Report

Atmospheric Dispersion Modelling of Discharges to Air from the Flaring of Fracturing Fluid

Prepared for

Taranaki Regional Council

By

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Figures and Tables

1 Introduction

A study carried out during 1998 (Abbott, Backshall, & Bingham, 1998) investigated emissions to air as a result of gas and condensate flaring from flare pits in Taranaki. The flare was sampled from the side using an extended probe, and samples were collected and analysed for selected toxic air pollutants. In addition, the sample gas was continuously analysed for products of combustion.

These results were then used in an atmospheric dispersion modelling study in order to assess possible effects on ambient air quality. Ambient ground level concentrations predicted by the model were well below the corresponding air quality guidelines.

Fluid fracturing is a technique used to increase the production from gas wells. Liquid under pressure is injected down the well, with excess fluid returned to the wellhead. This fluid is usually disposed of by re-injection, but on occasion the fluid is pumped into a flare pit, where a gas flare evaporates the liquid. Most of the volatile organic compounds (VOCs) contained in the liquid will be combusted in the flare.

Emission tests were carried out at a flare pit in Taranaki during February 2012 to investigate possible effects from the flaring of fracturing fluid. The method was similar to that used for the 1998 study; the main difference being the use of a manifold so that a number of sample trains could be operated simultaneously. At the same time, ambient air quality was monitored downwind from the flare pit. Details of these studies can be found in reports by Source Testing New Zealand (STNZ) (Source Testing New Zealand Ltd, 2012) and the Regional Council (Taranaki Regional Council, 2012). However, not all pollutants, especially some toxic substances typically found at very low concentrations, can be readily measured in ambient air.

1.1 Scope

Taranaki Regional Council requested that Air Quality Management Ltd carry out a dispersion modelling study of discharges to air from the flare pit during disposal of the fracturing fluid. The aim of this study was to predict the maximum downwind concentrations of the pollutants discharged to air during the disposal of fracturing fluid.

1.2 Disclaimer

This review has been prepared by Air Quality Management on the specific instructions of the Taranaki Regional Council. It is solely for our Client’s use for the purpose for which it is intended in accordance with the agreed scope of work. Unless Air Quality Management has given prior written consent, any use or reliance by any person contrary to the above is at that person's own risk.

2 Methodology

Atmospheric dispersion modelling is typically used to relate mass emission rates of pollutants from a source(s) to the resulting concentrations in ambient air downwind from this source. These are mathematical models that use Gaussian air pollutant dispersion equations to determine downwind ground level concentrations. Most dispersion models use this approach, including the "puff" models such as CALPUFF.

The Regional Council requested that the modelling be carried out for a generalised situation rather than a specific drilling operation in Taranaki. This was because the terrain and meteorological conditions in Taranaki can vary widely, depending on the topography and the location relative to the mountain and coastline. So it was decided to carry out a screening study, which should give the maximum possible ground level concentrations. Screening models do not use site-specific meteorological data, but instead include a set of conditions that should result in worst-case plume dispersion. It should be noted that these conditions may not actually occur at a particular location, so screening studies are likely to be conservative.

The model selected for this study, and the reasons for this choice, are discussed below. The emission data used the model are then described in the remaining subsections.

2.1 The Dispersion Model

There are many atmospheric dispersion models, some of which are general models designed for a variety of source configurations whereas others are intended for specific applications, such as vehicle traffic on roads. The AUSPLUME model is intended for a variety of applications, and was used for the 1998 study. An updated version is still in use in Australia and New Zealand today.

This is a steady-state Gaussian model, where the meteorological conditions at the source are assumed to remain constant as the plume travels downwind. While non-steady state models can give more accurate predicted concentrations in complex terrain or at distances of more than a few kilometres downwind, models such as AUSPLUME are considered to be appropriate for situations where the terrain is reasonably flat and we are not concerned with long-distance transport of pollutants. Given that a flare pit is a low-level source, maximum concentrations would be expected a few hundred metres downwind.

Only a limited number of models include flares as a source option. These include some US EPA models, such as SCREEN3 (U. S. Environmental Protection Agency, 1995). As the name would suggest, this is a screening model. This type of model is primarily intended for use in situations where emissions from the source are unlikely to result in significant effects on ambient air quality. Such models usually include a set of meteorological data intended to estimate maximum possible downwind concentrations and incorporate conservative settings for the model options.

One disadvantage of screening models is that long-term average concentrations (greater than 24 hours) are difficult to determine with a reasonable degree of accuracy. This was one of the reasons why the AUSPLUME model was chosen for the 1998 study, as the evaluation criteria for some of the pollutants included long-term average concentrations.

However, the disposal of fracturing fluids in a flare pit will generally take only a few hours and is likely to be infrequent. So in this case, a screening model that included flares as a source type was considered to be more suitable than a general model such as AUSPLUME.

The SCREEN3 model with the flare source option is currently the federal standard for assessing flare discharges in the US, and this model with the default settings was used to predict ground level concentrations during fluid disposal. A recent study by Trinity Consultants (Boger & Kanchan, 2012) compared the flare modelling methodologies prescribed by state regulatory authorities in the US and Canada. They found that the U.S. EPA method generally predicted levels similar to the other methods.

Screen View version 3.5.0 (Lakes Environmental, 2011) was used for this study. This is a version of SCREEN3 developed by Lakes Environmental that includes a user-friendly interface.

2.2 Emission measurements: Flare zone

The STNZ emission monitoring study (Source Testing New Zealand Ltd, 2012) measured the concentrations of selected pollutants at the outer edge of the flare pit above the liquid. There are many practical difficulties in carrying out this type of testing. One of the key issues is the difficulty in placing the sample inlet, which is at the end of a long stainless steel probe, at a suitable position to collect representative samples.

If the inlet is too far from the flare, then the sample will be diluted by ambient air, and the mass of the pollutants collected will be comparatively low. However, if the probe inlet is within the combustion zone, then the flammable material in the flare may not be completely combusted and the samples are unlikely to be representative of the flare emissions.

This is discussed in the STNZ report, where it is noted that essentially no flare emissions were collected during the third sample due to a wind change.

Table 23 of the STNZ report summarises the results of the combustion gas monitoring. The mean oxygen levels during each of the three samples were 20.8%, 20.8% and 21.0%, which indicates a very high level of dilution of the sample. The levels for the first and second samples are similar to those measured during the 1998 ESR study.

The high dilution level had a number of effects on the results from the emission monitoring:

1. Compared to typical stack testing results, comparatively low emission concentrations were measured. In some cases, e.g. the dioxin / furan results, measured concentrations were below the limits of detection (LOD).
2. As the measured oxygen levels were typically a few tenths of a percent below 21%, measurement accuracy is reduced. For example, a measured value of 20.8% will be in the range 20.75 to 20.84%.
3. The combination of the above reduces the accuracy with which mass emission rates can be estimated.

2.3 Emission Measurements: Evaporation zone

As well as sampling from the edge of the flare, STNZ also measured concentrations of certain VOCs above the surface of the fracturing fluid in the flare pit. While these results are of interest, the sample inlet was positioned upwind of the combustion zone where air drawn in by flare combustion was likely to dilute the gas samples.

If the sampled gas had been representative of the VOC concentrations immediately above the liquid surface , then it may have been possible to directly estimate the total evaporation rate from the fracturing fluid. However, as with the combustion zone sampling, there were clearly practical difficulties in positioning the probe sample inlet.

2.4 Mass Emission Rates used in the Dispersion Model

One of the key inputs required for dispersion modelling is the mass emission rate of the pollutant being assessed. For point sources (stacks discharging above ground level), this is usually determined by multiplying the measured, or estimated, emission concentration by the gas flow rate.

An alternative method for combustion sources is to use standard combustion equations that relate fuel usage, the airflow required for combustion and the excess air (oxygen) to the flue gas emission rate. This approach was used in the 1998 ESR study to determine mass emission rates for the selected pollutants.

This approach has some deficiencies when applied to flare emissions. One of the key issues is that the combustion process is not contained within a boiler or incinerator. The amount of air entering the combustion zone is unknown, which means that the calculated combustion gas[1] emission rate is likely to be inaccurate. Combined with the difficulty of determining a representative temperature and effective diameter of the combustion zone, there is likely to be a significant margin of error in the calculated mass emissions. In the case of the STNZ emission tests, the sample dilution and LOD issues discussed above will compound this.

Given the issues in determining mass emission rates based on the results of the emission tests, other approaches were investigated. These included:

2. Using suitable emission factors.
3. Assuming that all of the VOCs in the fracking fluid were evaporated and not combusted in the flare.

All three approaches were used to determine the mass emission rates used in the dispersion model, as shown in Table 1. More than one method was used for some compounds, which increases confidence in the estimates. Details of each method and its application are given in the following sections.

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[1] The discharges to air from a boiler are normally referred to as flue gas. As there is no flue or stack in this case, then the discharges from flares will be referred to as combustion gas

Table 1: Mass emissions from flaring

Compound	Mass emission calculation method		
	Combustion equations	Emission factors	Fracking fluid VOC analysis
	Mass emission rate (g/s)		
TSP	1	0.24	
NOx	2.27	0.62	
VOC		1.27	
VOC (BTEX)	0.44		0.35
CO	8.53	3.37	
PAH (as BaP)	0.00013		
Formaldehyde	0.14		0.006
benzene			0.05
toluene			0.12
ethylbenzene			0.02
m&p-xylene			0.13
o-xylene			0.04
methanol			0.05
C7-C9			2.45
C10-C14			6.33
C15-C36			7.93
Total HC			16.50

2.4.1 Combustion equations with STNZ results

Combustion equations are often used to calculate emissions from stationary combustion sources such as boilers and incinerators. If the fuel composition and rate of use are known, then the amount of oxygen required for complete combustion can be calculated. The composition of the flue gas can then be determined if the amount of excess air is known or estimated.

This approach can only be used for products of combustion resulting directly from oxidation of the fuel. For example, sulphur dioxide emissions can be determined as this is formed from the sulphur contained in the fuel. However, oxides of nitrogen cannot be estimated this is primarily formed from nitrogen in the air during combustion.

This method is difficult to apply to uncontrolled combustion sources such as gas flares. This is mainly because large volumes of air are entrained into the combustion zone so it is not possible to accurately estimate parameters such as excess oxygen, combustion gas temperature, etc.

Notwithstanding these difficulties, this was the method used in the earlier ESR study. Excess oxygen was based on the combustion gas measurements during the sampling and was used to determine volumetric flow rate and vertical velocity for the dispersion modelling. As the dilution factor was very high, the estimates of the combustion gas parameters were expected to be result in conservative model predictions because the temperatures of the combustion gases and therefore plume buoyancy would be higher than modelled.

2.4.2 Emission factors

The US EPA in their compendium of emission factors, AP-42, has published some data for flares. Flare emissions were tested under controlled conditions using synthetic gas mixtures and the results used to derive the emission factors presented in Section 13.5 of AP-42. While only one set of emission tests was used to determine the emission factors, they have been rated as "B" (above average accuracy). However, factors were only published for the following 4 pollutants:

- Total hydrocarbons
- Carbon monoxide
- Oxides of nitrogen
- Soot

The emission factors are incorporated in the Flares model by Enviroware (Enviroware, 2011), which is a model designed to estimate the atmospheric impact of industrial flares, and was used to estimate mass emission rates. The output from the model is presented in Appendix A along with the model input settings.

2.4.3 Evaporation of VOCs from the fracturing fluid

As part of the study of emissions from the disposal of fracturing fluids, the Regional Council commissioned analyses of the fluid stored in the holding tank prior to flaring. One sample was taken and analysed for VOCs. These results are presented in Appendix C.

The volume of fluid pumped into the flare pit was recorded on each day of the sampling, so the mass of each VOC can be calculated from the analysis results, which were reported as µg/m^3. The well log for the emission testing period showed the gas fuel flow rate, but gave only a total for the fluid pumped into the flare pit.

As the fluid in the flare pit heats up during flaring, the VOCs will evaporate with most of the vapour entrained into the flare and combusted. Some of the vapour evaporated near the outside of the pit, especially on the downwind side, may mix with ambient air and disperse downwind.

The evaporation rates of the VOCs will vary during the flaring as the fluid is heated and the compounds vaporise.

Given the dynamics of the vaporisation process, it would be very difficult to estimate evaporation and combustion rates. A conservative estimate of the mass emission rates was made by assuming that all of the VOCs contained in the fluid evaporated at a constant rate during the flaring. While this approach is likely to underestimate the peak evaporation rate, the assumption that none of the VOCs were combusted is expected to be highly conservative.

It should be noted that this approach may underestimate the total mass emission rate of compounds such as formaldehyde that form as combustion products.

2.4.4 Accuracy of the Methods

Although each of the methods has some deficiencies as noted in the previous sections, it would be reasonable to assign the highest accuracy to the estimates made using the AP-42 emission factors as these are based on emission test results carried out under controlled conditions. The other two methods can be expected to be less accurate but the total mass emissions estimated from the fluid analysis are likely to be conservative, especially given that most of the VOC evaporated from the fluid surface is likely to be combusted in the flare.

Given the assumptions made to determine mass emission rates using combustion equations, the agreement between the results from this method and the emission factor approach is good, with the largest discrepancy being a factor of 4 for the TSP results. It is unclear why the STNZ results were higher in this case; one possibility is that higher levels of soot were present at the edge of the flare due to incomplete combustion. However, given the uncertainties in the mass emission rates calculated from the STNZ results, this should be considered as good agreement between the methods.

The largest discrepancy between methods was for formaldehyde mass emissions determined from the concentration in the fluid and the STNZ results, which were 6 and 143 mg/s respectively. The reasons for this are unclear, although as noted previously, formaldehyde may have formed in the flare as a product of combustion, which would increase the amount of formaldehyde in the plume relative to that vaporised from the fracturing fluid.

3 Assessment of Effects

The mass emission rates used in the model were those presented in Table 1 in Section 2.4. In the cases where more than one estimate was available, the highest was used.

SCREEN3 is a screening model and few options are available when setting up the model, especially when the flare source option is selected. As the gas flow rate varied during the emission testing period, the maximum value was used to determine the flare heat output for the model, which was also used to determine mass emission rates from the AP-42 emission factors (using the Flares model), and from the STNZ results.

Copies of the SCREEN 3 model setup and output files are provided in Appendix B.

Assessment criteria are discussed in Section 3.2 and the SCREEN3 model results are presented in Section 3.3 and discussed in the subsequent sub-sections.

3.1 Background Levels

In order to assess the possible effects of discharges to air of specific pollutants, it is necessary to measure or estimate ambient levels of these substances. However, all of the pollutants considered in this study are generally of anthropogenic origin, so the levels in rural environments are likely to be very low. $PM_{2.5}$ mostly results from combustion processes, although source apportionment studies in New Zealand have found sea salt and dust in this size fraction.

As ambient levels are likely to be minimal, background levels of zero have been assumed for all pollutants considered in this study.

In the case where a well is close to an industrial plant, it may be necessary to consider the effects of emissions from the plant of pollutants such as $PM_{2.5}$. However, this is not relevant to the screening study undertaken for this report.

3.2 Assessment Criteria

The following hierarchy of references was used to determine appropriate assessment criteria, as recommended in the Good Practice Guide for Assessing Discharges to Air from Industry.

1. New Zealand National Environmental Standards (NES)
2. New Zealand Ambient Air Quality Guidelines (NZ AAQG)
3. WHO Ambient Air Guidelines (WHO)
4. US EPA Reference Concentration for Chronic Inhalation Exposure (RfC)
   California OEHHA Acute Reference Exposure Levels (REL)

Due to the short-term and intermittent nature of the disposal of fracturing fluid by flaring, only short-term exposure criteria were considered, with one exception. Assessment criteria for particulate matter (PM) are generally 24-hour averages or longer, so the WHO 24-hour average guideline of 25 $\mu g/m^3$ for $PM_{2.5}$ was compared to the maximum model prediction for PM[2].

Long-term exposure criteria (annual averages) are included in the table of exposure criteria. For some compounds the annual average guideline is far lower than the short-term exposure criteria. Using benzene as an example, the 6-hour average of 1.3 mg/m^3 is more than 300 times the annual average.

Table 2: Assessment criteria

Compound	**Exposure criteria**		**Reference**
	Acute (short-term)	Chronic (long-term)	
benzene	1.3 mg/m^3, 6-hour		OEHHA
		3.6 $\mu g/m^3$, annual	NZ AAQG
toluene	37 mg/m^3, 1-hour	-	OEHHA
ethylbenzene	-	2,000 $\mu g/m^3$, annual	OEHHA
m&p-xylene, o-xylene	22 mg/m^3, 1-hour	700 $\mu g/m^3$, annual	OEHHA
m&p-xylene, o-xylene	22 mg/m^3, 1-hour	700 $\mu g/m^3$, annual	OEHHA
formaldehyde	100 $\mu g/m^3$, 30-minute	-	NZ AAQG
$PM_{2.5}$	25 $\mu g/m^3$, 24-hour	-	NZ NES
NO_2	200 $\mu g/m^3$, 1-hour	-	NZ NES
CO	10 mg/m^3, 8-hour	-	NZ NES

3.3 Dispersion model results

The output from SCREEN 3 is in the form of a graph showing ground level concentrations versus distances as presented in Figure 1. The model does not produce a 2-dimensional contour plot as it is a screening model and the set of meteorological conditions include wind speed but not direction.

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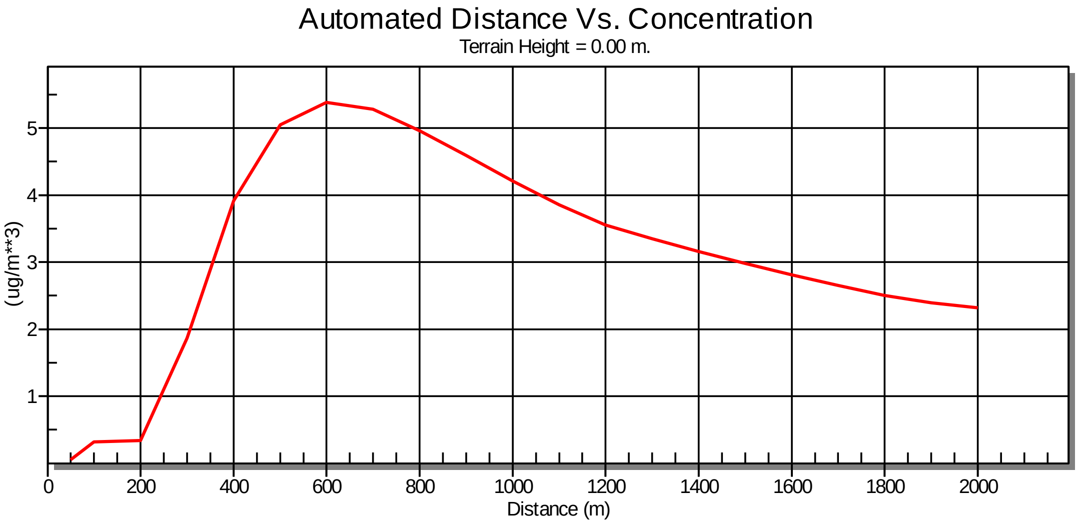
[2] A PM2.5 assessment criterion was used because PM from the flare is likely to be smoke, which is fine particulate less than 2.5 µm.

Figure 1: SCREEN 3 results



The maximum concentration is predicted to occur at a distance of 615 m downwind from the source. As there is only a single source, the model was run with a mass emission rate of 1 gram per second (g/s) and the resulting maximum ground level concentration of 5.4 µg/m^3 was scaled according to the mass emission rate of the specific compound.

The maximum predicted concentrations are given in Table 3 along with the short-term assessment criteria. The results are assessed in the following Sections.

Table 3: Dispersion model results

Compound	Maximum ground level concentration (µg/m^3, 1-hour avg.)	Exposure criteria	
		Acute (short-term)	Chronic (long-term)
benzene	0.26	1.3 mg/m^3, 6-hour	3.6 µg/m^3, annual
toluene	0.63	37 mg/m^3, 1-hour	-
ethylbenzene	0.09	-	2,000 µg/m^3, annual
m&p-xylene	0.69	22 mg/m^3, 1-hour	700 µg/m^3, annual
o-xylene	0.23	22 mg/m^3, 1-hour	700 µg/m^3, annual
formaldehyde	0.89	100 µg/m^3, 30-minute	-
$PM_{2.5}$	5.4	25 µg/m^3, 24-hour	-
NO_2	12.3	200 µg/m^3, 1-hour	-
CO	46	10 mg/m^3, 8-hour	-

3.3.1 Assessment of VOCs

All of the maximum predicted 1-hour average ground level concentrations for VOCs are well below the corresponding short-term assessment criteria. In fact, benzene, ethylbenzene and xylene are also well below the corresponding annual average guidelines.

It can be concluded that VOC emissions from the flaring of fracturing fluid should not result in adverse effects beyond the flare pit.

3.3.2 Assessment of Formaldehyde

The New Zealand guideline for formaldehyde of 100 µg/m^3 is given as a 30-minute average. The model prediction for this compound can be converted to a 30-minute average by following the procedure described in the MfE Good Practice Guide, which is to multiply by a factor of 1.15. This gives a maximum predicted level of 0.89 µg/m^3, which is still well below the guideline and should not result in adverse effects beyond the flare pit.

3.3.3 Assessment of Combustion Gases

The maximum predicted concentration of NOx as a 1-hour average is 12 µg/m^3 based on the STNZ results. Even making the conservative assumption that this a

100% NO_2, this level is well below the New Zealand guideline of 200 µg/m^3 as a 1-hour average.

The New Zealand guideline for CO is 10 **mg**/m^3 as an 8-hour average. The maximum predicted concentration of 9 µg/m^3, even as a 1-hour average, is well below this.

3.3.4 Assessment of Particulate

Using the STNZ results for TSP (total suspended particulate) gives a maximum predicted concentration of 5.4 µg/m^3 as a 1-hour average. The WHO PM2.5 guideline is a 24-hour average, which is difficult to estimate from a 1-hour predicted average. However, even making the conservative assumption that the 24-hour average is the same as the 1-hour average, this is still well below the guideline value of 25 µg/m^3.

As discussed previously, the mass emission rate estimated from the US EPA emission factors is about 25% of the estimate from the STNZ results. Using this value gives a maximum predicted concentration of 1.3 µg/m^3.

4 Discussion and Limitations

Maximum predicted concentrations from the screening study are well below the relevant guidelines and standards for the pollutants assessed. Many of the assumptions that were made to determine ground level concentrations are conservative or highly conservative, so it is reasonable to conclude that the disposal of fracturing fluid by flaring should not result in any adverse effects beyond the well site.

The Taranaki Regional Air Quality Plan (Taranaki Regional Council, 2011) includes rules that cover discharges from exploration well sites. One requirement is that the well must be at least 300m from the nearest dwelling.

Some limitations apply to this study as follows. However, these are unlikely to affect the above conclusion.

1. Some of the data used in the study was from an actual well (Turangi B) in Taranaki. While some parameters may be different at other well sites, the conclusion that there will not be adverse effects should remain valid.
2. The accuracy of the methods used to determine mass emission rates is not very high. Estimates were made where possible using more than one method and the highest emission rate was used in the assessment. With the exception of formaldehyde, the agreement between the methods was good considering the likely inaccuracies.
3. Flat terrain was assumed in the dispersion modelling. As the maximum ground level concentrations occurred reasonably close to the flare pit (615 m downwind), then it is unlikely that typical topography will significantly affect the model results.

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5 Appendices

5.1 Appendix A: *Flares model settings*

Model option	**Value or setting**	**Notes**
Fuel gas mass flow rate	1741 kg/h	Maximum from well log
Excess air	250%	Model default
Reference oxygen	3%	Model default
Flare smoking type	Light smoking	Conservative setting as no smoke observed
Emission factor for VOC	AP-42 default	Model default
Stream composition	CO2: 5% CH4: 90% C2H6: 5%	Typical of NZ natural gas
Air composition	Model default	

```
----------------------------------------------------
Flares is copyright Enviroware srl (2012)
             http://www.enviroware.com
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File created on 23/11/2012 11:28:21 a.m.

-----------------------
SUMMARY OF INPUT VALUES
-----------------------
Fuel gas stream mass flow rate:  1741.00 kg/h
Excess air: 250.0 %
Reference oxygen: 3.0 %
Flare smoking type: LightSmoking
Emission factor for VOC: AP42 value used

Stream composition
H2S: 0 %
H2: 0 %
CO2: 5 %
N2: 0 %
NH3: 0 %
CH4: 90 %
C2H6: 5 %
C3H8: 0 %
iC4H10: 0 %
nC4H10: 0 %
iC5H12: 0 %
nC5H12: 0 %
nC6H14: 0 %
C6H6: 0 %
C7H8: 0 %
C8H10: 0 %
oC8H10: 0 %
nC7H16: 0 %
nC8H18: 0 %
nC9H20: 0 %
nC10H22: 0 %
nC11H24: 0 %
nC12H26: 0 %
nC15H32: 0 %
COS: 0 %
CS2: 0 %
CH4S: 0 %
C2H6S: 0 %
C3H8S: 0 %
C4H10S: 0 %
C2H6S2: 0 %
C6H14S: 0 %
S: 0 %
O2: 0 %
H2O: 0 %

Air composition
N2: 77.3 %
O2: 20.7 %
CO2: 0.031 %
Ar: 0.934 %
```

```
H2O: 1.035 %

---------------------
OUTPUT VALUES: STREAM
---------------------
Molar weight: 18.14
VOC mass flow rate: 1529.8 kg/h
Normal density: 0.810 kg/Nm3
Volume flow rate: 2149.7 Nm3/h
Heat released: 21157.9 kW
Lower Explosive Limit: 5.085 %
Upper Explosive Limit: 17.511 %
Mols of C: 1.050000
Mols of H: 3.900000
Mols of O: 0.100000
Mols of N: 0.000000
Mols of S: 0.000000

-----------------------
OUTPUT VALUES: FLUE GAS
-----------------------
Molar weight: 28.52
Units of flue gas produced for each unit of fuel gas: 54.287
Mass flow rate: 94513.1 kg/h
Normal density: 1.273 kg/Nm3
Volume flow rate: 74227.5 Nm3/h
Water content: 5.647 % vol
Oxygen content in dry flue gas: 15.539 % vol
Dry Volume flow rate: 70035.6 Nm3/h
Dry Volume flow rate @ reference O2: 21248.1 Nm3/h
SO2 concentration: 0.000 mg/Nm3
Dry SO2 concentration: 0.000 mg/Nm3
Dry SO2 concentration @ reference O2: 0.000 mg/Nm3
SO2 emission rate: 0.000 g/s
Dry NOX concentration @ reference O2: 104.817 mg/Nm3
NOX emission rate: 0.619 g/s
Dry CO concentration @ reference O2: 570.328 mg/Nm3
CO emission rate: 3.366 g/s
Dry VOC concentration @ reference O2: 215.800 mg/Nm3
VOC emission rate: 1.274 g/s
Dry PM concentration @ reference O2: 40.000 mg/Nm3
PM emission rate: 0.236 g/s

-------------------------------------------------------------------
OUTPUT VALUES: EFFECTIVE STACK PARAMETERS - SCREEN3 EPA METHODOLOGY
-------------------------------------------------------------------
Heat lost radiating: 55.0
Sensible heat: 2274060.0 cal/s
Buoyancy flux: 84.1 m4/s3
Effective diameter: 1.5 m
Additional geometrical height: 7.3 m
```

```
-------------------------------------------------------------
OUTPUT VALUES: EFFECTIVE STACK PARAMETERS - TCEQ METHODOLOGY
-------------------------------------------------------------
Heat lost radiating: 20.4
Sensible heat: 4020350.0 cal/s
Buoyancy flux: 148.8 m4/s3
Effective diameter: 2.0 m
Additional geometrical height: 0.0 m
```

5.2 Appendix B: *SCREEN 3 output*

```
11/29/12
                                                                      17:30:19
  ***  SCREEN3 MODEL RUN  ***
  *** VERSION DATED 96043 ***

 C:\Lakes\Screen View\Flare1.scr

 SIMPLE TERRAIN INPUTS:
    SOURCE TYPE            =        FLARE
    EMISSION RATE (G/S)    =      1.00000
    FLARE STACK HEIGHT (M) =       0.0000
    TOT HEAT RLS (CAL/S)   =     0.505688E+07
    RECEPTOR HEIGHT (M)    =       0.0000
    URBAN/RURAL OPTION     =        RURAL
    EFF RELEASE HEIGHT (M) =       7.3016
    BUILDING HEIGHT (M)    =       0.0000
    MIN HORIZ BLDG DIM (M) =       0.0000
    MAX HORIZ BLDG DIM (M) =       0.0000

 THE REGULATORY (DEFAULT) MIXING HEIGHT OPTION WAS SELECTED.
 THE REGULATORY (DEFAULT) ANEMOMETER HEIGHT OF 10.0 METERS WAS ENTERED.


 BUOY. FLUX =   83.845 M**4/S**3;  MOM. FLUX =   51.127 M**4/S**2.

 *** FULL METEOROLOGY ***

 **********************************
 *** SCREEN AUTOMATED DISTANCES ***
 **********************************

 *** TERRAIN HEIGHT OF    0. M ABOVE STACK BASE USED FOR FOLLOWING DISTANCES ***

   DIST     CONC             U10M   USTK  MIX HT   PLUME   SIGMA   SIGMA
    (M)   (UG/M**3)   STAB  (M/S)  (M/S)    (M)   HT  (M)  Y  (M)  Z  (M) DWASH
 -------  ----------  ----  -----  -----  ------  ------  ------  ------  -----
     50.  0.5382E-01    6     1.0    1.0 10000.0  115.25   27.24   27.19    NO
    100.  0.3196        6     1.0    1.0 10000.0  115.25   31.11   30.93    NO
    200.  0.3372        6     1.0    1.0 10000.0  115.25   31.80   31.11    NO
    300.   1.869        4    20.0   20.0  6400.0   33.41   23.05   12.90    NO
    400.   3.917        4    20.0   20.0  6400.0   33.41   29.95   16.21    NO
    500.   5.050        4    20.0   20.0  6400.0   33.41   36.69   19.35    NO
    600.   5.387        4    20.0   20.0  6400.0   33.41   43.31   22.37    NO
    700.   5.280        4    20.0   20.0  6400.0   33.41   49.82   25.30    NO
    800.   4.963        4    20.0   20.0  6400.0   33.41   56.13   27.92    NO
    900.   4.591        4    20.0   20.0  6400.0   33.41   62.38   30.50    NO
   1000.   4.213        4    20.0   20.0  6400.0   33.41   68.58   33.05    NO
   1100.   3.858        4    20.0   20.0  6400.0   33.41   74.73   35.02    NO
   1200.   3.551        4    15.0   15.0  4800.0   43.60   81.12   37.59    NO
   1300.   3.350        4    15.0   15.0  4800.0   43.60   87.16   39.43    NO
   1400.   3.159        4    15.0   15.0  4800.0   43.60   93.15   41.22    NO
   1500.   2.978        4    15.0   15.0  4800.0   43.60   99.10   42.98    NO
   1600.   2.809        4    15.0   15.0  4800.0   43.60  105.02   44.69    NO
   1700.   2.652        4    15.0   15.0  4800.0   43.60  110.91   46.37    NO
   1800.   2.506        4    15.0   15.0  4800.0   43.60  116.76   48.03    NO
   1900.   2.393        4    10.0   10.0  3200.0   62.50  123.14   51.02    NO
   2000.   2.317        4    10.0   10.0  3200.0   62.50  128.91   52.57    NO

 MAXIMUM 1-HR CONCENTRATION AT OR BEYOND    50. M:
    615.   5.392        4    20.0   20.0  6400.0   33.41   44.35   22.85    NO

  DWASH=   MEANS NO CALC MADE (CONC = 0.0)
  DWASH=NO MEANS NO BUILDING DOWNWASH USED
  DWASH=HS MEANS HUBER-SNYDER DOWNWASH USED
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 DWASH=SS MEANS SCHULMAN-SCIRE DOWNWASH USED
 DWASH=NA MEANS DOWNWASH NOT APPLICABLE, X<3*LB

     ***************************************
     *** SUMMARY OF SCREEN MODEL RESULTS ***
     ***************************************

 CALCULATION         MAX CONC     DIST TO    TERRAIN
  PROCEDURE         (UG/M**3)     MAX (M)     HT (M)
--------------    -----------    -------    -------
SIMPLE TERRAIN      5.392           615.         0.


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** REMEMBER TO INCLUDE BACKGROUND CONCENTRATIONS **
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5.3 Appendix C: *Fracturing fluid analysis results*