probable human carcinogen with an inhalation risk of 6.8×10^{-5} per $\mu g/m^3$. To achieve a one in a million risk for cancer, assuming constant exposure, the concentration would need to be 0.015 $\mu g/m^3$ (7 ppt). Acrylonitrile was only observed as an

emission from one air cleaner (P3) in Ye at al. and the emission was most likely associated with the air cleaner materials and not the electronic air cleaning mechanism [44]. Most other byproduct studies would not have detected acrylonitrile (or similar compounds), owing to the limitations of the chemical characterization methods used. Regardless, this is a concerning emission given the potential for health effects at low concentrations.

3.3.6. Propanediol, Acetic Acid, Ethanol, and Isopropanol Health Effects

Propanediol, acetic acid, ethanol, and isopropanol, all detected by Ye et al., have not been evaluated by the EPA, and limited relevant inhalation study data was found. For propanediol, no apparent toxic effects in rats through inhalation were observed at concentrations up to 1,800 mg/m^3 (578 ppm) [66]. For acetic acid, a mild irritative effect was observed at 10 ppm, including increased blinking and subjective ratings of nasal irritation and smell [67]. For ethanol, continuous 90-day exposure in animal studies showed no adverse effects at 86 mg/m^3 (45.6 ppm) [68]. For isopropanol, toxic effects were seen in rats only with a 13-week exposure at the highest dose group, 5,000 ppm, and were not seen at concentrations of 1,500 ppm and less [69]. In all cases, the adverse effects seen in these studies were at concentrations several orders of magnitude above the ppb exposures expected in indoor air with an electronic air cleaner operating.

3.4. Ultrafine Particles (UFP)

3.4.1. Ultrafine Particles Introduction

Particulate matter with an aerodynamic diameter less than 2.5 microns (PM 2.5) is a mass-based measurement regulated by the NAAQS with regulatory levels for outdoor air of 12 µg/m^3 (annual mean) and 35 µg/m^3 (98th percentile 24-hour exposure) [18]. Ultrafine particles (UFP) are a subset of PM2.5 and are particles defined has having an aerodynamic diameter of less than 0.1 µm. UFP may comprise a substantial fraction of an ambient particle distribution on a particle-count basis; however, UFP comprise a small fraction of the mass of PM 2.5, since particle mass is proportional to the cube of the diameter.

3.4.2. Ultrafine Particles Health Effects

It has long been thought that there may be greater health impacts resulting from exposure to ultrafine particles than exposure to PM 2.5, supported by the potential mechanisms driving the adverse health impacts of PM [70, 71]. In contrast to larger particles that are lost at various points in the respiratory system, UFP can reach the alveoli in the lungs, cross into the bloodstream, and diffuse into all organ systems [71]. The greatest increase in mortality from particles is from cardiovascular impacts, and ultrafine particles can cause pulmonary inflammation to a greater extent than PM 2.5 [70]. Exposure to ultrafine particles can impact the respiratory system, specifically induce cough and worsen asthma. They have also been linked with diabetes and low birthweight [70]. However, it is more difficult to conduct epidemiology studies on UFP than PM 2.5. Much of the epidemiology for PM2.5 is based on outdoor measurements of large regions and resulting health impacts from large databases. It is more difficult to determine exposure to UFP because it is more difficult to measure and there is considerably more spatial variability and greater differences between indoor and outdoor exposures. Based on a systematic review, adverse health impacts in the existing studies most consistently show short-term effects with pulmonary/systemic inflammation, heart rate variability, and blood pressure [71]. However,

for all but inflammatory and cardiovascular changes, impacts are not clearly linked independently to UFP, but may potentially result from co-exposures to other air pollutants.

3.4.3. Ultrafine Particle Emissions from Electronic Air Cleaners

The only identified source of UFP emissions resulting from electronic air cleaning technology is from reactions between ozone and terpenes. In this mechanism, the VOC byproducts of the ozone/terpene reaction nucleate and condense to form particles. UFP emissions are dependent upon the specific terpene, temperature, and other competing indoor chemistry reactions; however, typical yield rates range from 0.08 to 0.39 μg of UFP per μg of terpene [72]. In modeling indoor air chemistry, Corsi determined that in a worst-case scenario of a building with a terpene source and low air change rate, ozone emission rates indoors must be limited to 0.04, 0.13, and 0.45 mg/hr in an office, school, and residence respectively in order to keep UFP concentration increases resulting from ozone/terpene reactions below 2 μg/m^3 in each of those spaces [72]. These ozone emission rates are 1-2 orders of magnitude below the estimated 4 mg/hr of ozone that could potentially be emitted from devices compliant with UL 867 (see Section 3.1), indicating that UFP emissions could be a concern for certain air cleaning devices.

A variety of literature supports that operation of electronic cleaners with a terpene source present produces UFP. In a laboratory test by Waring et al., five ion generators generally decreased particle concentrations when operated with laboratory air, but increased particle concentrations by 636-2,332 #/cm^3 when an air freshener (terpene source) was placed in the chamber [47]. The particle increase was in the ultrafine range (4.6-157 nm) and was accompanied by a decrease in ozone concentration compared to the ion generator operating without the air freshener present. Even though the UFP generation consumed some of the ozone generated by the air cleaner, the ozone concentrations with the ion generator operating still exceeded the background concentration by approximately 10 ppb. These results were replicated by Waring and Siegal in a room of a home, although the increase in particles was only observed when the heating, ventilating, and air conditioning (HVAC) system was off, such that that air exchange with the outdoors was minimized (air exchange rate (AER) ~ 0.5 hr^{-1}) [48]. When the HVAC system was on, the air exchange rate was higher (AER ~ 1.3 hr^{-1}), and UFP was not formed. In a field test of an ion generator in three residential homes, Hubbard et al. demonstrated that introduction of a terpene source resulted in generation of particulate matter less than 1 μm (PM1) at rates of 160-1,149 μg/hr per terpene source [73]. Lastly, in the study previously discussed by Ye et al., the limonene challenge of air cleaner P2 (UVC+ ion generator + carbon filter) resulted in a peak particle mass concentration of 25 μg/m^3 in the 7.5 m^3 chamber [44].

3.5. Hypochlorous Acid

Potential concerns associated with HOCl generation from electronic air cleaners are impacts on occupant respiratory health from direct HOCl exposure and byproduct formation including oxygenated and chlorinated VOCs [74]. No data was found on expected HOCl concentrations resulting from use of electronic air cleaners that reportedly using this working mechanism. The only relatable data available on HOCl are from impacts of cleaning with chlorine bleach, but whether or not the HOCl levels from cleaning with bleach are relevant to HOCl-generating air cleaners is unknown. When using chlorine bleach, experimental evidence has found that HOCl can reach ppb levels indoors [75]. When HOCl reacts with indoor surfaces, Cl_2 gas, a known toxic compound, is formed. While the formation of HOCl supports evaluating epidemiologic evidence of adverse health impacts associated with chlorine bleach, it must

also be noted that several other pollutants other than HOCl can be produced from the use of chlorine [75]. Some of these compounds, including reaction products from chlorine and ammonia, ethanolamine, lauryl dimethyl benzyl ammonium chloride, and benzalkonium chloride, have also been associated with adverse health effects [76].

A review of epidemiology studies focused on workers found that there was an association between exposure to chlorine bleach and lower respiratory tract and asthma symptoms [77]. A recent study of 1,586 domestic cleaners in Belgium found significant associations with use of liquid chlorine bleach and work-related eye symptoms, work-related sore throat, work-related asthma, work-related cough, and chronic bronchitis [78]. Furthermore, time spent swimming in indoor pools, which use chlorine for disinfection, is suggestive of development of childhood asthma [79]. Given the wide range of evidence of adverse health impacts with chlorine, coupled with the fact that HOCl is formed after use of chlorine bleach, raises concern over the safety of HOCl-generating air cleaners. Testing of HOCl-generating air cleaners is needed to evaluate the expected human exposure.

3.6. Ions

A review by Alexander et al. on respiratory impacts of negative air ions included 23 studies published between 1933 and 1993 [80]. Overall, the review concluded that the literature did not demonstrate a clear benefit or risk for human exposure to air ions and that air ions do not appear to play an important role in respiratory function. More recently, Dong et al. studied cardiorespiratory effects in school children exposed to ionizer air purification in a randomized, double-blind crossover study [81]. Exposure measurements in the classroom demonstrated a 34 to 48% decrease in indoor PM and 50% decrease in black carbon concentrations, no change in ozone concentrations, and a large increase in negative air ions (NAI) concentrations (12 cm^{-3} to 12,997 cm^{-3}). The ionizer air purification was associated with two positive cardiorespiratory effects: a 4.4% increase in forced expiratory volume in 1 s and a 14.7% decrease in fractional exhaled nitrogen. However, ionizer air purification decreased heart rate variability (HRV), where decreased HRV in adults has been associated with increased lifetime risk of cardiovascular disease [82]. The observed reduction in HRV in children led Dong et al. to hypothesize that the NAI may have caused a negative impact on cardiac function [81]. However, there are many limitations to this study. First, no gas-phase pollutants other than ozone were measured, so ionizer impacts on indoor VOC chemistry, and whether changes in VOC concentrations may have impacted HRV, are unknown.

One potential concern is that the charge imparted on particles by ions emitted by electronic air cleaners (to increase particle deposition rates and reduce indoor concentration) could also potentially increase particle deposition rates in human respiratory systems. Deposition rates of charged particles have been found to be higher than those of uncharged particles, with the initial studies being conducted to understand the impact of radon and the charge put on particles in the presence of radon [83]. Studies involving charged particles have also been conducted to determine how to best optimize therapeutic aerosol treatments, and find higher deposition rates for charged particles [84]. However, it should be noted there are other sources of charged particles in our daily environment, including emissions from traffic sources, overhead power lines and cooking sources [85, 86]. Therefore, it is difficult to quantify the potential additional impacts of charged particles from air cleaners relative to other particle sources, but more work should be done to evaluate these impacts.

3.7. Direct UV exposure

Direct human exposure to UV radiation is an additional concern for UVGI air cleaners; the safety considerations well described by Raeiszadeh and Adeli and are summarized here [87]. UV radiation is classified as a carcinogen and has potential to damage human skin and eyes. Immediate consequences of UV exposure to the skin are sunburn, and long-term impacts are skin aging and increased risk of skin cancer due to the potential for UV to break DNA strands. Exposure to the eyes may result in photokeratitis, erythema of the eyelid, cataracts, solar retinopathy, and retinal damage. The most damaging wavelength to human cells is at a wavelength of ~270 nm; at this wavelength the International Commission on Non-Ionizing Radiation Protection (ICNIRP) recommends a maximum exposure of 3 mJ/cm^2 over an 8-hour period, with increased exposures allowable at other wavelengths (Figure 7) [88, 89]. These limits were developed for light-skinned individuals with the greatest risk for skin cancer. It's important to consider that these recommendations were developed for occupational exposures and must be used with caution for the general population, as there are photosensitive individuals for which these exposure limits may be inadequate. Photosensitivity can be caused by certain chemicals ingested or applied to the skin (e.g. vitamins, medications, and cosmetics). In 2021, the American Conference of Governmental Industrial Hygienists (ACGIH) published increased limits for UVC exposure with a recommended maximum exposure of 160.7 mJ/cm^2 over an 8-hour period for the 222 nm wavelength, which is approximately seven times the 2020 limit of 23 mJ/cm^2 that matched the ICNIRP recommendations in Figure 7 [90]. Buonanno et al. investigated using 222 nm wavelength light for air disinfection in occupied spaces (without any protection for the occupants against exposure) and concluded that some disinfection performance is achievable within the more conservative allowable exposure limit of 23 mJ/cm^2 [91]. However, ozone production is still observed at this wavelength, as discussed in Section 2.4 [92].

UVGI devices can be designed to prevent human exposure to the source, such as by drawing air through ductwork or device with an enclosed UV lamp. In this case, the risks to humans of direct UV exposure are negligible if the device is installed and operated as designed. Warning labels and safety switches that prevent UV lamps from operating when covers are removed can help protect consumers. In some cases, UVGI may be designed with lamps only partially enclosed, with the radiation directed away from occupants (e.g. pointed toward the ceiling). In this case, reflectance of surrounding surfaces must be considered to ensure that radiation is not reflected toward occupants [87].
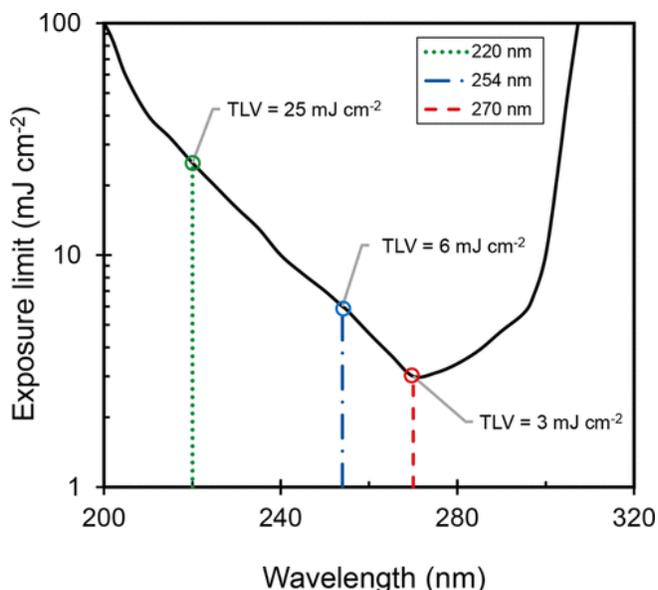
Figure 7: Occupational 8-hour exposure limit recommended for different wavelengths used in UVGI systems from Raeiszadeh and Adeli [87]. The wavelength most damaging to human cells is ~270 nm.

4. US Existing Test Standards and Regulations

California Assembly Bill 2276 (2006, Pavley) directed the State to regulate the amount of ozone emitted from indoor air cleaners for public health. CARB manages and enforces this regulation [93], requiring air cleaners to emit no more than 50 ppb ozone. One challenge to developing regulations on emissions is the need for robust and cost-effective test protocols to measure emissions that can be implemented by certification laboratories. Test protocols currently existing or under development in the United States covering air cleaner emissions are reviewed here. An effort was made to identify and understand and international air cleaner standards and regulations, however, language barriers and the high cost of purchasing standards for review rendered this effort unsuccessful.

4.1. Ozone Emissions Test Standards

4.1.1. UL 867

In 1988 Underwriters Laboratories (UL) published the first edition of UL Standard 867 – Electrostatic Air Cleaners [94] . This standard specified measurements for determining concentration of ozone emitted by air cleaners in a $26.9 - 31.1$ m^3 test chamber. The concentrations of ozone are required to be measured 2 inches (50 mm) from the air outlet of the air cleaner over a 24-hour period. No standard method of ozone measurement or maximum allowable concentration of ozone was specified. In 2011, UL published the fifth edition of UL 867 [94], which included specifications for the ozone measurement method, an overall chamber ozone removal rate of 1.33 h^{-1}, and a maximum ozone concentration of 50 ppb (i.e., 98 μg/m^3) measured 2 inches (50 mm) from the air outlet of the air cleaner over a 24 hour period. UL 867 does not directly measure an ozone emission rate; to directly measure the ozone emission rate the airflow rate through the device and the average ozone concentration in the air entering and leaving the device needs to be measured. However, the maximum ozone emission rate allowable for passing devices was estimated at 4 mg/hr (see Supplementary Information). The ozone

emission rate is needed to calculate the indoor ozone concentrations created by operation of the air cleaner in a modeled indoor space (e.g. home, classroom, office).

4.1.2. UL 2998

In 2016 UL published the first edition of UL Standard 2998 –Environmental Claim Validation Procedure for Zero Ozone Emissions from Air Cleaners [95]. The current third edition was published in 2020. This standard specifies the same ozone test method as UL 867, with a maximum ozone concentration of 5 ppb (i.e., 9.8 µg/m^3), which is 10 times lower than the UL 867 maximum. Air cleaners validated to UL 2998 are tested at least every three years, with surveillance audits occurring in non-testing years to ensure continued compliance and evaluate changes in the air cleaner materials and/or manufacturing that may alter ozone emissions. Like UL 867, UL 2998 does not directly measure ozone emission rate; it is a measure of the maximum ozone concentration measured 2 inches (50 mm) from the air outlet of the air cleaner. However, the maximum ozone emission rate allowable for passing devices was estimated at 0.4 mg/hr (see Supplementary Information). UL 2998 does not ensure "Zero Ozone Emissions from Air Cleaners" as the title of the standard implies, as some ozone production (~0.4 mg/hr) is permissible under the test criteria.

4.1.3. AHRI Standard 681

In 2017, the Air Conditioning, Heating, and Refrigeration Institute (AHRI) published AHRI Standard 681 - Performance Rating of Residential Air Filter Equipment [96]. Like UL 867 and UL 2998, this test method is a measure of the steady-state ozone concentration measured downstream of an air cleaner operating at the minimum design operational airflow rate in an ASHRAE 52.2 test duct. The reported ozone concentration is the steady-state ozone concentration measured downstream of the air cleaner minus the average pre and post-test background ozone concentrations with the air cleaner turned off. For a device to be compliant with the standard, the ozone concentration must not exceed 50 ppb (98 µg/m^3), similar to UL 867.

4.2. Chemical Assessment of Air Cleaners

4.2.1. ASTM-WK81750

Recently ASTM International convened a working group, WK81750, to draft a Standard Test Method for Chemical Assessment of Air Cleaning Technologies. This standard is expected to evaluate the emissions of byproducts of greatest health concern (ozone, formaldehyde, and ultrafine particles) created by portable air cleaners when indoor-relevant levels of challenge compounds are present. While the standard has not been finalized, a mix under consideration contains ozone, limonene, D$_5$ siloxane, tetrachloroethylene, formaldehyde, and o-xylene [97]. This standard is expected to provide a method of evaluating air cleaner emission rates of byproducts of greatest health concern when challenged with a mix of chemicals representative of the indoor environment.

4.2.2. AHAM-AC-4-2022

In 2022, the Association of Home Appliance Manufacturers (AHAM) published AHAM AC-4-2020, Method of Assessing the Chemical Reduction Rate of Chemical Gases by a Room Air Cleaner [98]. While this test method measures removal rates of gases (ammonia, formaldehyde, toluene, n-Heptane, and d-Limonene), the method does not report the generation of chemical byproducts generated by air

cleaners and is more limited in scope than ASTM-WK8170. This test method has potential to provide a more complete standard test method for electronic air cleaners with revisions to quantify byproducts and to challenge the air cleaner with a VOC (or VOC mixture) when ozone is also present.

4.3. UV Safety

4.3.1. IEC 62471:2006

International Electrotechnical Commission (IEC) Standard 62471:2006 "Photobiological safety of lamps and lamp systems" specifies a procedure for measuring optical radiation (total energy) and irradiance (energy per unit area reaching a surface) and comparing the results to exposure limits described in Figure 7. The results are used to classify lamps as low-risk, moderate-risk, or high-risk. The standard a broader scope than UV safety and evaluates wavelengths from 200 to 3,000 nm (which includes infrared).

5. Gaps in Research

Despite lack of comprehensive understanding of electronic air cleaner emissions and resulting indoor chemistry, compounds of clear concern are ozone, formaldehyde, and ultrafine particles, which were widely observed across studies from use of electronic air cleaners. Additionally, measurement of formaldehyde and ultrafine particle production provides a general indication of the extent to which other byproducts may be produced. Standard under development ASTM-WK81750 aims to evaluate emission rates of these three compounds from electronic air cleaners. However, subsequent research will be needed to model how emission rates for ozone, formaldehyde, and ultrafine particles from electronic air cleaners would then impact human exposure in terms of increased indoor air concentrations as a function of the building in which they are installed, where the primary variables driving exposure will be density of air cleaners installed, air exchange rate with the outdoors, mechanical filtration (for ultrafine particles only), and emission rates of VOC sources from building materials, furnishings, and occupants. To estimate the range of human exposure it will be important to analyze the expected distribution of exposure across building types) and identify populations that may be at risk for increased exposure (e.g., certain occupations). Combining this research approach with existing health effects information on ozone, formaldehyde, and ultrafine particles (with a particular focus on health effects in sensitive groups such as children and those with pre-existing lung conditions) could inform development of regulations for allowable emission rates of ozone, formaldehyde, and ultrafine particles for electronic air cleaners. Addressing this research gap could address, in the near team, several chemical compounds of concern that are known potential emissions of electronic air cleaners.

Other gaps in research that were identified in the literature review are:

- Emission rates of hydrogen peroxide from electronic air cleaners and health effects for continuous exposure to resulting concentrations for the general population.
- Emission rates of HOCl from HOCl-generating electronic air clears and health effects for continuous exposure to resulting concentrations for the general population.
- Health effects for exposure to ions and ionized particles.

- Health effects for direct exposure to UV light as a function of wavelength for the general population. Currently standards were developed for occupational health and safety, which does not consider sensitive members of the entire population.

It was also established that electronic air cleaners can produce myriad byproducts beyond ozone, formaldehyde, and ultrafine particles, but the identity of these byproduct compounds is in many cases not well established. This makes challenging any efforts to comprehensively assess the potential health risks of byproducts. Additional systematic studies of byproduct generation from electronic air cleaners challenged with individual chemicals or mixtures of chemicals, perhaps using techniques such as non-target analysis, is important to identify the wide range of byproducts that are likely produced, whether as individual compounds or classes of compounds based on functional groups. Compounds identified will likely needed to be complemented by health effects research for those compounds. Electronic air cleaner testing using advanced characterization methods such as non-target analysis is resource intensive and requires specialized equipment.

Although this whitepaper solely discusses the risks of byproduct formation, a well-informed decision to implement (or not implement) electronic air cleaning technology requires an assessment of the potential risks versus the potential benefits. In addition to research to quantify the risks of electronic air cleaners, additional research is needed to quantify performance in terms of contaminant removal under realistic and reproducible conditions so that the risks versus benefits can be considered. Such contaminants might include airborne pathogens, volatile organic compounds, or particulate matter.

6. Recommendations and Conclusions

The COVID-19 pandemic and worsening air quality from wildfires has led to increased interest in indoor air quality and indoor air cleaners. However, the risks versus benefits of electronic air cleaners have been inadequately studied and quantified. Comprehensive assessment and quantification of risks and byproduct formation from the use of electronic air cleaners in indoor environments is challenging given that many of the byproducts formed often depend on the initial presence of reactive compounds in the air, especially volatile organic compounds, and the abundance of such compounds will vary among buildings and over time in a given space. Moreover, the number of distinct byproducts formed from the reaction of even a single compound can number in the hundreds, which complicates the determination of potential health impacts of these byproducts. Our understanding of the risks of electronic air cleaner use is limited to the available literature on byproduct compounds produced, both in terms of identification and understanding of health impacts relative to the initial compounds from which they are formed.

Despite lack of comprehensive understanding of electronic air cleaner emissions and resulting indoor chemistry, compounds of clear concern are ozone, formaldehyde, and ultrafine particles, which were widely observed across studies from use of electronic air cleaners. We recommend addressing ozone, formaldehyde, and ultrafine particle emissions through:

1) testing air cleaners to a standardized test method when available (likely ASTM-WK81750),
2) analyzing the potential health impacts of those emissions through building modeling for various representative situations and conditions,
3) regulating formaldehyde and ultrafine particle emissions from electronic air cleaners (in addition to ozone) if emission testing and modeling demonstrates an unacceptable risk.

Additionally, we recommend California reduce ozone emissions from electronic air cleaners by requiring device compliance with UL2998, a more stringent, already existing, ozone emission standard of 5 ppb. This would reduce the allowable indoor ozone emissions by an order of magnitude which would provide a direct health benefit and subsequently reduce secondary formaldehyde and ultrafine particle formation that is driven by ozone chemistry.

In addition to these nearer-term actions designed to protect consumers from compounds of clear concern, we recommend CARB continue to further scientific understanding of electronic air cleaners, and indoor air chemistry and human exposure to indoor air compounds in general, by working to address gaps in research identified in this whitepaper. Future possible research to address these gaps includes:

- A modeling study to estimate increased human exposure to ozone, formaldehyde, and ultrafine particles from electronic air operation in different building types. The magnitude of increased exposure is expected to vary with density of air cleaners installed, building air exchange rate with the outdoors, mechanical filtration efficiency (for ultrafine particles only), and emission rates of VOC sources from building materials, furnishings, and occupants.
- A laboratory study to estimate emission rates of hydrogen peroxide from electronic air cleaners and emission rates of HOCl from HOCl-generating electronic air cleaners.
- Studies on health effects for indoor exposure to hydrogen peroxide, HOCl, ions, and ionized particles at air concentrations representative of the expected range resulting from electronic air cleaner operation.
- A study on health effects for direct exposure to UV light as a function of wavelength for the general population. Currently standards were developed for occupational health and safety, which does not consider sensitive members of the entire population.

Finally, continued effort to educate the public and institutional procurement staff on the risks associated with electronic air cleaners (to the extent allowable under CARB's purview) is helpful to build awareness and inform purchasing decisions.

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Supplementary Information

UL 867 and UL 2998 test procedures require testing an air cleaning device in a chamber of volume 26.9 to 31.1 m^3 with a total ozone loss rate from air exchange and surface deposition of 1.33 hr^{-1}; a passing device tested in this chamber must not produce an ozone concentration that exceeds 50 ppb for UL 867 and 5 ppb for UL 2998. In order to understand the impact a device may have in actual environments; it can be helpful to estimate the maximum ozone emission rate an air cleaner could produce while still passing this test using a well-mixed chamber model:

$$E_{ozone} = C_{ss} \; x \; MW_{ozone} \; x \; n_{air} \; x \; \lambda_{ozone}$$

Where:

C_{ss} – steady state concentration of ozone in the chamber [mol-ozone/mol-air]

MW_{ozone} – molecular weight of ozone [g/mol-ozone]

n_{air} – number of moles of in the chamber [mol-air]

λ_{ozone} – loss rate of ozone in the chamber [hr^{-1}]

For a maximum steady-state chamber concentration of 50 ppb (UL 867), the maximum ozone emission rate for the 31.1 m^3 chamber (containing 1271.9 mol-air using the ideal gas law) is:

$$E_{ozone} = 50x10^{-9} \; x \; 48 \; x \; 1{,}271.9 \; x \; 1.33 = 0.004 \; g/hr$$

For a maximum steady-state chamber concentration of 5 ppb (UL 2998), the maximum ozone emission rate for the 31.1 m^3 chamber (containing 1271.9 mol-air using the ideal gas law) is:

$$E_{ozone} = 5x10^{-9} \; x \; 48 \; x \; 1{,}271.9 \; x \; 1.33 = 0.0004 \; g/hr$$

Since the UL 867 standard measures ozone at the location with maximum concentration and ozone is very reactive, the measured concentration may be higher than the average test chamber concentration required to calculate the emission rate. Thus, 4 mg/hr is a theoretical upper bound on emission rate for UL 867 and 0.4 mg/hr is a theoretical upper bound on emission rate of UL 2998 (passing devices may be lower).