CONF--960273--2

UCRL-JC-122837
PREPRINT

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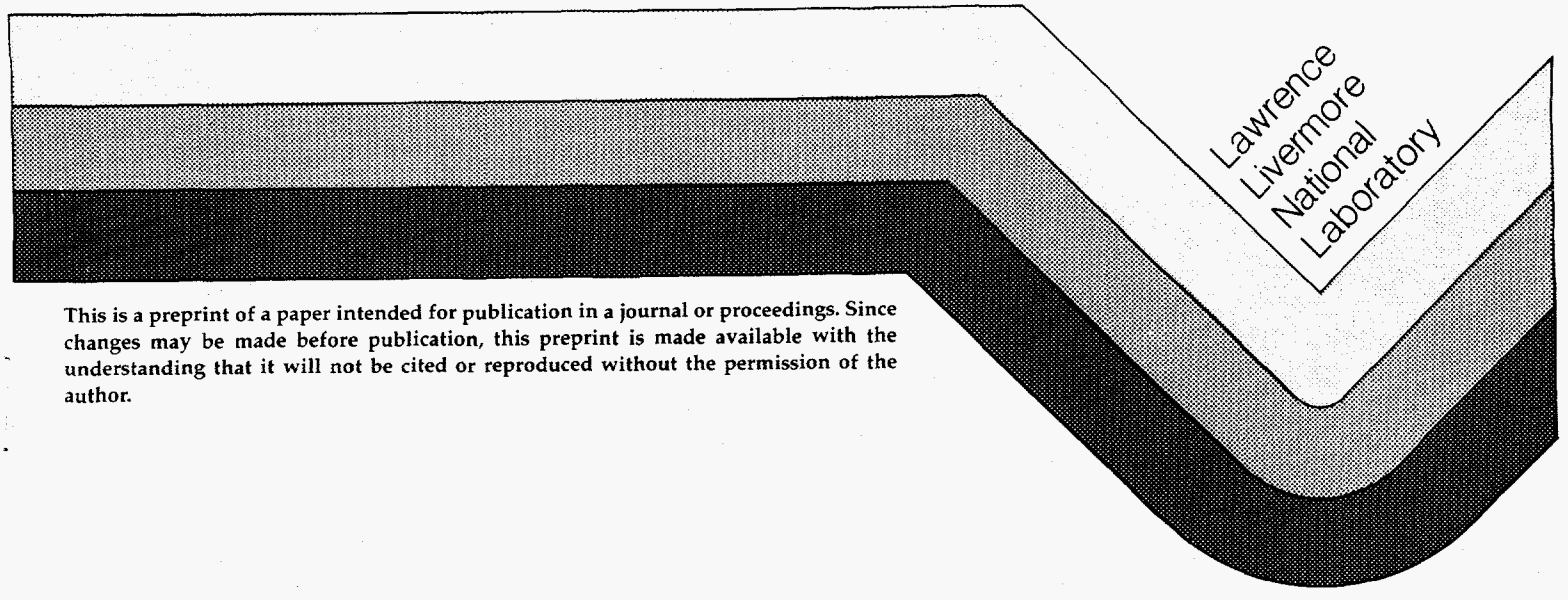
This paper was prepared for submittal to:
Emerging Solutions to VOC & Air Toxics Control Conference
Clearwater Beach, FL
February 28 - March 1, 1996

January 11, 1996



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ABSTRACT

Non-thermal plasma processing is an emerging technology for the abatement of dilute concentrations of volatile organic compounds (VOCs), nitrogen oxides (NO_x) and other hazardous air pollutants (HAPs) in atmospheric-pressure gas streams. Either electrical discharge or electron beam methods can produce these plasmas. Recent laboratory-scale experiments show that the electron beam method is remarkably more energy efficient than competing non-thermal plasma techniques based on pulsed corona and other types of electrical discharge plasma. Preliminary cost analysis based on these data also show that the electron beam method may be cost-competitive to thermal and catalytic methods that employ heat recovery or hybrid techniques.

INTRODUCTION

The control of emissions from dilute, large volume sources is a challenging problem. Conventional technologies for these, such as carbon adsorption or catalytic/thermal oxidation (or reduction), have high annual costs per ton of pollutant emissions controlled. Because of the large gas flow rate, operating costs over several years can greatly exceed the installed capital cost when conventional systems are evaluated.

Non-thermal plasma processing is an emerging technology for the abatement of dilute concentrations of volatile organic compounds (VOCs), nitrogen oxides (NO_x) or other hazardous air pollutants (HAPs) in atmospheric-pressure gas streams [1]. Either electrical discharge or electron beam methods can produce these plasmas. The basic principle that these techniques have in common is to produce a plasma in which a majority of the electrical energy goes into the production of energetic electrons, rather than into gas heating. Through electron-impact dissociation and ionization of the background gas molecules, the energetic electrons produce free radicals, ions and additional electrons which, in turn, oxidize, reduce or decompose the pollutant molecules. This is in contrast to the use of plasma furnaces or torches and several chemical techniques in which the whole gas is heated in order to break up the undesired molecules. For many applications, particularly in the removal of very dilute concentrations of air pollutants, the non-thermal plasma approach would be most appropriate because of its energy selectivity.

Electrical discharge and electron beam methods can both be implemented in many ways. There are many types of electrical discharge reactors, the variants depending on the electrode configuration and electrical power supply (pulsed, AC or DC). Two of the more extensively investigated types of discharge reactors are based on the pulsed corona and dielectric-barrier discharge, shown in Figure 1. In the pulsed corona method, the reactor is driven by very short pulses of high voltage, thus creating short-lived discharge plasmas that consist of energetic electrons, which in turn produce the free radicals responsible for the decomposition of the undesirable molecules. In a dielectric barrier discharge reactor, one or both of the electrodes are covered with a thin dielectric layer, such as glass or alumina. Whereas in the pulsed corona method the transient behavior of the plasma is controlled by the applied voltage pulse, the plasma that takes place in a dielectric-barrier discharge self-extinguishes when charge build-up on the dielectric layer reduces the local electric field.. Dielectric-barrier discharge reactors, also referred to as silent

discharge reactors, are now routinely used to produce commercial quantities of ozone. Unfortunately, the plasma conditions suitable for the generation of ozone are not the same plasma conditions optimum for the destruction of most VOCs and HAPs.

In the past, the high capital cost and x-ray hazard associated with conventional MeV-type electron beam accelerators have discouraged the use of electron beam processing in many pollution control applications. Recently, however, compact low-energy (<200 keV) electron accelerators have been developed to meet the requirements of industrial applications such as crosslinking of polymer materials, curing of solvent-free coatings, and drying of printing inks. Special materials have also been developed to make the window thin and rugged. Some of these compact electron beam sources are already commercially available and could be utilized for many pollution control applications.

In this paper we will present a comparative assessment of various non-thermal plasma reactors. The thrust of our work has been two-fold: (1) to understand the scalability of various non-thermal plasma reactors by focusing on the energy efficiency of the electron and chemical kinetics, and (2) to identify the byproducts to ensure that the effluent gases from the processor are either benign or much easier and less expensive to dispose of compared to the original pollutants. We will present experimental results using a compact electron beam reactor, pulsed corona reactor and dielectric-barrier discharge. We have used these reactors to study the removal of NO_x in both reducing and oxidizing environments, and to the removal of a wide variety of VOCs and HAPs, including carbon tetrachloride, trichloroethylene, methylene chloride, benzene, toluene, xylene, ethylene, propene, acetone and methanol. We have studied the effects of background gas composition and gas temperature on the decomposition chemistry. For all of the pollutants investigated, we find that electron beam processing is remarkably more energy efficient than pulsed corona or dielectric-barrier discharge processing. Preliminary cost analysis based on these recent data also show that the electron beam method may be cost-competitive to thermal and catalytic methods that employ heat recovery or hybrid techniques.

TEST FACILITY

All of our experiments were performed in a flow-through configuration. To characterize the energy consumption of the process for each VOC, the composition of the effluent gas was recorded as a function of the input energy density. The input energy density, Joules per standard liter, is the ratio of the power (deposited into the gas) to gas flow rate at standard conditions (25°C and 1 atm). The amount of NO_x or VOC was quantified using a chemiluminescent NO_x meter, an FTIR analyzer and a gas chromatograph/ mass spectrometer.

Our electron beam reactor, shown schematically in Figure 2, used a cylindrical electron gun designed to deliver a cylindrically symmetric electron beam that is projected radially inward through a 5 cm wide annular window into a 17 cm diameter flow duct. An electron beam of 125 keV energy was introduced into the reaction chamber through a 0.7 mil thick titanium window. The electron beam current was produced from a low-pressure helium plasma in an annular vacuum chamber surrounding the flow duct. This electron gun was designed to deliver a highly-uniform electron beam into the plasma processing chamber even at very high gas flow rates.

Our pulsed corona reactor is a 1.5 mm diameter wire in a 60 mm diameter metal tube 300 mm long. The power supply is a magnetic pulse compression system capable of delivering up to 15-35 kV output into 100 ns FWHM pulses at repetition rates from 15 Hz to 1.5 kHz. The power input to the processor was varied by changing either the pulse energy or pulse repetition frequency. For the same energy density input, either method produced almost identical results. The gas mixtures were set with mass flow controllers. The gas and processor temperatures can be maintained at a temperature that can be controlled from 25°C to 300°C.

We wanted to see if there are significant fundamental differences in the performance of various discharge reactors. We therefore investigated whether it is possible to improve the processing efficiency by taking advantage of transient high electric fields during the formation of the streamer plasma. To do this, the voltage pulse should be very fast-rising, but with a pulse length short enough so that most of the radical production occurs only during streamer propagation. One way of achieving this condition is by combining the fast-rising, strongly non-uniform applied electric field of a corona reactor with the self-extinguishing microdischarge pulses of a dielectric-barrier discharge reactor. We therefore used a reactor that is a form of hybrid between a pulsed corona reactor and a dielectric-barrier discahrge reactor. The reactor consisted of a wire (1.5 mm diameter) in a 300 mm long dielectric (alumina) tube with inner and outer diameters of 28 mm and 35 mm, respectively. The middle 150 mm of the dielectric tube has aluminum foil coating the outside to form the outer electrode.

RESULTS AND DISCUSSION

Whatever the type of reactor, the plasma can induce four basic types of reactions with the pollutant molecules, as shown in Figure 3. The electron mean energy in a plasma reactor is very important because it determines the types of radicals produced in the plasma and the input electrical energy required to produce those radicals. Figure 4 shows the dissipation of the input electrical power in a dry air discharge. Note that at low electron mean energies (< 5 eV) a large fraction of the input electrical energy is consumed in the vibrational excitation of N_2. Electron mean energies around 5 eV are optimum for the electron-impact dissociation of O_2, which is important for the production of O radicals. These oxidizing radicals play a key role in the generation of ozone and the initial decomposition of some types of VOCs. To implement the chemical reduction of NO to benign molecules such as N_2 and O_2, the important reducing species is the N atom, which is produced through the electron-impact dissociation of N_2. High electron mean energies are required to efficiently implement the dissociation of N_2. For VOCs that take advantage of electron-induced or ion-induced decomposition, high electron mean energies are also required to efficiently implement the ionization of the background gas.

In power plant flue gas treatment applications, the purpose of the plasma is to oxidize NO. The plasma produces OH radicals that play the major role in the simultaneous oxidation of NO and SO_2 to their respective acids. The presence of SO_2 lowers the power requirement of the oxidation process by recycling the OH radicals (see Figure 5). The presence of O radicals provide additional oxidation of NO to NO_2; the latter is then further oxidized by OH radicals to nitric acid. The desired products, in the form of ammonium salts, are then obtained by mixing ammonia with the formed acids. Some form of scrubbing is required to collect the final products. The application of electron beam irradiation for NO_x removal in power plant flue gases has been investigated since the early 1970's in both laboratory- and pilot-scale experiments [2-5]. Electrical discharge methods are relatively new entrants in the field of flue gas cleanup [6-8].

Recently there has been a growing interest in the application of non-thermal plasmas to the removal of NO_x from diesel engine exhaust gases [9]. The development of a technology for the chemical reduction of NO_x in oxygen-rich environments would represent a breakthrough in the transportation industry. The implication of such a technology is far reaching for mobile sources since it will allow fuel-efficient, lean-burn gasoline and diesel engines to be developed that decrease CO_2 greenhouse gas emissions yet still permit the reduction of harmful NO_x, carbon monoxide and hydrocarbons. Although such a technology exists for stationary combustion sources, a more feasible, cost-effective and environmentally sound approach for mobile sources does not exist.

For mobile engine applications, it is very important to make a distinction between NO removal by chemical oxidation and NO removal by chemical reduction. To avoid the need for scrubbing of process products, the desired method of NO removal is by chemical reduction; i.e. the conversion of NO to the benign products N_2 and O_2. For typical exhaust gases without additives, the only species that the plasma can produce to implement NO reduction is the N atom. The term "NO reduction" refers strictly to the reaction:

$$N + NO \Rightarrow N_2 + O. \quad (1)$$

The plasma produces N atoms through electron-impact dissociation of N_2 in the exhaust gas:

$$e + N_2 \Rightarrow e + N + N. \quad (2)$$

The development of a cost-efficient non-thermal plasma method for implementing electron-impact dissociation of N_2 in atmospheric pressure gas streams could therefore potentially lead to a NO_x reduction technology that works at ambient temperature and does not require additives or catalysts.

Figure 6 shows a comparison between electron beam, pulsed corona and dielectric-barrier discharge processing of 100 ppm of NO in N_2. The concentration of NO is presented as a function of the input energy density deposited into the gas. In the NO-N_2 mixture the removal of NO is dominated by the reduction reaction $N + NO \Rightarrow N_2 + O$. These experiments therefore provide a good measure of the electron-impact dissociation rate of N_2. Figure 6 shows that the energy consumption for NO reduction by electron beam processing is six times less than that of pulsed corona or dielectric-barrier discharge processing. The energy density required to reduce NO is around 20 Joules/liter and 120 Joules/liter by electron beam and electrical discharge processing, respectively. These experiments provide a good measure of the specific energy consumption for electron-impact dissociation of N_2. The specific energy consumption obtained by electron beam processing represents the minimum energy cost for N_2 dissociation that can be achieved in any type of atmospheric-pressure non-thermal plasma reactor [10, 11].

As evident in Figure 6, we see no significant difference in the energy efficiency of various types of electrical discharge reactors with respect to the reduction of NO. As shown in Figure 7, we also observe no significant difference between pulsed corona and dielectric-barrier discharge reactors with respect to the decomposition of other compounds such as methanol.

There is a need for reliable data concerning the decomposition mechanisms associated with VOC/HAP compounds. Optimization of the process requires understanding of the mechanisms responsible for the decomposition of the VOC and HAP molecules.

Chlorinated VOCs are some of the most common solvents used, and are now found in hazardous concentrations at many industrial and government installations. The electron beam method has been applied to the removal of trichloroethylene [1, 12-13], carbon tetrachloride [14- 16] and other types of volatile hydrocarbons from industrial off-gases [17]. Some of the electrical discharge reactors that have been investigated for VOC abatement include the pulsed corona [18-20], ferroelectric packed bed [19-20] dielectric-barrier discharge [21-27], surface discharge [28-29], gliding arc [30-31] and microwave [32].

Figure 8 shows a comparison between electron beam, pulsed corona and dielectric-barrier discharge processing of 100 ppm of carbon tetrachloride (CCl_4) in dry air (20% O_2 80% N_2) at 25°C. The rate limiting step in the decomposition of CCl_4 is determined by the dissociative attachment of CCl_4 to the thermalized electrons in the created plasma:

$$e + CCl_4 \Rightarrow Cl^- + CCl_3. \quad (3)$$

During the creation of the plasma, electron-ion pairs are produced through primary electron-impact ionization of the bulk molecules, such as $e + N_2 \Rightarrow e + N_2^+$ and $e + O_2 \Rightarrow e + O_2^+$, and the corresponding dissociative ionization processes for N_2 and O_2. An analysis of the rates of the reactions discussed above suggests that the energy consumption for CCl_4 removal is determined by the energy consumption for creating electron-ion pairs. Figure 8 shows that the energy consumption for CCl_4 decomposition by electron beam processing is around sixty times less than that of electrical discharge processing. The energy density required to decompose CCl_4 by 90% is around 20 Joules/liter and 1270 Joules/liter by electron beam and electrical discharge processing, respectively. This result demonstrates that for VOCs requiring copious amounts of electrons for decomposition, electron beam processing is much more energy efficient than electrical discharge processing. The main products in the plasma processing of CCl_4 in air are Cl_2, $COCl_2$ and HCl. These products can be easily removed from the gas stream; e.g. they dissolve and/or dissociate in aqueous solutions and combine with $NaHCO_3$ in a scrubber solution to form NaCl.

For the case of methanol, the electron beam method is more efficient because the decomposition proceeds mainly via a dissociative charge exchange reaction

$$N_2^+ + CH_3OH \rightarrow CH_3^+ + OH + N_2 \quad (4)$$

The OH radicals resulting from the initial decomposition reaction (4) in turn may lead to additional decomposition of methanol via $OH + CH_3OH$. To verify that the primary decomposition during electron beam processing does not proceed through an oxidation pathway using O radicals, we performed the experiment using N_2 as the background gas; the specific energy consumption for electron beam processing in dry air is almost identical to that in N_2.

Not all compounds have strong dissociative electron attachment or dissociative ion charge exchange rates. For methylene chloride, the dissociative attachment rate to electrons is many orders of magnitude lower compared to carbon tetrachloride. In this case, the electron beam method is also more efficient because the initial decomposition proceeds via a reaction with the N atom

$$N + CH_2Cl_2 \rightarrow \text{products} \quad (5)$$

The energy efficiency for dissociation of N_2 to produce N atoms is much higher in an electron beam reactor.

For the case of trichloroethylene (C_2HCl_3 or TCE), the initial decomposition pathway can proceed efficiently by reactions with either electrons (in the electron beam method) or O radicals (in the electrical discharge method). Figure 9 compares electron beam and pulsed corona processing of 100 ppm trichloroethylene in dry air at 25°C. The energy consumption for TCE removal is relatively small using either electron beam or electrical discharge methods. This is because of a chain reaction mechanism involving chlorine (Cl) radicals. The reaction of TCE with electrons or O radicals initiates the detachment of Cl radicals. Other TCE molecules then decompose by Cl radical addition to the carbon-carbon double bond

$$Cl + CHClCCl_2 \rightarrow \text{products} \quad (6)$$

The decomposition pathway (6) regenerates more Cl radicals, which react with other TCE molecules, causing a chain reaction. Our byproduct measurements and material balance analysis point to significant amounts of dichloroacetyl chloride (DCAC), phosgene, and hydrochloric acid in addition to smaller amounts of CO and CO_2 in the effluent.

The case of trichloroethane ($C_2H_2Cl_3$ or TCA) is interesting in comparison to TCE. TCA and TCE have very similar electron attachment cross sections, yet the energy required for decomposition of TCE by electron beam processing is more than 10 times less than for TCA. The TCA molecule decomposes primarily through hydrogen abstraction by chlorine and oxygen radicals, whereas the TCE molecule decomposes through chlorine and oxygen radical addition to the carbon-carbon double bond. The carbon-carbon single bond in TCA is not susceptible to chlorine radical attack. The chain reaction mechanism possible with chlorinated ethylenes therefore does not occur with chlorinated ethanes [33].

The above decomposition mechanisms provide examples of how the chemistry could strongly affect the economics of the process. In some cases it will be necessary to experimentally or theoretically obtain fundamental information on rate constants and branching ratios in order to understand the energy consumption and byproduct formation in the plasma process. Computer modeling of the plasma chemical kinetics serves as an important design tool for minimizing the energy consumption of the process and identifying all possible byproducts.

COST ESTIMATES

Table I shows the nominal air pollution control costs for various technologies according to the US EPA Handbook of Control Technologies for Hazardous Air Pollutants. This information is not specific to any particular application and therefore indicates approximate costs only [34-35].

Table II shows the comparison between pulsed corona and electron beam processing of various VOC/HAP in dry air at room temperature. As mentioned previously, we observe no significant difference in the performance of pulsed corona and dielectric-barrier discharge reactors. For all the compounds we tested, electron beam processing is more energy efficient than either pulsed corona or dielectric-barrier discharge processing.

Assuming a nominal energy cost of 10 Joules/liter to decompose a mixture of volatile organic compounds from 100 ppm to 10 ppm, the electron beam power required for an 80,000 cfm total gas flow rate application is 380 kilowatts. Some commercial electron beam generators now cost as low as $2 per beam watt. A 380 kilowatt electron beam system would therefore have a capital cost of $760,000. This corresponds to a capital cost of less than $10 per cfm. This is cheaper than thermal oxidation methods that use advanced heat recovery. Similarly, the five year operating cost (based on $0.05/kWh electricity cost with 4,000 hours operation/year) is less than $5 per cfm. Again the operating cost of the electron beam method is much lower than those of advanced thermal oxidation or carbon adsorption.

Straight forward engineering is the major advantage of electrical discharge methods. However, the electrical energy consumption of electrical discharge reactors are excessive, as can be deduced from Table II. If we assume that a pulsed corona or dielectric-barrier discharge reactor consumes only 5 times more energy per VOC/HAP molecule, i.e., 50 Joules/liter to decompose the VOC/HAP from 100 ppm to 10 ppm, then the power required is 1.9 megawatts. Even though the capital cost for discharge reactors may be low, the operating costs over several years can greatly exceed the capital cost because of the large electrical energy consumption.

Table III shows nominal air pollution control costs using electron beam, pulsed corona and dielectric-barrier discharge processing. An energy cost in the range of 10-30 Joules/liter was assumed to decompose 100 ppm of VOC/HAP using electron beam processing. A pulsed corona or dielectric-barrier discharge reactor consumes at least 5 times more energy per VOC/HAP molecule. For the capital costs, it was assumed that $2/watt for electron beam, $1/watt for pulsed corona, and $0.20/watt for dielectric-barrier discharge are required. The five year operating cost is based on $0.05/kWh electricity cost with 4,000 hours operation/year. For control of emissions from dilute, large volume sources of VOCs and other HAPs, this preliminary cost estimates show that

(1) the operating cost of electrical discharge methods such as pulsed corona or dielectric-barrier discharge is excessive,

(2) the electron beam method is the preferable non-thermal plasma technique, and

(3) the electron beam method may be cost-competitive to thermal and catalytic methods that employ heat recovery or hybrid techniques.

In order to provide a more rigorous cost analysis for comparison with other VOC and air toxics control technologies, it is imperative that we establish how the electrical energy consumption of these

plasma methods depend on exhaust stream parameters such as moisture level and VOC/HAP mixture. Our experiments were done for single VOCs/HAPs. For applications in which the exhaust gas consists of a mixture of VOCs/HAPs, it is not yet clear how the decomposition of one VOC/HAP will affect the decomposition of the other. These studies will require proper characterization of the emission source and need to be done on a case by case basis for each application.

CONCLUSION

In this paper we have presented the results of recent laboratory-scale experiments showing that the electron beam method is remarkably more energy efficient than competing non-thermal plasma techniques based on pulsed corona and other types of electrical discharge plasma. Preliminary cost estimates based on these data show that the electron beam method may be cost-competitive to thermal and catalytic methods that employ heat recovery or hybrid techniques.

ACKNOWLEDGMENTS

This work was performed in part at Lawrence Livermore National Laboratory under the auspices of the U.S. Department of Energy under Contract Number W-7405-ENG-48, with support from the Advanced Energy Projects Division of the Office of Energy Research. The electron beam processing equipment was developed under a National Science Foundation SBIR grant, Contract Number III-9122767.

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Table I. Nominal air pollution control costs according to the US EPA Handbook on Control Technologies for Hazardous Air Pollutants.

Technology	Capital Cost Range	Operating Cost Range
Thermal oxidation with regenerative heat recovery*	$30-450 / cfm	$20-150 / cfm
Thermal oxidation with recuperative heat recovery**	$10-200 / cfm	$15-90 / cfm
Carbon adsorption with steam regeneration	$15-120 / cfm	$10-350 / cfm
UV/ozone oxidation	$10-140 / cfm	not available

* Regenerative heat recovery utilizes large, heavy beds of ceramic materials for heat recovery and storage. Up to 95% heat recovery is possible.
** Recuperative heat recovery utilizes metallic shell and tube heat exchangers for direct heat recovery. Up to 70% heat recovery is possible.

Table II. Comparison between pulsed corona and electron beam processing of 100 ppm VOC/HAP in dry air at room temperature. Energy density (Joules per standard liter) required for 90% decomposition of the VOC/HAP. Based on experimental data taken at Lawrence Livermore National Laboratory and First Point Scientific, Inc.

VOC/HAP	Electron Beam	Pulsed Corona
Trichloroethylene	6	38
O-Xylene	10	370
Ethylene	15	83
Methanol	15	450
Carbon Tetrachloride	20	1277
Toluene	34	1586

Table III. Nominal air pollution control costs using non-thermal plasma process. Based on experimental data taken at Lawrence Livermore National Laboratory and First Point Scientific, Inc. (See Table II).

Technology	Capital Cost Range**	Operating Cost Range***
Electron Beam*	$9-27 / cfm	$4-12 / cfm
Pulsed Corona*	$4-12 / cfm	$20-360 / cfm
Dielectric-Barrier Discharge*	$1-3 / cfm	$20-360 / cfm

* Assuming a nominal energy cost of 10-30 Joules/liter to decompose 100 ppm of VOC/HAP using electron beam processing. A pulsed corona or dielectric-barrier discharge reactor consumes at least 5 times more energy per VOC/HAP molecule.
** Assuming $2/watt for electron beam, $1/watt for pulsed corona, and $0.20/watt for dielectric-barrier discharge.
*** Based on $0.05/kWh electricity cost with 4,000 hours operation/year.

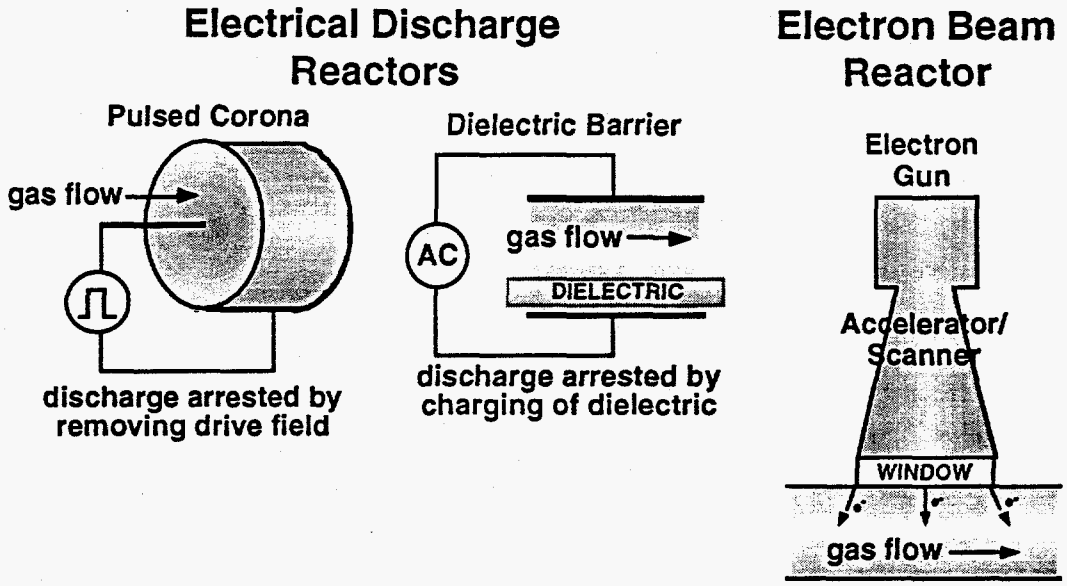

Figure 1. There are basically two types of non-thermal atmospheric-pressure plasma reactors: electrical discharge reactors and electron beam reactors.

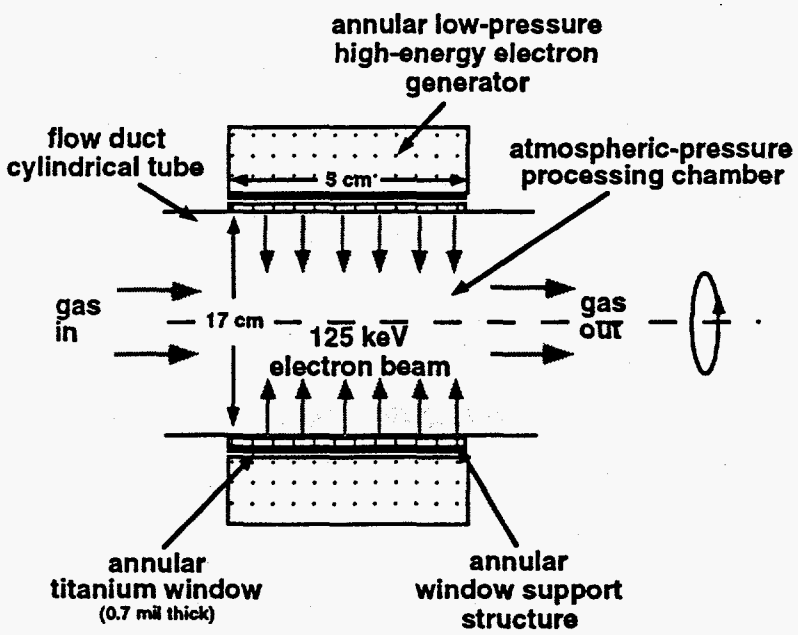

Figure 2. Compact electron beam source developed by First Point Scientific, Inc. The cylindrical electron gun is designed to deliver a cylindrically-symmetric highly-uniform electron beam that is projected radially inward through a 5 cm wide annular window into a 17 cm diameter flow duct. The non-thermal (ambient gas temperature) plasma produced by the electron beam is capable of decomposing VOCs and HAPs in the polluted gas stream even at very high gas flow rates.

- ***Oxidation***

$$e + O_2 \Rightarrow e + O(^3P) + O(^1D)$$
$$O(^3P) + NO + M \Rightarrow NO_2 + M$$
$$O(^1D) + H_2O \Rightarrow OH + OH$$
$$OH + NO_2 \Rightarrow HNO_3$$

- ***Reduction***

$$e + N_2 \Rightarrow e + N + N$$
$$N + NO \Rightarrow N_2 + O$$

- ***Electron-induced decomposition***

$$e + N_2 \Rightarrow e + e + N_2^+$$
$$e + O_2 \Rightarrow e + e + O_2^+$$
$$e + CCl_4 \Rightarrow CCl_3 + Cl^-$$

- ***Ion-induced decomposition***

$$N_2^+ + CH_3OH \Rightarrow CH_3^+ + OH + N_2$$

Figure 3. The plasma can induce four basic types of reactions with the pollutant molecules.

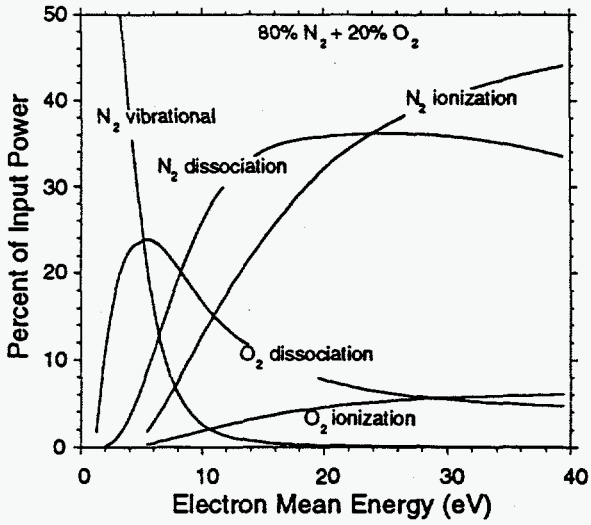

Figure 4. Electrical power dissipation in a dry air discharge, showing the percent of input power consumed in the electron-impact processes leading to vibrational excitation, dissociation and ionization of N_2 and O_2.

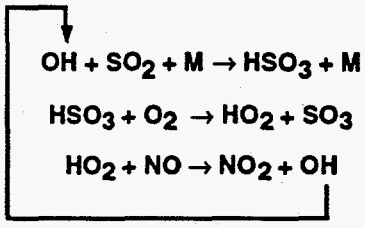

Figure 5. In flue gas treatment by non-thermal plasmas, the OH radical plays a key role in the simultaneous oxidation of NO and SO_2. The presence of SO_2 serves to lower the energy cost for oxidation of NO by converting OH to HO_2; the OH radical is then reproduced when NO is oxidized by HO_2.

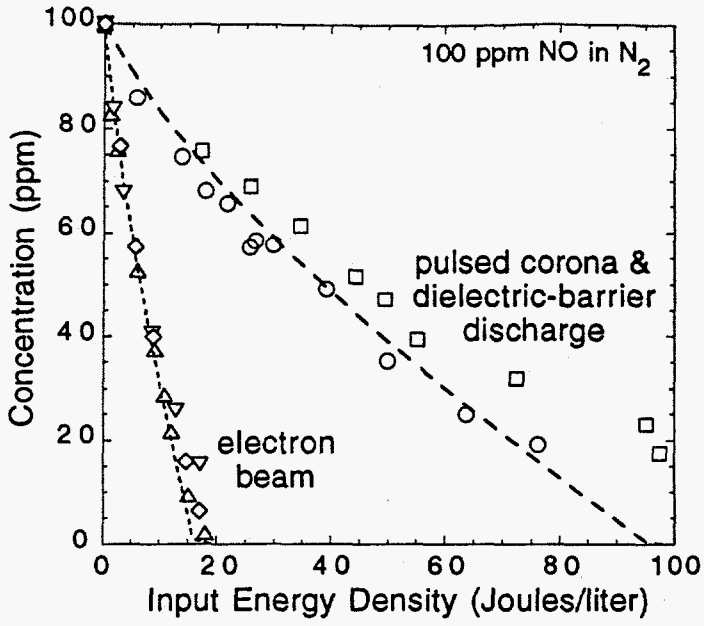

Figure 6. Electron beam, pulsed corona and dielectric-barrier discharge processing of 100 ppm NO in N_2. There is no significant difference in the performance of different types of electrical discharge reactors. Electron beam processing is six times more energy efficient compared to electrical discharge processing in reducing NO.

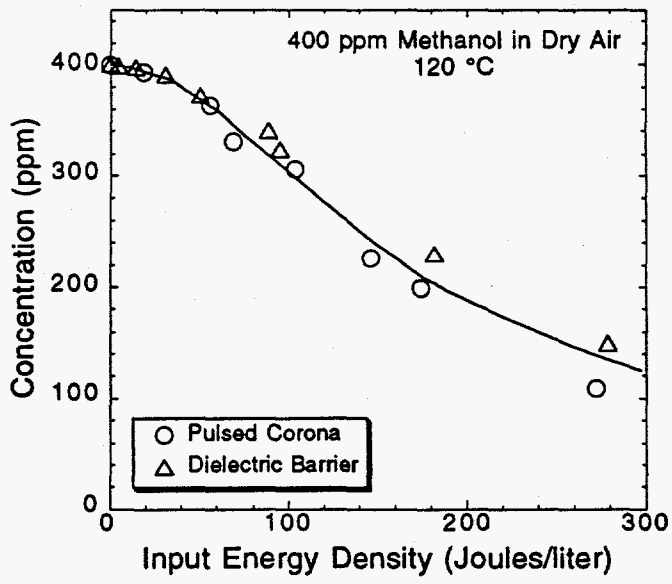

Figure 7. Pulsed corona and dielectric-barrier discharge processing of 400 ppm methanol in dry air at 120°C. There is no significant difference in the performance of different types of electrical discharge reactors.

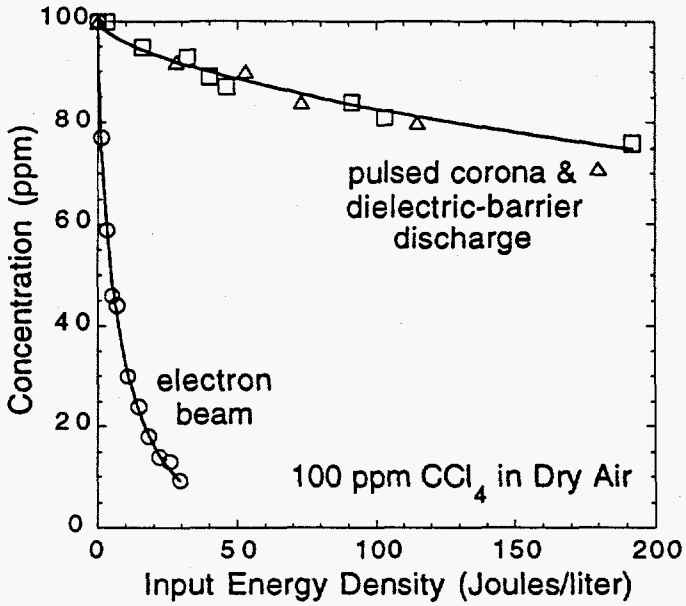

Figure 8. Electron beam, pulsed corona and dielectric-barrier discharge processing of 100 ppm of carbon tetrachloride in dry air. There is no significant difference in the performance of different types of electrical discharge reactors. Electron beam processing is more than sixty times more energy efficient compared to electrical discharge processing for 90% decomposition of carbon tetrachloride.

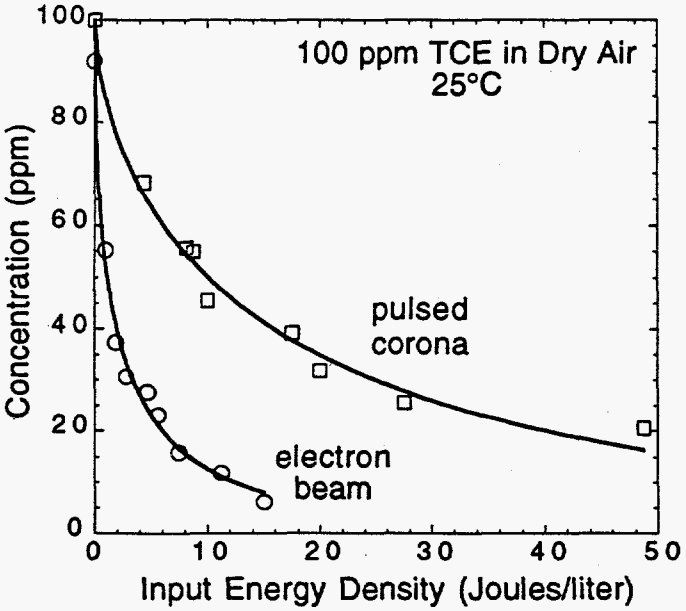

Figure 9. Electron beam and pulsed corona processing of 100 ppm of trichloroethylene in dry air. Electron beam processing is about six times more energy efficient compared to pulsed corona processing for 90% decomposition of trichloroethylene.